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Multifunctionality in agriculture: impact of hedgerows, grass strips and extensive grassland management on crops, regulating ecosystem services and biodiversity

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Content

Summary

Intensification of agriculture in the EU has boosted yield, food security and food quality, contributing to economic development and human well-being. On the other hand, land abandonment and intensification have resulted in a strong decline of farmland biodiversity. Seventy-six per cent of the Habitat Directive habitats linked to agro-ecosystems and 70% of the Habitat Directive species linked to agro-ecosystems have an unfavourable conservation status. In addition, the impact of agriculture on the ecosystem is often detrimental: nutrient surpluses and pesticides contaminate the surface water, intensive agricultural practices have led to soil degradation, compaction and erosion and agricultural activities account for 10% of total European greenhouse gas emissions. In order to put an end to further biodiversity losses and environmental degradation, measures have been proposed, for example in the European Common Agricultural Policy. These measures – in this research called nature-oriented measures – include the introduction of semi-natural elements and the inclusion of ecological principles in agricultural practices. It is expected that the costs related to nature-oriented measures will negatively affect farm income. Because most farmers already face economic challenges, it is crucial that the implementation of nature-oriented measures is effective and cost-efficient and that the efforts of the farmers are properly and fairly compensated.

Optimal implementation of nature-oriented measures requires insight into the simultaneous effects that can be expected, both on crop yield (a provisioning ecosystem service) and farm income as on the delivery of regulating and cultural ecosystem services and on biodiversity. The impact of a number of these measures on ecosystem services and biodiversity has already been studied, but generally within one study only one (or several related) response variables are evaluated at the time. Assessing multifunctionality by combining studies that focus on only one response variable may lead to overestimation of effects because monitoring sites are often selected to demonstrate a maximal impact of a measure on a specific response variable. Therefore, there is a need for research that considers multiple ecosystem service and biodiversity indicators simultaneously.

In this study, we assessed the simultaneous impact of three types of nature-oriented measures on one provisioning and multiple regulating ecosystem services and on biodiversity components. These nature-oriented measures include the implementation of i) hedgerows and ii) grass strips on arable field borders as well as iii) the extensification of grassland management. Crop yield, both biomass and quality, was the provisioning service and the regulating services that were considered are global climate regulation, chemical quality regulation of both surface and subsurface water, erosion regulation and natural pest control. Indicators for these regulating ecosystem services are respectively soil organic carbon sequestration, the interception of nitrogen and phosphorus from the surface and subsurface flow, the interception of soil particles from the surface water and the presence of natural predators. The considered indicators for biodiversity are the species number and species composition of plants and carabids.

By means of a meta-analysis, we quantified the effect relationship between hedgerow and grass strip characteristics and ecosystem service and biodiversity. Close to the hedgerow, until a distance of about twice the hedgerow height, crop yield was reduced by 21%, most probably as a result of competition for light and nutrients, and beyond this point until a distance of about 20 times the hedgerow height, crop yield was increased by 6%, potentially as a result of an improved microclimate. Near the hedgerow, until a distance of about four times the hedgerow height, soil organic carbon stock was increased by 8% compared to a parcel without a hedgerow. Also in the grass strip, soil organic carbon was 25% higher compared to the adjacent parcel. Both hedgerows and grass strips improved the water quality by the interception of nitrogen from surface (69% for

hedgerows, 76% for grass strips) and subsurface (34% for hedgerows, 32% for grass strips) water, phosphorus from the surface water (67% for hedgerows, 73% for grass strips) and reduction of erosion (91% for hedgerows, 90% for grass strips). Hedgerows increased the number of predator species, but not predator abundance. On parcels with grass strips, both the number of predator species and predator abundance were increased. Parcel-level estimations show that the trade-offs between provisioning and regulating ecosystem services and biodiversity primarily depend on dimensions of the hedgerow, grass strip and parcel.

Next, we monitored a set of ecosystem service and biodiversity indicators on arable parcels in Flanders with a hedgerow or grass strip along at least one of the field borders. Near the hedgerow, crop yield was reduced and thousand grain weight, soil organic carbon stock and activity-density of spiders were increased compared to further in the field. In the grass strip, soil organic carbon stock was increased, soil mineral nitrogen content was reduced and we found a different carabid species composition and higher spider activity-density, compared to the adjacent parcel. We concluded that hedgerows and grass strips have the potential to deliver a broad set of ecosystem services and to enhance biodiversity, but that this potential is not always realized, among other as a result of local management.

Additionally, we assessed the effect of grassland management type and intensity on ecosystem service delivery and biodiversity. The considered management types were regular, intensive management, meadow bird management and botanical management. Yield, crude protein content and soil mineral nitrogen content were higher in the regular, intensively managed grasslands. The number of plant species was higher in the more extensively managed meadow bird and botanical grasslands. From a literature review, we derived the same effect relationship between management intensity and ecosystem service and biodiversity indicators, but additionally, we found a positive impact of animal fertilizer application on soil organic carbon stock.

Finally, we developed a calculation tool that allows the prediction of income losses related to the implementation of nature-oriented measures, both at parcel and at farm level. Our results were integrated in the tool in order to estimate the effect of nature-oriented measures on income loss, ecosystem services and biodiversity at farm level. We concluded that the agricultural landscape has the potential to contribute to a wide range of services, but this requires the uptake of nature-oriented measures in the farm management. We end by formulating recommendations for further research, management and policy.

Samenvatting

Intensivering van de landbouw in de EU heeft gezorgd voor een enorme toename van de opbrengst, voedselzekerheid en voedselkwaliteit, wat heeft bijgedragen tot economische ontwikkeling en algemeen welzijn. Aan de andere kant hebben uitbreiding en intensivering van de Europese landbouw geleid tot een sterke afname van de biodiversiteit in het landbouwlandschap. Momenteel bevinden 67% van de onderzochte Habitatrichtlijnhabitats en 70% van de onderzochte Habitatrichtlijnsoorten zich in een ongunstige staat van instandhouding. Daarnaast oefent de landbouwsector een grote druk uit op onze leefomgeving: overschotten van nutriënten en pesticiden vervuilen onze oppervlaktewaters, intensieve landbouwmethoden zorgen voor een daling van de bodemkwaliteit, voor bodemverdichting en erosie, en landbouw is verantwoordelijk voor 10% van de broeikasgassen in Europa. Om zowel de negatieve impact op het milieu als verdere biodiversiteitsverliezen tegen te gaan, werden onder andere vanuit het Europese Gemeenschappelijk Landbouwbeleid (GLB) maatregelen genomen. Onder die maatregelen – in dit onderzoek naar gerefereerd als natuurgeoriënteerde maatregelen - bevinden zich de introductie van kleine landschapselementen en het integreren van ecologische principes in bestaande landbouwpraktijken. Er wordt echter verwacht dat, binnen de courante landbouwbedrijfsvoering, deze maatregelen zullen leiden tot een daling van het bedrijfsinkomen. De meeste landbouwers staan reeds voor een aantal economische uitdagingen en daarom is het belangrijk dat de implementatie van deze maatregelen gebeurt op een effectieve en kostenefficiënte manier en dat de inspanningen van de landbouwers op een rechtvaardige manier gecompenseerd worden.

Een optimale implementatie van natuurgeoriënteerde maatregelen veronderstelt inzicht in de verschillende effecten die tegelijkertijd kunnen verwacht worden, zowel op opbrengst (een producerende ecosysteemdienst) en bedrijfsinkomen als op de levering van regulerende en culturele ecosysteemdiensten en biodiversiteit. Natuurgeoriënteerde maatregelen beïnvloeden onze leefomgeving door de levering van (regulerende) ecosysteemdiensten. Een groot aantal studies onderzocht reeds de impact van natuurgeoriënteerde maatregelen in de landbouw op de levering van individuele ecosysteemdiensten en op biodiversiteit, maar zelden werden meerdere responsvariabelen simultaan bemonsterd. Wanneer dan de integrale impact van de maatregelen bepaald wordt door de verwachte effecten per individuele responsvariabele samen te voegen, kan dit leiden tot een overschatting van de uiteindelijke multifunctionaliteit, aangezien studies vaak ontworpen worden en proeflocaties vaak gekozen worden om een maximaal effect aan te tonen. Er is dus onderzoek nodig waarbij meerdere ecosysteemdiensten- en biodiversiteitsindicatoren simultaan worden opgemeten.

In dit onderzoek beschrijven we de impact van drie natuurgeoriënteerde maatregelen op een producerende en verschillende regulerende ecosysteemdiensten en op de biodiversiteit. Deze maatregelen zijn de implementatie van i) houtkanten en ii) grasstroken langsheen de perceelsgrenzen van akkerbouwpercelen en iii) de extensivering van graslandbeheer. Gewasopbrengst, zowel biomassa als bepaalde eigenschappen, was de producerende ecosysteemdienst en de regulerende ecosysteemdiensten die we beschouwen zijn regulering van het klimaat, regulering van de chemische kwaliteit van het oppervlakte- en het grondwater, erosievermindering en natuurlijke plaagbestrijding. Indicatoren voor deze regulerende ecosysteemdiensten zijn respectievelijk koolstofopslag in de bodem, het afvangen van stikstof en fosfor uit het oppervlakte- en grondwater, het afvangen van bodempartikels uit het oppervlaktewater en de aanwezigheid van natuurlijke plaagbestrijders. De beschouwde indicatoren voor biodiversiteit zijn het aantal soorten en de soortensamenstelling van planten en loopkevers.

Op basis van een meta-analyse koppelden we een aantal karakteristieken van houtkanten en grasstroken kwantitatief aan de indicatoren voor ecosysteemdiensten en biodiversiteit. Dichtbij de houtkant, tot een afstand van ongeveer twee keer de hoogte van de houtkant, was de gewasopbrengst gereduceerd met 21%, waarschijnlijk als een gevolg van competitie voor licht en nutriënten, en voorbij dit punt tot een afstand van ongeveer 20 keer de hoogte van de houtkant, was de gewasopbrengst verhoogd met 6%, vermoedelijk als een gevolg van een gunstiger microklimaat. Nabij de houtkant, tot een afstand van ongeveer vier keer de houtkant, was het koolstofgehalte van de bodem 8% hoger in vergelijking met een soortgelijk perceel zonder houtkant. Ook in de grasstrook was het koolstofgehalte 25% hoger in vergelijking met het aangrenzende akkerbouwperceel. Zowel houtkanten als grasstroken droegen bij tot de waterkwaliteit door het afvangen van stikstof uit het oppervlaktewater (69% voor houtkanten, 76% voor grasstroken) en grondwater (34% voor houtkanten, 32% voor grasstroken), fosfor uit het oppervlaktewater (67% voor houtkanten, 73% voor grasstroken) en door het verminderen van erosie (91% voor houtkanten, 90% voor grasstroken). Op percelen met houtkanten was het aantal soorten natuurlijke plaagbestrijders verhoogd, maar niet het aantal plaagbestrijders. Op percelen met grasstroken was zowel het aantal soorten als het aantal plaagbestrijders verhoogd. Na het doorrekenen van deze effecten op perceelsniveau, concludeerden we dat de ‘trade-off’ tussen producerende en regulerende ecosysteemdiensten en biodiversiteit vooral bepaald wordt door de afmetingen van de houtkant, grasstrook en het perceel.

Vervolgens hebben we een aantal indicatoren voor ecosysteemdiensten en biodiversiteit opgemeten op Vlaamse akkerbouwpercelen met ofwel een houtkant ofwel een grasstrook langsheen de perceelsgrens. Dichtbij de houtkant was de gewasopbrengst verlaagd en waren duizendkorrelgewicht, bodemorganische koolstof en de activiteit-densiteit van spinnen verhoogd, in vergelijking met verderop in het perceel. In de grasstrook was de bodemorganische koolstof verhoogd en de minerale stikstof verlaagd en vonden we een andere soortensamenstelling van de loopkevers en een hogere activiteit-densiteit van spinnen, in vergelijking met het aangrenzende perceel. We concludeerden dat houtkanten en grasstroken het potentieel hebben om een brede waaier van ecosysteemdiensten en biodiversiteit te verhogen, maar dat dit potentiaal in de praktijk niet altijd gerealiseerd wordt, onder andere als gevolg van lokaal beheer.

Ook hebben we gemeten wat het effect is van het type en intensiteit van graslandbeheer op de levering van ecosysteemdiensten en biodiversiteit. De verschillende types graslandbeheer waren regulier beheer, weidevogelbeheer en botanisch beheer. Opbrengst, ruw eiwitgehalte en minerale stikstof in de bodem waren hoger in de reguliere en intensief beheerde graslanden en we vonden meer plantensoorten in de extensieve graslanden met weidevogel- of botanisch beheer. Uit een literatuurstudie konden we dezelfde relaties tussen beheerintensiteit en ecosysteemdienst- en biodiversiteitsindicatoren afleiden, maar daarnaast vonden we ook een positief verband tussen de toepassing van dierlijke bemesting en bodemorganische koolstof.

Vervolgens hebben we een rekenkader ontwikkeld dat toelaat om de inkomstverliezen gerelateerd aan de toepassing van natuurgeoriënteerde maatregelen op perceels- en bedrijfsniveau te voorspellen. Ten slotte hebben we alle resultaten geïntegreerd om op bedrijfsniveau een inschatting te maken van zowel inkomstverlies als effecten op regulerende ecosysteemdiensten en biodiversiteit. We concluderen dat een landbouwlandschap kan bijdragen tot een brede waaier van diensten, maar om dit te realiseren is het nodig dat landbouwers natuurgeoriënteerde maatregelen opnemen in hun bedrijfsvoering. We eindigen met een aantal aanbevelingen voor verder onderzoek en voor beleidsmakers.

List of abbreviations

AC	Alley cropping
AES	Agri-enviornment schemes
AIC	Akaike Information Criterion
AN	Ammonium-Nitrate
ANB	Agency for Nature and Forest
BAU	Business-as-usual
BD	Bulk density
BVA	Bos van Aa
C	Carbon
CAN	Calcium ammonium nitrate
CAP	Common Agricultural Policy
CICES	Common International Classification of Ecosystem Services
CMS	Cattle manure slurry
CON	Grasslands under intensive agricultural management
EFA	Ecological Focus Area
ES	Ecosystem services
EU	European Union
FYM	Grasslands under meadow bird management
GLB	Gemeenschappelijk Landbouw Beleid
GS	Grass strip
H	Relative distance from the hedgerow
HR	Hedgerow
IC	Inorganic carbon
INBO	Flemish Research Institution for Nature and Forest
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
LUI	Land Use Intensity
MAES	Mapping and Assessment of Ecosystems and their Services
MAP	Mest Actie Plan
MBD	Mineral bulk density
N	Nitrogen
NCP	Nature's Contributions to People
NMDS	Non-Metric Dimensional Scaling
NPC	Natural pest control
NPK	Nitrogen Phosphorus Potassium
OC	Organic carbon
OM	Organic matter
P	Phosphorus
PMS	Pig manure slurry
R	Response ratio
SO	Standard output
SOC	Soil organic carbon
SR	Sensitivity analysis
TC	Total carbon
TSS	Total suspended solids
TVG	Turnhouts Vennengebied
ZER	Grasslands under botanical management

1. Introduction

1.1 Agriculture in the European Union and Flanders: sector characteristics, environmental issues and policy context

1.1.1 Characterization

Agricultural land accounts for 48% of the European territory and 60% consists of arable land with crops and vegetables, 34% of permanent grassland and 6% of permanent crops such as orchards and vineyards (Eurostat, 2017a).

The average farm in the EU-28 had 16.1 ha of agricultural land in 2013. Big differences remain between the EU-15 (28.1 ha per holding) and the 13 countries that joined the EU in 2004 or later (7.8 ha per holding). Sixty-six per cent of European farms were smaller than 5 ha and only 7% of the farms had more than 50 ha of agricultural land (European Union, 2017a). The majority of the small farms is situated in eastern Europe, and generally, input intensity goes along with farm size (Figure 1.1)

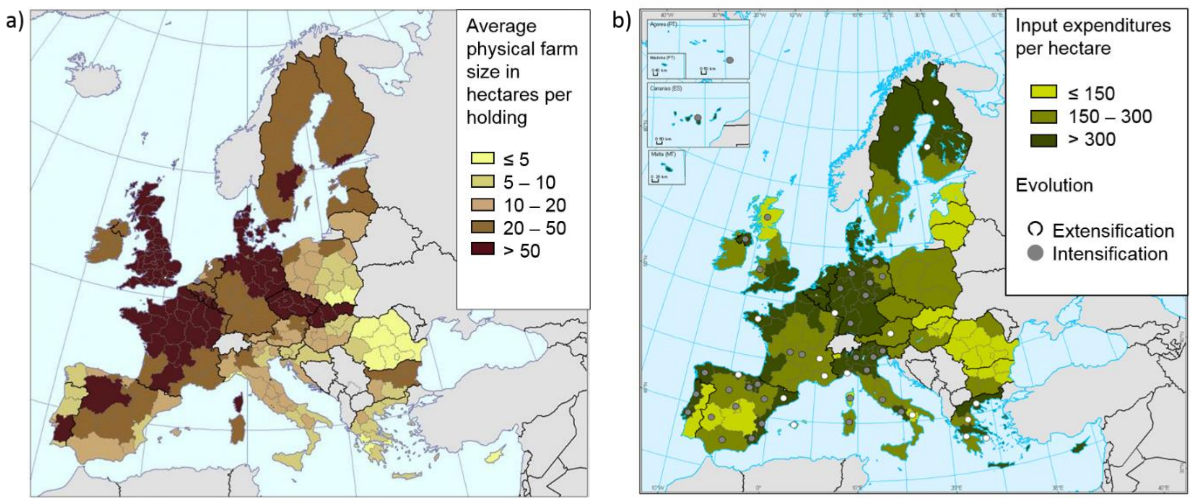

Figure 1.1. a) average physical farm sizes (ha) per agricultural region in 2013. Adapted from European Union (2017d) b) average expenditures (€ ha^{-1}) for inputs between 2005 and 2007. Inputs are fertilisers, crop protection products and purchased feeds. Extensification stands for a decrease of more than 15% and intensification stands for an increase of more than 15%, compared to the average of expenditures between 1995 and 1997. Adapted from Eurostat (2017)

Worldwide, the EU is among the greatest agricultural producers and wheat and other grain yields per hectare are the highest in the world. Also for sugar, beef, pig meat, poultry, sheep, milk, cheese and butter, the EU is ranked in the top 3 of the worldwide producers. Productivity of all crops is increasing over time, but the growth rate depends on the crop. For example, the annual growth rate of maize between 1993 and 2015 was 1.5%, while it was only 0.9% for rapeseed. However, in recent years, productivity increases have been stagnating: between 1993 and 2007, average productivity increase of wheat was 0.9% $year^{-1}$ in EU-15, between 2008 and 2015 it was 0.5% $year^{-1}$ and an annual growth of 0.4% between 2017 and 2025 is predicted (European Union, 2017b). This is due to modernization of technology and production systems of the past decades, resulting in yields already approaching the theoretical maximal yield (leaving little opportunities for further productivity increases), to a lower (allowed) use of fertilisers and to climate change (European Union, 2017b; Moore and Lobell, 2015).

The stagnating yield increase will most likely raise the pressure on farms, while they already face economic challenges. Poverty rates in the agricultural sector are among the highest in Europe and gross domestic product per capita in rural areas is only 73% of the overall EU average (European Union, 2017c). Over the last three years (2014-2016), farm income has decreased as a result of increasing production costs (energy, fertilisers) and agricultural prices that are lagging behind. Between 1997 and 2008, energy and fertiliser prices rose by 300%, while agricultural prices only increased by 29%. Over the last three years (2014-2016) average farm income declined by 8% (European Commission, 2016; European Union, 2017d).

In Flanders, the northern part of Belgium, 45% of the total territory consists of agricultural land. Farms are increasingly characterized by specialisation, intensification and increasing farm sizes, especially as a result of disappearance of small farms. Since 2001, average farm size increased from 16.2 ha to 25.5 ha in 2017. On the other hand, with its average size of 1.34 ha, parcels are still relatively small. Livestock farming dominates agriculture in Flanders (55% of the farms) and arable farming accounts for 25% of the farms. Grains, potatoes and sugar beets are the most important arable crops (Departement Landbouw en Visserij, 2014; FOD Economie, 2017).

1.1.2 Agricultural practices, rural biodiversity and environmental quality

Many species depend on agricultural landscapes for their habitats and food resources, and more specifically on traditionally managed, small-scale, extensive agricultural landscapes (Kleijn et al., 2011; Sutherland, 2004). However, since the 1950s, land abandonment on the one hand and intensification of European agriculture by increased use of chemical inputs, homogenization of the landscape, mechanisation, altered crop cycles and improved cultivars on the other hand, have resulted in a continuous decline of farmland biodiversity. For example, 76% of the habitats linked to agro-ecosystems have an unfavourable conservation status, mostly as a result of intensification and land abandonment. Twenty-five per cent of the mammals of European interest linked to agricultural landscapes, more than 80% of the amphibians and 42% of the birds are threatened (European Environment Agency, 2015).

Despite of declining greenhouse gas emissions and less pesticide use (since 1990), the impact of agriculture on the ecosystem has been mostly detrimental. In 2012, agricultural activities accounted for 10% of total European greenhouse gas emissions and pesticide and nitrate concentrations in groundwater exceeded the quality standards in respectively 7% and 13% of the monitoring stations (Eurostat, 2017c). Intensive agricultural practices have led to soil degradation, and in particular to water, wind and tillage erosion, a decrease in soil organic carbon, compaction, salinization, contamination and declining soil biodiversity (SoCo Project Team, 2009). Together with urbanization, agriculture is the main responsible for the pollution of freshwaters, and in 2015 only 53% of all freshwater bodies were in good ecological condition. This is mainly due to the loss of excessive nutrients and pesticides (Gilbert, 2015).

1.1.3 Policy responses

The European Common Agricultural Policy (CAP) was founded in 1962, originally as a response to the incapacitated agricultural sector, destroyed food markets and persistent tensions between countries after the second world war. As a result of modernization and price support, from the 1970s onwards, more food was produced than needed and in the early 1980s, the first production quota and set-aside programmes were introduced. In 1992, the CAP underwent a first transformation and support shifted from price support to direct payments related to area (for crop production) or headage (for livestock). Also, the first agri-environment schemes (AES) were introduced, providing compensations to farmers for voluntarily implementing specific environmentally friendly management practices. Current AES focus on water quality (via adapted fertilization or the implementation of grass strips along water courses), erosion control (via the implementation of grass strips or grassland), the management of (semi-)natural elements in the landscape like hedgerows, botanical management of grassland, protection of habitats of certain species and the development of several field margins in order to enhance pollination, protect vulnerable elements and create habitats. Since 2018, the Flemish government promotes a regional implementation of AES, in order to stimulate the uptake of measures where the greatest benefits are expected.

In 2003, the second major reform of the CAP consolidated the shift to income support and the payments were related with environmental protection, food safety and animal health and welfare conditions, the so-called cross-compliance conditions (Hill, 2012). The latest reform in 2014 aimed to tackle both economic, territorial and environmental challenges (European Commission, 2013a). As for the latter, since this last reform, 30% of the direct payments is now subject to greening requirements: i) the implementation of ecologically beneficial elements, so-called Ecological Focus Areas (EFAs) on 5% of the arable land, ii) crop diversification and iii) maintenance of permanent grassland. Within the constraints of a member state's specific list of options, farmers are free to choose how they fill in the EFAs, e.g. with hedgerows, buffer strips, alley cropping agroforestry, fallow land, nitrogen fixing crops, catch and cover crops. According to the ecological value of the chosen option, a conversion and weighting factor is used to convert the lengths/areas of the elements into equivalent focus areas: elements with a lower ecological value, will have a lower weighting factor compared to elements with a higher ecological value (e.g. hedgerows have a weighting factor of two). The greening of the CAP is meant to consolidate the incorporation of the delivery of environmental and climate benefits into the general agricultural activities and the main goal of the EFAs is 'to safeguard and improve biodiversity on farms'[1] (European Commision, 2017). Currently, the next reform of the CAP is debated on various levels and a wide consultation has been performed on simplification and modernisation of the CAP. The next reform of the CAP is aimed at tackling decreasing agricultural prices and market instability, the development of trade negotiations and the contribution to climate change commitments and the UN's Sustainable Development Goals (European Commission, 2017a).

Already before the actual implementation of the new CAP in 2014, the greening was perceived critically and it was feared that the real environmental benefits would be negligible: since the first greening proposals by the European Commission in 2010, negotiations and lobbying had severely weakened and diluted the original ambitions (Matthews, 2013; Pe'er et al., 2014). A recent report from BirdLife Europe and the European Environmental Bureau confirmed that the contribution of the greening to biodiversity conservation is negligible (Underwood and Tucker, 2016). In 2015, 70% of the EFAs comprised of nitrogen fixing crops and catch crops and in Flanders, this was even 85% (European Commision, 2017; VILT, 2016). It has been shown that under the current EFA management rules and regimes, both nitrogen fixing crops and catch crops will contribute very little to species diversity, mostly as a consequence of the use of fertilizers and pesticides, cutting regime and species selection. Also, the effect of land lying fallow would be minimal, because the fallow lasts only up to 8 months, while a minimum period of 1 year is needed to generate positive effects on biodiversity (Underwood and Tucker, 2016). The greatest biodiversity wins are expected from non-productive, permanent elements such as hedgerows, field margins and ponds (Underwood and Tucker, 2016; Westhoek et al., 2012) and this has been acknowledged in the EFA requirements by the introduction of the weighting and conversion factors (European Commission, 2014). Given de

[1] Recital 44 of Regulation (EU) No 1307/2013.

low adoption rate of these elements (European Commision, 2017), it seems though that the weighting and conversion factors do not suffice to encourage the implementation of elements with higher biodiversity value.

In order to protect water quality by limiting nitrate pollution from agricultural practices, the EU implemented the Nitrates Directive in 1991. The Nitrates Directive is part of the Water Framework Directive and is one of the key instruments to protect surface water and groundwater from agricultural pressures. Both for surface freshwaters as for groundwater, the maximum allowed nitrate concentration is 50 mg L^{-1} and Member States are required to set up action programmes to ensure that the threshold is not exceeded. In Flanders, this is formalized in the Mestdecreet or Mest Actie Plan (MAP). Currently, MAP 5 is in operation, with a focus on both nitrate and phosphate concentrations. Measures included in MAP 5 are related to among other, maximal fertilization intensity, application of fertilizer type and storage of manure (Vlaamse Gemeenschap, 2006; VLM, 2018a). Since 2018, a non-cropped strip next to all watercourses is required. In this strip, no tillage or application pesticides or fertilizers is allowed (VLM 2018).

Also in Flanders, the development of erosion control plans on the level of municipalities is stimulated via subsidies. This approach allows to take into account specific and local context. On fields that are very prone to erosion, farmers are obliged to take erosion control measures. These measures depend on the type of crop that is planted and are mainly related to the direction of ploughing and the fallow period. On fields with moderate erosion sensitivity, farmers are encouraged to take several measures like reduced tillage, direct sowing, the implementation of grass strips and hedgerows along field borders, etc. (Van Gossum, 2012)

Another instrument is the Natura 2000 network, composed of the Birds and Habitats Directives. The Birds Directive was founded in 1979 and had a sole focus on bird conservation. In 1992, the Habitats Directive was added, and this extends to both plants, animals and habitat types. For every Natura 2000 site, specific conservation targets and measures are developed. The network consists of areas that exclude human activities but also areas that depend on human management for their continued existence such as agricultural landscapes. Requirements for the agricultural sector in the Natura 2000 network are mostly related to nutrient input and more generally to the intensity of agricultural practices (European Commission, 2008). In Flanders, 12% of the total area was designated as Natura 2000 network and one third is in agricultural use. Like for every Member State, general conservation measures are defined for Flanders, indicating the minimum area and environmental quality required for successful continuation of both protected habitats and species. For every Natura 2000 site, specific conservation measures are prescribed both for habitats and species. To realise the required areas, the first focus is on the habitats that are already present within the site but if needed, land management or land use changes are recommended. For agricultural lands, prescribed management practices are mainly related to the prevention of excessive nutrient use (on grasslands) and losses (from arable lands) (ANB, 2017).

Apart from compulsory measures, several voluntary initiatives exist to realise ecological, environmental or landscape goals. In Flanders, partnerships between farmers (such as the so-called ‘agrobeheergroepen’) are setup on a broad range of topics such as farmland and meadow birds, botanic grassland management, hedgerow management, erosion etc. In this context, collaboration allows to work more efficiently and to have a bigger, regional impact (Defrijn et al., 2010).

1.2 Agriculture and the ecosystem

1.2.1 Ecosystem services

Human life is fundamentally dependent on and in strong interaction with the ecosystem. Ecosystem services are the structures, functions and processes that contribute to the human well-being. Human capital, built capital and social capital affect both directly and indirectly the benefits that can be derived from the ecosystem services (Figure 1.2) (Costanza et al., 2017).

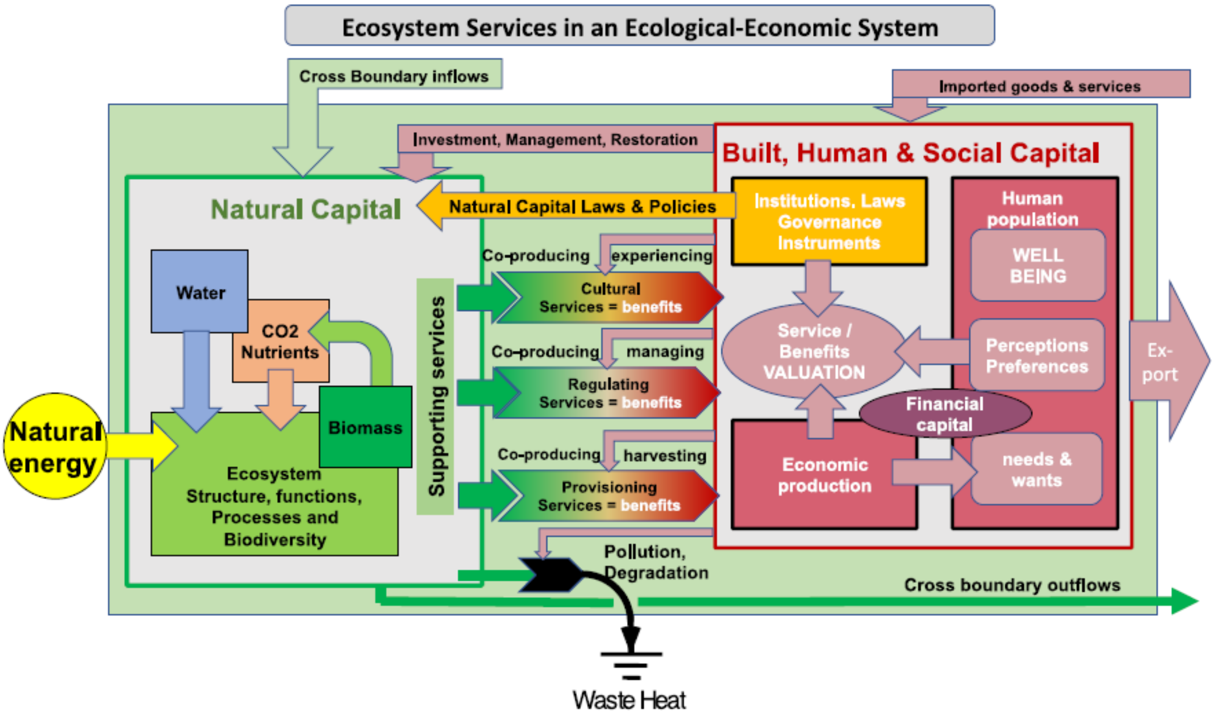

Figure 1.2: Simplification of the interactions between the ecosystem and human well being (Costanza et al., 2017)

A first categorization of ES into four broad types was proposed in the Millennium Ecosystem Assessment (MEA, 2005) and this structure was also used in The Economics of Ecosystems and Biodiversity (TEEB, 2010). These categories comprise:

- Provisioning ES: the material output of ecosystems
  E.g. food, fodder, water, wood, medicines
- Regulating ES: the regulating processes of ecosystems
  E.g. the regulation of local climate, air quality and water quality, carbon sequestration, erosion prevention, maintenance of soil fertility, pollination and biological control
- Supporting or Habitat ES are the basis of all other ES: habitats for species and maintenance of genetic diversity
- Cultural ES: the non-material benefits that can be obtained from ecosystems
  E.g. recreation, tourism, mental and physical health, spiritual experiences

In the Common International Classification of Ecosystem Services (CICES) (https://cices.eu/cices-structure/), provisioning, regulating and cultural ES are retained. Supporting ES are not included, and they are considered as a part of the ecosystem functioning, underlying the other ES. The benefits of the supporting ES are obtained indirectly via other ES.

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has introduced the term 'Nature's Contributions to People' (NCP) and classification of NCP is very similar to the abovementioned, comprising material contributions, non-material contributions and regulating contributions (IPBES, 2017). More than previous approaches of the ES concept, IPBES recognizes multidisciplinary and incorporates different scientific disciplines, stakeholders and knowledge systems. Additionally, the role of institutions and governance is included in the framework (Figure 1.3).

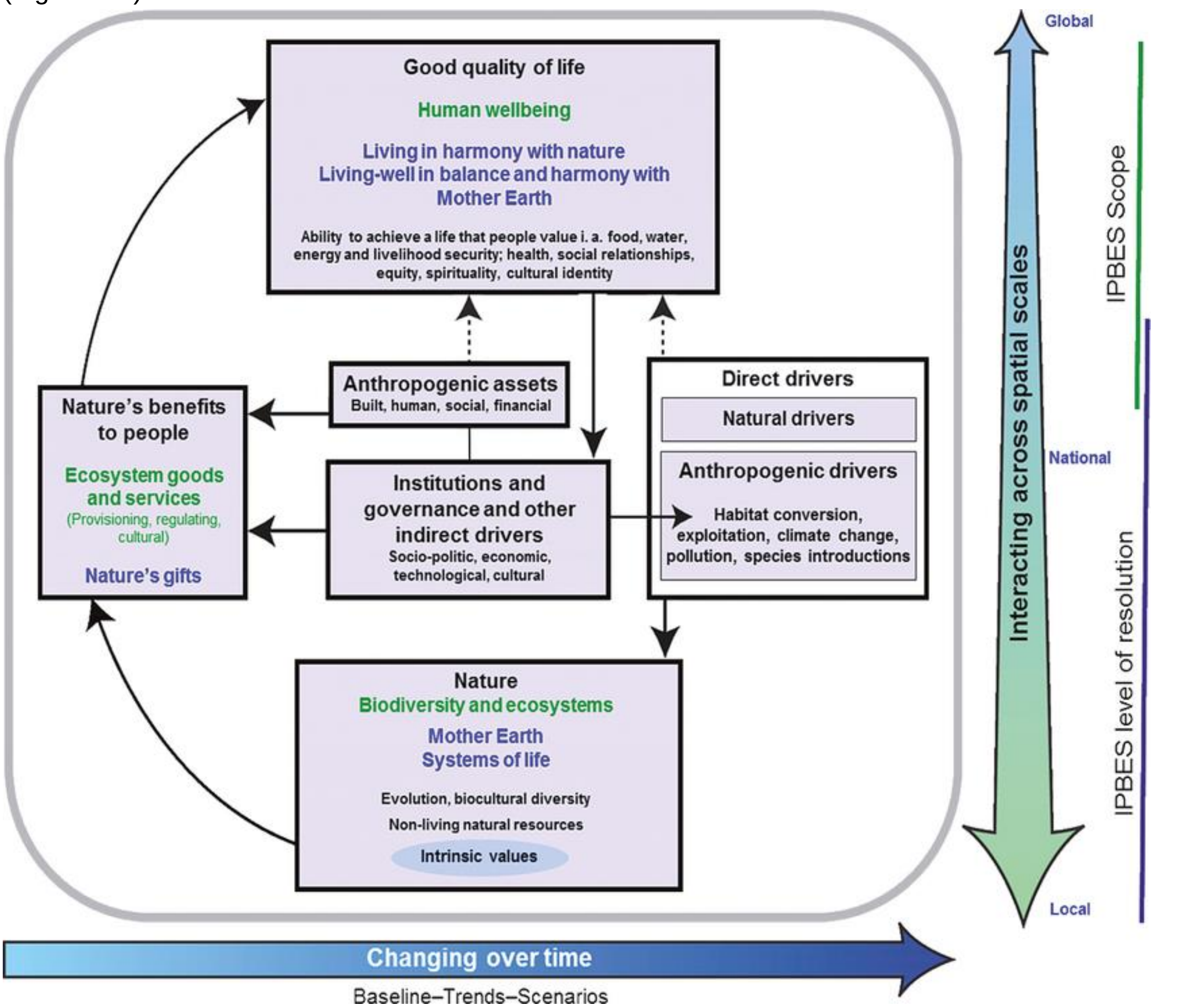

Figure 1.3: IPBES conceptual framework representing the main elements and their interlinkages. The different colours represent different knowledge systems (western science, indigenous and local and practitioners' knowledge) by the various stakeholders (Díaz et al., 2015).

In a world with limited space and resources, trade-offs in ecosystem management have to be made, involving different priorities held by different stakeholders. In order to reveal these trade-offs, the various benefits of ecosystems need to be valued. In this regard, the choice of valuation method is crucial. The valuation method has a major impact on the range and priorities of values that is perceived and different stakeholders wield different value systems (Costanza et al., 2017; IPBES, 2015; Jacobs et al., 2017, 2016). The holistic and multidisciplinary approach of the IPBES framework requires the consideration of multiple value systems, ranging from the individual to the community level, from short-term to long-term and from local to global. Generally, three types of values are discerned in the IPBES framework: intrinsic values, relational values and instrumental values (Díaz et al., 2015; IPBES, 2015). It has been shown that the choice of valuation method allows for the identification of various values and value types and that the use of a synthesising method offer the greatest chances for capturing a broad set of values (Jacobs et al., 2017).

1.2.2 Agriculture and ecosystem services

Intensification of agriculture has boosted yield, food security and food quality in the EU, contributing to economic development and human well-being (European Union, 2012; Hill, 2012). The rate of undernourishment in the EU is currently among the lowest in the world and strict producing and processing requirements are enforced to guarantee a high food quality (Hill, 2012; Our World in Data, 2018). Additionally, in many parts of the EU, agricultural activity lies at the basis of rural economic development (European Union, 2016). Because food security is predicted to be one of the major global challenges in the next decades, the agricultural production potential remains one of the main focusses of the EU agricultural policy (European Commission, 2017b).

Simultaneously, agricultural intensification has given rise to a broad range of environmental problems. The high fertilizer and pesticide use has caused a deterioration of the chemical water quality, up to 40% of all croplands are subject to severe erosion and reduced soil fertility and habitats for pollinators and natural pest predators are disappearing from the rural landscape. Both the EU and European citizens call for a more environment-friendly agricultural management (Figure 1.4) (European Commission, 2017b). EU initiatives to reduce the environmental impact of intensive agriculture can be found in, among others, the CAP (both in pillar I and pillar II) and the Natura 2000 network. Society's involvement is reflected in the growing success of organic agriculture, the short food supply chain and community structured agriculture (Canfora, 2016; Eurostat, 2018; Lang, 2010).

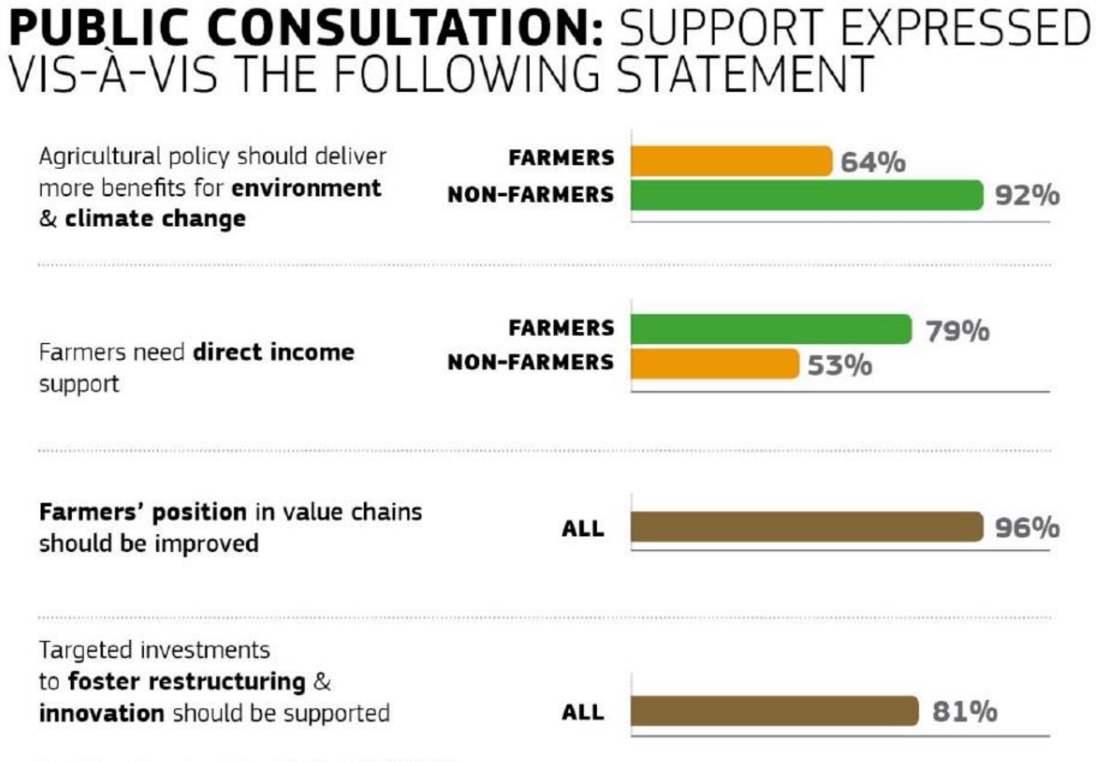

Figure 1.4: Presentation of some of the results of the public consultation 'The CAP: Have your say' (European Commission, 2017b)

Despite the current focus on provisioning services, agricultural landscapes can be designed and managed in such a manner that a broad range of ecosystem services, such as carbon sequestration and water quality regulation, can be delivered (Bennett et al., 2009; Power, 2010; Smukler et al., 2010). At the same time, this can address and minimize the adverse environmental side effects of agricultural practices (Power, 2010; Rey Benayas and Bullock, 2012). If agricultural production remains the main function of the land, the broadening of the delivered set of ecosystem services can be realized through the implementation of nature-oriented measures, or so-called ecological intensification, which entails the introduction of semi-natural elements in the agricultural landscape or by the uptake of ecological principles in the applied agricultural practices. The first option includes, among others, the implementation of vegetated field margins, isolated trees, hedgerows and flower strips. The second category consists of organic farming, environment-friendly practices, agroforestry systems etc. (Bommarco et al., 2013; Tittonell, 2014).

Agricultural landscapes, and more specifically extensive, traditional and/or small-scale agricultural landscapes are among the most important ecosystems for biodiversity. This is illustrated by the Natura 2000 network, which is for 31% composed of agricultural lands. Especially extensive agricultural landscapes are among the most important ecosystems for biodiversity. Semi-natural grasslands are the preferred habitat of more than two thirds of all butterfly species and almost 20% of Europe's endemic vascular plant species (Habel et al., 2013) and farmland acts as a breeding or winter habitat for nearly 120 bird species of European interest (Donald et al., 2001). However, agricultural intensification has resulted in great biodiversity losses and puts a threat on remaining species in agricultural landscapes. Among the measures to restore agricultural biodiversity are the introduction of semi-natural elements and the uptake of ecological principles (Kleijn et al., 2011).

Simultaneously, agricultural land use accounts for almost half of the European territory and, especially in densely populated areas, the agricultural landscape creates recreational and other non-material opportunities such as aesthetic beauty, cultural heritage, spirituality and inspiration. It has been shown that these benefits are higher in complex, traditional and/or extensive agricultural landscapes. For example, a landscape with more semi-natural elements is visually more attractive, and landscapes with a more diverse vertical structure, for example with hedgerows, were most preferred for outdoor activities (Assandri et al., 2018; Junge et al., 2015; van Berkel and Verburg, 2014).

Finally, agricultural activities and farm management contribute to farmers' identity and farmers attach value to, for example, the state of the farm and the land, fertility and quality of the soil, health of the animals, freedom and independency, contact with nature etc. Depending on farmers' preferences, different values will be obtained from these aspects and intensive agriculture will generate different values than extensive, traditional or ecological agricultural practices (Andersson et al., 2015).

In Table 1.1, an overview is given of the ES that can be obtained in an agricultural landscape. ES are classified according to CICES (version 5.1, https://cices.eu/resources/) and IPBES (IPBES, 2017).

Table 1.1: Overview of the ES that can be obtained in an agricultural landscape

ES section (in CICES)	CICES	IPBES	Example
		Habitat creation and maintenance	Botanical diversity in extensive grasslands
Regulating	Pollination	Pollination and dispersal of seeds and other propagules	Contribution to the yield of fruit crops
Regulating	Regulation of chemical composition of atmosphere and oceans	Regulation of air quality	Sequestration of carbon in biomass and soil
Regulating	Smell and noise reduction		Shelter belts that filter particulates that carry odours
Regulating	Wind protection		Hedgerows protecting crops from wind damage
Regulating	Regulation of the chemical condition of freshwaters by living processes	Regulation of freshwater quality	Use of buffer strips along water courses to remove nutrients in runoff
Regulating	Control of erosion rates	Formation, protection and decontamination of soils and sediments	The capacity of vegetation to prevent or reduce the incidence of soil erosion
Regulating	Weathering processes and their effect on soil quality		Inorganic nutrient release in cultivated fields
Regulating	Decomposition and fixing processes and their effect on soil quality		Decomposition of plant residue; N-fixation by legumes
Regulating	Pest control (including invasive species)	Regulation of organisms detrimental to humans	Reduction in pest damage to cultivated crop
Regulating	Disease control		Reduction in disease damage due to harvested fruit or vegetables
Provisioning	Cultivated plants (including fungi, algae) grown as a source of energy	Energy	Miscanthus
Provisioning	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	Food and feed	Harvested crops
Provisioning	Wild plants (terrestrial and aquatic, including fungi, algae) used for nutrition		Wild berries
Provisioning	Fibres and other materials from cultivated plants, fungi, algae and bacteria for direct use or processing (excluding genetic materials)	Materials and assistance	Timber
Provisioning	Seeds, spores and other plant materials collected for maintaining or establishing a population		Grass seed
Cultural	Characteristics of living systems that that enable activities promoting health, recuperation or enjoyment through active/passive or immersive/observational interactions	Physical and psychological experiences	Recreation, fitness; de-stressing or mental health; nature-based recreation
Cultural	Characteristics of living systems that are resonant in terms of culture or heritage	Supporting identities	Local identity

Cultural	Characteristics or features of living systems that have an option or bequest value	Maintenance of options	Rare species
Cultural	Characteristics of living systems that enable aesthetic experiences	Learning and inspiration	A beautiful landscape
Cultural	Characteristics of living systems that enable scientific investigation or the creation of traditional ecological knowledge		Participation in community-structured agriculture

1.3 Scope of this study

1.3.1 Selection of nature-oriented measures and ecosystem services

The EU Biodiversity Strategy to 2020 has called all member states to map and assess the state of the ecosystems and their services in their national territory. For this, a working group on Mapping and Assessment of Ecosystems and their Services (MAES) was set up. In this context, the Flemish Research Institution for Nature and Forest (INBO) has identified and assessed the status of 16 ES in Flanders. The ES that were considered were: production of food, game, wood, energy crops and water, pollination, pest control, maintenance of soil fertility, regulation of air quality, regulation of noise disturbance, regulation of erosion, regulation of flood risk, coastal protection, regulation of the global climate, regulation of air quality, regulation of water quality and a green environment for outdoor activities. Classification of the ES was based on CICES, adapted to the Belgian context (CICES-BE). Apart from pollination, all of these ES ware in an unfavourable state, meaning that the demand for each of the ES is higher than what could be supplied by the ecosystem (Jacobs et al., 2014b; Van Reeth et al., 2014).

This study focusses on a subset of provisioning and regulating ES and biodiversity aspects. More specifically, crop production was retained as the provisioning ES and pest control, regulation of erosion, regulation of the global climate and regulation of water quality were the selected regulating ES. Crop production was retained because food security remains a priority in the European CAP. Selection of regulating ES was based on both European, Flemish and local concerns (see section 1.1.3, section 1.5 and description of the study areas in chapter 3 and chapter 4) and hypothesized parcel-level and short-term effects of the introduction of nature-oriented measures.

This study does not take into account any of the cultural ES. However, if a truly holistic and multidisciplinary approach as proposed by IPBES is desired, the range of considered ES needs to be broadened. Cultural ES and all relevant provisioning and regulating ES and stakeholders on a local and community level ideally should be included. In this context, the current study aims to be a first stepping stone towards the future development of a full assessment framework for ecosystem services by nature-oriented measures in agriculture.

1.3.2 Research gaps

Both in the context of the AES, the EFAs and Natura 2000, nature-oriented measures were originally introduced to enhance biodiversity (or at least put an end to further losses) or to reduce environmental pressure of agricultural practices. Several studies have already aimed at investigating the impact of nature-oriented measures on biodiversity. The results were patchy. In the review of

Kleijn and Sutherland (2003), the contribution of AES to botanical diversity was only positive in only 55% of the studies, the effect on arthropod diversity was more consistent (a positive effect in 70% of the studies) and an improvement of bird species richness of abundance was only recorded in 45% of the studies. The review of Batáry et al. (2015) showed a general moderate positive effect of AES on species diversity and abundance of common species.

Despite the great potential of nature-oriented measures for ecosystem service delivery, we still lack insight into the impact of nature-oriented measures on multiple ecosystem services (Batáry et al., 2015). Many studies have demonstrated the effect on a single ecosystem service, but this approach has been questioned and several authors (Gamfeldt et al., 2008; Hector and Bagchi, 2007; Reiss et al., 2009) call for a more holistic approach considering multifunctionality and complementarity among species and among ecosystem services. More concrete, **there is a need for more insight into the simultaneous response of both biodiversity and multiple ecosystem services to nature-oriented measures and into potential synergies and trade-offs** (Bommarco et al., 2013; Bullock et al., 2011; Reiss et al., 2009).

It is expected that costs related to nature-oriented measures, and more specifically to non-productive elements or a reduction of fertilizer input, will negatively affect regular yields. At the same time, they may result in a diversification of farm products and improve the quality and sustainability of the agro-ecosystem, both enhancing economic resilience of the farm (Geertsema et al., 2016; Tittonell, 2014). It is crucial that (1) the implementation of nature-oriented measures in an agricultural context happens in an efficient and cost-effective way, with maximisation of related benefits, and (2) that the efforts of the farmers (in terms of production losses and/or resource investments) are either covered by an alternative, income-generating production.

Thus, apart from the identification of synergies and trade-offs in biodiversity and regulating ecosystem service delivery, the link with crop production and farm income remains to be assessed (Bommarco et al., 2013). The latter is especially important for the development of optimal implementation schemes at policy-level, as the economic consequences of the measures directly affect farm income and thus economic feasibility (Bommarco et al., 2013). To avoid far too severe trade-offs and in order to promote nature-oriented measures as a proper element of farm management, estimates of the full range of expected effects, including those related to farm income, is necessary.

1.4 Objectives and outline of the thesis

With this study, we aim to enhance the understanding of the simultaneous impact of a selection of nature-oriented measures on the delivery of multiple (both provisioning and regulating) ecosystem service and biodiversity, both at parcel level and at farm level, and under a variety of conditions. By doing so, we hope to contribute to the further establishment of the concept of agriculture and farming practices as a type of multifunctional land management, providing not only food, feed, fibre and fuel, but also a wide range of ecosystem services and biodiversity.

- Our first objective is to **quantify the potential impact** of nature-oriented measures for a chosen set of ecosystem services and biodiversity components. To do so, we perform a systematic literature review and determine quantitative effect-relationships between measure characteristics such as width and height and ecosystem service and biodiversity indicators.

- Second, we investigate whether this potential is realised **under real field conditions** by means of a monitoring campaign on fields in Flanders. Linked to this, we evaluate which factors might affect the actual impact of these measures, e.g. by identifying the role of local management.
- This monitoring campaign further serves to evaluate the **possibilities and limitations of field monitoring**, and more specifically of assessing more than one response variable at the same time.
- Subsequently, through the development and application of a calculation tool, we assess farm-level financial impact of the measures studied, under a range of potential scenarios.
- Ultimately, using the same tool but now integrating also the studied impacts on a range of ecosystem serviced and biodiversity, we assess multifunctionality and trade-offs at farm level.

An overview of the structure of this study is given in Figure 1.5.

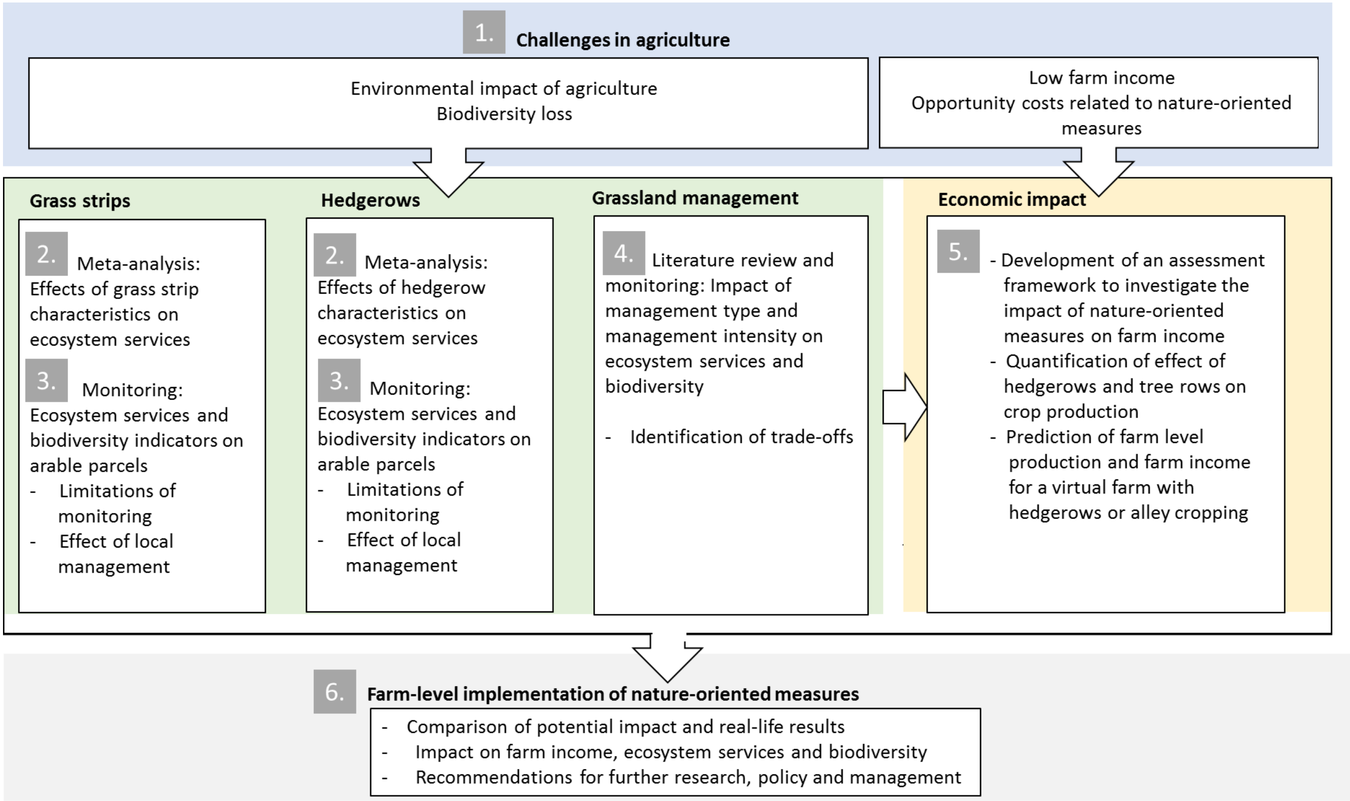

Figure 1.5: Outline of the thesis and the interactions between the various chapters. Chapter numbers are in the grey boxes.

We do this for three nature-oriented measures: the implementation of i) hedgerows and ii) grass strips on arable field borders and iii) extensification of grassland management. The first two fall into the category of semi-natural elements as described in section 1.2.2 and the third measure is based on the inclusion of ecological principles in agricultural management.

In the **first chapter**, we described the challenges the agricultural sector currently faces and we describe the measures that have been proposed to counter these challenges. We conclude that for an efficient and cost-effective implementation of the measures, more research both into the multiple environmental and biodiversity effects and into the economic impact is necessary.

In **chapter 2**, we quantitatively assess the impact of hedgerows and grass strips bordering parcels with annual arable crops on the simultaneous delivery of a set of ecosystem services and biodiversity indicators. To do so, a meta-analysis is performed and effect relationships linking hedgerows and grass strip characteristics to ecosystem service and biodiversity indictors are developed. From this we derive synergies and trade-offs on virtual parcels.

In **chapter 3**, we present the result of parcel-level, straightforward ecosystem service and biodiversity indicator measurements on Flemish arable parcels with either hedgerows or grass strips on the field borders. We question a multifunctionality and trade-off assessment that is based on the accumulation or combination of results from studies that focus on only one response variable. Also, we evaluate the possibilities and limitations of investigating more than one response variable at the same time and we identify the potential impact of local management on hedgerow and grass strip performance.

In **chapter 4**, the third nature-oriented measure, grassland management, is linked to ecosystem service delivery and biodiversity. Based on own monitoring data, the effect of management type (meadow bird targeted or biodiversity targeted) and management intensity (based on fertilization, mowing and grazing intensity) is related to the observed ecosystem service and biodiversity indicators. Second, after a literature review and data extraction, a general effect relationship between grassland management intensity and ecosystem service and biodiversity indicators is developed. This allows comparison of our own data to trends described by literature. Finally, we explored whether there are trade-offs in grassland management with respect to ecosystem service delivery and biodiversity and whether these trade-offs could be minimized.

In **chapter 5**, we develop an assessment framework to quantify the farm-level economic consequences of the implementation of nature-oriented measures. This is done by combining crop production information on tree-crop interactions with farm data. In this chapter, the assessment framework is applied in the context of the new CAP, with hedgerows and alley cropping being two EFA implementation options to increase the presence of permanent, woody vegetation in the agricultural landscape. The calculation tool that is developed in this chapter, will be used in chapter 6 to investigate multiple ecosystem service delivery and biodiversity at farm level.

Finally in **chapter 6**, results of the literature review and our own monitoring are compared to distinguish between potential impact and real-life impact of the measures. We also described the potential multiple ecosystem service delivery and biodiversity at farm level. Similar to chapter 5, this is performed in the context of the EFAs, for a varying set of hedgerows and grass strips. In this research, farm-level calculations are confined to hedgerow and grass strip scenarios because the calculation tool was developed specifically for measures that are implemented on a part of the parcel. We discuss the potential of the concept of ecosystem services for enhancement of multifunctionality in agriculture. Finally, we will formulate recommendations for further research, management and policy. By doing so, we hope to improve the applicability of our research and to facilitate a more easy implementation of the nature-oriented measures for the farmers.

1.5 Description of the study areas

The monitored parcels were situated in Flanders (see chapter 3 and chapter 4), the northern part of Belgium (Figure 1.6). To monitor ecosystem service and biodiversity indicators, four study regions were selected: one region for the monitoring of hedgerow impact, one for grass strip impact and two for extensive grassland management impact. Study region selection was based on prevalence of the measures (as AES) and on local environmental issues and hence relevance of these measures.

For hedgerow monitoring, West-Vlaams Heuvelland was selected, a hilly region in the south-west of Flanders. The area is characterized by a diverse landscape with forests, springs, valleys and numerous semi-natural elements, among which hedgerows. About 50% of the total area is currently in agricultural use. Environmental problems related to agriculture are eutrophication, acidification, erosion and disappearance of the semi-natural elements. Hedgerows play an important role as corridors between the forest patches (Bot et al., 2010).

Grass strips were monitored in the Polder region in the north of Flanders. This region is completely flat and land use is predominantly agriculture. Within the few natural areas that remain and in the agricultural matrix, valuable vegetation types (especially permanent grasslands) offer important habitats for a number of important (bird) species as a breeding ground or wintering place. Main threats coming from agriculture are a lowering of the water level, a deterioration of the water quality as a result of leaching and drift of fertilizers and ploughing of the grasslands (Loose and Bot, 2011).

Monitored grasslands were situated in Turnhouts Vennengebied in the Campine region in the north of Flanders and in Bos van Aa, located centrally in Flanders. Turnhouts Vennengebied and Bos van Aa are both designated as a Natura 2000 site. Turnhouts Vennengebied is a nature reserve with a total size of 285 ha and the area is characterized by species rich oligotrophic ponds and wed and try heathlands. Main threats for the area are caused by intensive farming, and more specifically eutrophication, acidification, intensive mowing and change of the water level (Natuurpunt, 2011). The area is managed by Natuurpunt, the largest nature conservation organisation in Belgium and by the Flemish Agency for Nature and Forest (ANB). Both organisations grant concessions to farmers for the management of the grasslands under specific conditions. While ANB focusses on meadow bird management, Natuurpunt aims at enhancing both botanical diversity and meadow bird management in the grasslands. Bos van Aa has a total area of 113 ha. The nature reserve is owned by the Flemish Waterways and Sea Canal Agency and managed by Natuurpunt and ABC Eco², a non-profit organisation which stimulates and supports Flemish farmers to cooperate on landscape, nature, water and soil management. Because of its former use for sand extraction, the reserve has a high potential for the development of unique pioneer vegetations such as poor heathlands. Until only recently, the majority of the area was under regular grassland management. Environmental problems related to agriculture are eutrophication, acidification and a change of the water level (ANB, 2011). The main focus of Natuurpunt is to restore botanical diversity.

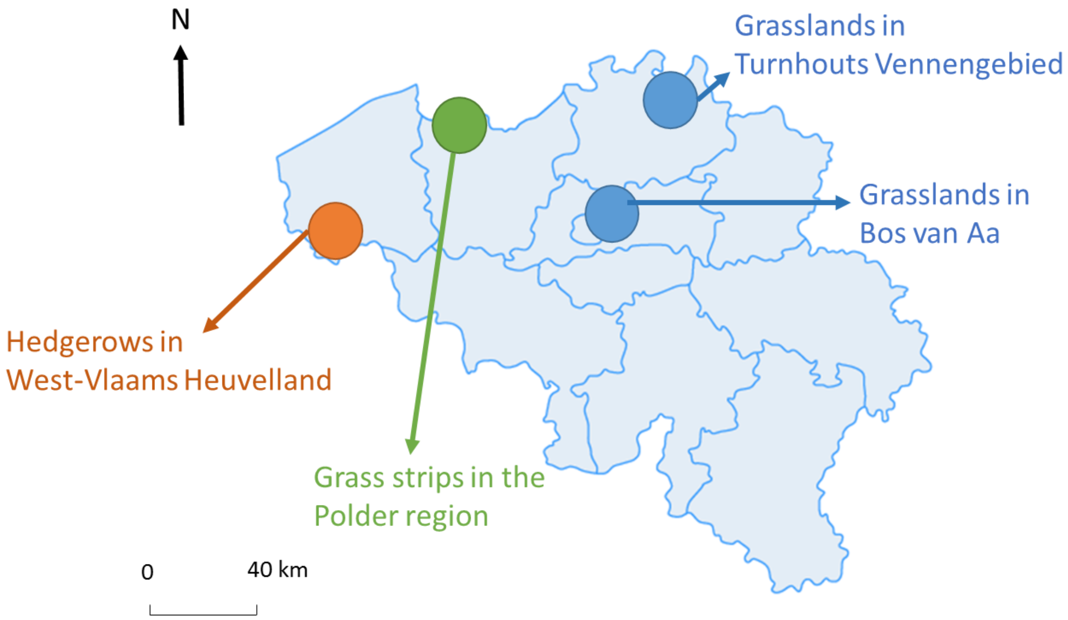

Figure 1.6: location of the monitored study areas in Flanders, Belgium. Study regions were selected based on prevalence of the nature-oriented measures and local environmental concerns

2. Ecosystem service delivery of agri-environment schemes: a synthesis for hedgerows and grass strips on arable land

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2.1 Abstract

In north western Europe, agricultural systems are managed to maximize the delivery of provisioning ecosystem services. This has been at the expense of other ecosystem services. Because the demand for most ecosystem services is increasing, multifunctionality in agriculture is vital. In this paper, we assessed the impact of hedgerows and grass strips on the simultaneous delivery of a set of ecosystem services and we identified synergies and trade-offs on virtual parcels. After a systematic literature search, mixed models were applied on observations from 60 studies and quantitative effect relations between ecosystem service delivery and hedgerow or grass strip characteristics were developed. Until a distance of two times the tree height, crop yield was reduced. Beyond this distance, until twenty times the tree height, crop yield increased. Soil carbon stock was higher in the hedgerow, in the adjacent parcel next to the hedgerow (until a distance of six times the tree height) and in the grass strip. Both hedgerows and grass strips removed nitrogen from the surface and subsurface flow and phosphorus from the surface flow. Erosion was reduced both by hedgerows and grass strips. More natural predator species were found on parcels with hedgerows, but the number of predators was unaffected. On parcels with grass strips, both predator density and diversity was higher and pests were reduced. Our calculations on parcel level indicate that effectiveness and trade-offs depend on hedgerow, grass strip and parcel characteristics.

2.2 Introduction

Agricultural systems are generally managed to maximize the delivery of provisioning ecosystem services (ES), such as food, forage, fibres, bioenergy and pharmaceuticals (Power, 2010). The pursuit of these services by agricultural intensification and expansion across the globe has resulted in high biodiversity loss (Tsiafouli et al., 2015) and ecosystem degradation (Foley et al., 2005; Ogle et al., 2005; Pimentel & Kounang, 1998). Like other ecosystems, agroecosystems have the potential to deliver multiple ES (Bennett et al., 2009) and to sustain a certain level of biodiversity (Rey Benayas and Bullock, 2012), but the focus on provisioning services has in many cases been at the expense of other services. Because the demand for most ES is increasing (Millennium Ecosystem Assessment, 2005) and in order to address the adverse side effects of intensive agriculture, a multifunctional land use and land management has been called for (Bennett et al., 2009; Gordon et al., 2010).

Measures have been proposed to combine agricultural production with the delivery of other ecosystem services and the conservation and restoration of biodiversity. Examples can be found in the agri-environment schemes in the context of the EU Common Agricultural Policy (Kleijn et al., 2011). Some of these measures imply the introduction of non-crop habitats in the agricultural landscape (Rey Benayas and Bullock, 2012). Extensive research on the effects of non-crop habitats on the delivery of individual ES and on biodiversity has been performed. For instance, Falloon et al. (2004) calculated that conversion of arable land to grass strips or hedgerows increases soil organic carbon (SOC) by 1.30% year^{-1} and 1.23% year^{-1}, respectively. In the review of Dorioz et al. (2006), high trapping efficiencies for nitrogen, phosphorus and sediment were reported for grass strips. Marshall et al. (2006) found that grassy field margins have a positive effect on abundance and diversity of plants, bees and grasshoppers. Holland & Fahrig (2000) concluded that landscape-level carabid diversity increases with the amount of woody field borders.

Despite the existing knowledge on the delivery of individual ES, there is an urgent need for an integrated evaluation of the simultaneous changes in multiple ES. This will allow us to identify synergies and trade-offs between services (Bennett et al., 2009) and is key for the optimization of the potential benefits of non-crop habitats on agricultural land (Power, 2010). Additionally, we need to examine the extent of the effect on ES delivery into the adjacent parcel and the role of vegetation and parcel characteristics such as hedgerow width or parcel slope. To address these research gaps, quantitative, spatial relationships that describe the effects of non-crop habitats on ES delivery need to be derived. In this study, we present an integrated overview of the effects of two types of non-crop habitats, i.e. hedgerows and grass strips, on multiple ES. These measures were selected because they entail parcel level interventions that can easily be adopted by individual land users, such as farmers, and because they are abundant and popular in numerous European and other temperate areas (Baudry et al., 2000; Marshall & Moonen, 2002). Ecosystem services considered were the provisioning ES crop yield, and the regulating ES were global climate regulation, regulation of chemical water quality, erosion regulation and pest regulation. We performed a systematic literature review and quantified the size of the effect of hedgerows and grass strips on the delivery of these ES using quantitative meta-analysis techniques. Next, we investigated the role of hedgerows and grass strip characteristics on ES delivery. Finally, we integrated the delivery of multiple individual ES into an overall assessment in order to identify trade-offs and describe the multifunctional role of hedgerows and grass strips on agricultural parcels.

2.3 Material & Methods

2.3.1 Definitions and scope of literature search

Hedgerows are defined here as unfertilized, perennial, linear, woody structures, established on agricultural field borders and consisting of shrubs and/or trees. Both hedges and tree rows are considered and we will investigate whether both hedgerow types affect the result differently. The distinction between hedges and tree rows is based on management; if the stems are pruned and thus branchless and if no shrubs are present under the trees, the row is considered as a tree row. Otherwise, the row is considered a hedge. Grass strips are defined here as linear areas that are never fertilized, sprayed, or tilled and consisting of perennial structures, established on agricultural field borders and consisting of graminoids, often in combination with other herbaceous species (but no woody species). Flower strips only consisting of annual species are not included given the focus on perennial elements.

The systematic literature search is performed conform the PRISMA guidelines (Moher et al., 2009) and the process is described in section 7.1. Candidate papers were selected for further reading based on their title and abstract, when they met the following criteria: i) the study region is situated within the temperate regions of the globe (as defined by Olson et al. (2001)), ii) empirical data of the indicator of interest are available (modelling studies are thus excluded), iii) true controls are present allowing indicator comparison with and without hedgerow or grass strip. Additionally, the reference lists of the retained studies were searched. If the experimental setup or data were unclear, additional information was searched for in other papers of the authors. When results were only given in figures, the data were extracted using WebPlotDigitizer v3.10 (Rohatgi, 2014).

2.3.2 Ecosystem service indicators

Hedgerow impact on **crop yield** is expressed as relative crop yield ($R_{HR\text{-}yield}$), which is the ratio of the crop yield influenced by the hedgerow to the crop yield without hedgerow influence. We only withheld yield data specifically linked to the distance from the hedgerow. If the distance was not specifically mentioned, we did not retain this data point. To allow comparison between different experiments, the distance is expressed in relative terms of the height of the hedgerow. For this, we use H, which is the ratio of the distance from the hedgerow to the height of the hedgerow. For example, a plot on a distance of 10 m from a 2-m-high hedgerow corresponds to a H value of 5. Hedgerow height was given in all studies. Own empirical measurements (see chapter 3 and section 7.2) from 2014 and 2015 on crop yield, were included in this dataset. Because we assumed no effect of grass strips on crop yield, apart from the arable surface loss, we did not further investigate this.

The indicator for global climate regulation is carbon sequestration, expressed as the relative **soil carbon stock.** On hedgerow parcels, an effect extending into the cropped area is expected and relative soil carbon stock is the ratio ($R_{HR\text{-}C}$) of the amount of carbon stored within the influenced parcel zone to the amount of carbon stored in the unaffected zone. Similar to crop yield, soil carbon data were related to H in order to estimate the effect size. If not given, hedgerow height was estimated based on species and age. Carbon stock in the grass strip was compared to carbon stock in the adjacent parcel ($R_{GS\text{-}C}$). Again, own measurements from 2014 and 2015 (see chapter 3 and section 7.2) were added to the hedgerow dataset. Preferably, carbon stock data were extracted from the retained papers. When only carbon concentration was given, data on bulk density and sampling depth were needed to calculate the stock. When bulk density was not given, this was estimated based on organic matter and mineral bulk density (see equation 2.1, Post & Kwon 2000). This was done for 4 out of 20 retained studies.

$$BD = \frac{100}{\frac{\%\,OM}{0.244}+\frac{100-\%OM}{MBD}} \qquad \textit{(Equation 2.1)}$$

BD stands for bulk density (g cm^{-3}), OM for organic matter and MBD for mineral bulk density. MBD typically has a value of 1.64 g cm^{-3} (Post and Kwon, 2000).

Water quality regulation was quantified as the amount of nitrogen and phosphorus interception from the water flow. **Nitrogen interception** was calculated based on the ratio of nitrogen inflow into the hedgerow or grass strip to the nitrogen outflow out of the hedgerow or grass strip (R_{HR-N} and R_{GS-N}). Surface and subsurface flow data were analysed separately. We used the same approach as Mayer et al. (2007) and thus did not distinguish among different N forms (e.g. ammonium, nitrate, etc.). **Phosphorus interception** was calculated based on the ratio of P inflow into the hedgerow or grass strip to the P outflow out of the hedgerow or grass strip (R_{HR-P} and R_{GS-P}). Because we found only one study reporting data from subsurface flow and most P is transported in the surface flow (Vought et al., 1995), we limited ourselves to surface flow data. We did not distinguish among different P forms. Section 7.3 shows N and P interception indicators for all N and P forms. The indicator for **erosion reduction** was calculated as the ratio of total suspended solids (TSS) inflow into the hedgerow or grass strip to the TSS outflow out of the hedgerow or grass strip (R_{HR-E} and R_{GS-E}).

Preferably, N, P and TSS mass was extracted from the papers. When flow volume and concentration were given, mass was calculated. If no mass or water volume was given, N, P and TSS concentrations were used in the analysis. If dilution would occur, the concentration in the outflow would decrease while the mass would remain unaffected. However, Mengis et al. (1999) showed that dilution is not the cause of lower concentrations in the outflow and that lower outflow concentrations are mainly the result of interception processes. Therefore, we assume that concentration and mass data will indicate the same trend. We realize that the use of concentrations could underestimate the interception of N, P and TSS, because water flow reduction in the outflow will increase the outflow concentration accordingly. All studies that were used for the calculation of N interception from the surface flow reported mass data (8 studies for hedgerow systems and 14 studies for grass strip systems). Phosphorus interception calculation is based on mass data for 8 studies out of 9 in hedgerow systems and on 14 studies out of 15 for grass strip systems. From the 9 and 19 studies describing erosion reduction in hedgerow and grass strip systems, respectively, all reported mass data. Nitrogen interception from the subsurface flow was mostly reported as concentrations (2 out of 3 studies for both hedgerow and grass strip systems), probably because subsurface flow hydrology is complex and difficult to quantify.

Regarding the ES **pest regulation**, we investigated both potential for natural pest control (NPC), based on increased predator abundance (predator density) and species number (predator diversity) and actual NPC, based on reduced pest abundance. When studying hedgerow and grass strip effects on predators, we considered species' seasonal dispersal patterns. Many predators use hedgerows and grass strips for hibernation (Holland et al., 2016), aestivation (Garcia et al., 2000) or as an additional food source during non-predatory life stages (Holland et al., 2016), followed by colonisation of arable fields (Geiger et al., 2009). Therefore, decreased or increased predator abundance can be found close to the hedgerow or grass strip or in the field centre, depending on the moment of sampling and species life cycle. For this reason, we did not include summer presence data gathered within the same parcel. Instead, we focused on i) predator overwintering within the hedgerow or grass strip vs overwintering in the adjacent parcels and ii) predator summer presence in parcels with hedgerow or grass strip versus predators in parcels without hedgerow or grass strip. This is comparable to the inclusion criteria used in the review of Poveda et al. (2008). Potential pest control was calculated as the ratio of predator density or diversity influenced by the hedgerow or grass strip to the predator density or diversity without hedgerow or grass strip influence (R_{HR-NPC} and R_{GS-NPC}). We did not distinguish among predator species. To investigate actual pest control, we compared pest abundance on parcels with hedgerows or grass strips to pest abundance on parcels without hedgerow or grass strip. Actual pest control is calculated as the ratio of pest density without hedgerow or grass strip influence to pest density with hedgerow or grass strip influence.

2.3.3 Effect modelling

We applied mixed-effect models to define an effect relationship for each hedgerow or grass strip and ES combination. In this relationship, the dependent variable was the natural logarithm of the ratios (ln(R)) as described above. The ln(R) is commonly used in ecological multilevel data analysis because this linearizes the ratio and thus ln(R) will be affected equally by changes in the numerator or denominator. Furthermore, ln(R) is more likely to be normally distributed, especially in small samples (Hedges et al., 1999). In the mixed models, “study” was always included as a random factor to account for potential pseudo-replication problems of data points from the same original publication. We did not perform a traditional, weighted meta-analysis because most studies did not report variances. Neglecting variances could negatively affect the preciseness of the result (Koricheva and Gurevitch, 2014). Therefore, when enough data with the required statistics were available, a mixed model as well as the traditional, weighted meta-analysis method was applied and compared. As represented in Appendix D, both approaches yield very similar results.

A general effect of hedgerow or grass strip presence on ES delivery is calculated with intercept-only mixed-effect models. Next, potential explanatory variables were added to the model on a one-by-one basis. Explanatory variables were hedgerow and grass strip characteristics and were individually tested for every ES. An overview of the tested variables is given in Table 2.1 and in Table 2.2.

Table 2.1: Model results for every hedgerow – ES combination. For significant model outcomes, 95% confidence intervals (between square brackets) and p-values of the according coefficients are given. The number of studies (N°) and the number of observations (n) used in every model is given. NS = non significant. Empty cells indicate that combination of the corresponding explanatory variable and ES is not tested because it was not relevant and no data were available in the retained studies.

	Crop yield	SOC stock	N interception: surface	N interception: subsurface	P interception	Erosion reduction	Pest regulation: predator density	Pest regulation: predator diversity
General effect Intercept-only model	**-0.16** [-0.22, -0.09] (p<0.001) N° = 11 n = 343	**0.14** [0.08, 0.19] (p<0.001) N° = 10 n = 80	**1.16** [0.93, 1.50] (p<0.001) N° = 8 n = 49	**0.42** [0.10, 0.75] (p=0.0112) N° = 3 n = 71	**1.12** [0.63, 1.62] (p<0.001) N° = 9 n = 36	**2.44** [1.21, 3.67] (p<0.001) N° = 9 n = 25	NS N° = 5 n = 13	**0.53** [0.06, 1.00] (p=0.0762)[2] N= 4 n= 10
Explanatory variables								
Distance $\alpha \times \log^2(H) + \beta \times \log(H) + \gamma$ $\delta \times e^{(\varepsilon \times H)}$	α: **-0.38** [-0.70, -0.06] (p=0.0210) β: **0.62** [0.28, 0.97] (p<0.001) γ: **-0.16** [-0.28, -0.05] (p=0.0059) N° = 11 n =343	δ: **0.20** [0.11, 0.30] (p=0.0820) ε : **-0.86** [-1.81, 0.10] (p=0.0001) N° = 10 n = 80						
Width $\alpha \times \log(\text{width}) + \beta$			α: **1.81** [0.35, 3.28] (p=0.0166) β: **-0.53** [-1.96, 0.89] (p=0.4545) N° = 8 n = 49	NS N° = 3 n = 71	NS N° = 9 n = 36	NS N° = 9 n = 25		
Type	NS N° = 11 n = 343	NS N° = 10 n = 80	NS N° = 8 n = 49	NS N° = 3 n = 71	NS N° = 9 n = 36	NS N° = 9 n = 25		
Crop type	NS N° = 11 n = 347							
Orientation	NS N° = 7 n =247							
Slope			NS N° = 8 n = 49	NS N° = 3 n = 71	NS N° = 9 n = 36	NS N° = 9 n = 25		

[2] Significant at the p=0.1 level

Age		NS N° = 5 n = 20	NS N° = 8 n = 49	NS N° = 3 n = 71	NS N° = 8 n = 31	NS N° = 9 n = 25
Inflow concentration			NS N° = 7 n = 48	NS N° = 3 n = 71	NS N° = 8 n = 35	NS N° = 7 n = 22
Sampling depth		NS N° = 10 n = 80		NS N° = 3 n = 71		

Table 2.2: Model results for every grass strip – ES combination. For significant model outcomes, 95% confidence intervals (between square brackets) and p-values of the according coefficients are given. The number of studies (N°) and the number of observations (n) used in every model is given. NS = non significant. Empty cells indicate that combination of the corresponding explanatory variable and ES is not tested because it was not relevant and no data were available in the retained studies

	SOC stock	**N interception: surface**	**N interception: subsurface**	**P interception**	**Erosion reduction**	**Pest regulation: predator density**	**Pest regulation: predator diversity**	**Pest regulation: aphid density**
General effect Intercept-only model	**0.22** [0.14, 0.30] (p<0.001) N° = 10 n= 108	**1.42** [0.88, 1.97] (p<0.001) N° = 14 n= 90	**0.38** [0.07, 0.70] (p=0.0204) N° = 3 n= 22	**1.30** [0.76, 1.84] (p<0.001) N° = 15 n= 116	**2.32** [1.66, 2.98] (p<0.001) N° = 19 n= 103	**1.53** [0.77, 2.29] (p=0.0075) N° =3 n=10	**0.90** [0.40, 1.40] (p=0.0074) N° =1 n=4	**-0.52** [-0.85, -0.19] (p=0.0121) N° = 2 n=6
Explanatory variables								
Depth α × log(depth) + β	α: **-0.29** [-0.39, -0.19] (p<0.001) β: **0.52** [0.37, 0.67] (p<0.001) N° = 10 n= 108							
Width α × width + β γ × log(width) + δ		α: **0.17** [0.13, 0.21] (p<0.001) β: **-0.25** [-0.94, 0.44] (p=0.4803) N° = 14 n= 90	γ: **0.50** [0.05, 0.96] (p=0.0307) δ: **0.08** [-0.61, 0.77] (p=0.8065) N° = 3 n= 22	α: **0.13** [0.09, 0.17] (p<0.001) β: **0.10** [-0.48, 0.69] (p=0.7246) N° = 15 n= 116	γ: **2.66** [1.59, 3.72] (p<0.001) δ: **0.02** [-1.10, 1.13] (p=0.9766) N° = 19 n= 103			
Slope		NS N° = 14 n= 90	NS N° = 3 n= 22	NS N° = 13 n= 99	NS N° = 18 n= 91			
Age	NS N° = 10 n= 108	NS N° = 9 n= 63	NS N° = 3 n= 22	NS N° = 8 n= 59	NS N° = 12 n= 63			

Inflow concentration $\alpha \times$ inflow + β		NS N° = 8 n= 54	α: **0.03** [0.00, 0.05] (p=0.0282) β: **0.15** [-0.22, 0.53] (p=0.4075) N° = 3 n= 22	α: **0.19** [0.07, 0.30] (p=0.0017) β: **1.14** [0.28, 1.99] (p=0.0099) N° =10 n=81	NS N° = 10 n= 66
Sampling depth			NS N° = 3 n= 22		

Tested hedgerow characteristics included H, width, type (hedge or tree row), orientation and age (in years). Tested grass strip characteristics were width and age. We also investigated whether the effect of hedgreow on crop yield was influenced by crop type, whether parcel slope or inflow concentration affected N interception, P interception or erosion reduction and whether sampling depth changed the effect on carbon stock and N interception from the subsurface flow. The crops that were covered in the retained studies are classified into four types: maize, cereals, leguminous crops and mustard. Orientation of the hedgerow ranged from north-south over northeast-southwest, northwest-southeast to east-west.

Based on Van Vooren et al. (2016) (see chapter 5), a polynomial function of the second order with $\log_{10}(H)$ as the explanatory variable was fitted to $\ln(R_{HR\text{-}yield})$. For spatial variables (H and width) related to the remaining ES, inclusion of the original data and of the data after $\log_{10}$ transformation was compared and based on the Akaike Information Criterion (AIC), the preferred model was selected. Because both in the hedgerow and grass strip case we retained only five studies to investigate NPC, the dataset is too limited for variable testing. Therefore, the analysis is restricted to the quantification of the general effect, calculated with the intercept-only model. Because the retained data on predator presence include different species, the model is extended and a multilevel random structure is included for species and study level.

2.3.4 Assessment of trade-offs and multifunctionality

The derived effect models allow to simulate ES delivery by different types of hedgerows and grass strips on different types of parcels. To investigate the simultaneous delivery of multiple ES, a set of virtual parcels is constructed. For each of these parcels, the delivery of selected ES is quantified and trade-offs are identified. We compare two hedgerow types and two grass strip types on two virtual parcels. Dimensions of the first virtual parcel were always 50 m x 100 m. The width of the second virtual parcel was always doubled, while the length stayed exactly the same. Each hedgerow and grass strip scenario was then compared with a control parcel. Ecosystem service indicator levels were set at 0% for every indicator in the control parcel. Different hedgerows and grass strips were then virtually implemented on one long side of the parcel. The first hedgerow type (HR 1) had a width of 7.5 m and height of 10 m (Deckers et al., 2004). Dimensions of the second hedgerow type (HR 2) were halved. Similarly, GS 1 had a width of 7.5 m and GS 2 had a width of 3.75 m. Crop yield on hedgerow parcels was always affected in two ways: the area occupied by the hedgerow was not productive and the hedgerow influence extends into the parcel as defined by the model described above. Using this model, relative crop yield in the parcel was calculated for every H value (increasing with steps of 0.01) that lies within the parcel borders. Crop yield on grass strip parcels was reduced because the grass strip area was not productive. Pruning material from the hedgerow or grassy biomass yield (e.g. for bioenergy applications or fodder) were not considered. Soil carbon stock calculation within the hedgerow was based on the effect relationship we found, with H set at 0.01. Similar to crop yields, relative carbon stock in the adjacent parcel is calculated for every H value that lies within the parcel borders. Carbon stock on grass strip parcels is affected within the grass strip area and is unaffected in the adjacent parcel. The calculation of relative N interception, P interception and erosion reduction on parcel level is based on the effect relation we found and on the assumption that all N, P and erosion flows are directed through the hedgerow or grass strip. Because no information was found for aphid reduction on hedgerow parcels, only predator density and diversity were retained to describe impact on pest control.

2.4 Results and discussion

Data were retrieved from 60 studies (given in section 7.1). An overview of the studies (N°) and number of observations (n) for every hedgerow or grass strip and ES combination is given in Table 2.1 and Table 2.2. Prior to all statistical analyses, outliers with residuals greater than four standard deviations were detected and eliminated. This was the case for five observations in the hedgerow – crop yield analysis, one observation in the hedgerow – N interception (subsurface) analysis and one observation in the grass strip – N interception (surface) analysis. In section 7.6, models with and without outliers are compared.

2.4.1 Modelled effects of hedgerows

An overview of the tested explanatory variables for each ES is presented in Table 2.1. When the explanatory variable significantly affects ES delivery, the coefficients, 95% confidence interval and p-value derived from the model are given. Effect relations between explanatory variables and ES delivery were constructed based on the coefficients. Observations that were used in the analysis are represented in Figure 2.1. When significant, the effect relation between ln(R) and H or width is visualized.

All authors reported a similar trend, consisting of lower **crop yield** close to the hedgerow and a gradually restoring crop yield when H increases. In eight studies, positive ln(R) values were found for higher H values. In the other three studies (Chirko et al., 1996; Gao et al., 2013; Stamps et al., 2009), no positive values were found but this can be attributed to the small H values (maximum H=1.7) in these studies. Based on the model, crop yield was negatively affected by the hedgerow between H=0.0 and H=2.1. Average relative crop yield in this zone was 79%. At H=0.5, relative crop yield was 68% and at H=1, relative crop yield was 85%. This means that for a hedgerow of 20 m high, on a distance of 10 m, relative crop yield was 68% and at 20 m, relative yield was 85%. Between H=2.1 and H=20.4, crop yield was positively affected by the hedgerow: at H=2.5 (25 m from a 10-m-high tree row), relative crop yield was 103% and at H=5.0 (50 m from a 10-m-high tree row), relative crop yield was 109%. Average relative crop yield in this area was 106%. Over the entire affected zone (up to H=20.4), the net effect on crop yield was 103%, meaning that without considering the loss of arable area, the model indicates that hedgerow have an overall positive effect on crop yield. The effect relation between hedgerow and crop yield is in line with the trends found by Kort (1988), who reported a 50% crop yield reduction between H=0.5 and H=1 due to competition and a net yield increase up to H=15 on account of the shelter effect, and by Nuberg (1998), who described crop yield increases between 104% and 220% as a result of improved microclimate and shelter effect. Contrary to the findings of Kort (1988) and Nuberg (1998), hedgerow type did not significantly affect the analysis and neither did crop type or hedgerow orientation. This inconsistency might indicate that the impact of hedgerow systems on crop yield is complex, crop and context-specific and affected by a broad range of variables and that the range in different crop types or orientations we considered was not broad enough. In contrast to most hedgerow studies, we did not include the search terms ‘windbreak’ or ‘shelterbelt’ because we wanted the experimental conditions to be as representative as possible to the typical situation in regions characterized by relatively small agricultural parcels and high urbanization. Because in most regions, wind damage to crops is exceptional in these conditions, we expect no strong positive effect of reduced wind velocity on crop yield. Several models were fitted to the data, i.e. an intercept-only model, a polynomial model and two asymptotic models. Model selection was based on statistic criteria and biophysical relevance (see section 7.5).The polynomial model was retained, and therefore the function first increases, reaches a maximum and then decreases again. This trend corresponds to what might be expected based on causal mechanisms in the affected crop zone and the findings of the review of Kort (1988) and Nuberg

(1998). Beyond H=20.4, the decreasing function indicates that crop yield is negatively affected by the hedgerow compared to a treeless situation. In reality, it is to be expected that the hedgerow has no impact at this point and it is recommended not to use the equation beyond H=20.4. Additionally, it is important to acknowledge that a positive effect on hedgerows is not guaranteed and that the magnitude of this positive effect varies among the applied models.

With exception of three data points with a sampling depth of 70 cm, all studies reported **soil carbon stocks** over a depth varying between 10 and 40 cm. The mean sampling depth was 27 cm. This was confirmed by Nair (2012), who stated that very few studies measure carbon in soil layers below 30 cm depth. This knowledge gap can be problematic because tree roots extend far beyond 30 cm and thus an effect can be expected in the deeper soil layers as well (Nair, 2012). However, Cardinael et al. (2015) reported that the greatest soil carbon addition in agroforestry systems is found in the 0-30 cm layer in the inter-row and in the 0-50 cm layer in the tree row. Therefore, we assume that the range of sampling depths we considered in the analysis allows us to detect the main trends. Between H=0.01 and H=6.3, hedgerow presence resulted in a higher soil carbon stock: at H=0.5 (5 m from a 10-m-high hedgerow), relative carbon stock was 112% and at H=1 (10 m from a 10-m-high hedgerow), relative carbon stock was 108%. Beyond H=6.3, the model asserted that less soil carbon would be stored on the hedgerow parcel compared to a parcel without hedgerow. Again, this has no physical meaning and is only a consequence of the model design. Interpretation of the equation should therefore be limited to H values between 0.1 and 6.3. Beyond H=6.3, it is very likely that the effect is negligible. Neither hedgerow type nor hedgerow age or sampling depth significantly affected the result of the analysis. However, any increase in SOC stock is the result of a yearly built-up and most likely, the lack of effect of hedgerow age is due to the fact that almost all hedgerows that were included in the analysis had been there for more than 10 years. Additionally, the subset of studies where hedgerow age was known, was very limited (see Table 2.1). Based on this subset, hedgerow presence resulted in a yearly increase of SOC stock of 7%. The lack of effect on sampling depth can be attributed to the limited range of considered depths in the analysis and confirms the statement of Nair (2012).

Nitrogen interception from the surface flow was positively affected by the hedgerow: the average N interception was 69%. In the model, hedgerow width was a significant explanatory variable: the wider the hedgerow, the more N was trapped. Nitrogen interception in a 2-m-wide hedgerow was 42% and 72% in a 5-m-wide hedgerow. Only one study (Schoonover et al. 2005) reported two observations with negative ln(R) values. These values are related to narrow hedgerow, confirming the trend described in the model. The model suggests that N interception asymptotically approaches 100% for a hedgerow width of 1000 m. This trend seems plausible, but has no practical value. Based on the data visualization, it appears to be advisable to apply the equation on hedgerow with widths between 1 m and 10 m. The effect of more narrow strips (<1 m) is uncertain. Beyond 10 m, the number of observations is relatively low and it seems more appropriate to assign the N interception rate that was modelled for a 10-m-wide hedgerow.

Average **N interception** from the subsurface flow was 34%. Two studies (Salazar et al., 2015; Wang et al., 2012) reported each one observation with negative ln(R) values. Average **P interception** was 67%. Three studies (Schoonover et al., 2005a; Uusi-Kämppä and Jauhiainen, 2010a; Yang et al., 2015) reported four observations with negative ln(R) values. Uusi-Kämppä and Jauhiainen (2010) explained this adverse effect based on vegetation decomposition and sorption/desorption processes resulting in P losses. Therefore, vegetation management is crucial and removal of decaying plant material can increase P interception efficiency. Average **erosion reduction** by the hedgerow was 91% and no study reported negative ln(R) values, indicating that hedgerow are very effective under a wide range of circumstances. None of the tested explanatory variables appeared to be significant for subsurface N interception, P interception or erosion reduction. Rather than actually demonstrating the limited influence of the explanatory variables, non-significance can be the result of the low amount of studies retained for analysis, making it harder to unravel complex underlying processes such as subsurface hydrology (Hefting et al., 2006). Still, data visualization for P interception and erosion reduction indicates a positive effect of hedgerow width. To quantify this potential effect, more data and experimental studies should be added to the analysis. The range of slopes considered in the analysis ranges from 1% to 30%. Because gentle slopes (1%) are included in every analysis and this does not affect the general result, we assume the models to be applicable to relatively flat landscapes. However, this is subject to the water flow direction; if the water flow is dispersed and the hedgerow is bypassed, the system will be less effective. Therefore, we recommend to interpret and use these relationships with caution; parcel-level characteristics and hydraulics should always be considered.

The model indicated that predator density on the parcel was higher in hedgerow systems, but this result was not significant. On the significance level of p=0.1, predator diversity was significantly increased. This means that more predator species are overwintering in the hedgerow compared to the adjoining parcel and that more predator species are found on parcels with hedgerow compared to parcels without hedgerow. In the review of Chaplin-Kramer et al. (2011) predator density and diversity are linked to landscape complexity. Both are significantly correlated, but a stronger effect was found for predator diversity. Our results are in accordance with this conclusion and the lack of significance for predator density may be attributed to the low number of observations in the analysis, reducing the statistical power of the test. In order to quantify and predict pest control on agricultural parcels, a very comprehensive analysis of both species mobility and lifecycle, parcel level and landscape level characteristics is necessary (Bianchi et al., 2006; Geiger et al., 2009; Holland et al., 2016). The study selection we performed aimed at eliminating as much as possible the species level and landscape level effects. Then again, the applied selection criteria resulted in a very low number of retained studies. This confines us to apply a limited quantitative approach and results have to be interpreted with great caution. The increased predator diversity we found could result in higher pest control, especially because higher diversity can augment functional complementarity (Holland et al., 2016). However, other processes such as intra-guild predation (Straub et al., 2008) or the provision of additional food resources for pests can induce the opposite effect.

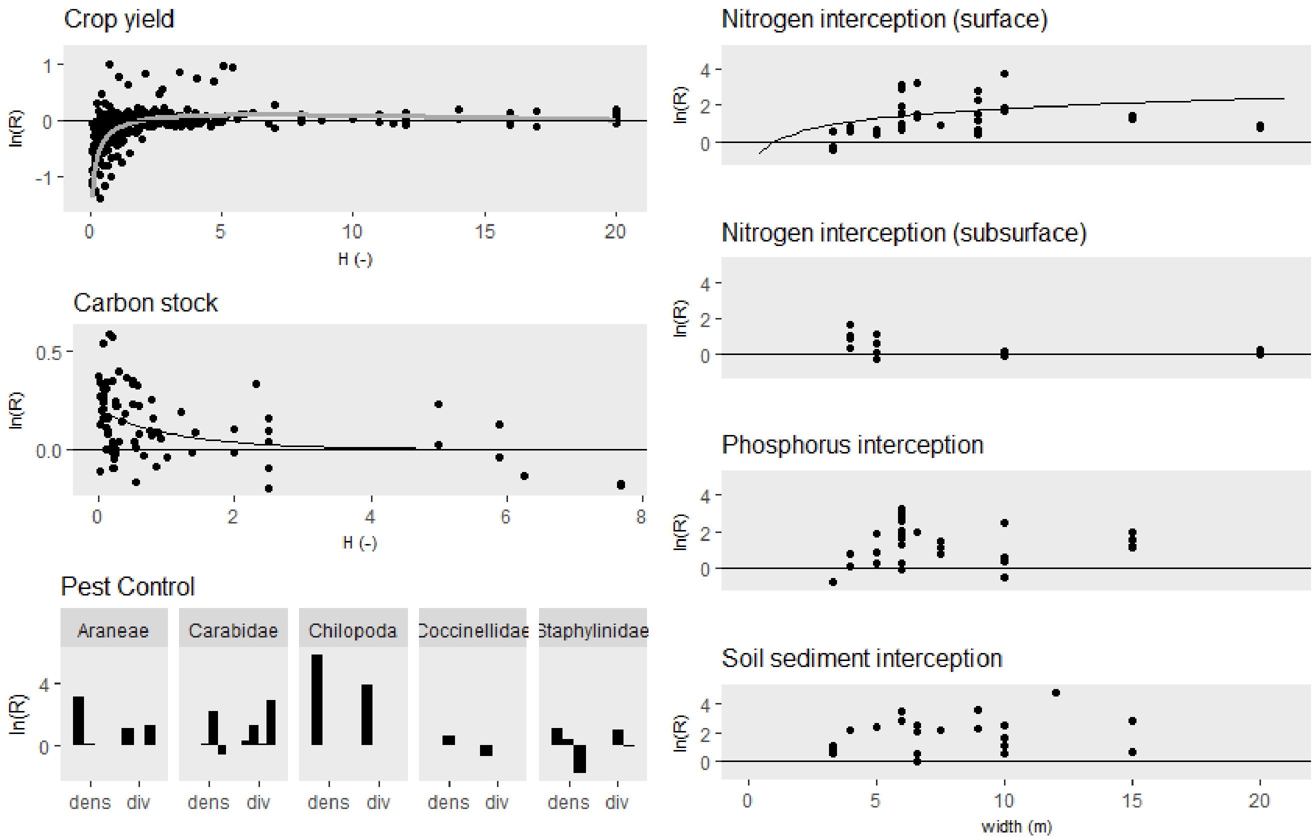


Figure 2.1: Effects of hedgerows on ecosystem service delivery. All observations used in the analysis for each hedgerow – ES combination are represented. Each dot or bar represents an individual observation. Displayed are the significant effects of the distance over height of the hedgerow (H; left, up) and width of the hedgerow (right) on ecosystem service delivery. The effect of hedgerows on pest control (left, down) is not linked to H or width. Dens stands for density, div stands for diversity.

2.4.2 Modelled effects of grass strips

An overview of the tested explanatory variables for each ES is presented in Table 2.2. When the explanatory variable significantly affects ES delivery, the coefficients, 95% confidence interval and p-value derived from the model are given. Effect relations between explanatory variables and ES delivery were constructed based on the coefficients. Observations that were used in the analysis are represented in Figure 2.2. When significant, the effect relation between ln(R) and depth or width is visualized.

Within the grass strip, average **soil carbon stock** was increased with 25%. Depth was a significant explanatory variable and the model indicates a carbon stock increase until a depth of 62 cm. Beyond this point, the equation resulted in negative ln(R) values, indicating that less carbon is present compared to the adjacent parcel. Similar to what we concluded for hedgerow-crop yield and hedgerow-soil carbon stock, this is probably the consequence of the equation design. It is very doubtful whether this strokes with reality. Additionally, because most observations (101 out of 108) are situated in the upper 30 cm, the model has greater reliability within this range and we advise to confine application of the equation to the upper soil layer. We expect to cover the main effect within this range because the majority of the grass roots biomass is found in the upper soil layer (Reeder et al., 1998). In the upper 30 cm, total soil carbon stock is increased with 37%. We expected grass strip age to have a positive impact on the carbon stock ratio (Guzman and Al-Kaisi, 2007) but this did not emerge from the model. This could be due to the high variability in ln(R) values found for young grass strips (up to 10 years).

Average **N interception** from the surface flow was 76%. Width was a significant explanatory variable: the wider the grass strip, the more N is intercepted. For a grass strip width of 2 m, N interception was 29% and for a width of 5 m, N interception was 58%. Three observations (out of 91) from two studies (Hay et al., 2006; Magette et al., 1989) reported negative ln(R) values. Two of these values were related to narrow grass strips, confirming the trend described in the model. All observations indicated a positive effect of grass strips on N interception from the subsurface flow and average N interception was 32%. Width was a significant explanatory variable: the wider the grass strip, the more N was trapped. For a grass strip width of 2 m, N interception was 14% and for a width of 5 m, N interception was 29%. Average **P interception** by the grass strip was 73%. Grass strip width was a significant explanatory variable: the wider the grass strip, the more P was trapped. For a 2-m-wide grass strip, P interception was 23% and for a 5-m-wide grass strip, P interception was 48%. Out of 116 observations, four (Duchemin and Hogue, 2009; Patty et al., 1997; Sheppard et al., 2006) reported negative ln(R) values. Two of these values were approximately zero. The two most negative values are linked to relatively narrow grass strips, confirming the trend that is described in the model. Average **erosion reduction** by the grass strip was 90%. Grass strip width was a significant explanatory variable: the wider the v, the more erosion was reduced. For a grass strip width of 2 m, erosion reduction was 55% and for a width of 5 m, erosion reduction was 84%. Out of 103 observations, two (Hay et al., 2006; Yang et al., 2015) reported negative ln(R) values. The value found by Yang et al. (2015) was approximately zero. Hay et al. (2006) recorded a stronger negative value, but for this grass strip more inconsistent results were found, indicating complex and unknown underlying processes. According to the equations, a 1-m-wide grass strip will in all cases result in positive ln(R) values and thus N interception, P interception and erosion reduction occurs. However, also for wider grass strips, observations with negative ln(R) values have been found. Therefore, in order to increase the grass strip effectiveness, a minimum width of 6 m is recommendable. Beyond this distance, no strongly negative values (below ln(R)=-0.1) were observed. Parcel slope and grass strip age appeared to be non-significant for N interception, P interception and erosion reduction and inflow concentration was only significant for N interception from the subsurface flow and P interception. Rather than actually demonstrating the limited influence of these variables, this could indicate a high variability between experimental conditions. The range of slopes considered in the N, P and erosion analyses varied between 0.3% to 30%. Because gentle slopes (≤ 1%) were included in each analysis and slopes were non-significant, we assume the outcome of the analyses to be applicable to relatively flat landscapes. However, this is subject to the water flow direction; if the water flow is dispersed and the grass strip is bypassed, the system will be less effective. Therefore, we recommend to interpret and use these relationships with caution; parcel-level characteristics and hydraulics should always be considered.

The model indicated that predator diversity and density are significantly higher. This means that more predators and predator species are using the grass strip for overwintering compared to the adjoining parcel and that more predators and predator species are found on parcels with grass strips compared to parcels without grass strips. Additionally, pests were significantly reduced, indicating that less pests are found on parcels with grass strips. Similar to the hedgerow – pest control analysis, the study selection resulted in a very low number of retained studies, obstructing further quantitative analysis. A spatial relation with distance from the grass strip is to be expected, but this could not be deduced from the model. Because this spatial distribution of pest species is of major importance for determining optimal pesticide application, further research into this topic is necessary.

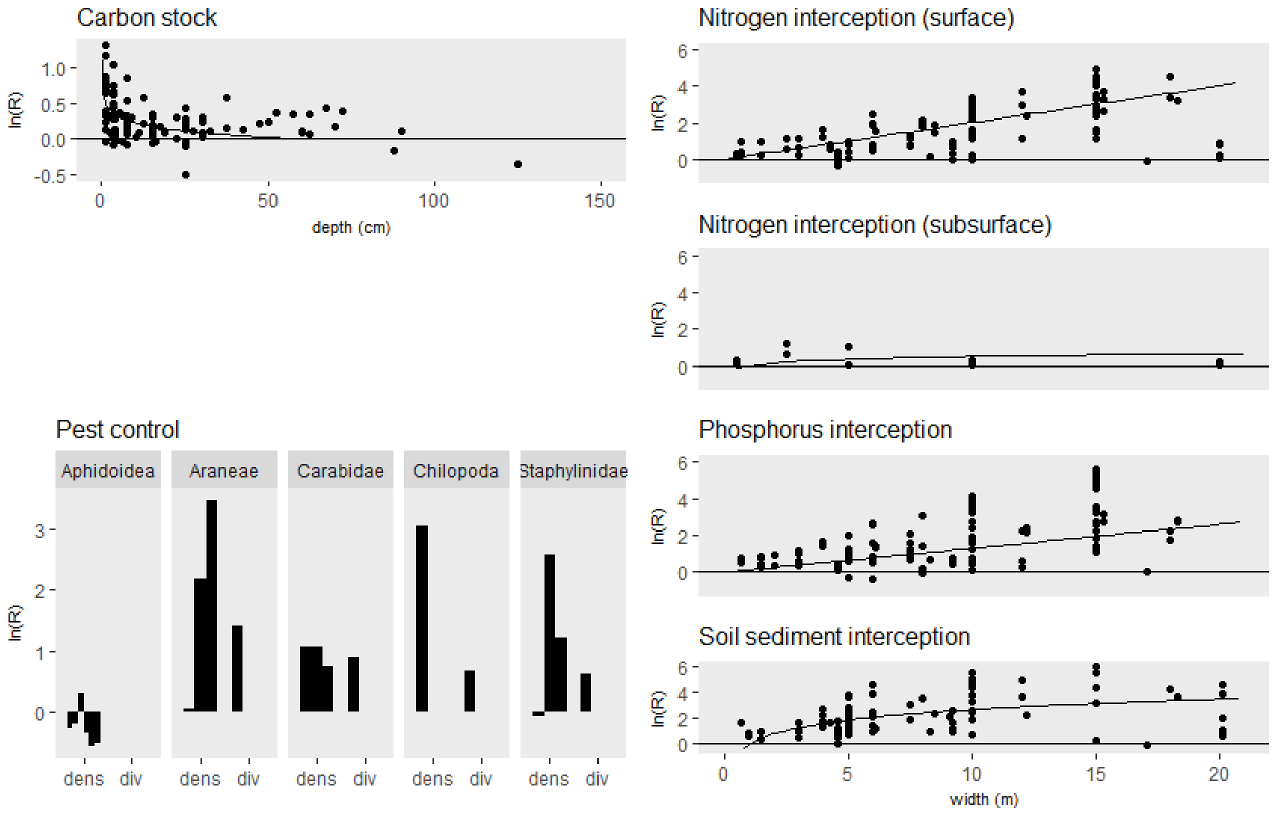


Figure 2.2: Effects of grass strips on ecosystem service delivery. All observations used in the analysis for each grass strip – ES combination are represented. Each dot or bar is an individual observation. Displayed are the significant effects of the sampling depth (left, up) and width of the grass strip (left) on ecosystem service delivery. The effect of grass strips on pest control (left, down) is not linked to width. Dens stands for density, div stands for diversity

2.4.3 Assessment of trade-offs and multifunctionality

Figure 2.3 illustrates the virtual hedgerow systems and ES delivery (except for NPC). In Figure 2.4, the indicator level of each studied ES on the virtual parcels HR 1, GS 1 and control 1 are visualized. Based on the results presented in section 2.4.1, the hedgerow influence zone for crop yield theoretically extends up to a distance of H=20.4. On the virtual parcel 1, the maximal H within the parcel borders was however 4.25 (HR 1) and 9.25 (HR 2). As a consequence, the theoretical hedgerow influence zone continues beyond the parcel borders and the positively affected zone is cut off. Up till H=2.1, crop yield is reduced by hedgerow presence. The narrower the parcel or the higher the hedgerow, the higher the relative weight of this negatively affected zone. Therefore, crop yield reduction is more distinct on parcel 2 compared to parcel 1. Because hedgerow height in HR 1 is higher, the effect is stronger and yields are more reduced.

Soil carbon stock within the hedgerow is 32% higher compared to the control parcel. This is similar to the difference between 0-30 cm soil carbon stocks in the tree rows of alley cropping systems and in monocultures (Cardinael et al. 2015). Hedgerow impact on soil carbon stock extends into the parcel, until a distance of H=6.3. However, on parcel 1 with HR 1, maximum H within the parcel borders is 4.25 and thus the full potential effect of hedgerow on soil carbon stock is not reached. On parcel 1 with HR 2 and on parcel 2 with both hedgerow types, the entire influence zone (until H=6.3) lies within the parcel borders. Beyond H=6.3, soil carbon stock remains unchanged.

Sampling depth is a significant variable in the grass strip – carbon stock analysis, so we first define the range that will be considered. As suggested in section 2.4.2, this range is confined to the 0-30 cm soil layer. Integration of the equation over this range yields an increase of 37% in the grass strip. This is slightly more than the relative carbon stock in the hedgerow row. On the other hand, hedgerow influence extends into the parcel, resulting in an additional carbon stock increase compared to the grass strip systems.

When N interception, P interception or erosion reduction was significantly affected by hedgerow or grass strip width, the effect relation based on this explanatory variable was used to distinguish among HR 1 and HR 2 and among GS 1 and GS 2. If width was not a significant variable, the average effect (based on the intercept-only model) was used and the impact of HR 1 and GS 1 is equal to the impact of respectively HR 2 and GS 2.

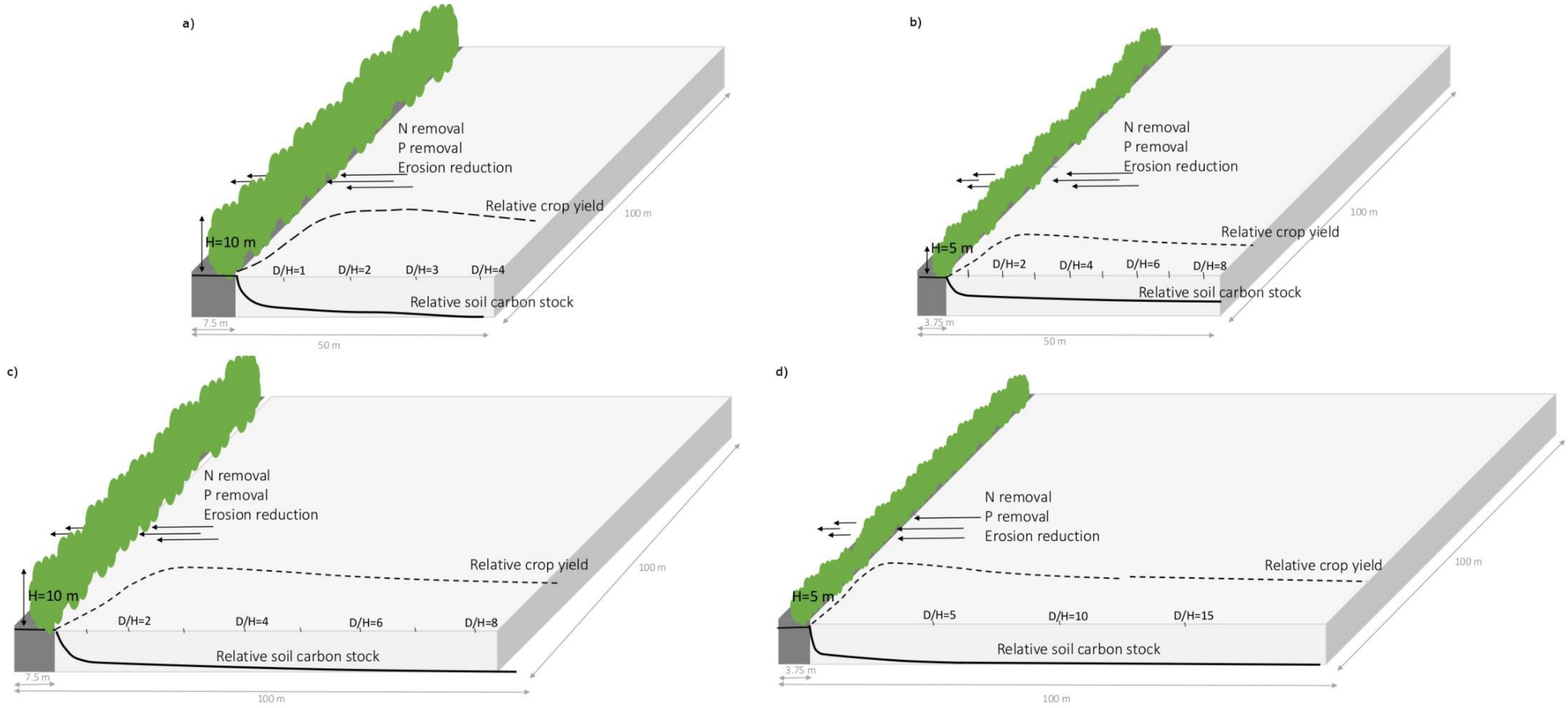
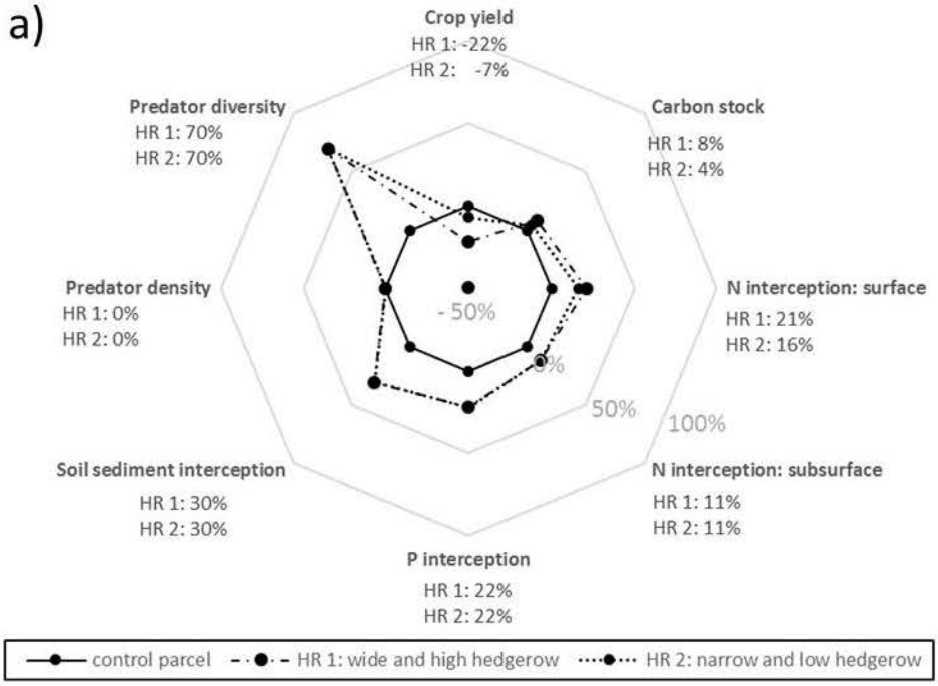
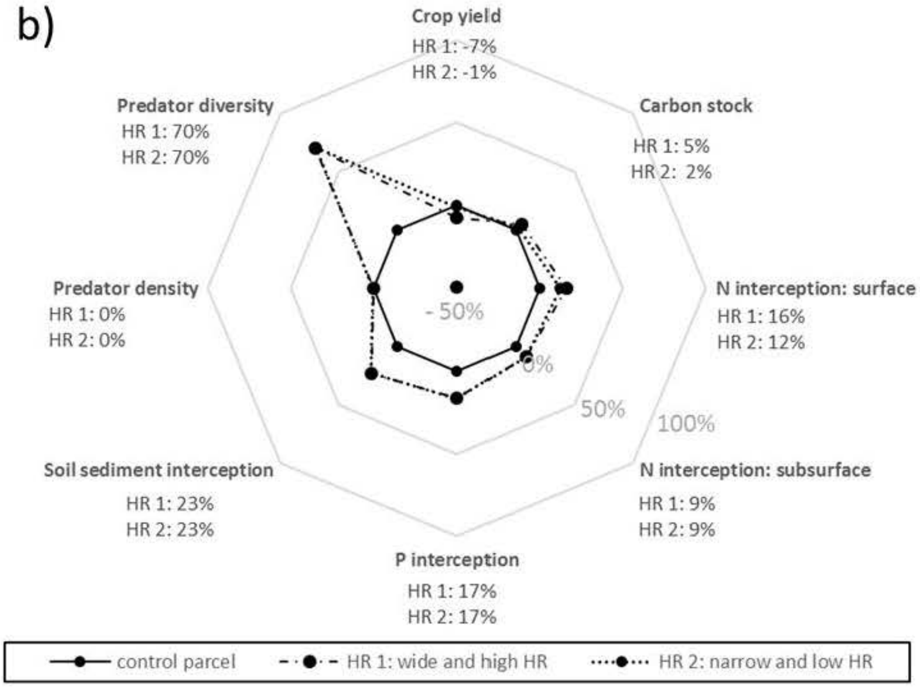


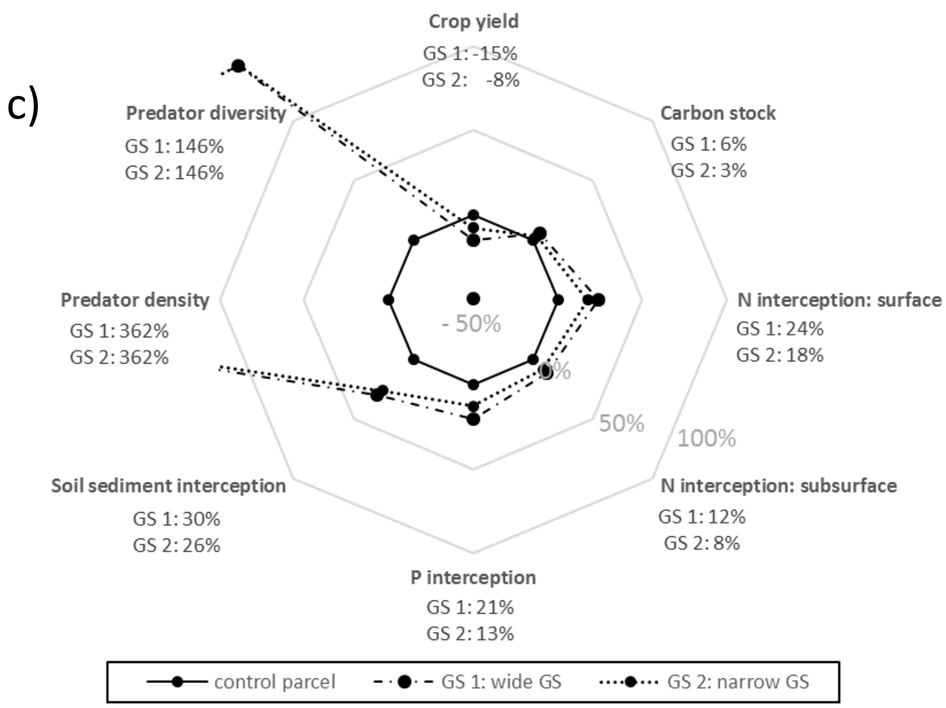
Figure 2.3: ES delivery (except for natural pest control) on virtual parcel 1 by HR 1 (a) and HR 2 (b) and on virtual parcel 2 by HR 1 (c) and HR 2 (d). Crop yield and carbon stock depended on distance from the hedgerow (D) and height (H). Both trends were qualitatively visualized for every H-value. No y-axis is set, so the trend line has no quantitative meaning.

a)


Crop yield
HR 1: -22%
HR 2: -7%
Carbon stock
HR 1: 8%
HR 2: 4%
N interception: surface
HR 1: 21%
HR 2: 16%
N interception: subsurface
HR 1: 11%
HR 2: 11%
P interception
HR 1: 22%
HR 2: 22%
Soil sediment interception
HR 1: 30%
HR 2: 30%
Predator density
HR 1: 0%
HR 2: 0%
Predator diversity
HR 1: 70%
HR 2: 70%
- 50%
0%
50%
100%
control parcel
HR 1: wide and high hedgerow
HR 2: narrow and low hedgerow

b)


Crop yield
HR 1: -7%
HR 2: -1%
Carbon stock
HR 1: 5%
HR 2: 2%
N interception: surface
HR 1: 16%
HR 2: 12%
N interception: subsurface
HR 1: 9%
HR 2: 9%
P interception
HR 1: 17%
HR 2: 17%
Soil sediment interception
HR 1: 23%
HR 2: 23%
Predator density
HR 1: 0%
HR 2: 0%
Predator diversity
HR 1: 70%
HR 2: 70%
- 50%
0%
50%
100%
control parcel
HR 1: wide and high HR
HR 2: narrow and low HR



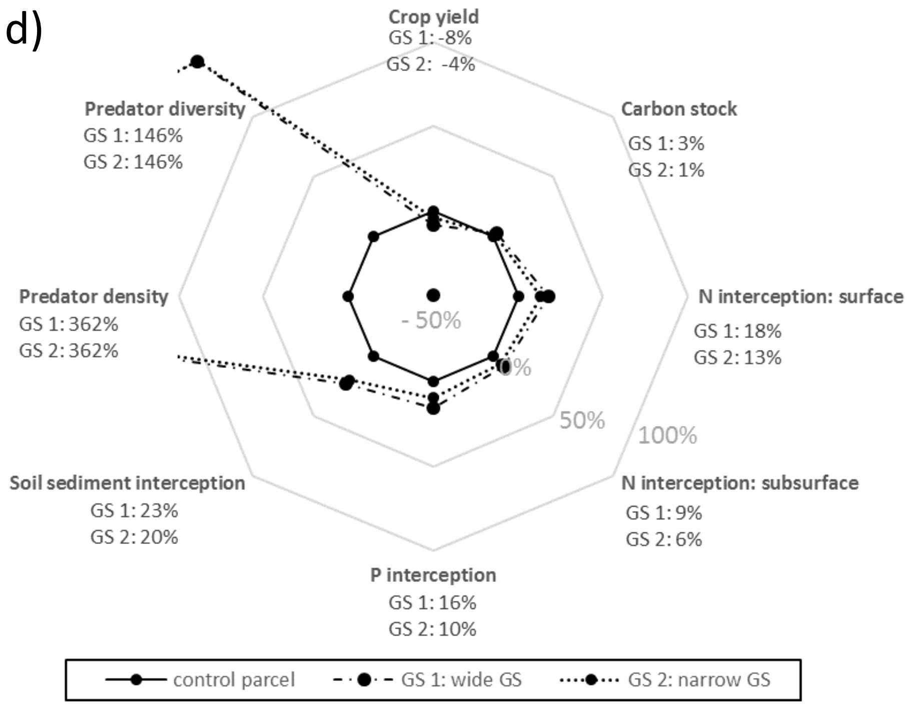


Figure 2.4: ES indicator delivery by the control parcel, HR 1 and HR 2 on virtual parcel 1 (a) and on virtual parcel 2 (b) and by the control parcel, GS 1 and GS 2 on virtual parcel 1 (c) and on virtual parcel 2 (d).

In contrast with N interception, P interception and erosion reduction, the effect on crop yield and carbon stock depended on the location within the parcel. Therefore, the parcel-level impact of hedgerows and grass strips on soil carbon stock strongly depends on parcel layout and is markedly less distinct compared to the other ES. This is due to averaging at the parcel level. Close to the hedgerow or in the grass strip, the effect is considerable. Further away in the parcel, the effect decreases and approaches zero. The averaged parcel-level impact is then an intermediate value. The modelled N interception, P interception and erosion reduction on parcel level assumes that all surface and subsurface flows are directed through the hedgerow of grass strip and that no other nutrient or erosion losses occur. In case of considerable parcel slopes and adequate hedgerow or grass strip location, this might be the case. However, if the parcel is rather flat or when the hedgerow of grass strip is not implemented perpendicular to the direction of the water flow, surface water might leave the parcel via other pathways. Also, subsurface flow hydrology is very complex (Hefting et al., 2006) and hedgerow and grass strip bypassing is likely. As a result, theoretical interception rates for both surface and subsurface flow could overestimate the true effect. Therefore, parcel-level calculations for N and P interception and erosion reduction were adjusted in accordance with the relative share of the hedgerow of grass strip compared to the other parcel borders. This means that on parcel 1, the modelled N and P interception and erosion reduction are multiplied with 1/3, because hedgerow and grass strip length is 1/3 of total parcel border length. On parcel 2, modelled interception rates are then multiplied with 1/4.

Based on the intercept-only model, hedgerow presence does not affect the number of predators on the parcel. However, the number of predator species is increased with 70%. On grass strip parcels, the number of predators is 4.5 times higher and the number of predator species increased 2.5 times. As expected, there is a trade-off between crop yield and the delivery of regulating ES. On both virtual parcels, this yield reduction is more extreme for scenario HR 1. In comparison with HR 2, the delivery of the other ES is only slightly increased or the same. We can thus deduct that wider and higher hedgerows have a considerable impact on crop yield, but a rather limited additional effect on the delivery of the other studied ES compared to narrow and lower hedgerows. In order to maximize ES delivery, low, narrow hedgerows seem therefore preferable to higher and wider hedgerows. Increasing the width of grass strip systems causes a relatively small yield reduction and soil carbon stock increase, but simultaneously results in considerably higher potential N interception, P interception and erosion reduction. As a consequence, if water quality and/or erosion reduction is of high priority, wider grass strips are preferred over narrow grass strips. As demonstrated on virtual parcel 2 for both hedgerows and grass strips, if parcel width (and thus size) increases, crop yield reduction diminishes but so do N and P interception and erosion reduction.

2.5 Conclusion

Despite the modelled net positive effect of hedgerows on crop yield, the calculations for the virtual parcels indicate a trade-off between crop yield and regulating ES. However, a thoughtful and appropriate hedgerow design and implementation can minimize crop yield loss and thus trade-offs. The delivery of regulating ES is increased by grass strip presence, but due to crop area loss, there is a trade-off with crop yield. Increasing grass strip width causes more crop yield loss but at the same time results in higher regulating ES delivery. Depending on local priorities, an optimal grass strip width can be deduced. The set of ES we considered enabled us to perform a quantitative multipurpose evaluation of the studied measures and a trade-off analysis between these ES, which is of interest not only to farmers and land managers but also to policymakers. The effect relationships we developed were based on black-box models and we did not investigate causal mechanisms. Because the equations do not take into account underlying processes, it is not advisable to extrapolate beyond the ranges proposed in sections 2.4.1 and 2.4.2. Also, we acknowledge that the shape of the effect relationships was determined by the equation type that was fitted. Although the modelled effect relationships seem to be representative for the trends in the data, other models may produce (slightly) different results. Reliability of the results can be improved by adding more observations to the analyses, in order to better and more extensively identify the influence of hedgerow and grass strip characteristics and by fitting more equation types. Many explanatory variables did not significantly affect the outcome of the analyses, but it is very plausible that this is due to the limited amount of observations and studies that were retained. More experimental, parcel-level studies in temperate areas are therefore needed to fine-tune the results. Additionally, hedgerows and grass strips can deliver more ES than those considered in this study. For example, different types of field margins can have a positive effect on the presence of pollinating insects (Lagerlöf et al., 1992). Therefore, it would be interesting to extend the evaluation framework and include more ES that are potentially affected by hedgerows or grass strips.

3. Monitoring the impact of hedgerows and grass strips on the performance of multiple ecosystem service indicators

After: Van Vooren L., Reubens, B., Ampoorter, E., Broekx, S., Pardon, P., Van Waes, C., Verheyen, K. Monitoring the impact of hedgerows and grass strips on the performance of multiple ecosystem service indicators. Environmental Management, in press.

3.1 Abstract

The importance of semi-natural vegetation elements in the agricultural landscape is increasingly recognized because they have the potential to enhance multiple ecosystem service delivery and biodiversity. However, there is great variability in the observed effects within and between studies. Also, little is known about the simultaneous delivery of multiple ecosystem services and biodiversity because most studies focus on monitoring one service at a time and in conditions specifically suited to observe this one service. This study presents the results of one year of monitoring of a set of parcel-level and straightforward ecosystem service and biodiversity indicator on Flemish parcels with grass strips or hedgerows, two very common types of semi-natural vegetation elements. In the grass strips, an increase in soil organic carbon stock was found, a decrease in soil mineral nitrogen content, a different carabid species composition and a higher spider activity-density, compared to the adjacent arable parcel. These results indicate a contribution to climate regulation, the regulation of water quality, an increase of beta diversity and potential for pest control. Next to hedgerows, crop yield was reduced and winter wheat thousand kernel weight, soil organic carbon stock and spider activity-density were increased. These indicators show an effect of the hedgerow on food production, climate regulation and potential for pest control. The study concludes that both grass strips and hedgerows have the potential to increase multiple ecosystem service delivery, but that an increase of every service is not assured and trade-offs will probably have to be made through management choices. Also, we suggest an improved experimental setup in order to enhance ecosystem service monitoring.

3.2 Introduction

Intensification of agricultural systems has generally resulted in higher productivity of food and forage. This focus on provisioning ecosystem services has generally been at the expense of other services (Foley et al., 2005) and the agricultural sector faces the challenge of combining the delivery of provisioning ecosystem services with the delivery of regulating ecosystem services and the maintenance or increase of biodiversity (Rey Benayas and Bullock, 2012). The presence of semi-natural vegetation elements within an agricultural landscape enhances the potential to deliver multiple regulating ecosystem services and their positive effect on biodiversity is widely recognized (García-Feced et al., 2014). This research focusses on two types of semi-natural vegetation elements, i.e. hedgerows and grass strips, because they are very common in the Flemish agricultural landscape. Previous research has shown that both hedgerows and grass strips affect the delivery of ecosystem services, for example by carbon sequestration and nitrogen (N) and phosphorus (P) interception from water flows (Van Vooren et al., 2017). However, results on the effectiveness and extent of the impact of these vegetation elements can be variable. For instance, within a distance of 2 m from a hedgerow, soil organic carbon (SOC) stock increases between 17% and 45% were reported (Cardinali et al., 2014; D'Acunto et al., 2014; Paudel et al., 2011). Below grass strips, SOC stock increase ranged from 10% to 45% (Bowman and Anderson, 2002; Culman et al., 2010). In the study of Schmitt et al. (1999), removal rates for nutrients (N and P) vary between 48% and 96%. This variability is not surprising because actual ecosystem service delivery will depend on specific characteristics (such as location, age, species composition, dimensions, etc.) of the hedgerow and grass strip, but also on local management, specific soil and environmental characteristics etc. Therefore, for any local context, it is important to gain insight into which aspects influence the extent of actual ecosystem service delivery.

Besides the variable delivery of individual ecosystem services, another issue is the lack of understanding of the simultaneous delivery of multiple ecosystem services, biodiversity and their interactions (Bennett et al., 2009). Field measurements that assess multiple services at the same site are rare. Most published monitoring campaigns focus on a single variable or on multiple variables representing one ecosystem service (e.g. N and P concentrations as indicators for chemical water quality). Assessing multifunctionality and trade-offs by combining several studies, each focusing on individual ecosystem services or specific species, might result in an overestimation of multiple ecosystem service delivery, because monitoring sites are often specifically selected to demonstrate maximal delivery of one specific service. For instance, SOC stock increases with grass strip age (Conant et al., 2001) whilst due to saturation, the P retention capacity of grass strips can decline after a certain timeframe (Dorioz et al., 2006b).

Main goals of this study were to gain insight into (1) the extent of multiple ecosystem service delivery and biodiversity increase by hedgerows and grass strips and (2) the potential causes of variability in their effects. A secondary objective was the evaluation of possibilities and limitations of assessing more than one ecosystem service at the same time by simultaneously measuring a range of parcel-level and simplistic indicators on the same study site.

Thereto, the results of a set of ecosystem service and biodiversity indicator measurements on Flemish agricultural parcels with either hedgerows or grass strips are presented. The data result from a one-year-monitoring campaign, and therefore provide a snapshot of the ongoing processes. The services targeted were food production, climate regulation, maintenance of chemical water quality and potential for pest control (classification according to IPBES (2017)). Ecosystem services selection was based on relevance for the Flemish context and parcel-level measurability. For each of the abovementioned ecosystem services, indicators which could provide information on the impact of grass strips or hedgerows on a very local level were selected. Hence this study focusses on the immediate working environment of and consequences for the farmer, making abstraction of potential impact at a larger scale beyond the field borders. Crop yield and thousand kernel weight (Zecevic et al., 2014) were two indicators for food production. Soil organic carbon stock was the indicator for climate regulation because a higher SOC stock generally means that more CO_2 has been captured in the soil (Smith et al., 2000). Soil N and P were chosen as indicators to describe the effect on chemical water quality. A higher N and P content in the soil entails a higher risk for leaching (Dhondt et al., 2002; Etana and Bergstro, 2013) and thus a negative effect on the chemical water quality. Activity-density of natural predators (carabids, spiders and rove beetles) was selected as an indicator for potential for pest control. Biodiversity was described by means of the number of carabid species (alpha diversity) and carabid species composition (beta diversity). While alpha diversity is generally defined by the number of species in a sampling unit, beta diversity considers the difference in species composition among sampling units (Anderson et al., 2006).

This study focusses on within-parcel changes induced by the implementation of either a hedgerow or grass strip, but it should be clear that it is beyond the scope of this paper to make a comparison between both measures. As a consequence of focussing on within-parcel, local changes in ecosystem services, slightly different approaches were used for hedgerows and grass strips: the monitored grass strips are assumed to make up a proper part of the parcel and ecosystem service delivery is affected mostly within the grass strips, while hedgerows are typically planted on the field borders and the impact is situated both in the hedgerow itself and in the adjacent parcel. However, because this study focusses on changes in the arable parcel, SOC stocks, N and P content and natural predators within the hedgerow were not investigated. Therefore, ecosystem service indicators were monitored in the grass strip and in the adjacent arable field and both systems were compared, while for the hedgerow parcels, indicator values in the arable field close to the hedgerow were compared to indicator values further in the arable field, following a transect approach. For grass strips, the hypotheses are that (1) the permanent soil cover and grass biomass decomposition (Nelson et al., 2008) will increase SOC stock compared to the adjacent arable land, (2) the combination of zero fertilization with nutrient uptake and removal by the grass (Van Beek et al., 2007) will reduce soil N and P concentrations in the grass strip and (3) the grass strips will act as a suitable habitat and alternative food resource for natural predators (Bianchi et al., 2006), resulting in an increased activity-densities and species number of natural predators in the grass strip. For hedgerow systems, it is hypothesized that close to the hedgerow (1) due to competition (Nuberg, 1998b), crop yield will be lower, (2) litter fall and root exudates (Lorenz and Lal, 2014) will increase SOC stock, (3) the hedgerow root system will take up N and P from the soil (Lehmann, 2003) and thus decrease N and P concentrations and (4) that hedgerows will act as a suitable habitat for natural predators (Bianchi et al., 2006), resulting in higher activity-densities and species numbers close to the hedgerow and that both will decrease further away from the hedgerow.

3.3 Material & Methods

3.3.1 Study areas

Measurements were performed in 2014 in two Flemish (northern Belgium) regions with either grass strips or hedgerows as the most common type of semi-natural vegetation elements alongside arable land. The parcels that were retained for further monitoring were in the first place representative for the region and its agricultural landscape. There was no specific search for parcels with a high potential for the delivery of one or more ecosystem services. In 2014, the mean annual temperature in (Ukkel) Flanders was 11.9 °C and annual precipitation was 784.3 mm (KMI, 2018).

Six arable parcels bordered by grass strips were monitored in the polder region in the north of Flanders (51°15'8" N, 3°45'52" E). Every grass strip was located adjacent to a ditch and had a minimum width of 12 m. The soil is clay and classified as Gleyic Fluvic Cambisol according to the World Reference Base (IUSS Working Group WRB, 2006). A part of the area has been designated as Natura 2000 site. Apart from the development and maintenance of existing specific natural grassland habitats and wetlands, specific Natura 2000 guidelines require an improvement of the general environmental quality of the site, among others by enhancing the chemical water quality. Currently in this area, 72% of the Natura 2000 site is in agricultural use. As was stated by the farmers, all grass strip parcels (GS) have been in cropland for several generations. The site is completely flat. In Table 3.1, parcel details are given. Management of the grass strips is in accordance with the prescriptions of the agri-environment measures: no fertilizers or pesticides are intentionally applied in the strip, the soil is not tilled and mowing is only allowed after the 15th of June.

Table 3.1: Overview and characteristics of the monitored grass strip parcels (GS)

Parcel	Crop 2014	Grass strip age (years)	Grass strip width (m)	Dominant plans species (survey on 12.06.2014)	pH-KCl arable parcel	pH-KCl grass strip
GS 1	Winter wheat	7	12	*Dactylis glomerata, Arrhenatherum elatius, Agrostis capillaris, Holcus lanatus, Lolium perenne, Rumex obtusifolius*	7.56	7.34
GS 2	Sugar beet	10	12	*Dactylis glomerata, Arrhenatherum elatius, Agrostis capillaris, Holcus lanatus, Lolium perenne, Symphytum officinale, Ranunculus repens, Cirsium arvense, Trifolium repens, Vicia cracca*	7.64	7.46
GS 3	Winter wheat	10	12	*Dactylis glomerata, Agrostis capillaris, Holcus lanatus, Lolium perenne, Ranunculus repens, Trifolium repens, Trifolium dubium, Taraxacum campylodes, Vicia hirsuta, Vicia sativa, Persicaria maculosa*	7.64	7.40
GS 4	Potato	10	12	*Agrostis capillaris., Holcus lanatus, Lolium perenne, Cirsium arvense, Trifolium repens, Trifolium dubium, Vicia hirsuta, Vicia sativa*	7.68	7.45
GS 5	Potato	10	15	*Agrostis capillaris., Holcus lanatus, Lolium perenne, Symphytum officinale, Ranunculus repens, Trifolium repens, Taraxacum campylodes, Vicia sativa*	7.71	7.51
GS 6	Potato	10	12	*Agrostis capillaris., Holcus lanatus, Rumex obtusifolius, Vicia sativa*	7.71	7.48

The second set consists of four arable parcels situated in Heuvelland (50°46'44" N, 2°49'44" E), a hilly region in the south west of Flanders. The area is designated as Natura 2000 site and one of its main characteristics is the presence of hedgerows as a part of the typical "bocage" landscape. The hedgerows create corridors between forest fragments with a protected status and are valuable specific habitats (ANB, 2011). All parcels were bordered at least on one side by a hedgerow, here defined as perennial, linear woody structures established on agricultural field borders and consisting of shrubs and trees. Width of the hedgerows varied between 1 m and 3 m. The soil is sandy loam or loam and classified as a Gleyic Luvisol (IUSS Working Group WRB, 2006). In Table 3.2, hedgerow parcel (HR) details are given. Management of the hedgerow is in accordance with the prescriptions in the agri-environment measures: no fertilizers or pesticides were intentionally applied in the hedgerow. The parcels received fertilizer and pesticides in accordance with common practice and legislation (Vlaamse landmaatschappij, 2013).

Table 3.2: overview and characteristics of the monitored hedgerow parcels (HR)

Parcel	Crop 2014	Hedgerow age (years)	Hedgerow height (m)	Hedgerow width (m)	Hedgerow pruning management	Hedgerow orientation	Parcel position relative to the field	Hedgerow woody species	Management	Parcel slope + hedgerow position	pH-KCl parcel
HR 1	Winter wheat	8	5	1.2	Pruned every 8 years	north-south	east	*Crataegus monogyna, Sambucus nigra, Corylus avellana, Fraxinus excelsior*	conventional	< 1%, upslope	6.48
HR 2	Winter wheat	10	6	1.3	Not (yet) pruned	east-west	north	*Carpinus betulus*	conventional	6%, parallel to the slope	6.91
HR 3	Oats + peas	13	2	1	Yearly pruned to the same height	north-south	west	*Crataegus monogyna*	organic	2%, downslope	5.69
HR 4	Winter wheat	14	5	3	Pruned every 7 years	north east – south west	north	*Sambucus nigra, Corylus avellana, Acer campestre, Fraxinus excelsior*	conventional	2%, downslope	6.48

3.3.2 Data collection

3.3.2.1 Grass strips

Soil samples were collected in November 2014 to assess SOC stock (ton ha^{-1}), soil mineral N content (kg ha^{-1}) and soil P concentration (mg kg^{-1}). Nitrate-N (NO_3^--N) and ammonium-N (NH_4^+-N) (mg kg^{-1}) were sampled in the 0-30 cm, 30-60 cm and 60-90 cm layers and at three different locations within each parcel. Ten samples were collected in the grass strip, parallel and next to the ditch (position -10), a second row of ten samples was collected in the grass strip, parallel and next to the parcel (position -1) and a third row of samples was collected in the parcel, parallel to and at a distance of 30 m from the grass strip (position 30) (**Error! Reference source not found.**Figure 3.1 a). Samples f rom the same row were pooled for N because N is relatively mobile and we expect small variations within the same row. This way, nine samples were obtained, originating from three different depths at three different locations. NO_3^- - N and NH_4^+ - N were extracted in 1M KCl (extraction ratio 1:2) and determined colorimetrically. NO_3^-- N and NH_4^+ -N content (kg ha^{-1}) was calculated based on bulk density and sampling depth. Mineral N was calculated for every sample as the sum of NO_3^- -N and NH_4^+-N. For every plot, mineral N, NO_3^- -N and NH_4^+-N was summed up over the different soil layers (0-30 cm, 30-60 cm and 60-90 cm). Separate samples were taken from the 0-10 cm and 10-20 cm soil layers for the determination of total carbon (TC) (%), inorganic carbon (IC) (%), bulk density (g cm^{-3}) and soil moisture content (%). Soil bulk density was determined on multiple undisturbed soil samples taken from each experimental plot using Kopecky rings (height = 5.1 cm, diameter = 5 cm). TC (%) was determined using a Vario MACRO cube Elementary analyser. For soil samples with pH values above 6.4, IC (%) was determined with a SKALAR Primacs SLC analyser and organic carbon (OC) concentration was calculated as the difference between TC and IC concentration. If pH was below 6.4, the share of IC was considered negligible and TC and OC concentration were assumed to be equal. Soil organic carbon stocks (ton ha^{-1}) were calculated based on OC concentration (%), BD and sampling depth. For every plot, SOC stocks were summed up over the different soil layers (0-10 cm and 10-20 cm). In-calcium-chloride-extractable P (P-$CaCl_2$) (mg kg^{-1}), P-Olsen (mg kg^{-1}) and pH-KCl were also determined for the 0-10 cm soil samples. Soil pH was measured in a 1 M KCl solution (extraction ratio 1:5) (ISO 10390). Because P dynamics are very complex and strongly depends on its form, we applied two test procedures on the samples. P-$CaCl_2$ was measured after 1:10 soil extraction with 0.01 M $CaCl_2$ (NEN5704). P-$CaCl_2$ is often used as a predictor for P-leaching (Maguire and Sims, 2002). P-Olsen was extracted with 0.5 M $NaHCO_3$ (pH=8.5). P-Olsen is an agronomic soil test that is often used to assess fertility status (Messiga et al., 2010). Three separate 0-10 cm and 10-20 cm soil samples were collected within the same rows that were set up for N sampling (position -10, position -1 and position 30), but samples were not pooled because we expect to observe local variations in soil P and SOC (Figure 3.1 a).

Three pitfall traps (9 cm opening diameter, volume 300 ml), half filled with a 50% propylene glycol (antifreeze) solution with detergent were installed in the middle of the grass strip (position -5), three were installed in the adjacent parcel at a distance of 10 m from the grass strip (position 10) and three at a distance of 30 m (position 30) (Figure 3.1 a). This setup allows to compare predator activity-densties, species number and species composition in the grass strip with the adjacent arable parcel. Pitfall traps were set up in the first half of May 2014 and were emptied three times, approximately every two weeks. Collected specimens were stored in 70% ethanol. For every individual pitfall trap, the number of carabids (Carabidae), spiders (Aranae) and rove beetles (Staphylinidae) was determined. Additionally, carabids were determined to species level according to Boeken et al. (2002).

3.3.2.2 Hedgerows

Three transects were installed on the HR parcels. Each transect was perpendicular to the hedgerow and consisted of three experimental plots, at a distance of 1 m, 10 m and 30 m from the hedgerow (Figure 3.1 b). In order to measure crop yield (ton ha^{-1}), plots of 1 m x 8 m were harvested with a Wintersteiger small plot combine (on the winter wheat parcels) or a cutter bar mower (on the oats & peas parcel) in each experimental plot in July or August. On every winter wheat sample (samples from HR 1, HR 2 and HR 4), thousand kernel weight (g) was determined. To assess SOC stock (ton ha^{-1}), mineral N content (kg ha^{-1}) and P concentration (mg kg^{-1}), soil samples were collected in November 2014. Nitrate-N (NO_3^--N) and ammonium-N (NH_4^+-N) concentration (mg L^{-1}) was measured in the 0-30 cm, 30-60 cm and 60-90 cm layer. One row of 10 samples was collected at 1 m from the hedgerow, another row at 10 m and another row at 30 m from the hedgerow. Samples from the same row were pooled. This way, nine samples, coming from three different layers (0-30 cm, 30-60 cm, 60-90 cm) on three different locations, were collected on every parcel. Mineral N (kg ha^{-1}) was calculated for every sample as the sum of NO_3^--N and NH_4^+-N. Total carbon (TC) (%), inorganic carbon (IC) (%), bulk density (g cm-3) and soil moisture content (%) were determined in the 0-10 cm and 10-20 cm layer. In the same 0-10 cm soil samples, in-calcium-chloride-extractable P (P-$CaCl_2$) (mg kg^{-1}), P-Olsen (mg kg^{-1}) and pH-KCl were analysed. All 0-10 cm and 10-20 cm soil samples were collected in the experimental plots and they were not pooled. All samples were collected in the adjacent parcel (and not in the hedgerow itself) in order to detect a potential gradient of hedgerow effect into the neighbouring parcel. To monitor activity-density and diversity of natural predators, pitfall traps were installed in May 2014 in the experimental plots on the same transects that were installed for yield measurements and they were emptied three times, every two weeks.

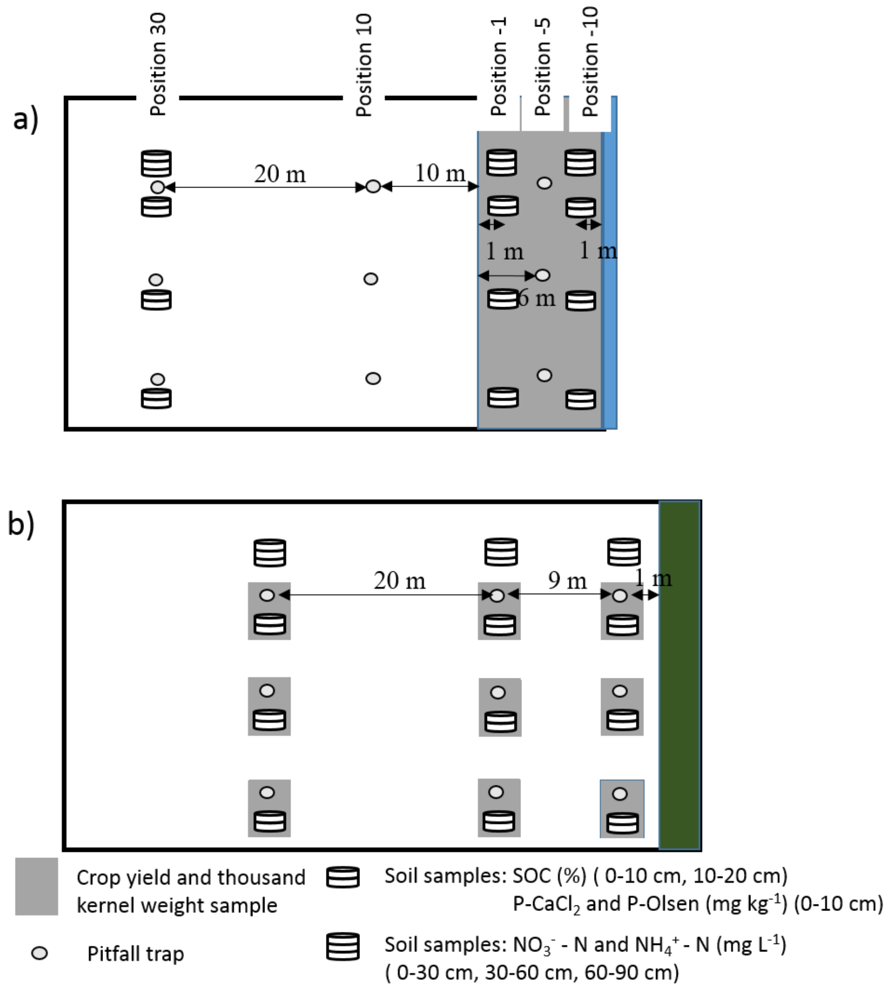


Figure 3.1: Experimental setup on the grass strip parcels (a) and hedgerow parcels (b)

3.3.3 Statistical analysis

To test whether the ecosystem service indicators differed significantly between the sampled positions and to account for the nested structure of the data, a mixed effects model (Zuur et al., 2009) to each of the ecosystem service indicators for the GS and HR parcels separately. Activity-density of carabids, spiders and rove beetles were described by means of a mixed model with a Poisson error distribution and log-link function. If overdispersion was significant, an observation-level random effect was added to the model (Elston et al., 2001). The consecutive pitfall trap collections were averaged before uptake in the model. All other indicators were analysed with a linear mixed effects model. To account for pseudo-replication, parcel was included as a random effect. Position of the plot in relation to the grass strip or hedgerow was included as a categorical fixed effect. Post-hoc Tukey tests were used to determine the statistical differences among the different positions ($p<0.05$). For the HR parcels, a second mixed effects model type was fitted. In this model, parcel was again included as a random effect, and distance from the hedgerow was included as continuous fixed effect. This model allows us to quantify a potential gradient of the indicators in the parcel next to the hedgerow. To allow comparison of different parcels with varying hedgerow heights, distance from the hedgerow is expressed in terms of its height. For this, H is used, which is the ratio of the distance from the hedgerow to the height (Van Vooren et al., 2016). This means e.g. that for a hedgerow height of 10 m and a plot on a distance of 5 m from the hedgerow, H equals 0.5. The Akaike Information Criterion (AIC) was used to test whether H of $\log_{10}$(H) was a better explanatory variable. The modelling was performed using the lme4 package (Pinheiro et al., 2016) in R, version 3.3.1 (R Development Core Team, 2016).

In order to identify potential indicator species for the different positions on the parcels, the Indicator Value (IndVal) index as defined by Dufrêne & Legendre (1997) is used. The IndVal index measures the association between a species and the positions where it is found. Association values can range from 0 (when the species was not found in any of the plots of a specific position) to 1 (when the species was only present in the plots of a specific position). Statistical significance of this association is calculated based on a permutation test (replicated 1000 times). The IndVal analysis was performed using the multipatt function in the indicspecies package. Finally, the effect of position on the parcel on carabid species composition (beta diversity) was assessed. Therefore, a dissimilarity matrix using the Bray-Curtis dissimilarity index based on activity-density data was calculated. The significance of the compositional differences was tested with a permutational multivariate analysis of variance (PERMANOVA) with 999 permutations using the adonis function in the vegan package (Anderson, 2001). To account for the multilevel structure of the data, permutations were constrained within parcels. Pairwise differences were tested using the multiple comparison tests with the Bonferroni correction. Because PERMANOVA can not distinguish among location (ordination in the 2-dimensional non-metric space (non-metric dimensional scaling, NMDS), determined by the species composition) and dispersion effects (around the centroid in the 2-dimensional non-metric space) (Anderson, 2001), multivariate heterogeneity of dispersion among the tested species compositions at the different positions was tested seperately by means of a multivariate analogue of Levene’s test for homogeneity of variances (Anderson, 2006) using the function betadisper in the vegan package. A significantly heterogeneous multivariate dispersion means that the PERMANOVA result cannot be solely attributed to a difference in the species composition but also to the variability within the positions.

3.4 Results

3.4.1 Grass strips

The results of the ecosystem service indicators measured on every GS parcel are presented in Figure 3.2 and in section 7.7.

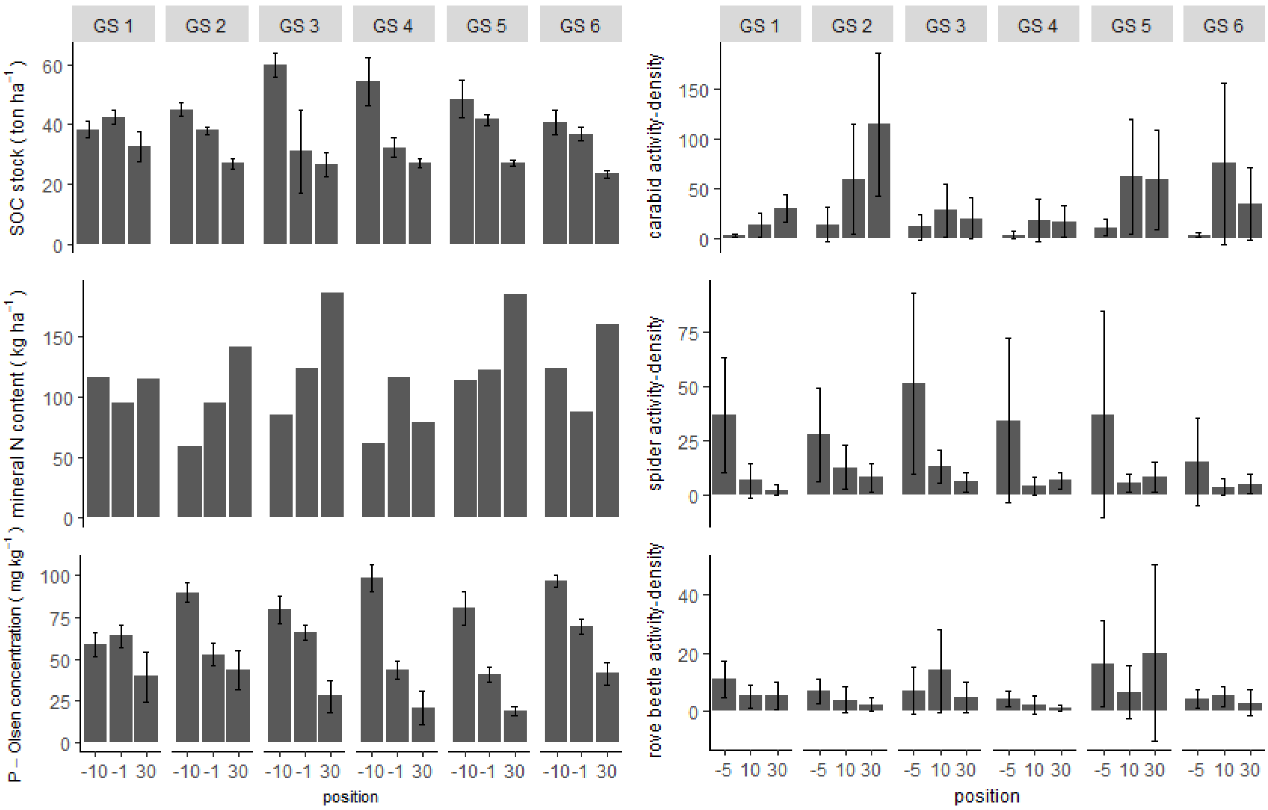


Figure 3.2: Mean ecosystem service indicator values measured on the grass strip parcels (GS 1 up to GS 6): soil organic carbon (SOC) stock (ton ha-1) (0-20 cm), soil mineral nitrogen (N) content (kg ha-1) (0-90 cm), P-Olsen concentration (mg kg-1) (0-10 cm), carabid activity-density, spider activity-density and rove beetle activity-density. Position -10 is in the grass strip next to the ditch, position -5 is in the middle of the grass strip, position -1 is in the grass strip next to the parcel, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel. Error bars represent standard deviations among samples from the same row. Because the N samples were pooled within the same row, no error bars are given.

On all parcels, total SOC stock (0-20 cm) in the grass strip was higher compared the adjacent parcel. Mean SOC stock was highest in the plot closest to the ditch.

Average mineral N and NO_3^- - N content (0-90 cm) were significantly lower in the plots next to the ditch (position -10) compared to the plots in the adjacent parcel (position 30) and intermediate values were found in the plots next to the parcel (position -1). In section 7.9, NO_3^- - N and NH_4^+ - N for each of the different positions on every GS parcel are presented.

Both P-Olsen and P-$CaCl_2$ concentration were significantly higher in the plots next to the ditch (position -10) and decreased in the plots next to the parcel (position -1) and in the plots in the parcel (position 30) (Table 3.3). In section 7.9, P-$CaCl_2$ concentrations are presented.

Table 3.3: Mean ecosystem service indicator values measured on the grass strip parcels. Position -10 is in the grass strip next to the ditch, position -5 is in the middle of the grass strip, position -1 is in the grass strip next to the parcel, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel.F-values with n.d.f. (numerator degrees of freedom) and d.d.f. (denominator degrees of freedom) indicate the significance of the tested model. Different letters within a row show significant differences among positions ($p < 0.05$, Tukey-test)

Ecosystem service	Ecosystem service indicator	In the grass strip			In the parcel		F-value of the tested model	n.d.f., d.d.f.
		Position -10	Position -5	Position -1	Position 10	Position 30		
Climate regulation	SOC stock (ton ha^{-1}) (0-20 cm)	47.73±8.80 a (n=18)		36.99±6.71 b (n=18)		27.17±3.63 c (n=18)	42.12	2, 46
Maintenance of chemical water quality	Mineral N content (kg ha^{-1}) (0-90 cm)	93.09±28.89 a (n=6)		106.51±15.78 a (n=6)		144.41±42.09 b (n=6)	5.56	2, 10
	NO_3^- - N content (kg ha^{-1}) (0-90 cm)	52.17±27.31 a (n=6)		64.92±19.86 ab (n=6)		98.73±37.04 b (n=6)	7.93	2, 46
	NH_4^+ - N content (kg ha^{-1}) (0-90 cm)	40.92±8.92 a (n=6)		41.59±8.45 a (n=6)		45.67±10.93 a (n=6)	82.99	2, 46
Maintenance of chemical water quality	P-$CaCl_2$ (mg kg^{-1}) (0-10 cm)	2.30±1.59 a (n=18)		1.08±1.08 b (n=18)		0.78±0.75 b (n=18)	15.05	2, 144
Maintenance of chemical water quality	P-Olsen (mg kg^{-1}) (0-10 cm)	84.40±14.96 a (n=18)		53.71±13.94 b (n=18)		34.37±15.72 c (n=18)	9.79	2, 144
Potential for pest control	Carabid activity-density		8.12±10.58 a (n=54)		43.38±52.53 b (n=54)	46.49±51.13 b (n=54)	28.88	2, 144
Biodiversity	Number of carabid species		3.26±2.37 a (n=54)		5.19±2.25 b (n=54)	4.73±2.53 b (n=54)	0.59	2, 144
Potential for pest control	Spider activity-density		33.72±34.64 a (n=54)		7.77±7.45 b (n=54)	6.22±5.33 b (n=54)	42.12	2, 46
Potential for pest control	Rovebeetle activity-density		8.22±8.59 a (n=54)		6.25±8.12 a (n=54)	6.45±14.39 a (n=54)	5.56	2, 10

On the GS parcels, 5370 carabids and 35 different carabid species were collected. *Bembidion tetracolum* (1227 individuals) and *Ptereostichus melanarius* (2246 individuals) comprised more than 64% of all individuals in the sampled population. Both species are typically found in open areas (Turin, 2000). Based on the IndVal indices, five species were found to be significant indicator species for the grass strips. Activity-density of carabids collected in the parcel was significantly higher compared to the grass strip. The average number of carabid species per trap was significantly lower in the plots in the grass strip (section 7.10). Both PERMANOVA and betadisper yielded significant p-values (p=0.001 and p=0.003, respectively) for the effect of position on the parcel on carabid species composition. More specifically, the carabid composition in the grass strips (position -5) differed significantly from the arable field (position 10: p=0.003, position 30: p=0.003). Based on PERMANOVA, it can not be concluded whether the carabid species composition differs among various positions on the parcel. However, the NMDS plot (section 7.11), indicates a difference in species composition between the plots in the grass strip and the plots in the arable parcel. On all GS parcels, 2854 spiders were found and 1852 individuals were collected in the grass strips. Activity-density of the spiders was significantly higher in the grass strip compared to the adjacent parcel. In total, 1229 rove beetle individuals were found and 420 individuals were collected in the grass strips. No trends in the activity-density of rove beetles was found.

3.4.2 Hedgerows

The results of the ecosystem service indicators measured on every HR parcel are presented in Figure 3.3 and in section 7.8.

Yields were significantly lower in the plots closest to the HR. Compared to the plots at 30 m from the hedgerow, relative yield was 72% in the plots at 1 m from the hedgerow and 96% in the plots at 10 m from the hedgerow. On the wheat parcels (HR 1, HR 2 and HR 4), 1000-grain weight (g) was significantly higher next to the hedgerow (Table 3.4). Both for crop yield and thousand kernel weight, H and $\log_{10}$(H) were significant predictors (Table 3.5).

Compared to the plots at 30 m from the hedgerows, average SOC stock in the 0-20 cm soil layer was slightly higher in the plots at 1 m from the hedgerow (8% higher) and in the plots at 10 m from the hedgerow (1% higher). The linear mixed effects model reports a decreasing SOC stock when H increases (Table 3.5). Average mineral N and NO_3^- - N content (0-90 cm) did not differ among different distances from the hedgerow and average NH_4^+ - N was higher in the plots on 10 m from the hedgerow (Table 3.4). In section 7.9, NO_3^- - N and NH_4^+ - N on the different positions on every HR parcel are presented.

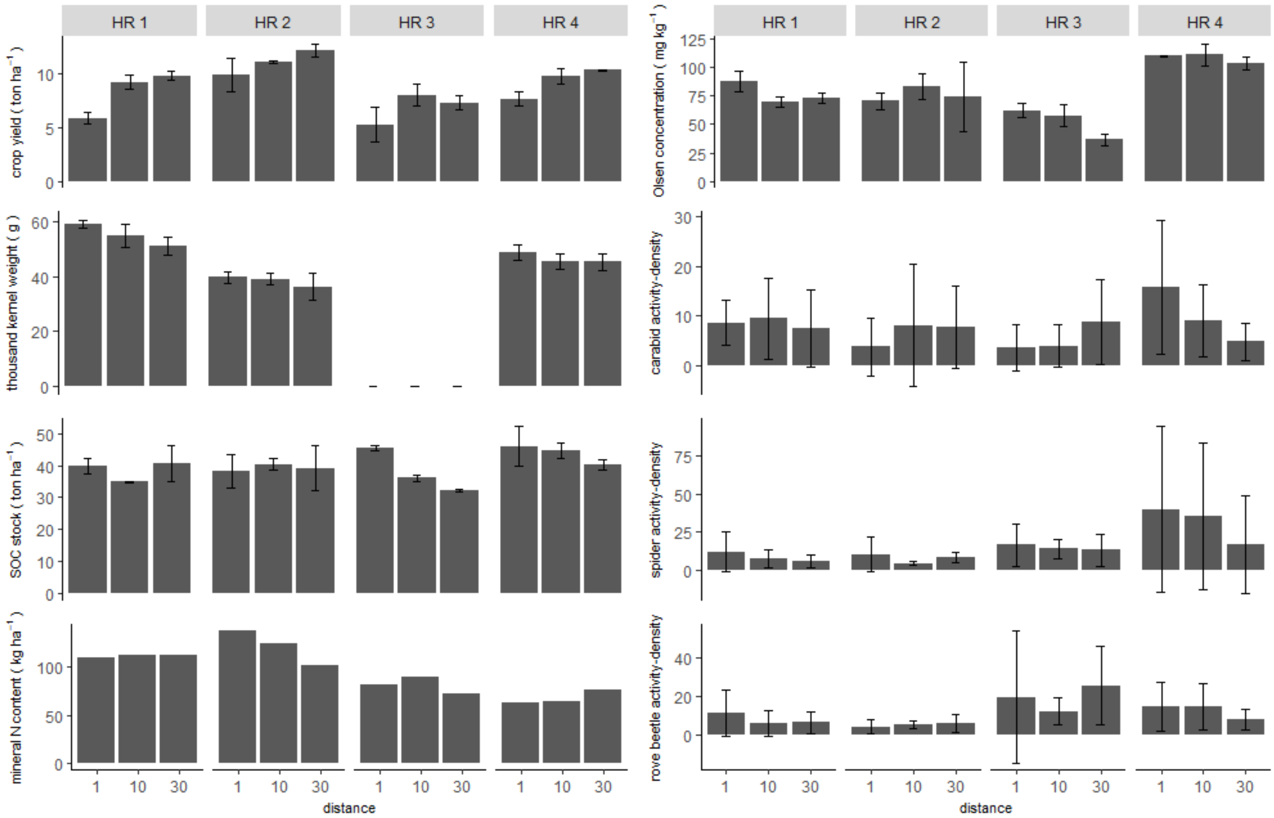


Figure 3.3: Mean ecosystem service indicator values measured on the hedgerow parcels (HR 1 up to HR 4): crop yield (ton ha-1), thousand kernel weight (g), soil organic carbon (SOC) stock (ton ha-1) (0-20 cm), soil mineral nitrogen (N) content (kg ha-1) (0-90 cm), P-Olsen concentration (mg kg-1) (0-10 cm), carabid activity-density, spider activity-density and rove beetle activity-density. Samples were collected on a distance of 1 m, 10 m and 30 m from the hedgerow. Error bars represent standard deviations among samples from the same row. Because the N samples were pooled within the same row, no error bars are given.

Observed P-CaCl2 concentrations were low and both P-CaCl2 and P-Olsen did not differ significantly across the various distances (Table 3.4). Only P-Olsen was significantly affected by H (Table 3.5). In section 7.9, P-CaCl2 concentrations are presented.

On the HR parcels, 577 carabids and 28 different carabid species were collected. Five species (*Anchomenus dorsalis, Bembidion properans, Bembidion tetracolum, Poecilus cupreus, Pterostichus melanarius*) comprised 85% of the total population that was sampled. These are all species typically found on open and cultivated lands (Turin, 2000). The IndVal indices did not return significant indicator species. Carabid activity-density and number of species per trap were not affected by distance from the hedgerow (Table 3.4, section 7.10). PERMANOVA did not indicate a difference in carabid species composition. Over all HR parcels, 1121 spiders and 812 rove beetle individuals were collected. Spider activity-density was significantly affected by distance from the hedgerow (Table 3.5). No trend was found in rove beetle activity-density among the different trap locations (Table 3.4, Table 3.5).

Table 3.4: Mean ecosystem service indicator values measured on the hedgerow parcels. F-values with n.d.f. (numerator degrees of freedom) and d.d.f. (denominator degrees of freedom) indicate the significance of the tested model. Different letters within a row show significant differences among distances from the hedgerow ($p < 0.05$, Tukey-test)

Ecosystem service indicator	Ecosystem service indicator	Distance from the hedgerow			F-value of the tested model	n.d.f., d.d.f.
		1 m	10 m	30 m		
Food production	Crop yield (ton ha^{-1})	7.17±2.14 a (n=12)	9.51±1.27 b (n=12)	9.90±1.86 b (n=12)	32.87	2, 30
Food production	Thousand kernel weight (g)	49.30±0.75 a (n=9)	46.52±1.08 ab (n=9)	44.34±0.96 b (n=9)	5.96	2, 22

Climate regulation	OC stock (ton ha^{-1}) (0-20 cm)	41.74±4.91 a (n=12)	38.76±4.15 a (n=12)	38.49±5.43 a (n=12)	1.53	2, 30
Maintenance of chemical water quality	Mineral N content (kg ha^{-1}) (0-90 cm)	97.91±32.34 a (n=4)	97.48±26.33 a (n=4)	90.61±19.68 a (n=4)	0.54	2, 6
Maintenance of chemical water quality	NO_3^- - N content (kg ha^{-1}) (0-90 cm)	47.80±29.28 a (n=4)	34.25±15.31 a (n=4)	36.68±14.73 a (n=4)	0.44	2, 30
Maintenance of chemical water quality	NH_4^+ - N content (kg ha^{-1}) (0-90 cm)	50.11±8.45 a (n=4)	63.22±15.34 b (n=4)	53.94±8.23 ab (n=4)	1.58	2, 30
Maintenance of chemical water quality	P-$CaCl_2$ (mg kg^{-1}) (0-10 cm)	3.15±1.82 a (n=12)	2.95±0.97 a (n=12)	2.58±1.96 a (n=12)	0.06	2, 64
Maintenance of chemical water quality	P-Olsen (mg kg^{-1}) (0-10 cm)	81.88±18.75 a (n=12)	79.70±20.57 a (n=12)	72.25±26.89 a (n=12)	0.05	2, 64
Potential for pest control	Carabid activity-density	8.03±9.11 a (n=36)	7.71±8.28 a (n=36)	7.15±6.83 a (n=36)	0.81	2, 64
Biodiversity	Number of carabid species	3.04±2.37 a (n=36)	3.25±2.31 a (n=36)	3.09±2.45 a (n=36)	0.26	2, 64
Potential for pest control	Spider activity-density	19.97±29.63 a (n=36)	15.53±25.77 a (n=36)	11.63±16.93 a (n=36)	32.87	2, 30
Potential for pest control	Rovebeetle activity-density	12.14±18.84 a (n=36)	9.42±8.27 a (n=36)	11.80±13.80 a (n=36)	5.96	2, 22

Table 3.5: Mixed model results for the prediction of ecosystem services on hedgerow parcels. For every ecosystem service indicator, the significance of the relative distance from the hedgerow (H and log_{10}(H)) is tested as a fixed effect. Indicators with (*) were predicted by means of a mixed model with a Poisson error distribution and log-link function. Other indicators were predicted by means of a linear mixed effects model.

Ecosystem service	Ecosystem service indicator	Model	AIC	Fixed effect p-value
Food production	Crop yield (ton ha^{-1})	8.09+0.21*H 8.46+1.91*log10(H)	142.3 116.0	**0.00** **0.00**
Food production	Thousand kernel weight (g)	48.92-0.85*H 47.05-3.27*log10(H)	151.3 148.0	**0.00** **0.00**
Climate regulation	OC stock (ton ha^{-1}) (0-20 cm)	41.74-0.56*H 40.23-2.42*log10(H)	180.8 179.9	**0.02** 0.08
Maintenance of chemical water quality	Mineral N content (kg ha^{-1}) (0-90 cm)	97.00+0.38*H 99.03+0.25*log10(H)	151.6 147.8	0.41 0.95
Maintenance of chemical water quality	NO_3^- - N content (kg ha^{-1}) (0-90 cm)	26.86-0.52*H 25.25-2.64*log10(H)	40.9 37.3	0.38 0.58
Maintenance of chemical water quality	NH_4^+ - N content (kg ha^{-1}) (0-90 cm)	55.30+0.13*H 54.89+4*log10(H)	92.8 88.3	0.87 0.36
Maintenance of chemical water quality	P-$CaCl_2$ (mg kg^{-1}) (0-10 cm)	3.26-0.10*H 2.98-0.38*log10(H)	117.5 115.1	0.17 0.35
Maintenance of chemical water quality	P-Olsen (mg kg^{-1}) (0-10 cm)	84.30-1.65*H 79.56-6.12*log10(H)	244.7 244.7	**0.02** 0.11
Potential for pest control	Carabid activity-density (*)	1.97+0.01*H 2.02-0.07*log10(H)	750.0 749.2	0.44 0.26
Biodiversity	Number of carabid species	2.94+0.06*H 3.13+0.05*log10(H)	322.4 319.5	0.36 0.90
Potential for pest control	Spider activity-density (*)	2.79-0.04*H 2.69-0.35*log10(H)	1421.0 1396.3	**0.00** **0.00**
Potential for pest control	Rovebeetle activity-density (*)	2.21+0.03*H 2.33-0.0.1*log10(H)	1033.2 1052.2	0.94 0.91

3.5 Discussion

3.5.1 Grass strips

Compared to an average SOC stock of 27.17 ton ha^{-1} in the adjacent parcel, conversion of arable land into a grass strip has resulted in a yearly SOC stock increase of 1.60 ton ha-1 year-1 (based on the average age of the sampled grass strips and the average SOC stock in the adjacent parcels). Among others, Poeplau et al. (2011), Goidts and van Wesemael (2007) and Kämpf et al. (2016) all reported yearly SOC stock increases in the upper soil layer (0-30 cm) after conversion from arable land to grassland in temperate areas. Rates varied between 0.44 ton ha^{-1} year^{-1} and 1.99 ton ha^{-1} year^{-1}. The findings of this study are very much in line with these results and SOC stock seems an appropriate indicator to describe the contribution of grass strips to climate regulation by carbon sequestration in the soil. Within the grass strips, in the plots next to the ditch, a higher SOC increase was observed compared to the plots next to the adjacent parcel. This might be caused by the deposit of sludge after ditch clearing or by enhanced grass biomass and root growth due to higher water availability during dry periods. As GS 1 was the only parcel without sludge storage and no higher SOC stock was observed next to the ditch of GS 1, sludge is assumed to be the cause of the observed increase in SOC stock next to the ditch.

In order to reduce NO_3^--N leaching and consequent eutrophication of surface water, Flemish legislation imposes a maximum NO_3^- -N content of 90 kg ha^{-1} in the 0-90 cm soil layer between October 1 and November 15 (Vlaamse Gemeenschap, 2006). Mean NO_3^- -N content of the GS parcels was 99 kg ha^{-1}. Four out of six parcels exceeded the maximum NO_3^--N content in the plots at 30 m from the grass strip and thus theoretically entail a higher risk for NO_3^--N leaching. In the grass strips, mineral N content was 34% lower next to the ditch and 26% lower next to the parcel compared to the adjacent parcel. Accordingly, NO_3^- -N content was 48% and 34% lower, respectively. A lower N content in the grass strip in flat regions, as in our case, is a strong indicator for reduced N leaching (Dhondt et al., 2002). Van Beek et al. (2007) investigated nitrogen dynamics in drained grass strip parcels in the Netherlands, which are very comparable to the parcels in our study, by measuring NO_3^- - N concentrations, Cl/ NO_3^- - N ratios and ϱ^{15}N values over two growing seasons. They deduced that the dominant water flow was towards the ditch and through the grass strip and that NO_3^- -N was removed from the grass strip due to denitrification and grass uptake. Based on the findings of Dhondt et al. (2002) and Van Beek et al. (2007), the gradient in measured N content was assumed to indicate a positive effect of the grass strip on the reduction of N leaching. Because of the zero fertilization and removal of biomass, we expected to find lower soil P concentrations in the grass strip. In contrast, P-Olsen and P-$CaCl_2$ were higher in almost all grass strips and this was more distinct next to the ditch. The abovementioned deposit of sludge next to the ditch could explain the higher P concentrations. This hypothesis was confirmed by the results from GS 1, where no sludge was stored and no higher P concentrations next to the ditch (compared to P concentrations next to the parcel) were observed. Higher P-Olsen concentrations in the grass strip in the row next to the parcel (position -1) could be caused by fertilizer misplacement (E. J. P. Marshall and Moonen, 2002), fertilization of the arable land before conversion to grass strips (Schelfhout et al., 2015) or changed physicochemical soil processes (Dorioz et al., 2006b). For example, higher SOC reduces the availability of soil P sorption capacity (Daly et al., 2001). Bhattarai et al. (2009) found that P concentrations in the upper 15 cm soil layer of a drained parcel decreased in the grass strip, that dissolved P moves through the soil profile and that P in the subsurface outflow is reduced. Surprisingly, no lower P concentrations were found in the upper soil layer in the grass strip. In order to estimate grass strip effectiveness for P removal in flat, drained regions, it is therefore recommended to measure water flows, as local management or historical land use can affect soil P.

Semi-natural vegetation elements like grass strips and hedgerows are very important as a hibernation habitat for predatory arthropods (Geiger et al., 2009; Pfiffner and Luka, 2000a) and after hibernation, they provide additional food resources for both parasitoids and predators (Bianchi et al., 2006). During their search for prey, predators spread in the surrounding crops (Bianchi et al., 2006; Marrec et al., 2015; Varchola and Dunn, 2001) and thus we would have expected to find higher activity-densities and species numbers in the grass strip and decreasing numbers with increasing distance to the grass strips. This was only confirmed by spider (and rove beetle, but not significant) activity-densities. This might indicate that spiders are more susceptible to disturbances and thus rely more on undisturbed habitats. Carabid activity-densities and species numbers did not confirm the hypothesis, possibly because in summer, arable parcel colonization starting from the grass strip is rare. Additionally, it should be noted that activity-density data are affected by soil surface vegetation and roughness, with a more smooth soil surface (e.g. arable land) resulting in higher activity-density (Thomas et al., 2006). Thus, the hypotheses regarding the contribution of grass strips to pest control and alpha diversity were only partially confirmed. Future trials should install pitfall traps in the grass strips and in the adjacent parcel in early spring, allowing the follow-up of predator post-hibernation emergence and colonization. Also, a more direct way of measuring pest control by measuring pest species or crop damage could be considered. Three of the five indicator species that resulted from the Indval method typically live in grassy riverbanks (*Amara plebeja*, *Pterostichus vernalis* and *Chlaenius nigricornis*). Also, one individual of *Panagaeus cruxmajor* and one of *Trechoblemus micros* were found in the grass strip. These species both have a habitat preference for riversides and banks. The Bray-Curtis indices indicate that grass strips support a different carabid species composition compared to the adjacent arable parcels and therefore contribute to a higher beta diversity of the agroecosystem as a whole.

While grass strips seem to enhance the delivery of multiple (regulating) ecosystem services, their implementation results in a loss of arable land and thus a parcel-level reduction of crop yield. When the grass biomass from the grass strips cannot be adequately valorised, this entails a true trade-off between provisioning and regulating ecosystem services (Van Vooren et al., 2017). On the other hand, as grass strips are expected to enhance the presence of natural predators, it may be expected that pest damage to crops will be reduced next to the grass strip, which may increase yield. However, the potential of this service may be undermined by the use of pesticides in conventional farming in certain crops (Sutter et al., 2017), as was the case for the monitored parcels. Hence, in this particular situation, where the input of pesticides is unaffected by the grass strip, this service is not delivered and the impact of the grass strip on crop yield by pest control may be non-existing or negligible.

3.5.2 Hedgerow measurements

The review papers by Kort (1988), Van Vooren et al. (2016) (see chapter 5) and Van Vooren et al. (2017) (see chapter 2) described the reduction of crop yield next to tree rows and hedgerow and the magnitude of this effect was similar to the reductions that were measured in this study. Song and Wei (1991) investigated the effect of separate shelter-induced microclimate components on winter wheat. They found that solar radiation reduction results in lower thousand kernel weight, but wind speed reduction, air saturation deficit reduction and increased soil moisture seemed to increase thousand kernel weight. In our study, it seems that the thousand kernel weight increasing processes are dominating. Overall, crop yield is an appropriate indicator for food production but crop characteristics need to be considered as well. For example, the shift in thousand kernel weight could negatively or positively affect the quality of the product, depending on its later use.

A SOC stock increase in the upper soil layers of arable parcels next to tree rows has been described by Pardon et al. (2017) and the order of magnitude of the increase is similar to our own measurements. Cardinael et al. (2015) reported that next to tree rows, the largest share of SOC increase can be found in the upper soil layer (0-30 cm). Upson and Burgess (2013) found a similar increase in the 0-40 cm soil layer of an agroforestry system, but they found lower SOC stocks in the deeper soil layers next to tree rows (up to 150 cm). Also Nair (2012) stated that shallow soil layer sampling is problematic and inadequate to describe the impact of tree rows on SOC because tree roots extend to the deeper soil layers. Because we only sampled the 0-20 cm soil layer, we assume that we have detected the main positive effect on SOC stock but that we have missed potential (negative) changes in the deeper soil layers. Also, SOC stock seems an appropriate indicator to describe the contribution of hedgerows to climate regulation, but it does not capture the effect completely, because additional carbon can be stored in the woody biomass of the hedgerow (Falloon et al., 2004).

None of the parcels exceeded the maximum allowed NO_3^- - N content of 90 kg ha^{-1} in 0-90 cm soil layer between October 1 and November 15 (Vlaamse Gemeenschap, 2006), entailing no risk for NO_3^--N leaching. Our experimental setup allowed us to detect whether nutrient uptake by the hedgerows reduced soil N and P and hence whether hedgerows contribute to the maintenance or improval of the chemical water quality. We found no direct evidence for nutrient uptake by the hedgerow and more surprising, the relationship between H and P-olsen was significant, with higher P concentrations closer to the hedgerow. Because P concentrations were sampled in the 0-10 cm soil layer, due to ploughing the effect of nutrient uptake on soil P concentrations could probably not be detected. Pardon et al. (2017) found a similar limited but significant increase of ammonium lactate–extractable P closer to tree rows planted on arable parcels. Input of litterfall and reduced crop-uptake were suggested to be at the base of this increase. Because the correlation between P-Olsen and AL-P is high (Carmo Horta et al., 2010), results of this study are in line with Pardon et al. (2017). In order to investigate the contribution of hedgerows to the maintenance of chemical water quality, complexity of the nitrogen and phosphorus cycle might call for a more integrated experimental setup. For example, interception of atmospheric wet and dry N deposition and subsequent N throughfall and litter decomposition (Remy et al., 2016) might neutralize soil N uptake by the hedgerow or even increase soil N content near the hedgerow. Several studies (Borin et al., 2005; Duchemin and Hogue, 2009; Salazar et al., 2015; Schmitt et al., 1999; Schoonover et al., 2005) reported a positive impact of hedgerows on the maintenance of chemical water quality. In these studies, N and P concentrations were monitored in surface and subsurface water flows instead of the soil. Therefore, it seems that the limited experimental setup we applied is not appropriate and in order to further investigate and quantify potential N and P removal by hedgerows, the experimental setup should be more comprehensive and hydrological processes should be considered, for example by sampling the soil both upslope and downslope from the hedgerow or by sampling the water flow.

Hedgerows are important for predatory arthropods as a hibernation habitat and for the provision of additional food resources (Bianchi et al., 2006; Pfiffner and Luka, 2000). Therefore, more predators were expected near the hedgerows. Only spider activity-density data confirmed the hypothesis. Carabids and rove beetles seemed not to be affected by the hedgerows. In order to further assess the role of hedgerows in the lifecycle of predators, pitfall trapping in early spring, allowing the follow-up of predator post-hibernation emergence, is recommended. Five individuals of *Limodromus assimilis*, one of *Notiophilus quadripunctatus* and eleven of *Pterostichus strenuus* were collected on the HR parcels. All three species are typically found in deciduous forests. Their presence on the arable parcels might be explained by the hedgerows next to the parcels. Still, none of these species were related to the plots closer to the hedgerow. Similar to grass strips, the follow-up of post-hibernation emergence would help to identify the role of hedgerows for natural predators and the measurement of pest species could be a more direct indicator for pest control.

3.6 Conclusions

The monitored indicators suggested a positive contribution of grass strips to climate regulation, the maintenance of chemical water quality (by reduction of N leaching), potential pest control (by increased spider activity-density) and beta diversity but their implementation results in a reduction of parcel-level yield. We found no evidence for the reduction of phosphorus leaching nor for increased carabid and rove beetle activity-density or alpha diversity. On the hedgerow parcels, close to the hedgerow, we found evidence for a change in food production (a decrease in crop yield and increase in thousand kernel weight), a positive effect on climate regulation and a potential contribution to pest control (by increased spider activity-density). We did not measure a contribution to the maintenance of chemical water quality nor to increased potential pest control by carabids or rove beetles nor to alpha and beta carabid diversity. It should be stressed that the data are the result from a one-year-monitoring campaign, therefore providing only a snapshot of potential impact on ecosystem service delivery. Both grass strips and hedgerows have a great potential for the delivery of multiple ES (Van Vooren et al., 2017) but results from this study suggest that management of the grass strip, hedgerow and close environment will affect to what extent an effect can be expected. For example, ploughing next to the hedgerow enhances carbon mineralisation and hampers hedgerow root growth in the upper soil layers and infiltration of surface water nutrients, impeding both an increase of SOC stock and nutrient removal from the water flow. Similarly, mowing of grass strips will encourage nutrient uptake and removal, contributing to the regulation of chemical water quality, but will simultaneously increase disturbance in the grass strip, which may have a negative effect on natural predator abundance and diversity. Therefore, in order to increase effectiveness of semi-natural vegetation for ES delivery, specific management schemes, targeting the prioritized ES, are required and trade-offs have to be made. To unravel whether some of the hypotheses were not confirmed due to a failed change in ES delivery or due to inappropriate indicator choice or experimental setup, further research is needed. Therefore, we suggested a number of improvements for the experimental setup of future research. All this stresses the importance of the simultaneous assessment of a well thought-through set of multiple ecosystem service indicators on real-life parcels.

4. Assessing the impact of grassland management extensification in temperate areas on multiple ecosystem services and biodiversity: monitoring and a literature review

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4.1 Abstract

In order to halt further biodiversity loss in the agricultural landscape, measures for grassland management extensification have been proposed and implemented. Apart from biodiversity conservation and enhancement, these measures are expected to affect a range of ecosystem services delivered by these grasslands. It is well-known that grasslands have the potential to contribute to the delivery of multiple ecosystem services, but there generally is a trade-off between provisioning services and regulating services, which is strongly linked to grassland management. This study investigated the effect of grassland management type and intensity on multiple ecosystem service and biodiversity indicators. To do so, two sets of grasslands in Flanders with varying management types were monitored: a regular, intensive management, a meadow bird management and a botanical management. For every monitored grassland, a land use intensity index was calculated and linked to the ecosystem service and biodiversity indicators. The results showed that biomass yield, forage quality, soil mineral N content and number of plant species differed among the various management types and that increasing land use intensity resulted in higher biomass yields, forage quality and soil mineral N content and in a lower number of plant species. However, it was observed that other factors such as the timing of the first cut affected these variables as well. A literature review was subsequently performed to quantify the link between land use intensity of other temperate grasslands and the same response variables. Results of the literature review confirmed the trends that were found in the monitoring data, but an additional effect of animal fertilizer application on soil carbon stock was noted. Taken together, the results suggest that the impact of grassland management in terms of fertilization, mowing and grazing on the selected ecosystem service delivery and biodiversity indicators can be predicted, but that other management components should be considered as well.

4.2 Introduction

Semi-natural grasslands are among the most diverse ecosystems in Europe. They can host up to 80 plant species per square meter, many bird species depend on grasslands for feeding, nesting and wintering, and almost 50% of all European butterfly species are typically found in grasslands (Habel et al., 2013; Vickery et al., 2001; WallisdeVries and Van Swaay, 2009). Semi-natural grasslands are maintained by human management, especially mowing and grazing (European Environment Agency, 2015). In north western Europe, almost all grassland management is intensified in order to increase productivity, for example through grassland renewal, the application of fertilizers and pesticides, intensive mowing and grazing and drainage. This has resulted in a significant drop in grassland biodiversity (Batáry et al., 2015; Habel et al., 2013; Plantureux et al., 2005). Measures have been proposed to put an end to biodiversity loss, for example via agri-environment schemes or the Natura 2000 network (EU Birds and Habitats Directive). In Flanders, 30% of the agricultural area is in grassland management (FOD Economie, 2016). Seventy percent of these grasslands are intensively managed and highly productive. The remaining grasslands are more extensively managed because they are less suitable for agricultural production or because they are subject to biodiversity conservation targets (Meiresonne and Turkelboom, 2012). In this case, their management is restricted in terms of fertilization intensity and/or timing and intensity of mowing and grazing.

Apart from hosting biodiversity, all grasslands have the potential to contribute to the delivery of multiple ecosystem services (ES), such as climate regulation, water quality regulation and soil quality regulation (Maes et al., 2011). However, there seems to be a trade-off: negative relationships have been described between high forage yield (a provisioning ES) on the one hand and many other regulating ES, biodiversity conservation and landscape quality on the other hand (Maes et al., 2011; Pilgrim et al., 2010). Thus, in almost all cases, extensifying grassland management in order to increase biodiversity is expected to have a negative effect on provisioning ES and a positive effect on several regulating ES.

Grassland management consists of many aspects (fertilization amount, fertilization type, number of cuts, livestock management etc.) which could all affect provisioning and regulating ES and biodiversity. Despite of a growing interest in the effect of grassland management on multiple ES delivery, quantification of the expected effects of varying management is still missing. Several studies have compared presence and absence of a management practice, for example grazed versus ungrazed grasslands (Cichota et al., 2016; Grandchamp et al., 2005) or mown versus unmown grasslands (Callaham et al., 2003; Eriksen-Hamel and Whalen, 2006), but these studies do not capture the impact of varying management intensity (Blüthgen et al., 2012). The effect of fertilization intensity on several ES has been described (Malhi et al., 2005; Müller et al., 2011), but there are few studies that have investigated the association between the intensity of fertilization or more broadly the intensity of grassland management and a broad set of ES and biodiversity simultaneously. Consequently the trade-offs and interactions between these different components remain insufficiently understood (Batáry et al., 2015; Pilgrim et al., 2010).

The objective of our study was to evaluate the impact of management type (regular, meadow bird or botanical management) on a set of provisioning and regulating ES and on biodiversity components, based on an on-field assessment on two sets of grasslands in Flanders. We also quantify the relationship between management intensity and the same set of response variables. The definition of management intensity was based on a fertilization, mowing and grazing component (Blüthgen et al., 2012). A second objective is to examine whether the observed trends were in line with the results from other studies on temperate grasslands. Thereto, a systematic literature review was performed and relationships between management intensity and provisioning and regulating ES and biodiversity components were quantified. Finally, we explored whether there are trade-offs in grassland management with respect to ES delivery and biodiversity.

4.3 Materials and study area

4.3.1 Study area

In 2014, twelve grassland parcels, which could be categorized into three different management types, were monitored (Table 4.1). These grasslands were located in Turnhouts Vennengebied (TVG), in the Campine region in the north of Flanders (51°21'48.2"N 4°54'50.6"E). Soils are Gleyic Podzols and the texture is sandy (IUSS Working Group WRB, 2006). Four grasslands were not subject to specific management restrictions or prescriptions apart from general Flemish legislation. They were in conventional agricultural use and will be referred to as CON grasslands. Four grasslands were owned by the Flemish Agency for Nature and Forest (ANB), who granted concessions to farmers for management under specific conditions. Management focuses on the promotion of meadow bird populations. Meadow birds typically thrive under a postponed first cut or grazing activity in order to reduce egg and chick mortality. The soil should have a sufficiently high organic carbon content, enhancing availability of invertebrates which make up an important part of the diet of meadow birds (Breeuwer et al., 2009). These grasslands were therefore fertilized with farmyard manure with a total application restricted to 120 kg N/ha. Grazing was permitted after June 15th and mowing after July 15th. Application of pesticides was not allowed. Based on the fertilization type that was applied, these grasslands are referred to as FYM grasslands. Finally, four grasslands were owned by Natuurpunt, the largest nature conservation organisation in Belgium. Similar to the FYM grasslands, Natuurpunt granted concessions to farmers for the exploitation of the grasslands. They were managed to increase botanical diversity by means of nutrient depletion. No fertilizers or pesticides were applied on these grasslands. Both grazing and mowing were only allowed after July 15th. These grasslands are referred to as zero input (ZER) grasslands. In 2015, six grasslands, corresponding with two management types, were monitored (Table 4.1). Grasslands were situated in Bos van Aa (BVA), located centrally in Flanders (50°59'20.2"N 4°23'59.6"E). Soils are Gleyic Lumisols and the texture is sandy loamy (IUSS Working Group WRB, 2006). Three grasslands were not subject to specific management restrictions or prescriptions, apart from general Flemish legislation. They were in conventional agricultural use and will also be referred to as CON grasslands. Three grasslands were managed by Natuurpunt. They were mown after June 15th and no fertilizers or pesticides were applied on these grasslands (ZER grasslands). Within both study sites, all selected grasslands were in close vicinity in order to reduce heterogeneity in terms of surrounding landscape, soil conditions, etc. Before the implementation of meadow bird or botanical management, all grasslands were intensively managed. Grasslands were only selected for monitoring if they had received the same management for at least three years. The mean annual temperature in (Ukkel) Flanders was 11.9 °C in 2014 and 11.3 °C in 2015 and annual precipitation was 784.3 mm in 2014 and 742.4 mm in 2015 (KMI, 2018).

Table 4.1: Overview of the monitored grasslands in both study regions in terms of fertilization applied (type and N amount), the number of cuts by the farmer and the number of livestock unit grazing days. Also, the number of years since ploughing

(grassland age), the pH-KCl and LUI values are given. NM stands for not measured, as the topsoil of this grassland was removed before samples were collected.

Study region	Parcel	Fertilization type	Application of available N (kg/ha)	Number of cuts by the farmer	Number of livestock unit grazing days	Grassland age (years)	pH-KCl	LUI
TVG	CON1	Cow slurry	203	6	0	4	4.71	4.54
	CON2	Cow slurry	203	6	0	3	5.13	4.54
	CON3	Cow slurry	154	3	0	18	5.24	2.77
	CON4	Cow slurry	154	1	89	18	5.40	4.13
	FYM1	Farmyard manure	135	1	100	15	4.78	4.23
	FYM2	Farmyard manure	135	1	133	15	4.85	5.07
	FYM3	Farmyard manure	128	1	62	24	5.08	3.21
	FYM4	Farmyard manure	128	1	97	20	4.95	4.07
	ZER1	No fertilization	0	2	0	3	4.81	0.86
	ZER2	No fertilization	0	2	0	4	4.81	0.86
	ZER3	No fertilization	0	2	0	14	NM	0.86
	ZER4	No fertilization	0	2	0	18	4.78	0.86
BVA	CON1	Cow slurry	92	4	0	10	4.79	4.18
	CON2	Cow slurry	92	1	199	10	4.90	5.36
	CON3	Cow slurry	92	0	225	10	4.51	5.19
	ZER1	No fertilization	0	2	0	16	3.95	1.09
	ZER2	No fertilization	0	2	0	16	4.07	1.09
	ZER3	No fertilization	0	2	0	16	4.04	1.09

4.3.2 Experimental data collection

We measured dry biomass yield and forage quality as provisioning ES and climate regulation and regulation of chemical water quality as regulating ES. Grasslands play a role in the process of climate regulation, because they can store a considerable amount of carbon, especially compared to arable lands. Furthermore, the impact of grassland management is stated to have an effect as well (D'Hose and Ruysschaert, 2017; Nelissen et al., 2016). Both for arable land as for grassland, fertilization restrictions (related to fertilizer type and intensity) have been imposed in order to protect both surface water and groundwater from nitrate pollution and to enhance the chemical water quality (VLM, 2018a). Finally, number of carabid and plant species were monitored as biodiversity components. Measurements were performed in Flanders on two sets of grasslands with varying management.
For every ES and for biodiversity, parcel-level indicators were selected and monitored. Total grass yield (ton ha^{-1}), crude protein concentration (%) and yield (ton ha^{-1}) were the indicators for forage productivity and quality. To enhance interpretation of the crude protein data, organic matter digestibility (OMD) (%) was additionally measured. Soil organic carbon (SOC) stock (ton ha^{-1}) was an indicator for climate regulation because a higher soil carbon stock implies that more CO_2 has been captured (Smith et al., 2000). Soil mineral nitrogen (N) content (kg ha^{-1}) was selected as an indicator for chemical water quality, because soils with more N entail a higher risk for N leaching (Dhondt et al., 2002) and thus a negative effect on the maintenance of chemical water quality. Biodiversity was expressed in terms of the number of carabid and plant species (alpha diversity) and in terms of the difference among carabid and plant species compositions under different management types (beta diversity).

On every grassland parcel, three plots were selected for ES and biodiversity indicator monitoring. When the grasslands were grazed, plots were fenced. Fencing excludes the direct impact of cattle, but because the grasslands were under the same management for at least three years, we expected to measure the potential impact of the cattle of previous years. Grazing of the grasslands was rotational. In order to measure forage yield, the grass was mown with a cutter bar mower (1.4 m wide) over a length of 8 m. Mowing of the plots was done just before the farmer mowed the rest of the parcel. When the parcel was grazed, the plots were mown every month or every two months, depending on grass (re-)growth. In TVG, CON1 and CON2 plots were mown six times, CON3, CON4 and all FYM grassland plots were mown four times and all ZER grassland plots were mown twice. On FYM2, yield could not be measured because the fences were destroyed early in summer, most probably by the cattle. In BVA, CON grassland plots were mown four times and ZER grassland plots were mown twice. Fresh herbage yield was recorded in the field. Herbage samples of about 300 g were taken per plot and oven-dried at 65°C to calculate dry biomass yield. Dry matter samples were used for the determination of crude protein concentration and OMD. This was done using the near-infrared spectroscopy (NIRS) method (Corson et al., 1999; De Boever et al., 1996), employing the NIRS regressions developed at the Research Institute for Agriculture, Fisheries and Food (ILVO). Crude protein yield (ton ha^{-1}) was determined by multiplying the protein concentration with biomass yield of every cut and every plot.

A pitfall trap (9 cm opening diameter, volume 300 ml), half filled with a 50% propylene glycol (antifreeze) solution with detergent, was installed in every plot. Pitfall traps were set up in the first half of May and were emptied three times, approximately every two weeks. The collected species were stored in 70% ethanol. For every individual pitfall trap, carabids were determined to species level according to Boeken et al. (2002). Between 13/06 and 20/06, a 2 m x 2 m quadrat was set out in every plot and vascular plants within the quadrat were surveyed according to van der Meijden (1996). The cover of all species was recorded using the Braun-Blanquet method.

Soil samples were collected between October 1st and November 15th to assess SOC stock and mineral N content. To determine SOC stock, samples were taken from the 0-10 cm and 10-20 cm layer and organic carbon concentration (%), bulk density (BD) (g cm^{-3}), pH-KCl and soil moisture content (%) were measured. Soil BD was determined on multiple undisturbed soil samples taken from each experimental plot using Kopecky rings (height = 5.1 cm, diameter = 5 cm). Total carbon concentration was determined using a Vario MACRO cube Elementary analyser. Because all soil samples had pH-KCl values below 6.4, inorganic carbon (carbonate) was considered negligible and total and organic carbon concentration were assumed to be equal. SOC stocks (ton ha^{-1}) were calculated based on organic carbon concentration, BD, soil moisture content and sampling depth. Nitrate-N (NO_3^--N) and ammonium-N (NH_4^+-N) concentrations (mg L^{-1}) were sampled in the 0-30 cm, 30-60 cm and 60-90 cm layer. Per depth layer, samples were pooled for every grassland. NO_3^- - N and NH_4^+ - N were extracted in 1M KCl (extraction ratio 1:2) and determined colorimetrically. NO_3^--N and NH_4^+ -N content (kg ha^{-1}) was calculated based on BD and sampling depth. This procedure was in accordance with the soil sampling method prescribed by Flemish regulations, allowing the monitoring of NO_3^-- N leaching risk on agricultural soils (Vlaamse Gemeenschap, 2006) and performed in the BELAC accredited lab for Plant, Soil and Substrates (ILVO). In order to reduce NO_3^- - N leaching and consequent eutrophication of surface water, Flemish legislation imposes a maximum NO_3^-- N content of 90 kg ha^{-1} in the 0-90 cm soil layer between October 1 and November 15. Mineral N content was calculated as the sum of NO_3^-- N and NH_4^+-N content. On grassland Z2 in TVG no soil samples were collected, because the 0-10 cm upper soil layer had been removed in October 2014 in the context of a biodiversity restoration project.

4.3.3 Calculation of LUI and link with indicators

In order to develop generally applicable relationships between grassland management intensity and ES and biodiversity, a compound, quantitative definition of management intensity is needed, allowing for comparison across different regions and studies. The definition of grassland management intensity used in this study is based on Blüthgen et al. (2012) and will be referred to as the Land Use Intensity (LUI) index. It is a quantitative, additive index and consist of the standardized intensity of three components, namely fertilization (LUI_F), mowing (LUI_M) and livestock grazing (LUI_G) with F_i being the nitrogen fertilization level (kg ha^{-1} $year^{-1}$), M_i the number of cuts per year and G_i the number of livestock unit days of grazing per year. Grazing cattle was converted to livestock units as presented by Fischer et al. (2010). Cow slurry was assumed to have a nitrogen content of 5.2 kg ton^{-1} and farmyard manure a nitrogen content of 8.5 kg ton^{-1} (Coppens, 2009) and nitrogen availability in both fertilizer types was estimated to be 60% and 30%, respectively (VLM, 2017). For each plot, F_i, M_i and G_i was standardized relative to the mean of the study region (F_R, M_R and G_R, respectively) (Formula 1).

$$LUI_i = LUI_{Fi} + LUI_{Mi} + LUI_{Gi} = \frac{F_i}{F_R} + \frac{M_i}{M_R} + \frac{G_i}{G_R} \quad \text{(Formula 1)}$$

In our calculations, TVG and BVA were considered as two different regions (thus resulting in different F_R, M_R and G_R values). Despite the fertilization restrictions on the FYM parcels, average LUI in TVG on the FYM grasslands was higher (4.15±0.76) compared to the CON grasslands (3.99±0.84). In TVG, minimum LUI was 0.86 (ZER parcels) and maximum LUI was 5.07 (FYM). In BVA, LUI varied between 1.09 (ZER parcels) and 5.36 (CON2).

We developed quantitative relationships between LUI indices of the monitored grasslands and standardized indicator values. Calculation of standardized ES indicator values was similar to LUI calculation: for example for yield, the ratio was taken of the total yield in a plot to the average of all total plot yields in the study region (either TVG or BVA). Using standardized ES indicators allows us to compare ES across different situations and studies (e.g. environmental conditions, cultivation history, weather etc.). For example, different weather patterns resulted in generally higher yields in 2014 than in 2015, but standardization allows to link relative differences between grasslands to a variation in LUI.

4.3.4 Literature review

On Web of Science, a literature research was conducted based on the following search terms: grassland AND fertiliz* AND (yield OR carbon OR nitrogen OR biodiversity). The search was updated until 17.08.2017. Studies were retained if they met the following criteria: (i) the study region is situated within the temperate regions of the globe (as defined by Olson et al. (2001)), (ii) empirical data of the indicator of interest are available (modelling studies are thus excluded), (iii) the study compares the effect of varying grassland management intensities (LUI), (iv) before the start of the experiment, all grasslands had the same management and soil characteristics, and (v) in case of grazing, information on number of livestock units is given. Within every study, varying fertilizer applications were a required component of management intensity and we did not retain studies describing only the effect of varying grazing intensities or cutting regimes. Mowing and grazing without fertilization are typically disturbance measures with a positive effect on biodiversity (van der Maarel, 1993). However, the presence or absence of fertilization will most presumably affect the way grazing and cutting influences ES and biodiversity indicators. Because the focus of this paper is on grassland management within an agricultural context, fertilization was considered a mandatory component of management intensity and thus LUI.

Because the original search did not return any relevant study describing carabid diversity, an additional search was performed based on the following terms: (grass* OR meadow OR pasture) AND carabid*. Despite our additional literature search, we found no studies describing the impact of varying grassland management intensity on carabid diversity in grasslands. However, we did retain the study by Allan et al. (2014), describing the number of Coleoptera species on grasslands with different LUI indices. Additionally, the study by Siemann (1998), linking the number of arthropod species to grassland management intensity, was included.

When results were only given in figures, the data were extracted using WebPlotDigitizer v3.10 (Rohatgi, 2014). When the experiment reported varying phosphorus or potassium fertilization levels linked to the same level of N fertilization, only the most intensive fertilization management (with the highest P and K levels) was retained for further analysis. When a study did not report the number of cuts, it was assumed to be equal for all plots. Finally, we retained 14 studies on grass yield, 11 studies on crude protein yield, 17 studies on SOC stock, 17 studies on soil N content, 2 studies on carabid diversity and 9 studies on plant diversity (Table 4.2).

For every individual study, LUI of the experimental plots was calculated based on the fertilization, mowing and grazing of the specific plot (F_i, M_i and G_i, respectively) and on the average fertilization, mowing and grazing of all plots (F_R, M_R and G_R, respectively) within the same study, region and year. Relative ES indicators were calculated the same way. Both crude protein and nitrogen concentrations from grassland biomass were retained as forage quality indicators. Because we consider relative indicator changes and because crude protein concentration equals 6.25 times the total nitrogen concentration (De Cauwer et al., 2006), this will not affect the result. For the regulating ES, both concentrations and stocks were used in the calculations. Also, we used the same approach as Mayer et al. (2007) and thus did not distinguish among different mineral N forms.

Table 4.2: Overview of the studies used in the literature review. For every study, the study location, the applied fertilizer type, nitrogen (N) application level, the number of cuts and the number of livestock units is given. NPK = nitrogen phosphorus kalium, CMS = cattle manure slurry, PMS = pig manure slurry, CAN = calcium ammonium nitrate, AN = ammonium nitrate, FYM = farmyard manure. NG stands for not given.

Indicator	Author	Year	Location	Fertilizer type	N Fertilization level (kg ha^{-1})	Number of cuts	Number of livestock units
Biomass yield	Bobbink	1991	The Netherlands	NPK	0, 100	NG	0
	Cop	2009	Slovenia	NPK	0, 50, 100, 150, 200	2, 3, 4	0
	Eriksen-Hamel	2006	Canada	NPK, CMS	0, 75	1	0
	Fornara	2016	Ireland	NPK, CMS, PMS	0, 78, 96, 162, 192, 200, 324, 384	3	0
	Harty	2017	Ireland	CAN	0, 200	5	0
	Hejcman	2012	Czech Republic	NPK	0, 300	2	0
	Lkhagvasuren	2011	Canada	NPK	0, 56, 112, 224	NG	0
	Nevens	2003	Belgium	NPK	0, 100, 200, 400	3, 4, 5	0
	Schils	2003	The Netherlands	CAN	0, 75, 243	NG	0
	Schroder	2007	The Netherlands	CAN	0, 170, 340, 510	5	0
	Sochorova	2016	Germany	CAN	0, 100	2	0
	Spohn	2016	Austria	NPK	0, 120	3	0
	Trott	2004	Germany	CAN	0, 100, 200, 300	4	0
	van Eekeren	2009	The Netherlands	CAN, CMS	0, 150, 162.8	4	0
Crude protein yield	Bobbink	1991	The Netherlands	NPK	0, 100	NG	0
	Cop	2009	Slovenia	NPK	0, 50, 100, 150, 200	2, 3, 4	0
	Eriksen-Hamel	2006	Canada	NPK, CMS	0, 75	1	0
	Harty	2017	Ireland	CAN	0, 200	5	0
	Hejcman	2012	Czech Repulic	NPK	0, 300	2	0
	Lee	2007	South Dakota	AN, FYM	0, 112, 224	1	0
	Lkhagvasuren	2011	Canada	NPK	0, 56, 112, 224	NG	0
	Nevens	2003	Belgium	NPK	0, 100, 200, 400	3, 4, 5	0
	Schils	2003	The Netherlands	CAN	0, 75, 243	NG	0
	Schroder	2007	The Netherlands	NPK	0, 300	4, 5	0
	Schroder	2010	The Netherlands	CAN	0, 170, 340, 510	5	0
Soil carbon stock	Bélanger	1999	Canada	NPK	0, 90, 180, 270	NG	0
	Fornara	2016	Ireland	NPK, CMS, PMS	0, 78, 96, 162, 192, 200, 324, 384	3	0
	Gudmundsson	2004	Iceland	NPK	0, 75, 120	NG	0
	Hassink	1994	The Netherlands	Mineral N	0, 200, 400	NG	0
	Kidd	2017	UK	NPK, FYM	0, 35, 100, 120	NG	0
	Lee	2007	South Dakota	AN, FYM	0, 112, 224	1	0
	Lkhagvasuren	2011	Canada	NPK	0, 56, 112, 224	NG	0
	Malhi	1997	Canada	AN	0, 56, 112, 168, 224, 336	2	0
	Malhi	2002	Canada	AN	0, 112	1	0
	Nyborg	1999	Canada	Mineral N	0, 112	NG	0
	Olson	2006	Canada	CMS	0, 100, 200, 300, 400	NG	0
	Reeder	1998	Wyoming	AN	0, 34	NG	0
	Riggs	2015	US Central Great Plains	NPK	0, 100	NG	0
	Schwab	1990	Kansas	CAN	0, 67, 157, 224	NG	0
	Sochorova	2016	Germany	CAN	0, 100	2	0
	Spohn	2016	Austria	NPK	0, 120	3	0
	van Eekeren	2009	The Netherlands	CAN, CMS	0, 150, 162.8	4	0
Soil mineral N content	Eriksen-Hamel	2006	Canada	NPK, CMS	0, 75	1	0
	Fornara	2016	Ireland	NPK, CMS, PMS	0, 78, 96, 162, 192, 200, 324, 384	3	0

	Gudmundsson	2004	Iceland	NPK	0, 75, 120	NG	0
	Hassink	1994	The Netherlands	Mineral N	0, 200, 400	NG	0
	Kidd	2017	UK	NPK, FYM	0, 35, 100, 120	NG	0
	Lkhagvasuren	2011	Canada	NPK	0, 56, 112, 224	NG	0
	Malhi	1997	Canada	AN	0, 56, 112, 168, 224, 336	2	0
	Malhi	2002	Canada	AN	0, 112	1	0
	Nevens	2003	Belgium	NPK	0, 100, 200, 400	3, 4, 5	0
	Nyborg	1999	Canada	Mineral N	0, 112	NG	0
	Reeder	1998	Wyoming	AN	0, 34	NG	0
	Riggs	2015	US Central Great Plains	NPK	0, 100	NG	0
	Schroder	2007	The Netherlands	NPK	0, 300	4, 5	0
	Schroder	2010	The Netherlands	CAN	0, 170, 340, 510	5	0
	Sochorova	2016	Germany	CAN	0, 100	2	0
	Spohn	2016	Austria	NPK	0, 120	3	0
	van Eekeren	2009	The Netherlands	CAN, CMS	0, 150, 162.8	4	0
Number of arthropod species	Allan	2014	Germany	The study links LUI values to number of Coleoptera species			
	Siemann	1998	Minnesota	AN	0, 54, 170	2	0
Number of plant species	Bobbink	1991	The Netherlands	NPK	0, 100	NG	0
	Cole	2008	UK	AN	0, 120, 240	NG	0
	Cop	2009	Slovenia	NPK	0, 50, 100, 150, 200	2, 3, 4	0
	Foster	1998	Michigan	AN	0, 480	3	0
	Hejcman	2012	Czech Repulic	NPK	0, 300	2	0
	Hejcman	2014	Germany	AN	0, 60, 120, 160	3	0
	Hejcman	2007	Germany	NPK	0, 100	2	0
	Honsova	2007	Czech Republic	NPK	0, 50, 100, 150, 200	2	0
	Kidd	2017	UK	NPK, FYM	0, 35, 100, 120	NG	0
	Muller	2016	Germany	NPK, CAN	0, 6, 15, 28, 31, 40, 50, 52, 60, 70	2	0
	Sochorova	2016	Germany	CAN	0, 100	2	0

4.3.5 Statistics

Statistical analyses were performed in R, version 3.3.1 (R Development Core Team, 2016). All applied models were mixed models. To account for the multilevel structure in the data, parcel (in the analysis of our own data) or study and region (in the analysis of the literature data) were included as a random factor. First, we tested whether there was a difference in ES indicators between the different management types (CON, FYM and ZER grasslands in TVG and CON and ZER grasslands in BVA). To do this, we fitted a mixed effects model to each of the indicators that were measured in TVG and BVA separately and management type was included as a categorical fixed effect. Post-hoc Tukey tests ($p<0.05$) were used to determine the statistical differences among the different management types. A second set of mixed models was fitted to the combined data from TVG and BVA. With these models we tested whether LUI was a significant predictor for the standardized indicators. In these models, study region (TVG or BVA) was included as random effect and LUI was the fixed effect. Correlations between the various ES indicators were tested using a Principal Components Analysis (PCA). Additionally, LUI was included as an additional variable in the PCA analysis.

Finally, an additional analysis was performed on the carabid and plant species data from TVG and BVA. We assessed the effect of management type on community composition of the carabids and plants. Therefore, we calculated a dissimilarity matrix using the Bray-Curtis dissimilarity measure based on activity-density data of the carabids and cover data of the plants. The significance of the compositional differences was tested with a permutational multivariate analysis of variance (PERMANOVA) with 999 permutations using the adonis function in the vegan package (Anderson, 2001). To account for the multilevel structure of the data, permutations were constrained within grasslands. Pairwise differences were tested using the multiple comparison tests with the Bonferroni correction. Because PERMANOVA cannot distinguish among location (ordination in the 2-dimensional non-metric space determined by species composition) and dispersion effects (around the centroid in the 2-dimensional non-metric space) (Anderson, 2001), we tested separately for multivariate heterogeneity of dispersion among the tested management types by means of a multivariate analogue of Levene's test for homogeneity of variances (Anderson, 2006) using the function betadisper in the vegan package. A significantly heterogeneous multivariate dispersion means that the PERMANOVA result cannot be solely attributed to a difference in the species composition among management types but also to the variability within the management types.
Similar to the analysis on the experimental data, the link between LUI and standardized indicators was tested on the data from the literature review. Because both mineral and animal fertilizers were included, fertilizer type was included as an additional categorical fixed effect.

4.4 Results

4.4.1 Field monitoring

Both in TVG and in BVA, biomass yield differed significantly among the different management types (Table 4.3) and yield was reduced on the FYM and ZER grasslands. However, also within the management types, total yields varied considerable (Figure 4.1, Figure 4.2, section 7.12). In TVG, dry biomass yield varied between 11.99 and 15.11 ton ha^{-1} on the CON grasslands, between 7.65 and 12.62 ton ha^{-1} on the FYM grasslands and between 4.82 and 7.33 ton ha^{-1} on the ZER grasslands. Yield on ZER3 and ZER4 in TVG was considerably lower compared to ZER1 and ZER2 in TVG, potentially as a result of a longer time span since last fertilization. In BVA, yields were between 9.94 and 11.18 ton ha^{-1} on the CON grasslands and between 5.37 and 6.11 ton ha^{-1} on the ZER grasslands.

Both in the first cut as in the regrowth, crude protein concentration was significantly higher on the CON grasslands than on the FYM and ZER grasslands (Table 4.3). In TVG, crude protein yield varied between 0.61 and 2.34 ton ha^{-1} and in BVA between 0.50 and 1.49 ton ha^{-1}. Both in TVG and in BVA, crude protein yield was significantly affected by management type (Table 4.3). In TVG, also grasslands with the same management type showed considerable differences: crude protein yield was markedly higher on CON1 and CON2 compared to CON3 and CON4 and also within the FYM grasslands, protein yield varied.

In general, OMD (%) was significantly lower on the ZER grasslands. Only the regrowth in BVA had the same digestibility on the CON grasslands as the ZER grasslands. Whereas digestibility of the first cut and the regrowth was comparable on the CON grasslands, on the ZER grasslands, the regrowth was more easily digestible compared to the first cut and on the FYM grasslands in TVG, the reverse trend was detected (Table 4.3, section 7.12).

In the 0-20 cm soil layer, lowest SOC stock values were 35.89 ton ha^{-1} and 70.69 ton ha^{-1} and highest SOC stock values were 72.53 ton ha^{-1} and 89.18 ton ha^{-1} in TVG and BVA, respectively. SOC stock was unaffected by management type (Table 4.3), but considerable differences were noted between the parcels with the same management type. For example, the lowest SOC stock was measured on CON1 in TVG and the highest stock on CON3 in TVG.

In TVG, mineral N content of the 0-90 cm soil layer was between 61 and 184 kg ha^{-1} and in BVA between 43 and 160 kg ha^{-1}. Soil mineral N and NO_3^--N were significantly lower on the FYM and ZER grasslands, while NH_4^+-N was unaffected by management type (section 7.13). Despite of the difference in fertilization, none of the N forms differed significantly between FYM and ZER grassland (Table 4.3). In TVG, NO_3^--N was considerably higher on CON1 and CON2 compared to CON3 and CON4 (section 7.13).

Table 4.3: Average ecosystem service and biodiversity indicator values measured in both study regions (TVG and BVA) on grasslands under regular management (C), farmyard manure application (M) and zero fertilization (Z). The survey quadrat for plant species was 2 m x 2 m. Different letters within a row and within every study region show significant differences among management types (p<0.05, Tukey-test)

	TVG C	TVG M	TVG Z	F-value	n.d.f., d.d.f.	BVA C	BVA Z	F-value	n.d.f., d.d.f.
Biomass yield (ton ha^{-1})	13.55±1.38 a	10.32±2.32 b	6.23±1.20 c	19.8693	2, 9	9.23±1.07 a	6.33±0.95 b	32.3843	1, 4
Crude protein concentration (first cut) (%)	18.94±2.34 a	12.93±2.80 b	9.72±1.74 c	25.877	2, 9	16.13±0.96 a	7.90±1.72 b	102.124	1, 4
Crude protein concentration (regrowth) (%)	21.22±4.02 a	14.75±4.09 b	13.34±1.81 b	16.5167	2, 9	15.17±2.80 a	11.47±0.71 b	8.7994	1, 4
Crude protein yield (ton ha^{-1})	2.55±0.74 a	1.48± 0.35 b	0.68±0.09 b	19.7938	2, 9	1.41±0.07 a	0.57±0.06 b	138.8645	1, 4
Organic matter digestibility (first cut) (%)	81.46±1.93 a	77.53±3.19 a	57.87±8.76 b	19.759	2, 9	75.75±4.28 a	60.02±5.68 b	36.802	1, 4
Organic matter digestibility (regrowth) (%)	82.54±5.75 a	63.12±8.02 b	66.81±3.67 b	20.9469	2, 9	71.86±6.04 a	70.55±3.27 a	0.378	1, 4
Soil carbon stock (ton/ha)	52.75±18.36a	52.96±9.06 a	52.01±7.78 a	0.00467	2, 8	83.11±12.64 a	75.06±9.92 a	2.2566	1, 4
Soil mineral N content (kg/ha)	141.45±26.34 a	74.44±9.43 b	59.73±8.27 b	22.62	2, 8	95.34±50.63 a	43.53±4.55 b	3.122	1, 4
Soil NH_4^+ - N content (kg/ha)	90.04±20.56 a	71.08±11.27 a	58.71±8.42 a	3.932	2, 8	73.66±51.86 a	40.60±4.81 a	1.209	1, 4
Soil NO_3^- - N content (kg/ha)	51.41±30.38 a	3.37±3.55 b	1.02±0.18 b	8.742	2, 8	21.73±12.85 a	2.93±0.42 b	6.42	1, 4
Number of carabid species	2.50±1.63 a	2.68±1.45 a	3.51±1.71 a	1.184	2, 9	1.81±1.10 a	1.36±1.18 a	1.000	1, 4
Number of plant species per survey quadrat	2.08±0.90 a	5.50±2.11 b	9.50±2.24 c	24.1326	2, 9	4.11±0.93 a	9.67±2.74 b	11, 688	1, 4

Table 4.4: Mixed model results based on the standardized indicator values of both study regions combined.

	Mixed model	Fixed effect p-value
Biomass yield	0.54+0.15*LUI	***
Crude protein yield	0.46+0.21*LUI	***
Soil carbon stock	1.01-0.00*LUI	NS
Soil mineral N content	0.47+0.17*LUI	**
Number of carabid species	0.91+0.02*LUI	NS
Number of plant species	1.59-0.19*LUI	***

Abbreviations: NS, not significant at p>0.05; * 0.01<p≤0.05; ** 0.001<p≤0.01; ***p≤0.001

Table 4.5: Mixed model results based on the literature review data. LUI and fertilizer type were included as fixed effects in the model. When the interaction with fertilizer type was significant, a distinction was made between mineral and animal fertilizer application.

	LUI	Fertilizer type	interaction	Fertilizer type	Mixed model	Fixed effect p-value
Biomass yield	***	NS	**	mineral	0.52+0.24*LUI	***
				animal	0.38+0.32*LUI	**
Crude protein yield	***	***	NS	mineral	0.48+0.26*LUI	***
				animal	0.46+0.29*LUI	***
Soil carbon stock	***	***	***	mineral	0.98-0.01*LUI	NS
				animal	0.86+0.11*LUI	***
Soil mineral N content	***	NS	**	mineral	0.61+0.19*LUI	***
				animal	0.71+0.15*LUI	***
Number of arthropod species	NS	Not tested				
Number of plant species	***	Not tested			1.22-0.11*LUI	***

Abbreviations: NS, not significant at p>0.05; * 0.01<p≤0.05; ** 0.001<p≤0.01; ***p≤0.001

In TVG, 772 carabids and 33 different carabid species were collected (section 7.14). *Poecilus versicolor* (179 individuals) and *Pterostichus melanarius* (330 individuals) comprised 66% of all individuals in the sampled population. One individual of *Acupalpus brunnipes*, one of *Amara tricuspidata* and one of *Amara kulti* were collected, all on the ZER grasslands. These species are present on the Belgian Red List of Endangered Species as vulnerable, critically endangered and vulnerable, respectively (INBO, 2017). *Aculpalpus brunnipes* and *Amara tricuspidata* have a habitat preference for wet heathlands and *Amara kulti* for dry, nutrient-poor grasslands (Turin, 2000). The number of carabid species was not affected by the management type (). In the PERMANOVA, the effect of management type on carabid species composition was significant (p=0.001). Dispersion among management types was homogenous (p=0.09). The significant p-value from PERMANOVA therefore indicated a location effect, hence a difference in carabid species composition among different management types. In the NMDS plot (section 7.14), we see that this difference mostly plays in the species composition on CON grasslands versus FYM and ZER grasslands. In BVA, 161 carabids and 17 different carabid species were collected (section 7.14). *Poecilus versicolor* (98 individuals) comprised 61% of all individuals in the sampled population. The number of carabid species was not affected by the management type (Table 4.3). The effect of management type on carabid species composition was significant (p=0.005) and dispersion among management types was homogenous (p=0.134), indicating a significant difference in carabid species composition among management types. All pairwise comparisons indicated significant differences in carabid species composition between management types. In the NMDS plot (section 7.14), the difference in species composition between CON grasslands and ZER grasslands is confirmed.

Both in TVG and in BVA, *Lolium perenne* was the dominating grass species on the CON grasslands. On the FYM and ZER grasslands, the species composition shifted towards a *Holcus lanatus* dominated sward and the number of flowering herbaceuous species considerably increased (section 7.14). The average number of plants per plot was lowest in the CON grasslands, increased in the FYM grasslands and was the highest in the ZER grasslands and differences were significant (Table 4.3). Despite of the plant species increases on the FYM and ZER grasslands, none of the species are present on the Belgian Red List of Endangered Species (INBO, 2017). Both in TVG and in BVA, the significant result of PERMANOVA (p_{perm}=0.001 in both regions) indicated both a location and a disperion effect (p_{disp}≤0.001 and p_{disp}=0.002, respectively) on plant species composition and the pairwise comparisons showed significant differences in plant species compositions for the comparison of all mangement types. Also, the NMDS plots of both regions (section 7.15) show a clear distinction between plant species compositions among the different management types.

LUI of the monitored grasslands varied between 0.86 and 5.36 (Figure 4.3). LUI was a significant predictor for biomass yield, crude protein yield, soil N and number of plant species (Table 4.4). This means that increasing grassland management intensity (by increasing N fertilization, the number of cuts and/or mowing intensity) will result in an increase of biomass yield, an increase of crude protein yield, an increase of soil N and a decrease of the number of plant species in the grassland in a predictable way. Note that these relationships were developed in an agricultural context (with fertilization), and hence do not apply on grassland management with nutrient depletion (only mowing). Additionally, because the effect relationships were based on black-box models and thus do not take into account underlying processes, it is not advisable to extrapolate beyond the ranges that were considered. To give a numerical example: the effect-relationships developed indicate that if the fertilization dose of a grassland with yield, soil and biodiversity characteristics and an LUI similar to FYM1 would be increased with 50 kg N ha^{-1} and the grazing period was extended with 15 days, biomass yield is supposed to increase with about 2.6 ton ha^{-1}, crude protein yield with about 0.6 ton ha^{-1}, soil mineral N content with about 27.8 kg ha^{-1} and the grassland would lose approximately two plant species. Similarly, if the fertilization dose of a grassland with yield, soil and biodiversity characteristics an LUI similar to ZER1 would be increased with 50 kg N ha^{-1} and would be mowed one additional time, biomass yield is supposed to increase with about 1.4 ton ha^{-1}, crude protein yield with about 0.3 ton ha^{-1}, soil mineral N content with about 14.7 kg ha^{-1} and the grassland would lose approximately one plant species.

The trade-offs between various ES indicators and the link with LUI is represented in Figure 4.4. In TVG, there is a strong correlation between yield, crude protein yield, soil N content and floral diversity and in BVA, SOC stock is added to the list of correlated ES indicators. In both study regions, LUI varies along the first axis of explained variance. It seems that increasing management intensity will enhance the provisioning ES, but this is at the expense of the regulation of freshwater quality and floral diversity. Carabid diversity is unexplained by LUI, and so is SOC in TVG. Generally, the management types 'CON' and 'ZER' differ in the contribution to delivery of provisioning ES and the regulation of freshwater quality and floral diversity. The management type 'FYM' is not correlated with any of the selected ES indicators.

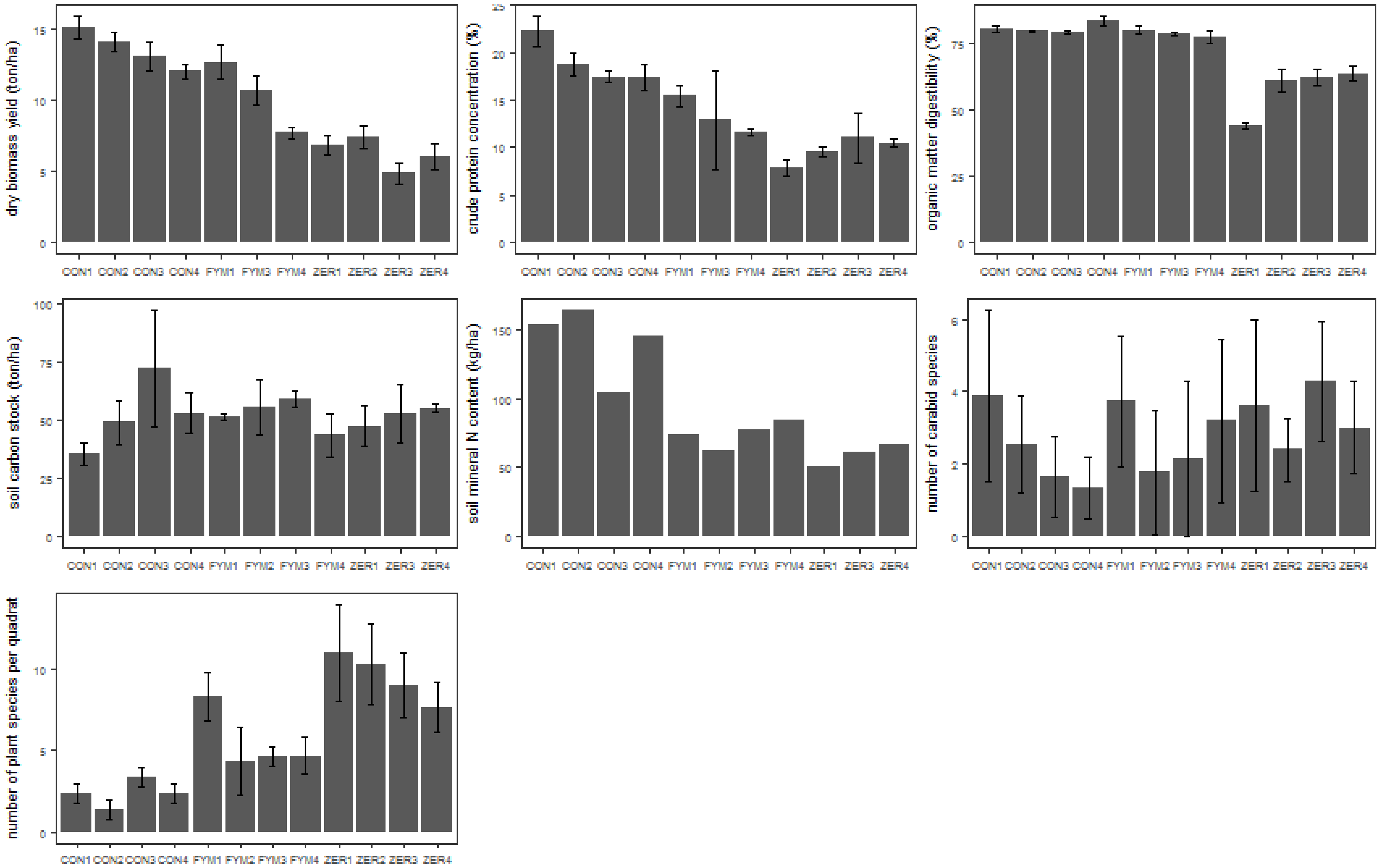


Figure 4.1: Results from the grassland monitoring in TVG. CON grasslands were under a regular management. FYM grasslands received farmyard manure and had a delayed first cut and grazing. ZER grasslands received no fertilizers and had a delayed first cut and grazing. Soil carbon stocks were measured in the 0-20 cm soil layer and soil mineral nitrogen content was measured in the 0-90 cm soil layer. Carabid and plant diversity stand for the average number of species that was found. The survey quadrat was 2 m x 2 m. Error bars indicate standard deviations

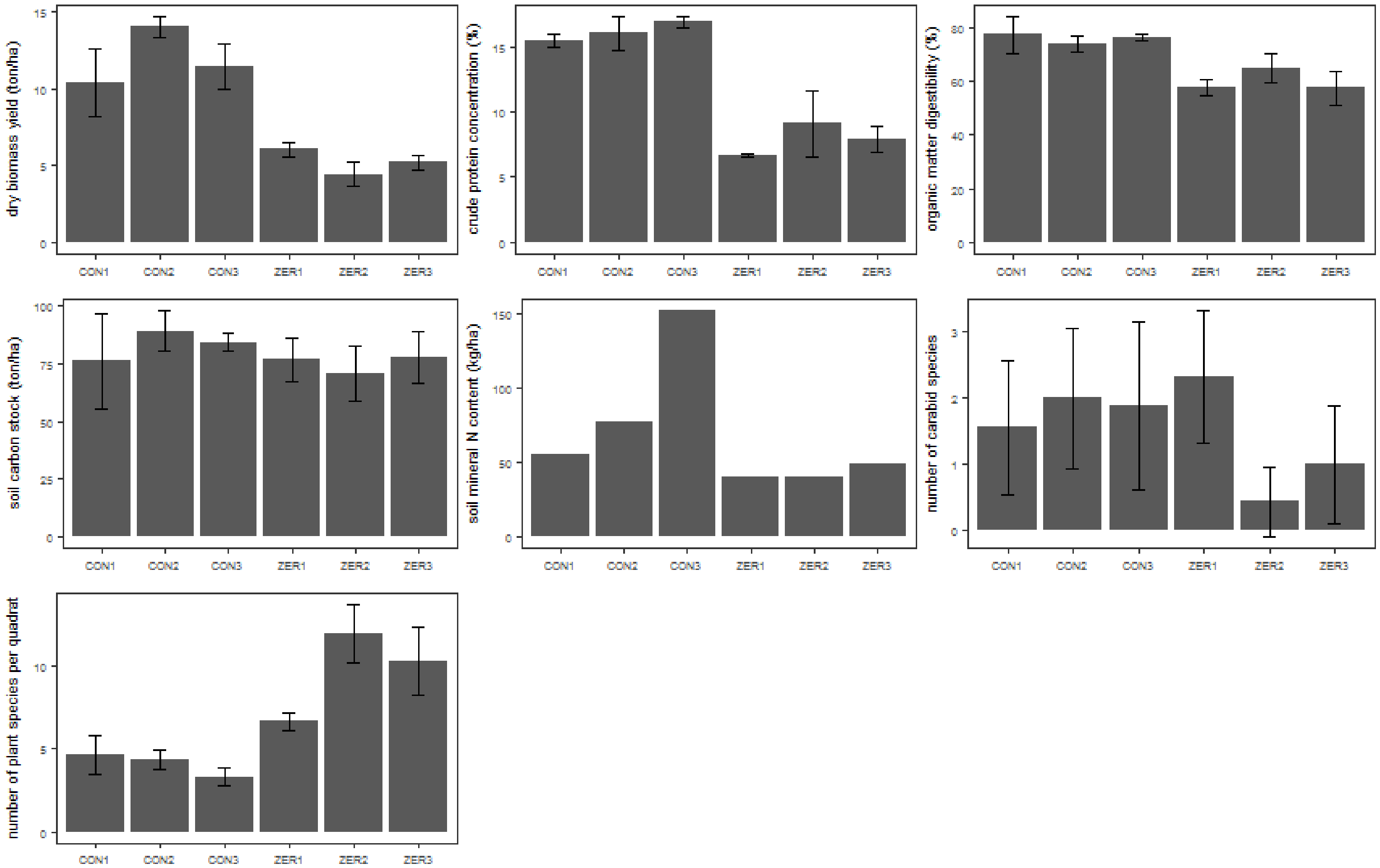


Figure 4.2: Results from the grassland monitoring in BVA. CON grasslands were under a regular management. FYM grasslands received farmyard manure and had a delayed first cut and grazing. ZER grasslands received no fertilizers and had a delayed first cut and grazing. Soil carbon stocks were measured in the 0-20 cm soil layer and soil mineral nitrogen content was measured in the 0-90 cm soil layer. Carabid and plant diversity stand for the average number of species that was found. The survey quadrat was 2 m x 2 m. Error bars indicate standard deviations.

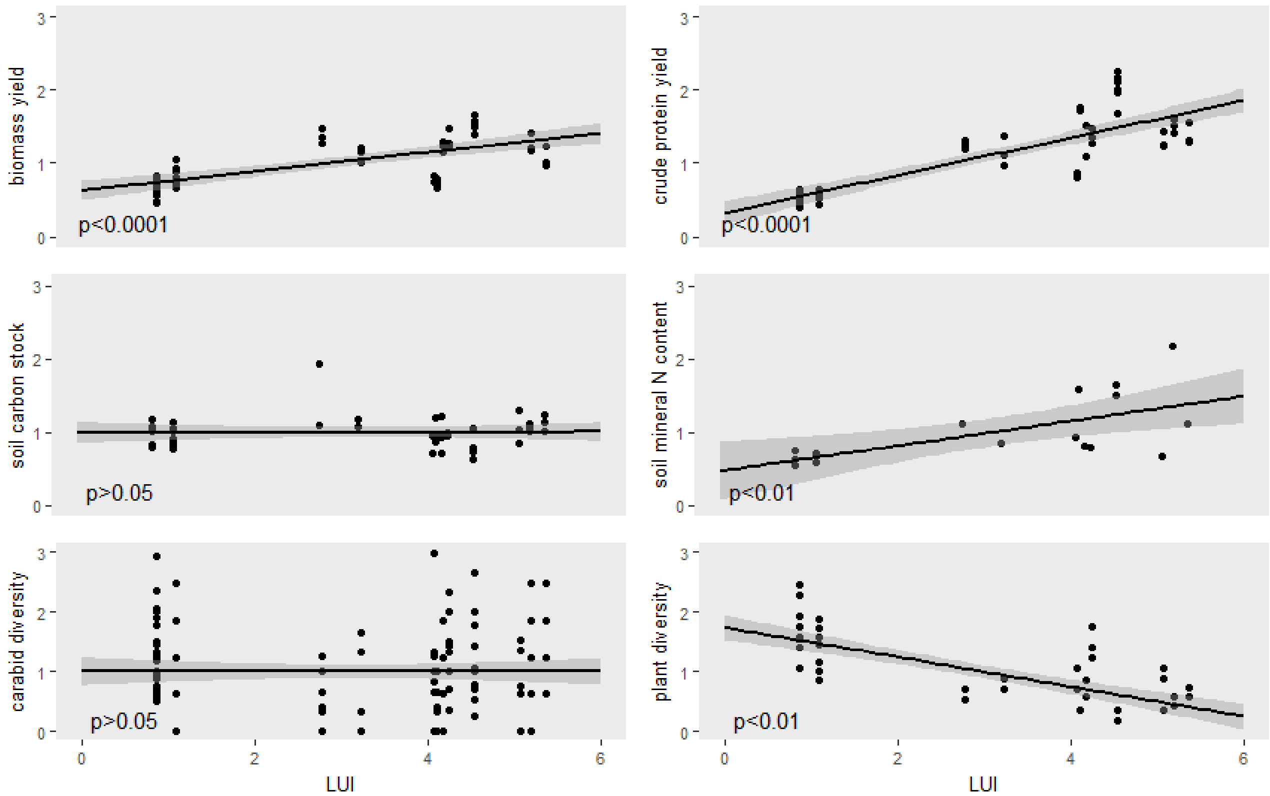


Figure 4.3: Standardized ecosystem service and biodiversity indicators as a function of grassland management intensity (LUI) based on data from both study regions. Lines represent fitted regression lines. Shadings show 95% confidence intervals. P-values indicate significance of the relationship between LUI and indicators

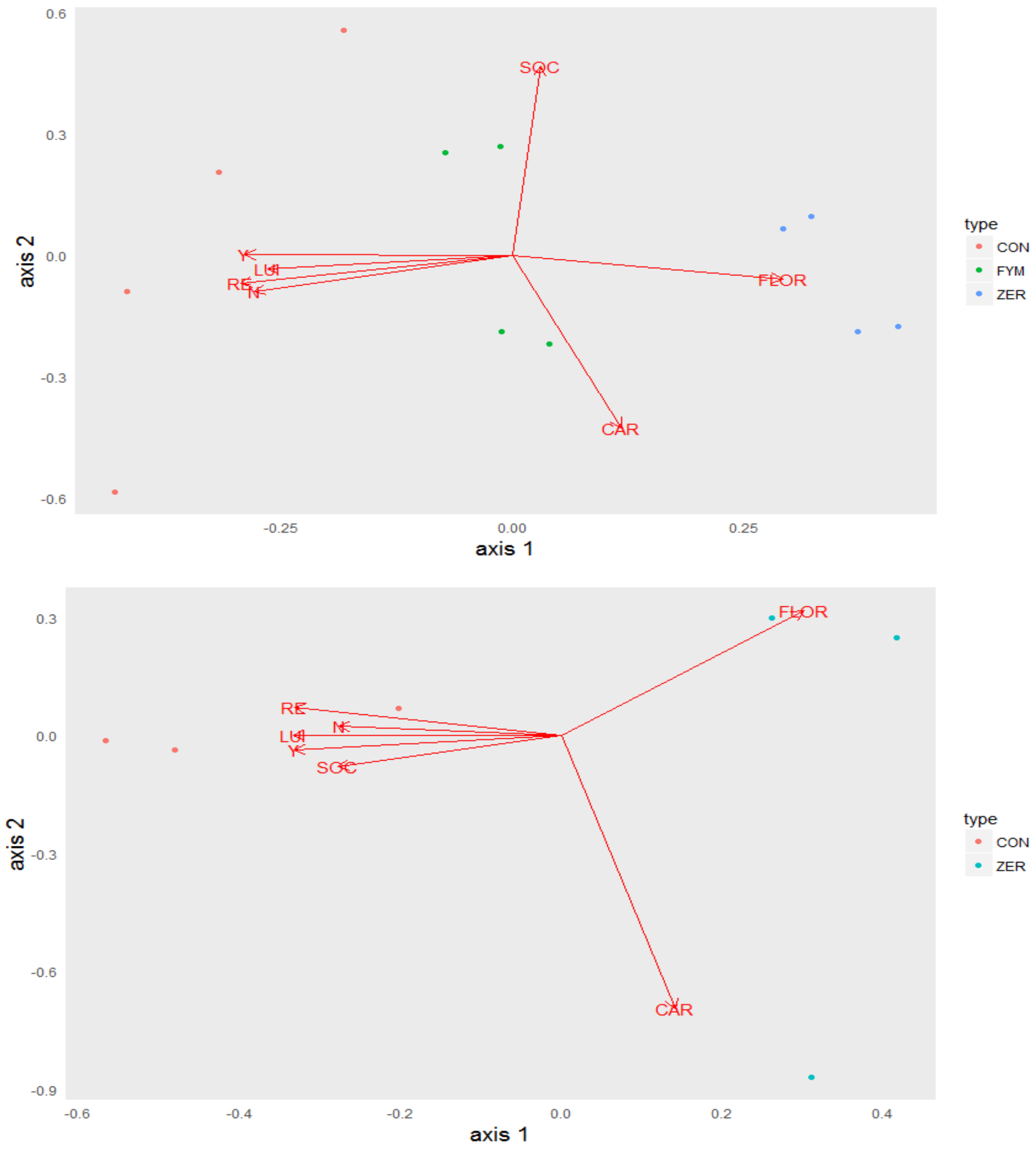


Figure 4.43: Graphical representation of the first two PCA axes computed on the ES indicators measured in the grassland parcels in TVG (top) and in BVA (down). The first two PCA axes for the TVG PCA explain 61% and 23% of total variance. The first two axes for the BVA PCA explain 74% and 14% of total variance. CON grasslands were under a regular management. FYM grasslands received farmyard manure and had a delayed first cut and grazing. ZER grasslands received no fertilizers and had a delayed first cut and grazing. Y is total yield. RE is total protein yield, SOC is soil carbon stock (0-20 cm) (kg ha^{-1}), N is soil mineral N content (0-90 cm) (kg ha^{-1}), FLOR is number of plant species and CAR is number of carabid species.

4.4.2 Literature review

In the studies retained from the literature review, maximum fertilization dose was 510 kg N ha^{-1} for the biomass yield, protein yield and soil mineral N content dataset, 400 kg N ha^{-1} for the SOC stock dataset and 480 kg N ha^{-1} for the plant species dataset. Compared to the grasslands monitored in TVG and BVA, maximum fertilization intensity in the literature review datasets was twice as high. Animal fertilizer included both the application of slurry and of manure. The number of cuts varied between 1 and 5 (Table 4.2). No studies on the impact of grazing were retained because none of the studies reported enough data (number and age of the grazing animals and number of grazing days) to calculate LUI_G. LUI of the grasslands in the retained studies varied between 0.48 and 4.5.
LUI affected biomass yield, protein yield, SOC stock, soil mineral N content and number of plant species (Table 4.5). Only the number of arthropod species was not affected by LUI (Figure 4.5).

When grassland management increased, biomass yield, protein yield, SOC stock and soil mineral N content increased and the number of plant species decreased. Similar to the effect relationships based on the monitoring data, it is advisable not to extrapolate beyond the considered LUI range. When tested, fertilizer type (mineral or animal) influenced the relationship between LUI and the indicators (Figure 4.5). Because only two studies were included in the arthropod analysis and only one study reported the effect of animal fertilizer on the number of plant species, the effect of fertilizer type on number of arthropod and plant species was not tested. Biomass yield, protein yield and soil mineral N content were increased both by mineral and animal fertilizer application but SOC stock was only affected by animal fertilizer application and increased when more fertilizer was applied.

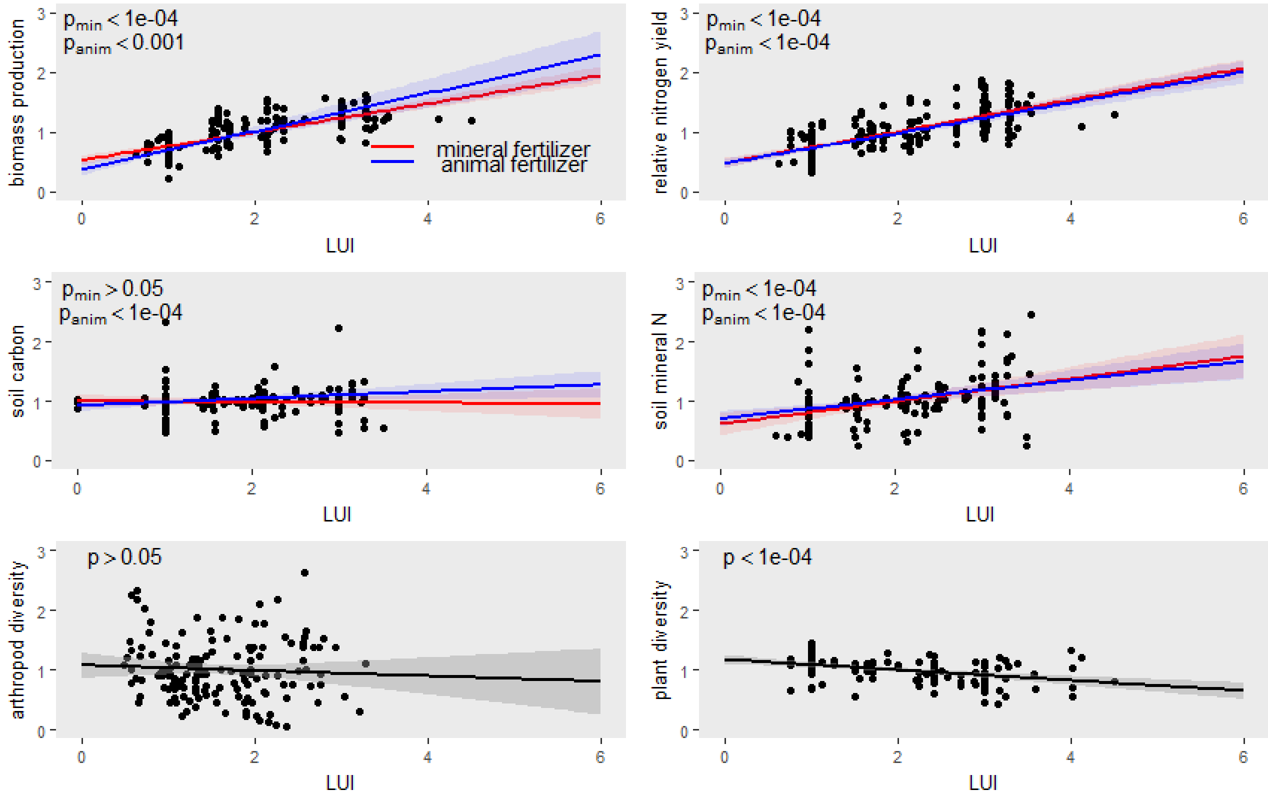


Figure 4.54: Standardized ecosystem service and biodiversity indicators as a function of grassland management intensity (LUI) based on data from the literature review. Lines represent fitted regression lines. When significant, a distinction was made between the application of mineral fertilizers (red) and animal fertilizers (blue). Shadings show 95% confidence intervals. p-values indicate significance of the relationship between LUI and indicators. p_{min} applies for mineral fertilizer and p_{anim} for animal fertilizer

4.5 Discussion

On the monitored grasslands, average LUI was highest on the FYM grasslands in TVG, despite the fertilization restrictions on this type of grassland. It seems thus that in the LUI as defined by Blüthgen et al. (2012), the grazing component was assigned a relatively high weight. The same is suggested by comparison of the fertilization doses and LUI ranges of the monitoring data and the literature review dataset: despite a maximal fertilization dose that was twice as high, maximal LUI of the data from the literature review was lower compared to the monitoring data.

Compared to the CON grasslands, dry biomass yield was reduced by 24% on the FYM grasslands (in TVG) and between 31% and 55% on the ZER grasslands (in BVA and TVG, respectively). Similar reductions were found by Bommelé et al. (2003) on Flemish grasslands with the same management (19% reduction on FYM grasslands and 46% reduction on ZER grasslands). LUI was a significant predictor for yield both in the own monitoring data and in the literature review data, the yield reduction on FYM grasslands (compared to CON grasslands) was not reflected in the corresponding LUI values and thus for this type of grassland management, LUI was not an appropriate predictor for biomass yield. These results further support the idea that the grazing component in the LUI index has been assigned too great a weight.

Crude protein concentration reduction on the FYM and ZER grasslands was strongest in the first cut, but in TVG also the next cuts showed a decline in crude protein concentration on the FYM and ZER grasslands. Generally, crude protein concentration was higher in the regrowth cuts, which was similar to the results of Pontes et al. (2007). Total crude protein yield was reduced by 40% on the FYM grasslands (in TVG) and between 60% and 71% on the ZER grasslands (in BVA and TVG, respectively). Similar reductions were found by Bommelé et al. (2003) and by Fiems et al. (2004). This cannot be (solely) attributed to the varying sward compositions, as the crude protein concentration in *Lolium perenne*, the dominant species on the CON grasslands, is similar to or less than most other grassland species such as *Holcus lanatus* and *Dactylis glomerata*, which are two common species on the FYM and ZER grasslands (Pontes et al., 2007). On the effect of N fertilization on crude protein concentration, literature is ambiguous. Aavola and Kärner (2008) found increasing N (and thus protein) concentrations in grasses with increasing N fertilization, while other studies found no effect (Pontes et al., 2007; Turner et al., 2006). Finally, as cutting affects the morphological development of grasses, mowing and grazing increase the crude protein concentration and a late first cut or grazing results in a lower protein yield (Čop et al., 2009; Pontes et al., 2007). Our data suggest both an effect of N fertilization and of a delayed first cut. Also, it should be noted that CON1 in TVG consisted of a mixture of *Lolium perenne* and *Trifolium repens,* the latter increasing crude protein concentration of the sward (Enriquez-Hidalgo et al., 2016) . This most probably explains the high concentrations that were measured on CON1. Finally, it should be noted that the presence of herbaceous species other than grasses on the ZER grasslands, and to a lesser extent on the FYM grasslands, might have affected crude protein concentration of the sward, positively or negatively.

Compared to the CON grasslands, digestibility reduction on the ZER grasslands was between 19% and 29%. Digestibility of grasses is highest during the vegetative stage and a decrease of digestibility is mainly caused by a decrease in leaf:stem ratio (Pontes et al., 2007). Therefore, it is very likely that digestibility decreases with a delayed first cut and that it increases with mowing intensity. Additionally, *Lolium perenne* typically is better digestible than other grass species such as *Holcus Lanatus* or *Dactylis glomerata* (Pontes et al., 2007), potentially contributing to the high digestibility of the CON grasslands in TVG.

The trends in crude protein concentration and digestibility both indicate a degradation in terms of forage quality on the FYM and ZER grasslands, most probably as a result of a delayed first cut, N fertilization and different grass species compositions.

Surprisingly, no differences were found in soil C stocks between CON, FYM and ZER grasslands. Cattle slurry and to a higher extent farmyard manure contain a considerable amount of carbon (Eekeren et al., 2009), and it was expected that the parcels with higher fertilization rates and especially those with application of farmyard manure would have higher SOC stocks. This was the case in the studies evaluated in the literature review. However, animal fertilizer application was markedly higher in the literature review studies: average animal fertilizer application in the literature review corresponded to 204 kg N ha^{-1} on average, while in TVG and BVA, fertilizer application corresponded with 138 kg N ha^{-1} on average. Note here that the two monitored grasslands with the highest fertilizer application (C1 and C2 in TVG, 203 kg N ha^{-1}) were tilled respectively 3 and 4 years before the monitoring, which most likely causes the soil C to decrease (Guzman and Al-Kaisi, 2007). It is possible, therefore, that increasing the animal fertilizer application would raise the SOC stock.

Both in TVG and in BVA, NH_4^+-N did not vary among management types and thus we assume that it is not affected by fertilization or any other human intervention. For this reason, discussion of the results will focus on NO_3^- - N. In order to reduce NO_3^--N leaching and consequent eutrophication of surface water, Flemish legislation imposes a maximum NO_3^- -N content of 90 kg ha^{-1} in the 0-90 cm soil layer between October 1 and November 15 (Vlaamse Gemeenschap, 2006). None of the monitored grasslands exceeded the legal maximum. In TVG, the differences in NO_3^- - N on the grasslands were very much in line with the N fertilization that was applied. For example, N application on CON1 and CON2 was 50 kg ha^{-1} higher than on CON3 and CON4, and so was the soil NO_3^- -N content. Similar trends can be found for the FYM grasslands. However, NO_3^- - N on the ZER grasslands was higher than on two FYM grasslands (FYM3 and FYM4), despite of fertilization. A similar result was found by Bommelé et al. (2003) and this was linked to the late first cut and limited biomass yield on grasslands without fertilization, resulting in lower N uptake and thus less N removal. In BVA, the low NO_3^- -N content on CON1 can not be explained by N fertilization or N uptake. The effect of N fertilization and/or N uptake on N leaching risk in grasslands is well known (Nevens and Reheul, 2003; Schröder et al., 2010; Tampere et al., 2014), but several studies point out other aspects as well, such as grassland age, mineralization of organic N after ploughing and grazing, all increasing the N leaching risk (Aavola and Kärner, 2008; Basso and Ritchie, 2005; Collins and Allinson, 2004; Eriksen et al., 2004; Sonneveld and Bouma, 2003). However, none of these factors could explain the lower NO_3^- -N content in CON1 in BVA.

Both in TVG and in BVA, the number of carabid species did not differ significantly between different management types, indicating that the application of one of the selected management types does not increase or decrease species number within the parcel (alpha diversity). However, in both monitored areas, Bray-Curtis indices differed significantly between grassland management types, indicating that different species compositions (beta diversity) are related to the different management types. It seems thus that the benefit of applying different management regimes lies in the range of habitats that are created (favouring beta diversity) rather than in the presence of one specific management type creating one habitat type (favouring alpha diversity). Grandchamp et al. (2005) came to a similar conclusion by investigating the effect of grassland management on carabid species. They demonstrated that both intensively and extensively managed grasslands sustain a different and diverse carabid species composition. Also, they found different species compositions on grasslands that were mown and on grasslands that were grazed. Finally, the presence of 3 carabid species that are on the Belgian Red List of Endangered Species on the ZER grasslands in TVG, indicates the importance of very extensively managed grasslands for nature and biodiversity conservation.

Nitrogen fertilization typically increases living plant biomass and reduces plant biodiversity (Socher et al., 2012), and this is reflected in the number of plant species found on the monitored grassland in TVG and BVA. Intensive mowing can be another cause of biodiversity reduction and except for very high stocking densities, grazing generally increases plant biodiversity (Plantureux et al., 2005). However, nor mowing nor grazing nor fertilization seemed to explain the difference between the number of plant species on CON3 and CON4 on the one hand and the FYM grasslands on the other hand. It has been proven that delaying the first cut or grazing increases plant diversity (Humbert et al., 2012; Smith et al., 1996), and we assume that together with fertilization, this is the main driver for the increased number of species on the monitored FYM and ZER grasslands. Despite of the increase of plant biodiversity on these grasslands, its true botanical conservation value may be questioned, as we found no vulnerable or endangered species in the monitored grasslands. It has been shown that grassland restoration may easily take a long time and that duration and success depend on many factors such as soil seed bank, landscape characteristics, soil characteristics etc. (Plantureux et al., 2005) suggesting the botanical conservation value of the ZER grasslands will increase over time. However, for example soil phosphorus after historical fertilization or nitrogen deposition might induce that restoration towards a grassland of high botanical value might not be realistic (Ceulemans et al., 2014; Isbell et al., 2013; Schelfhout et al., 2015).

In general, the effect relations that were derived from the literature review confirmed the trends that were monitored in TVG and in BVA. At the same time, we noticed that none of the retained studies investigated the effect of fertilization combined with grazing on ES and biodiversity. Also, because of the correlation between fertilizer intensity, mowing intensity and grazing intensity, we could not specify the effect of grazing on ES and biodiversity indicators. However, it is known that grazing will affect ecosystem services and biodiversity on grasslands. For example, SOC stock on grassland under grazing conditions is higher than under mowing conditions in Flanders (Mestdagh et al., 2006) and Nevens et al. (2003) found that grazing reduces the N leaching risk, even at high N application rates. Further investigations with more focus on the grazing component are therefore suggested.

4.6. Conclusion

The effect relations between LUI and ES and biodiversity indicators were remarkably similar for our own data and for the data from the literature review, indicating that in temperate grasslands, the impact of varying grassland management intensity on ES and biodiversity is relatively consistent. Also, trade-offs will generally be the same, and they are very clear for biomass yield and forage quality on the one hand and N leaching risk and number of plant species on the other hand. SOC stock was not or very little affected by LUI. Grasslands with a botanical management (ZER grasslands) had a higher number of plant species, but only very common species were found in the monitored grasslands and thus the botanical conservation value was low. This indicates that grassland management extensification does not guarantee botanical restoration and that other factors such as soil phosphorus conditions (Ceulemans et al., 2014; Schelfhout et al., 2015), nitrogen deposition (de Schrijver et al., 2011) or absence of a seed bank (Klaus et al., 2017) may hamper successful colonization of the grasslands.

Both biomass yield and forage quality were reduced by the extensification of grassland management, suggesting a negative impact on grassland performance, especially for dairy cows (Demeulemeester et al., 2012; Fiems et al., 2004). However, literature is not consistent and it has been shown that a diet with a limited share of biomass from extensive grasslands does not reduce milk production (Bruinenberg et al., 2006). For other production systems, such as sheep, suckler cows or beef cattle, biomass from extensive grasslands can be more widely applied (Demeulemeester et al., 2012; Hopkins, 2009). On the other hand, extensive grasslands may host poisonous plants, strongly reducing agronomic value of the silage (Wrage et al., 2011). Generally, results of this study show that extensification of grassland management has a considerable impact on the delivery of provisioning ES, but that the contribution to most of the other selected ES is rather limited, stressing the importance of a well-thought and effective management plan.

The relationship between LUI and ES and biodiversity indicators was often significant, but our own data showed that other management factors play a role as well. Timing of the first cut or grazing appeared to have an effect, especially on forage quality, soil mineral N content and number of plant species. In particular, delaying the first cut strongly reduced crude protein concentration, increased soil N content and increased number of plant species. Given the importance of delaying the first cut for meadow bird populations (Breeuwer et al., 2009), there seems to be a real trade-off, especially between provisioning ES and meadow bird management. Additionally, results from this study suggest that the definition of LUI can be improved and further research should explore whether assigning weights to the various components (fertilization, mowing and grazing) will improve the fit with the ES indicators. Also, it may be the case that the effect relationships between LUI and ES indicators are not linear. For example, increasing LUI may increase biomass yields, but rather asymptotically instead of linearly. In order to minimize trade-offs, a profound insight into this is needed.

5. Greening and producing: an economic assessment framework for integrating trees in cropping systems

After: Van Vooren, L., Reubens, B., Broekx, S., Pardon, P., Reheul, D., van Winsen, F., Verheyen, K., Wauters, E., Lauwers, L., 2016. Greening and producing: an economic assessment framework for integrating trees in cropping systems. Agric. Syst. 148, 44–57

5.1 Abstract

Nature-oriented measures in an agricultural context often lead to extra constraints for regular farm management. This suggests trade-offs between the ecological objectives and profitability. Whether trade-offs exist, or may be turned into win-wins, depends on creative farm options to comply with new constraints. This chapter concentrates on Ecological Focus Areas as a new EU Common Agricultural Policy greening requirement, and investigates profitability changes of two greening options with permanent woody elements, hedgerows and alley cropping. We predicted discounted gross margins for a hedgerow and alley cropping greening option and four market scenarios on a representative arable farm in Flanders (Belgium). Starting from the tree row, over a distance of 1.64 times the tree height, relative crop yield is 70% as compared to a treeless situation. Between 1.64 and 9.52 times the tree height, relative yield is 107%. Beyond that point, the effect is considered negligible. Discounted gross margins are calculated to account for the time horizon. Relative discounted gross margins at farm level, compared to the business as usual option, vary between 91% and 108%, depending on market conditions and policy support. The calculations show that fulfillment of the 5% ecological focus area greening requirement on arable farms with hedgerows and alley cropping only becomes economically competitive to the traditional cropping systems with extra financial stimuli (e.g. greening payments). We also show and discuss how the calculations can be fine-tuned and used in policy making, e.g. by i) getting better insights in the tree-crop interaction, ii) including the effect of e.g. crop type, tree species, tree line space and tree line orientation in the meta-analysis, iii) evaluating this conditional competitiveness and suggesting a better linking between subsidy level and ecological value and ecosystem services and iv) exploring novel valorization channels for wood products.

5.2 Introduction

A major part of the European countryside is shaped by agricultural land use. Farming creates habitats for wildlife and enjoyable landscapes and contributes to indirect benefits such as resilience to flooding. However, intensive agriculture has a negative impact on soil, water and air quality, as well as on biodiversity. Various measures to mitigate the negative impact of agriculture on the environment and to restore positive links between the environment and production are taken, including the recent greening measures in the Common Agricultural Policy (CAP). Since the 2013 reform, CAP direct payments consist of, among others, basic payments and greening payments. Thirty per cent of the direct payments to farmers is linked to greening requirements: the implementation of Ecological Focus Areas (EFA) on 5% of the arable land, crop diversification and the maintenance of permanent pasture at farm level (Matthews, 2013). Within the constraints of a member state's specific list of options, farmers are free to choose how they fill in the EFAs, e.g. with hedgerows, buffer strips, alley cropping agroforestry, fallow land, nitrogen fixing crops, catch and cover crops. According to the ecological value of the chosen option, a conversion and weighting factor is used to convert the lengths/areas of the elements into equivalent focus areas: elements with a lower ecological value, will receive a lower weighting factor compared to elements with a higher ecological value (e.g. hedgerows have a weighting factor of two) (European Commission, 2013b). In 2015, 85% of the EFAs in Flanders was composed of cover crops (data retrieved from the Flemish Department of Agriculture and Fisheries). However, it is known that this land use type contributes little to biodiversity (Pe'er et al., 2014). This suggests that the current EFA requirements and the weighting factors are not effective in reaching their primary target of biodiversity conservation. On the other hand, EFAs with permanent elements such as hedgerows and alley cropping, may have a positive impact on biodiversity (Westhoek et al., 2012).

Greening requirements will have an impact on farm economics: average decrease in overall farm income per worker is estimated between 1.4% and 3.2% (Matthews, 2013). Farmers that do not comply with the greening requirements may lose up to 125% of the greening payment (European Commision, 2013). Economic considerations play a role in farmers' decisions but these are hard to predict, in particular in the hedgerow and alley cropping case. Despite the crop yield loss due to cropland reduction and potential crop-tree competition for light, water and/or nutrients, alley cropping has the potential to deliver economic advantages such as wood production and diversification of farm income. However, profitability depends on many factors. A higher yield does not always result in more income and both are influenced by tree and crop type, tree density, orientation of the trees, interactions between crop and trees, and costs and prices of crops and wood (Dupraz and Liagre, 2008). Moreover, as alley cropping is a multiannual system, we face uncertainty in the changes in crop yields, costs and crop and wood prices. Besides data uncertainty, there is a considerable time lag between expected revenues and the decision to start-up alley cropping. Profit assessment then needs discounting the revenues and costs into a net present value.

In this paper, we design an assessment framework to combine crop yield information on tree-crop interactions with farm data in order to assess farm economic outcomes of greening measures. To do so, we i) quantify the effect of trees on crop yield in temperate regions and ii) assess the economic consequences of two farm level EFA choice options, a hedgerow and an alley cropping option, through comparison of discounted gross margins with the business-as-usual (BAU) option. A hedgerow can take many forms and dimensions; in this paper a hedgerow is defined as linear structure of unpruned trees and shrubs on the field boundary (Kuemmel, 2003). This option is seen as intermediate towards alley cropping because it is less far-reaching in terms of crop-tree mixing

(Borremans et al., 2016; Vandermeulen et al., 2012). In a tree row in the alley cropping system, we assume the trees to be pruned and the wood to be harvested.

5.3 Materials and methods

5.3.1 Effect of trees on crop yield

To investigate the effect of hedgerows and trees on crop yield, a double research question was defined: i) what is the spatial extent of the influence of the trees on crop yield and ii) what is the impact of tree-crop interaction on crop yield? Potentially relevant papers were searched on the ISI Web of Knowledge and Sciencedirect. Search terms were: trees, tree row, agroforestry, hedgerow, alley cropping, intercropping, woody edge, woody field margin, crop yield and productivity. Several combinations of these terms were searched. First, candidate papers were selected on title and abstract, meeting following conditions: i) data from areas with temperate climate, ii) actual field data are used (modelling studies are excluded), iii) true controls are present allowing yield comparison with and without tree-influence, iv) yield data are linked to the distance from the trees and v) interaction with arable crops, not with pasture. We focused on arable crops because we expect the effect of trees to be better measurable in crops compared to pasture. When necessary, the authors were contacted and asked to provide more information on the experimental setup or data statistics. The reference lists of the retained papers were used to search for additional papers. Ten studies (section 7.16) were retained. Own measurements from 2014 and 2015 on the effect of hedgerows and alley cropping on crops were added to this dataset. The experimental setup is described in section 7.22.

Considering measurements conducted in different years or on different locations as individual (but not independent) experiments, a set of 75 different experiments was used in the analysis. Relative yields (R) are used to express the effect of trees on crop yield and are calculated as the ratio of yield in the experiment group (plot with tree-influence) to the yield in the control group (plot without tree-influence). When R<1, yield is negatively influenced by the trees and when R>1, more is produced in the experimental plots than in control plots. R is related to the distance from the tree row. To allow comparison between different experiments, distance is related to height of the tree row. We therefore use H, which is the ratio of the distance from the tree row to the height of this tree row. This means that for a tree height of 20 m and experimental plots on a distance of 10 m from these trees, H = 0.5. The natural logarithm of R, ln(R), linearizes the response ratios and thus ln(R) will be affected equally by changing the numerator or denominator. Furthermore, ln(R) is more likely to be normally distributed, especially in small samples (Hedges et al., 1999).

A traditional meta-regression was performed with the metaphor-package (Viechtbauer, 2010), using the rma.mv-function. This was done in R, version 3.1.2 (R Development Core Team, 2016). Each ln(R) was weighted by the inverse of the corresponding standard deviation, giving a greater weight to studies with a lower standard deviation. However, standard deviations were only reported in 37 experiments. Only this subset was used in the meta-regression. A mixed-effects meta-regression model was applied, with H being the fixed effect. Due to the multi-level structure in the data, 'study' was included as a random variable, to account for non-independence between data from the same study. To include all experiments, a non-linear mixed model was applied on the dataset. Similarly, 'study' was included as a random variable. In the non-linear mixed model, data are not weighted and this could have a negative impact on the preciseness of the result (Koricheva and Gurevitch, 2014). Therefore, results of both models are compared in section 7.17.

5.3.2 Economic consequences of greening

The economic consequences of three choice options in the greening context are investigated. The first option is business-as-usual (BAU) without EFAs. The farmer does not benefit from greening payments and looses a part of the basic payments. This option is selected because it entails no additional costs or arable land loss. The second option is the hedgerow option: the EFA is entirely implemented with trees and shrubs on field boundaries. To meet the EFA requirements, the minimal hedgerow surface is 0.1 ha and maximum width is 10 m. In the EFA requirements, hedgerows are given a weighting factor of two. Therefore, only 2.5% of the arable surface (instead of 5%) should be filled in with hedgerows, because the hedgerow surface is doubled in the EFA calculations. The third option is the alley cropping option where trees are planted in lines on the field. The weighting factor of alley cropping is one. To meet the EFA requirements, the alley cropping parcel(s) should have an area of 5% of the arable farm land.

To assess the economic consequences of these EFA options, calculations are performed in Microsoft ® Excel. Besides an annual cash flow from the crops, trees provide an additional source of income from wood production after completion of the tree rotation cycle. To cope with a difference in time horizons, discounted cash flows are calculated. The total time-period covered is adjustable, depending on the rotation cycle of the tree. In our analysis we used a 20 year time-period. To compare the different choice options, similar starting conditions are assumed: all previous and extra costs and returns are supposed to be the same for each option. Differences in discounted cash flow then reflect only the economic consequences of farm's greening options. Information on tree-crop interactions (derived from the non-linear mixed model) is in a one-way flow transferred to the economic analysis. The parametrisation of our simulation is further discussed in section 7.18.

5.3.3 A case study in Flanders

Based on Belgian farm structure survey data (data from ADSEI 2010-2012) a farm representative for Flanders' arable cropping conditions is virtually constructed. The case farm is constructed based on the average of Flemish farms, selected as follows: i) two thirds of the farmland area is planted with the five most common arable crops (winter wheat (*Triticum aestivum*), winter barley (*Hordeum vulgare*), maize (*Zea mays*), sugar beet (*Beta vulgaris*) or potatoes (*Solanum tuberosum*)) and ii) farms exceed a minimum economic size which is based on the EU indicator Standard Output (SO). The SO is the average monetary value of the agricultural output at farm-gate price (euro ha^{-1} $year^{-1}$). The economic size is the sum of the SO per hectare of crop. Keeping the economic size threshold at € 25 000 $year^{-1}$ resulted in a subset of farms with on average 45 ha arable area, with 15.5 ha of winter wheat, 3 ha of winter barley, 11.5 ha of maize, 6 ha of sugar beet and 9 ha of potatoes.

In the BAU option, the farmer receives a basic payment without greening payment, but, due to not meeting the greening conditions, he additionally loses up to 25% of this basic payment (European Commision, 2013). Basic payment is estimated to be € 200 ha^{-1}. In this exemplary case, the hedgerow option is implemented on 1.125 ha (2.5% of 45 ha) of the case farm. Layout and management of the hedgerow option is described in section 7.20. We did not include hedgerow wood as a revenue, because in Flanders, hedgerow maintenance costs are barely compensated by the wood revenues. Also, opportunity and real costs related to administration are not included, as it was assumed that they are similar for every EFA option. In the alley cropping option, poplar trees (*Populus* sp.) are planted on a 2.25 ha parcel. Distance between the tree rows is 35.5 m and trees within the same row are 6 m apart. Poplar is often used in alley cropping because of its relative short rotation time, relatively narrow crown (Das and Chaturvedi, 2005) and suitability for agroforestry conditions in Flanders (Reisner et al., 2007). Layout and management of the alley cropping option is described in section 7.20. Both the hedgerow option, alley cropping option and farm structure are an exemplary case out of a wide range of potential arrangements or case farms which could be virtually constructed.

During the construction of the farm and the EFA options, we faced a lot of uncertainties and assumptions had to be made. For the most uncertain parameters, a sensitivity analysis is performed by varying the value of the parameter. Although other farm choices are possible to meet the EFA requirements, no other options are taken into account.

5.4 Results

5.4.1 Effect of trees on crop yield

Data from the literature search and own experiments are represented in Figure 5.1. When the distance from the tree and thus H increases, tree impact on crop yield decreases, and ln(R) approaches zero. Between H=0 and H=1.64, ln(R) values are negative and crop yield is negatively influenced by the trees. Between H=1.64 and H=9.52, ln(R) values are positive and crop yield is positively influenced by the trees. When H > 9.52, the effect of the trees on crop yield is negligible. In the meta-regression, a linear relation between ln(R) and log(H) was elaborated between H=0 and H=1.64:

$$ln(R) = -0.19 + 0.33 \times log(H) \qquad \text{(Equation 5.1)}$$

In the non-linear mixed model, a polynomial function of the second order was selected, based on the Akaike's information criterion and the preference for a straightforward model. This resulted in the equation:

$$ln(R) = -0.76 \times log(H)^2 + 0.91 \times log(H) - 0.16 \qquad \text{(Equation 5.2)}$$

In section 7.17, both models are compared. The non-linear mixed model is chosen for incorporation in the calculations because it takes into account the whole dataset and its range extends beyond H=1.64. In section 7.18, the difference between hedgerow and tree row impact is visually investigated, but since no strong trend was detected, we assume an equal effect and hence the final model is based on data from both hedgerows and tree rows.

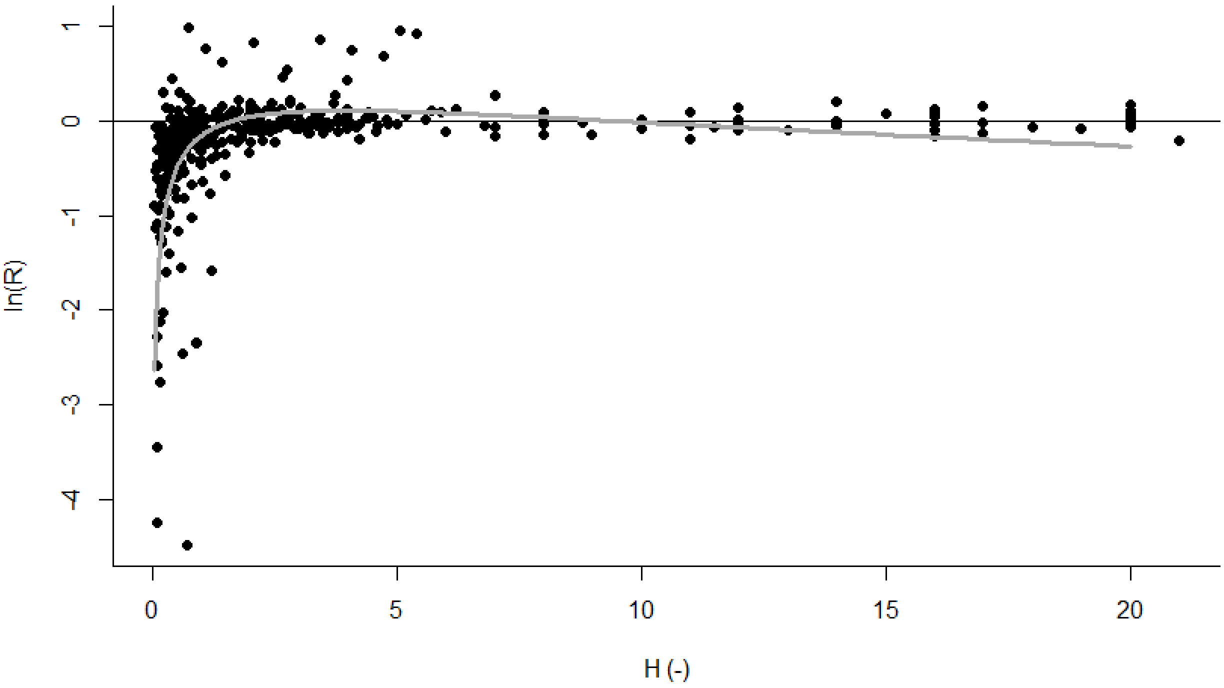


Figure 5.1: Relative crop yields (R) for varying relative distances (H) from the tree row and hedgerow. Data are both based on a literature review and own monitoring. The non-linear mixed model is fitted. Literature review data are retrieved from Rivest and Vezina (2014), Burgess et al. (2004), Chirko et al. (1996), Gao et al. (2013), Reynolds et al. (2007), Stamps et al. (2009), Chaves (2001), Senaviratne et al. (2012), Esterka (2008), Woodall and Ward (2002). Own data were measured by Pardon and Van Vooren.

5.4.2 Calculations

The calculations are suitable for both parcel level and farm level modelling. In section 7.19, the calculation components and mechanisms are described. Based on the tree-crop interactions obtained in the non-linear mixed model, we simulated crop productivities for the two greening options. Next to a hedgerow or tree row, crop yield at H=0.5 is 61% of BAU crop yield, at H=1 crop yield is 85% and at H=1.64, crop yield is 100%. Between H=0.01 and H=1.64, overall yield is 70%. At H=5, crop yield is 111% and at H=9.52, crop yield is 100%. Between H=0 and H=9.52, overall yield is 101%. Next, crop yield at plot level is examined. One hectare (100 m x 100 m) of winter wheat yields 8.6 ton ha^{-1} of grain (section 7.21). In the hedgerow and alley cropping option, yield is affected in two ways: i) loss of arable production surface and ii) tree – crop interaction. In Figure 5.2, wheat yield for each of the options is represented. In the hedgerow option, reduction of wheat yield is mainly due to crop surface loss. During the first five years after hedgerow planting or coppicing, the extent of the positive effect increases. After five years, hedgerow height causes the negative effect to overrule the positive effect. In the alley cropping option, the smaller distance between the tree rows causes the positive effect to be offset by the negative effect of the adjacent tree row.

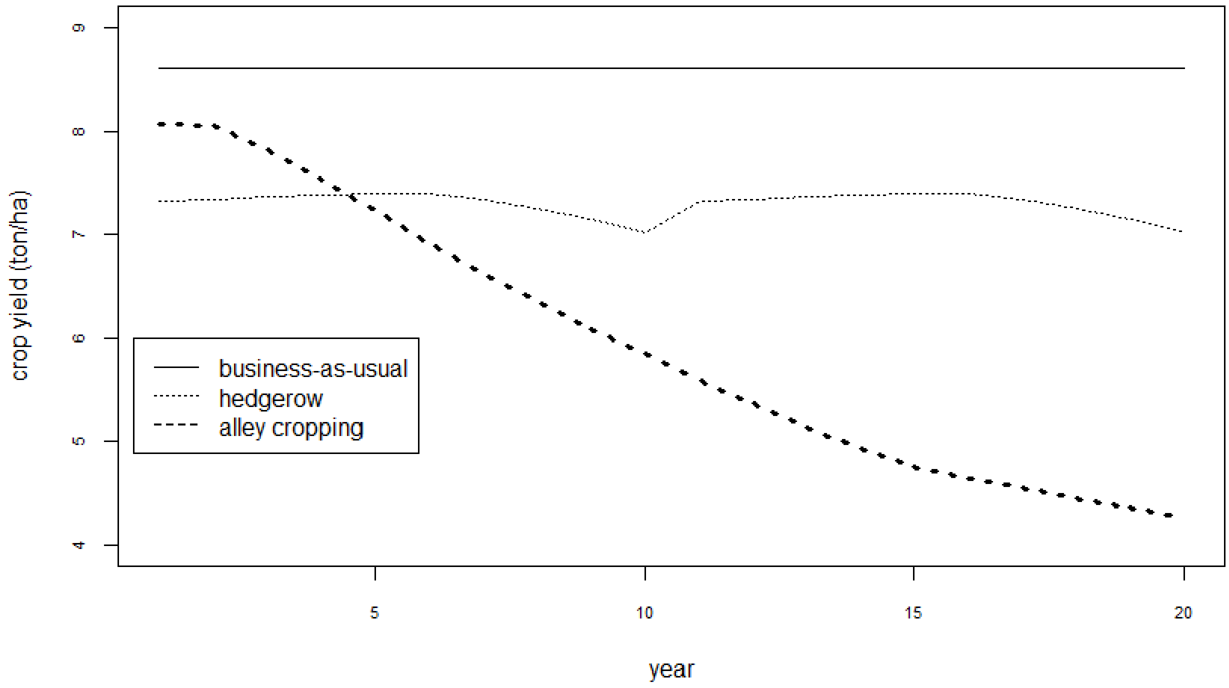


Figure 5.2: Wheat yield (ton ha^{-1}) in a business-as-usual, hedgerow and alley cropping option. In the business-as-usual option, no hedgerows or tree rows are planted. In the hedgerow option, a hedgerow is planted on two sides of the parcel. In the alley cropping option, two tree rows are planted on the parcel.

5.4.3 Economic consequences

Yield data are converted into monetary values based on crop prices and costs. Figure 5.3 shows the effect of crop species and represents the non-discounted crop gross margins for one hectare (100 m x 100 m) in the BAU, hedgerow and alley cropping option. Every year, a different crop is planted, resulting in a 5-year rotation. The gross margin highly depends on the crop: if the crop has a high financial return (€ ha^{-1}), the yield reduction results in a more severe impact on gross margins. A high financial return is affected by a combination of yield (ton ha^{-1}), selling price (€ ton^{-1}) and costs (€ ha^{-1}). Potatoes have a higher financial return, which compared to the other crops, results in a strongly reduced gross margin compared to the BAU option. As a result of tree growth, the difference between BAU and alley cropping increases over time. After 20 years, the discounted gross margin of the hedgerow option parcel is 76% of the discounted gross margin in the BAU option. For the alley cropping option, the relative discounted gross margin is 55%.

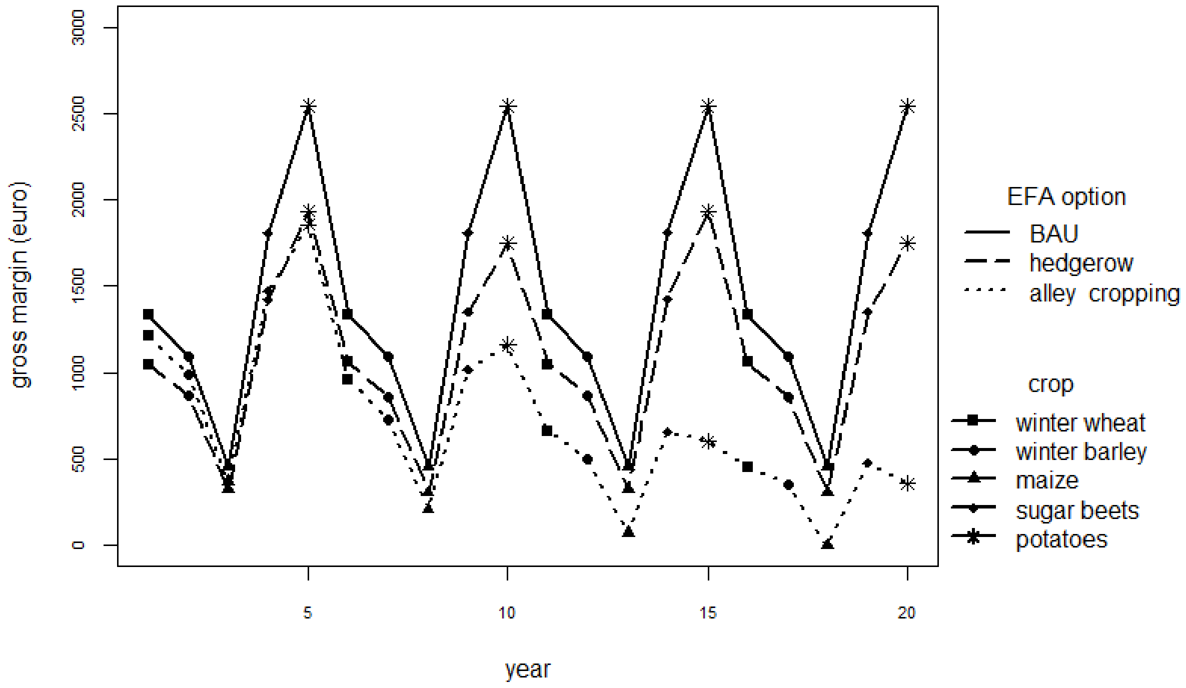


Figure 5.5: Yearly gross margins (€) of the business-as-usual (BAU), hedgerow and alley cropping option on a 1-hectare parcel. A rotation of five crops is implemented. No extra payments are included.

At farm level, all crops, crop revenues and costs are taken into account as well as hedgerow and alley cropping plantation costs and revenues (section 7.20 and 7.21). The differences between the discounted gross margins for the greening options and the BAU option at farm level, based on the farm we constructed and the assumptions described (column 'Initial assumptions'), are shown in Table 5.1. The other columns show the results of a sensitivity run (SR) that was performed to investigate the effect of initial assumptions.

Without subsidies, both the hedgerow and alley cropping options have a lower financial return than the BAU option. When accounting for the basic and greening payments, the results turn positive. In the alley cropping option, gross margin reduction is lower compared to the hedgerow option. This is due to EFA lay-out: in the alley cropping option, the whole parcel is designated as EFA while in the hedgerow option only the surface of the hedgerow is included. Despite the expected higher productivity, poplar wood revenues do not compensate for crop yield loss. This is in line with what was found by Dupraz & Liagre (2008). It should be noted that they did find positive economic results for alley cropping with high value timber, but this was not further investigated in our research. Additionally, we did not consider valorisation of the pruning material because not enough reliable information was available on this topic, but wood chip valorisation for e.g. bioenergy, mulch or composting (Viaene et al., 2016) may help to obtain a more promising picture.

Table 5.12: Difference (€) between the farm discounted gross margins for the hedgerow or alley cropping option and the discounted gross margin for the business-as-usual (BAU) option. The relative discounted gross margins (compared to the

BAU discounted gross margins) is shown between brackets. To assess the impact of agricultural prices, wood prices and the discount factor, a sensitivity run (SR) was performed by varying the value of the variables.

Difference in discounted gross margins	**Initial assumptions**	**$SR_{agricultural\ prices}$**	**$SR_{wood\ prices}$**	**$SR_{discount\ factor}$**
Hedgerow option, without subsidies	-70 972 (91%)	- 113 807 (94%)	-70 972 (91%)	-83 509 (91%)
Hedgerow option with subsidies	17 354 (102%)	-25 481 (99%)	17 354 (102%)	19 926 (102%)
Alley cropping option, without subsidies	-19 446 (98 %)	-38 162 (98%)	-16 900 (98%)	-23 166 (98%)
Alley cropping option, with subsidies	69 351 (108%)	50 635 (103%)	71 897 (108%)	80 741 (108%)

In $SR_{agricultural\ prices}$ agricultural prices follow the trend of the past years (2006-2012). Increases in winter wheat, winter barley, sugar beets and maize prices vary from 6% to 11%. Despite yearly fluctuations in the price of potatoes, no trend was detected for this crop. Price trends are given in section 7.21. When crop selling prices increase and costs remain constant, crop profitability rises, and thus the difference between BAU and the greening options increases, in favour of the BAU option. In $SR_{woodprices}$, wood prices are doubled. No true effect is observed, wood price does not notably influence the overall result because its contribution to the farm income is very small, compared to the crop revenues. In $SR_{discount\ factor,}$ the discount factor is set at 0.03, a factor that is often used in the forestry sector. A lower discount factor results in more divergent values, but the relative proportions remain the same.

Figure 5.4 represents the revenues and costs of the EFA options at farm level, compared to the BAU option. Due to reduced crop yield, crop gross margins are negative compared to the BAU option. The ‘startup costs’ category consists of plant purchase costs, planting costs and plant protection costs. In the ‘maintenance costs’ category, weed removal, tree row maintenance and pruning are included. Exact values of these costs can be found in section 7.21. The ‘subsidies’ category consists of the difference between the direct and greening payments for farms with 100% fulfilment of greening conditions and reduced direct payments for farms that do not fulfil greening requirements. In the alley cropping year, subsidies consist of greening payments and planting subsidies foreseen in EU Rural Development submeasure 8.2.

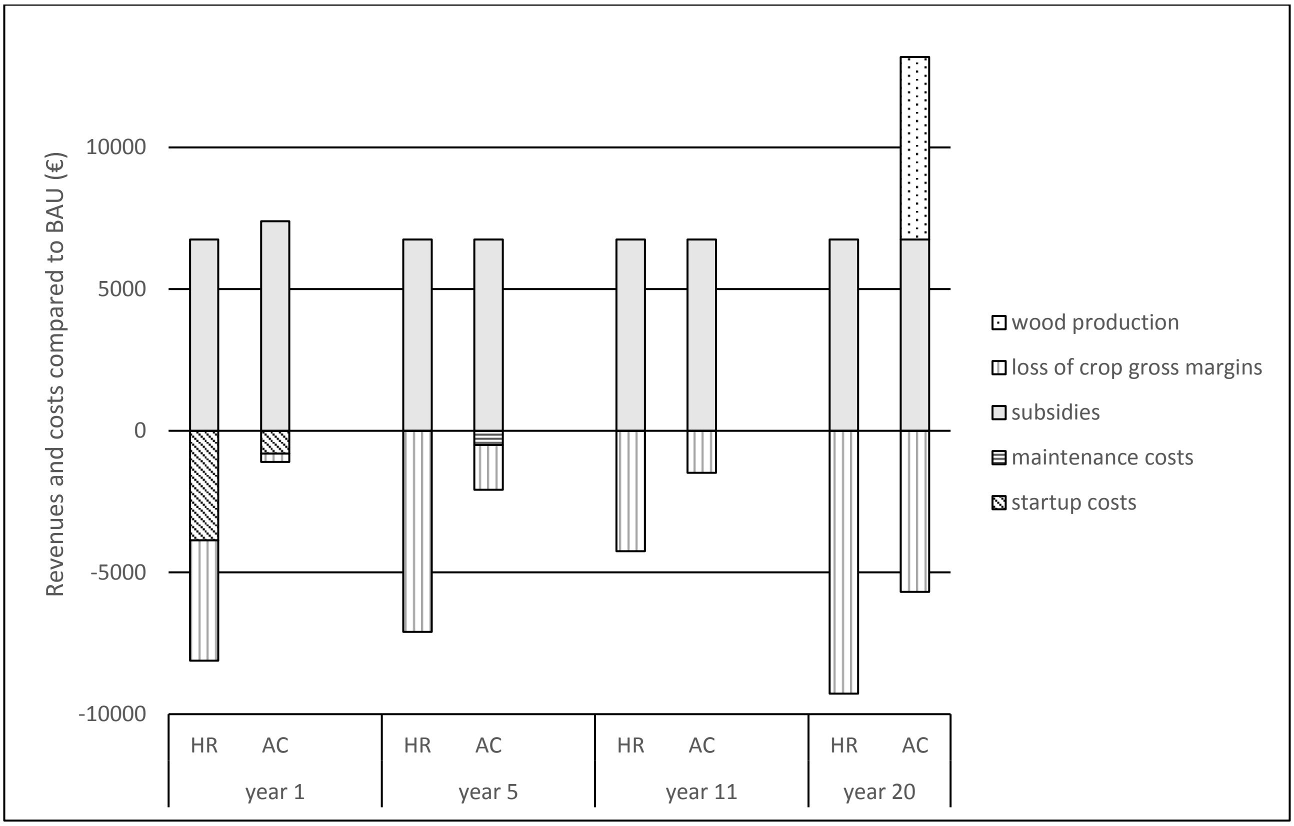


Figure 5.4: Difference in yearly revenues and costs (€) of the hedgerow (HR) and alley cropping (AC) option, compared to the business-as-usual option. The calculation is performed for a representative farm with 45 ha arable area.

5.5 Discussion

5.5.1 Effect of trees on crop yield

In a meta-analysis, results of different studies are aggregated to make a general statement and to gain more profound insight. This allows us, inter alia, to detect trends in the effect of a variable, to perform an analysis with a greater statistical power and to track knowledge gaps. However, a large amount of data is needed to obtain a reliable result (Borenstein et al., 2007; Koricheva and Gurevitch, 2014). Because we applied rather strict conditions (see above), the number of suitable studies was limited to twelve. This has repercussions on the validity of our analysis. As a rule of thumb, in a meta-regression it is recommended to include at least ten studies for each moderator (Borenstein et al., 2009). Because of the limited number of studies we used, the only moderator we included was H, although we know that crop type (Gao et al., 2013), tree species, tree line space and tree line orientation will have an impact on crop yield as well (Chirko et al., 1996; Dufour et al., 2013). Beyond H=5, the number of data and thus studies contributing to the model is rather limited, making the estimations more precarious.

During the literature search, we purposely did not search for ‘windbreak’ or ‘shelterbelt’ because we wanted the experimental conditions to be as comparable as possible to the agricultural conditions in Flanders. Because wind damage to crops is exceptional in Flanders given the relatively small parcel sizes and high urbanisation grade, we expected no strong positive effect of reduced wind velocity on crop yield.

Tree-crop interaction can reduce or enhance crop yield. Mainly competition for water, nutrients and light will result in a lower crop yield. Modification of the microclimate in terms of temperature, water distribution and wind speed can result in higher crop yield (Jose et al., 2004). It seems that between H=0 and H=1.64 the competition effect is stronger than the potential positive microclimate effect, resulting in lower crop yield. Beyond H=1.64, a higher yield is achieved, possibly due to microclimate modification. This is similar to what is found by Borin et al. (2010) and in several studies on windbreaks and shelterbelts (Kort, 1988). This could mean that, despite the fact that wind damage is not significant in the regions we studied, the microclimate is still improved by the hedgerow or tree row.

5.5.2 Calculations

In the Silvoarable Agroforestry For Europe (SAFE) project, a set of calculation tools was developed: i) Hi-sAFe, focussing on biophysical interactions between trees and crops, ii) Yield-sAFe, linking biophysical information to annual yield and iii) Farm-sAFe, allowing economic analysis of the system (Graves et al., 2011). Within the AGFORWARD project, Hi-sAFe and Yield-sAFe are currently being extended (Burgess et al. 2015). Both are biophysical, process-based and dynamic models. The Yield-sAFe model allows prediction of crop yield in different alley cropping and other types of agroforestry scenarios. In contrast to our calculations, Yield-sAFe requires a set of input variables linked to inter alia soil, crop and climate characteristics. These variables are not always known and often, there is no time or funding to further investigate them. In those cases, we believe that our calculations can be helpful to assess hedgerow and tree impact on crop yield. However, we realize that our approach is at the expense of nuance and precision. Farm-sAFe is a bio-economic model, combining Yield-sAFe and an economic model (Burgess et al., 2015).The mechanisms in Farm-sAFe are very similar to our own calculations, but as Farm-sAFe depends on input from Yield-sAFe, we could not use this model.

On a 1-ha parcel with alley cropping and a tree row distance of 35.5 m, we found a relative yield of 84% after 5 years, 68% after 10 years and 50% after 20 years. These results are strongly in line with results from the Yield-sAFe model applied on a Swiss casus (Sereke et al., 2015). In this research, the Farm-sAFe model was used to investigate profitability. In the baseline scenario, which was, except for the wood price, based on similar assumptions as our calculations, the net present value (NPV) of the arable alley cropping system ranged between 87 and 114% of the monoculture NPV. This is considerably higher than the parcel-level discounted gross margins we found, but this is due to the tree species and according lower wood prices we used.

5.5.3 EFA impact

Despite their numerous environmental advantages (Baudry et al., 2000), establishment of hedgerows is often not or only to a very limited extent an attractive EFA option from a farm-level economic point of view, because only the actual hedgerow surface is designated as EFA. Because of this, an extensive network of hedgerows needs to be present to fulfil the greening requirement, resulting in high losses of arable land. In the alley cropping option, the whole parcel is designated as EFA.

Next to hedgerows and alley cropping, the EFA requirements can be fulfilled with catch crops, nitrogen fixing crops, fallow land, buffer strips etc. Actual ecological benefits of these EFA options depend on spatial implementation but the greatest effects are expected for permanent measures (Westhoek et al., 2012). However, within the available member state options, farmers are free to choose how to implement the EFAs. This entails the risk that non-permanent measures with less ecological benefits will be broadly applied to reach the greening requirements, for example because they are more convenient for farmers (Matthews, 2013; Westhoek et al., 2012). This has shown to be the case in Flanders, where 85% of the EFAs consists of cover crops. Because of the considerable impact of hedgerows and alley cropping on farm economics and the few benefits of more easily implemented non-permanent measures, we suggest a better linking between subsidies and ecological benefits.

Additionally, research has shown that economic incentives alone do not determine the uptake of (voluntary) nature-oriented measures like agri-environmental schemes or erosion measures. Therefore, raising subsidies or other financial compensations alone will not suffice to stimulate the adoption of certain types of nature-oriented measures. Siebert et al. (2006) reviewed 160 publications on the uptake of nature-oriented measures in the EU and they identified three components affecting farmer's participation: farmers' willingness, farmers' ability and social influences. Farmers' willingness follows from interests, values, norms, problem awareness and self-perception. One of the main interests is economic motivation, but apart from profit maximization, these can be long-term viability and resilience and the risk minimization. Other specific aspects contributing to willingness are the will to contribute positively to the environmental quality, to maintain the farm for future generations, to have a satisfactory job and positive acknowledgement by society. Farmers' ability refers to objective factors in the context of the farmer (e.g. age, education level), type and organisation of the farm and biogeographical conditions. Social influences include direct social interaction and socio-cultural, political and juridical influences. Wauters et al. (2017) investigated farmers' intentions to adopt nature-oriented measures in Belgium. Like Siebert et al. (2016), they found that attitude, perceived behaviour control and both subjective and group norms (corresponding to willingness, ability and social influences, respectively) affect intention, but additionally, they identified a central role of self-identity and moral norms. Finally, Borremans et al. (2016) found that, despite a government-initiated subsidy program, the adoption of agroforestry in Flanders was low. They identified different groups of actors, i.e. researchers, entrepreneurs, intermediate actors, the government and society. Intermediate actors include for example environmental organizations and farmers' organizations. Each of these actors contributes to the adoption (or not) of agroforestry, and more generally nature-oriented measures. Also, they formulated the following recommendations to improve the uptake of agroforestry, including the development of a clear legal framework, the engagement of a wide range of private and societal actors and improving communication and education. To conclude, although profit maximization is of high importance for farmers, many other aspects play a role as well and a rather straightforward cost-benefit analysis, as presented in this paper, does not take into account any of the abovementioned aspects.

Other ecosystem services, apart from production, are not taken into account in the calculations, because under the current market conditions they do not contribute directly to the farm income. However, ecosystem services can supply many benefits to society (Jacobs et al., 2014a). For Belgium and other temperate areas, ecosystem services that can be provided by trees are (next to wood production): carbon sequestration, soil and water quality regulation, air quality regulation, mitigation of noise and visual impacts, mass stabilization and erosion control, hydrological cycle and water flow maintenance, pest and disease control, micro and regional climate regulation (Cardinael et al., 2015; Jacobs et al., 2014a; Tsonkova et al., 2014). Since ecosystem services are externalities, there is generally no market for them. A carbon market could benefit hedgerow and alley cropping systems (Toor et al., 2012), but we decided not to take into account the carbon payments, because the current EU Emission Trading System does not allow the use of credits from land use, land use change and forestry (LULUCF) systems. As farmers tend to maximize private benefits, an adapted land use with more attention for ecosystem services provision, as is the case for hedgerow and alley cropping implementation, is currently not very profitable (Van Hecken and Bastiaensen, 2010).

5.6 Conclusion

Although simulating the farm economic impact of greening options faces long term uncertainties, the framework and calculations in this study provide novel and quantitative insights in the feasibility of alley cropping and hedgerows in temperate regions with intensive arable farming. An important factor is the tree-crop interaction, which can be improved by adding information on tree and crop types, tree row orientation and distance etc. Given their flexibility, the calculations are adaptable to novel insights coming from crop, forest and technology science.

The first results of the EFA options calculated here, under the specific circumstances described in the paper, indicate that without greening payments and/or other financial supporting mechanisms, hedgerows and alley cropping, when implemented as the only fulfilment of the 5% EFA requirement on arable farms, are economically not attractive for farmers. Exceptional attractive situations, e.g. when the wood can be used on the farm for heating, are not considered. Given this importance of support from policy, such as the new CAP, we suggest that attention should be paid to the initial goals of EFA, namely to encourage biodiversity on agricultural fields. The discrepancy in profitability of greening options with or without subsidies must warn for too big dependency of greening options from subsidies, and for a competition for subsidies from cropping systems which are more similar to our BAU option. This puts a challenge to policy makers to ensure that effective ecological impact of the different greening measures is taken into account in the actual payment levels. Higher association of payment level to ecological benefits may increase relative competitiveness of alley cropping and hedgerows, also with respect to non-permanent EFAs. One option to meet this target is by widening the range of weighting factors and ensuring that the difference in weighting factors between measures with low expected ecological value and measures with high expected ecological value is big enough. In this case, this would mean that the weighting factor of hedgerows should be higher. Additionally, it could be interesting to consider different weighting factor according to alley cropping type, depending on for example tree species, tree species mixture, tree row maintenance etc.

Finally, in this paper, only direct yield losses due to the greening requirements were investigated. However, various aspects of farm management have been set to a simplifying assumption, but show potential for many indirect benefits. One example is the wood chips valorisation: when composted and used on the farm to improve soil quality and reduce the need for fertilization, a positive effect on yield and revenues can be expected in the long term. Also for this type of recommendations, the calculations are able to capture novel insights from practice or sciences.

6. General discussion and conclusion

To put an end to environmental degradation and biodiversity loss in rural landscapes, for the greater part caused by agricultural intensification and expansion (Foley et al., 2005; Rey Benayas and Bullock, 2012), specific measures are taken at different levels and scales. Greening measures proposed in the context of the Common Agricultural Policy are representative for these efforts. The measures taken – we call them nature-oriented measures here – are expected to increase the delivery of multiple ecosystem services and the enhancement of biodiversity in agricultural landscapes. They include the introduction of semi-natural elements and the inclusion of ecological principles in agricultural practices. The impact of a number of these measures on ecosystem services and biodiversity has already been studied (Cardinael et al., 2015; Schmitt et al., 1999; Yang et al., 2015), but generally within one study only one or several related response variables are evaluated at the time. Moreover, this type of research in our study area, i.e. Flanders, is limited. Also, so far not much research has focused on the economic and financial impact for farmers of the implementation of nature-oriented measures. This economic reality nevertheless should be taken into account in order to be able to tackle the challenges related to potential yield losses and resource investments.

In this research, we assessed the simultaneous impact of three types of nature-oriented measures on multiple ecosystem services (both provisioning and regulating services) and on biodiversity. These nature-oriented measures include the implementation of i) hedgerows and ii) grass strips on arable field borders as well as iii) the extensification of grassland management. First, we quantified the effect relationship between hedgerow and grass strip characteristics and ecosystem service and biodiversity by means of a meta-analysis (chapter 2). Because assessments based on the combination of results of studies that focus on one response variable will most likely result in an overestimation of multifunctionality (as discussed in chapter 1), we also performed an empirical study and monitored a set of ecosystem service and biodiversity indicators on parcels in Flanders with either a hedgerow or a grass strip (chapter 3). Next, relationships between grassland management type and intensity on the one hand and ecosystem service delivery and biodiversity on the other hand were assessed, both via own monitoring data and a literature review (chapter 4). For all measures, we explored whether there were trade-offs in the delivery of various ecosystem services and biodiversity, with particular attention for trade-offs between provisioning services (agricultural productivity) and regulating services and (climate regulation and regulation of chemical water quality) and biodiversity. In chapter 5, we developed a calculation tool to quantify the farm-level financial consequences of the implementation of nature-oriented measures. The assessment framework was applied in the context of the most recent (since 2014) CAP reform, with hedgerows and alley cropping being two options to reach the EFA (ecological focus area) requirements through increasing the presence of permanent, woody vegetation in the agricultural landscape. In this final chapter, we integrate the results of previous chapters (6.1 and 6.2) and we assess multifunctionality and trade-offs at farm level (6.3). Also, we discuss relevance on regional and global level (6.4), potential for further research (6.5) and formulate concrete management and policy recommendations (6.6).

6.1 Impact of hedgerows and grass strips on ecosystem services and biodiversity

The potential of hedgerows and grass strips for ecosystem service delivery and biodiversity has been demonstrated many times (Borin et al., 2005b; Cardinali et al., 2014; D'Acunto et al., 2014; Duchemin and Hogue, 2009; Falloon et al., 2004; Holland et al., 2012; Schmitt et al., 1999; Van Beek et al., 2007). Most research has focused on only one or several related ecosystem services and biodiversity indicators, such as N and P in runoff water. However, many authors have expressed the need for a more holistic approach, considering multifunctionality as the main goal of agricultural landscape management (Gamfeldt et al., 2008; Hector and Bagchi, 2007; Reiss et al., 2009). Optimal multifunctional landscape management requires insight into the simultaneous responses of ecosystem services and biodiversity to the management practices and into the synergies and trade-offs (Bommarco et al., 2013; Bullock et al., 2011; Reiss et al., 2009).

In chapter 2, a meta-analysis was performed to describe the impact of hedgerows and grass strips on a broad set of ecosystem service indicators. Also, the effect of relative distance from the hedgerow, hedgerow width and grass strip width was quantified. We found that close to the hedgerow, until a distance of twice the hedgerow height, crop yield was reduced. Beyond this point, until a distance of 20 times the hedgerow height, crop yield was increased. Also next to the hedgerow, SOC stock was higher compared to further into the parcel. Hedgerows intercepted N from the overland water flow (the surface flow) and from the flow beneath the surface (the subsurface flow), P and total suspended solids (TSS) from the surface flow. More species of natural predators were found on parcels with a hedgerow, but this did not result in an increased number of natural predators. These results indicate an effect of the hedgerow on the ecosystem services crop yield, climate regulation, water quality regulation and erosion regulation.

In a grass strip next to arable land, SOC stock was increased compared to the adjacent arable part of the parcel. The grass strip intercepted N, P and TSS from the surface flow, and N from the subsurface flow. On parcels with grass strips, both predator density and diversity were higher and aphid density was reduced. These results indicate an effect of the grass strip on climate regulation, water quality regulation, erosion regulation and potential for pest control.

Assessing multifunctionality by combining studies that focus on only one response variable may lead to overestimations because monitoring sites are often selected to demonstrate a maximal impact of a measure on a specific response variable (Bommarco et al., 2013). Our own experimental setup described in chapter 3 aims at tackling this concern. In this chapter 3, we present the results of a set of simultaneous, parcel-level and straightforward ecosystem service and biodiversity indicator measurements on a series of arable parcels with hedgerows or grass strips in Flanders. These data were the result of one year of monitoring, and when interpreting the trends, it is important to keep in mind that they only provide a snapshot of the ongoing processes. Also, as a consequence of focussing on within-parcel impacts, different approaches are used for hedgerows and grass strips: the monitored grass strips make up a considerable part of the adjacent parcel and ecosystem service delivery is affected mostly within the grass strips, while hedgerows are typically planted on the field borders and the parcel-level impact is situated mostly in the adjacent arable parcel. Next to hedgerows, crop yield was reduced and winter wheat thousand kernel weight, SOC stock and spider activity-density were increased, compared to results in the centre of the parcel. These indicators show an effect of the hedgerow on crop yield, climate regulation and potential for pest control (by increased spider activity-density). In the grass strips, we found an increase in SOC stock, a decrease in soil mineral N content, a different carabid species composition and a higher spider activity-density, all compared to the adjacent arable parcels. These results indicate a contribution of grass strips to climate regulation, water quality regulation (by the reduction of N leaching) and enhanced biodiversity. We did not find the expected effect of the hedgerow on soil mineral N content, soil P concentration, on carabid activity-density and number of species and on rove beetle activity-density. For grass strips, trends in soil P concentration, carabid activity-density and number of species and spider and rove beetle activity-density did not match our expectations. Both for hedgerows as for grass strips, these inconsistencies may be attributed to the effects of local management and timing of the measurements, but also to a less suitable experimental setup.

To investigate the simultaneous delivery of multiple ecosystem services, the derived effect relationships based on the meta-analysis data were used to simulate the change in ecosystem service delivery and biodiversity by different types of hedgerows and grass strips on different types of parcels. More specifically, two hedgerows (one narrow and low, one wide and high) and two grass strips (one narrow, one wide) were implemented on a virtual parcel of 50 m x 100 m and on a virtual parcel of 100 m x 100 m. Changes in ecosystem service delivery and biodiversity were compared to a similar parcel without hedgerow or grass strip. This simulation showed clear effects of both the hedgerow and grass strip on parcel-level ecosystem service delivery and biodiversity.
However, as stated above, we presume that these results are an overestimation or at least the maximum of the effects that can be expected in a real-life situation. Therefore, to compare the results from the meta-analysis (chapter 2) with our own monitoring data (chapter 3), we repeated the calculations, but instead used the results of our own monitoring. In order to show the strongest results, the calculations were performed for the implementation of a wide and high hedgerow (7.5 m width, 10 m height) and to a wide grass strip (12 m width) along the long side of a virtual parcel of 50 m x 100 m. While we assumed that grass strips do not affect crop yield, parcel level yield decreased as a result or arable land loss. In Table 6.1, the results of the parcel-level calculations based on the meta-analysis and on the monitoring data are presented. Because our experimental setup did not consider erosion and because our measuring methods for nutrients differed from the meta-analysis, not all variables can directly be compared.

Table 6.1: Prediction of parcel-level ecosystem service and biodiversity indicators based on results from the meta-analysis and own monitoring data. Response variables for a parcel with a hedgerow and a parcel with a grass strip are standardized relative to a parcel without hedgerow or grass strip. NE means that there was no effect.

	Hedgerow		Grass strip	
Ecosystem service indicators	Meta-analysis results	Monitoring data	Meta-analysis results	Monitoring data
Crop yield	-22%	-20%	-24%	-24%
SOC stock	+8%	+2%	+9%	+13%
Surface N interception/reduction[1]	+64%		+83%	
Subsurface N interception/reduction[1]	+34%	NE	+46%	+36%
Surface P interception[1]	+67%		+81%	
Subsurface P reduction[1]		-6% [3]		-195% [4]
Erosion interception	+91%		+94%	
Potential for natural pest control[2]				
∟ density of predators	NE		+362%	
∟ diversity of predators	+70%		+146%	
∟ density of carabids		NE		NE
∟ diversity of carabids		NE		
				NE
∟ density of spiders		+6%		+25%
∟ density of rove beetles		NE		
				NE

1 Meta-analysis results were based on sampling of water flows, indicating nutrient interception. Monitoring results were based on soil sampling, indicating nutrient reduction.

2 Meta-analysis results were based on predator overwintering in the hedgerow and grass strip and predator summer presence on parcels with a hedgerow or grass strip. The control was overwintering in the parcel or summer presence on a parcel without hedgerow or grass strip. Monitoring results were based on sampling of activity-density and number of species next to the hedgerow and in the grass strip. The control was activity-density and number of species further into the parcel.

3 P-Olsen concentration was significantly higher closer to the hedgerow, resulting in a negative effect on ecosystem service delivery.

4 P-Olsen and P-$CaCl_2$ concentrations were significantly higher in the grass strip, resulting in a negative effect on ecosystem service delivery. The highest concentration (P-$CaCl_2$) is reported.

Because the control plots were located at a distance of 30 m from the hedgerow (H=3) and because the potential impact of the hedgerow on crop yield and SOC stock extends to H=20.43 and H=4.30 respectively, according to the meta-analysis, it seems plausible that the calculated monitored effects are somewhat underestimated. Comparison of the predictions i) confirms the hypothesis that assessing multifunctionality by combining studies that focus on only one response variable will lead to an overestimation of multifunctionality and ii) shows that straightforward monitoring of ecosystem services and biodiversity has its limitations. More specifically, we found that crop yield and to a lesser extent SOC stock next to the hedgerow were comparable in the meta-analysis and our own data, but we did not monitor the predicted trends for N and P reduction. This indicates that N and P are not intercepted from the water flows through hedgerow root uptake in the arable parcel and thus that other processes lead to N and P interception. Also, only spider activity-density was slightly increased near the hedgerow and carabid and rove beetle activity-density were unaffected. This is similar to the results from the meta-analysis. We did found more species of natural predators on parcels with a hedgerow in the meta-analysis, and this is in contrast to the lack of carabid diversity on the monitored parcels. This may be due to the fact that we missed the post-hibernation colonization process. For grass strips, the monitored trends were generally consistent with meta-analysis. Only P in the grass strips was unexpectedly high, and we assumed that this was due to the deposit of sludge after ditch clearing. Carabids and rove beetles seemed unaffected by the grass strip on the monitored parcels, but again, we assumed that we missed the post-hibernation colonization process.

It seems thus, that both hedgerows and grass strips have the potential to increase the delivery of multiple ecosystem services and biodiversity, but that the predictions based on the meta-analysis are rather an indication of the maximal effect that can be expected and that in real-life situations, actual impact may be smaller. Also, we conclude that the adequate monitoring of ecosystem service and biodiversity indicators requires a substantiate and well-thought monitoring campaign, which imposes restrictions on the general feasibility of simultaneously measuring multiple response variables.

6.2 Impact of grassland management on ecosystem service delivery and biodiversity

Measures to enhance faunal or floral biodiversity associated with grasslands are expected to affect a wider range of ecosystem services. More specifically, very often there is a trade-off between biomass yield and quality (provisioning ES) and many regulating ES (Maes et al., 2011; Pilgrim et al., 2010). Grassland management consists of many aspects (fertilization amount, fertilization type, number of grass cuts, livestock management etc.) which can all affect provisioning and regulating ES and biodiversity in different ways. Few studies have investigated the effect of varying the full range of management aspects on a broad set of ES and biodiversity simultaneously, and consequently there is a lack of insight in the trade-offs and interactions (Batáry et al., 2015; Pilgrim et al., 2010).

In chapter 4, we investigated the effect of grassland management type and intensity on multiple ecosystem service and biodiversity indicators. To do so, we monitored two sets of grasslands in Flanders with varying management types: a regular, intensive management, a meadow bird management and a botanical management. Grasslands with meadow bird management are fertilized with farmyard manure with a total application restricted to 120 kg N/ha. Grazing was permitted after June 15th and mowing after July 15th. Application of pesticides was not allowed. Grasslands with a botanical management were not fertilized, no pesticides were applied and grazing and mowing were only allowed after July 15th. For every monitored grassland, a land use intensity (LUI) index was calculated and linked to the measured ecosystem service and biodiversity indicator values. We found that management type affected biomass yield, crude protein yield, soil mineral N content and number of plant species. More specifically, biomass and crude protein yield were lower on the grasslands with a meadow bird or botanical management. At the same time, soil mineral N content was higher on the grasslands with a regular management, entailing a higher risk for N leaching. The number of plant species was significantly lower on the regular grasslands. Soil organic carbon stock and number of carabid species were not affected by management type. Land use intensity of the grasslands appeared to be an appropriate predictor for the response variables: the higher the intensity index, the higher the biomass and crude protein yield and soil mineral N content and the lower the number of plant species. After performing a literature review, similar relationships were found. Additionally, analysis of the literature data revealed that increasing land use intensity with animal fertilization resulted in higher soil carbon stocks. In Table 6.2, effect relationships developed based on own monitoring data and on data from the literature review are compared. For an increase of land use intensity of one unit, the expected effects on ecosystem service and biodiversity indicators are calculated. An increase of land use intensity corresponds to an increase of fertilization dose and of mowing and/or grazing. Specific levels depend on the regional average fertilization, mowing and grazing (see chapter 4).

Table 6.2: Prediction of the change in ecosystem service and biodiversity indicators after an increase of one unit of land use intensity based on effect relationships from own monitoring data and from a literature review. Because all monitored grasslands received animal fertilizers, only effect relationships derived from studies in which animal fertilizer (slurry and manure) was used, were retained. NE means that there was no effect.

Ecosystem service and biodiversity indicators	Monitoring data	Literature review
Biomass yield	+ 15%	+ 32%
Crude protein yield	+ 21%	+ 29%
SOC stock	NE	+ 9%
Soil mineral N content	+ 17%	+ 15%
Number of arthropod species	NE	NE
Number of plant species	- 19%	- 11%

The predicted effect of increasing LUI was remarkably similar for our own data and for the data from the literature review, indicating that in temperate grasslands, the impact of varying grassland management intensity on ES and biodiversity is relatively consistent. Also, trade-offs will generally be the same, and they are very clear for biomass yield and forage quality on the one hand and N leaching risk and number of plant species on the other hand. However, only very common species were found in the monitored grasslands and thus the botanical conservation value was low. This indicates that grassland management extensification does not guarantee botanical restoration. Despite the significant relationships between LUI and ES and biodiversity indicators, own data showed that other management factors play a role as well. Timing of the first cut or grazing appeared to have an effect, especially on forage quality, soil mineral N content and number of plant species. In particular, delaying the first cut strongly reduced crude protein concentration, increased soil N content and increased number of plant species.

6.3 Farm-level impact

In chapters 2 and 3 we quantified ecosystem service delivery and biodiversity effects of hedgerow and grass strip implementation at parcel level. In order to assess multifunctionality and trade-offs also at farm level, we simulate the impact for a virtual, for Flanders typical arable farm. This virtual farm was similar to the one described in chapter 5 and was composed of seven parcels of winter wheat (2.38 ha per parcel), two parcels of winter barley (1.97 ha per parcel), seven parcels of maize (1.64 ha per parcel), two parcels of sugar beets (2.55 ha per parcel) and four parcels of potatoes (2.46 ha per parcel). Parcel size was based on the average parcel size per crop in Flanders. We assumed every parcel to be square, allowing a straightforward calculation of parcel length and width. For a set of scenarios (Table 6.3), we quantified ecosystem service delivery and biodiversity. Scenarios were developed to meet the 5% EFA area requirement at farm level. In the first three scenarios, the EFA consists entirely of hedgerows. One scenario (HR_big) is similar to the scenario that was developed in chapter 5. In the final three scenarios, the EFA is composed of grass strips. The total farm area is 47.04 ha, so in order to meet the greening requirements, the EFA should have a minimum area of 2.35 ha. In the hedgerow scenarios, the required hedgerow length depends on its width: when the hedgerow is more narrow, the hedgerow should be longer to reach the required area. Total hedgerow length was allocated to the parcels, and the allocation was proportionate to the share of the corresponding crop. This resulted in a set of parcels bordered by a hedgerow (N° +) and a set of parcels without a hedgerow (N° -) (Table 6.3). In every grass strip scenario, grass strip length is 2 613 m, as the standard grass strip width in the EFA prescriptions is 6 m, and any increase in width is not taken into account as EFA. For this reason, the number of parcels with a grass strip and without a grass strip does not vary over the different scenarios.

Table 6.3: Overview of the case study scenarios. In the HR scenarios, only hedgerows were implemented as EFAs. In the GS scenarios, the EFA is composed of grass strips. N°+ stands for the number of parcels with a hedgerow or grass strip on one of the borders, N°- is the number of parcels without any hedgerow or grass strip.

Scenario	Hedgerow/grass strip width (m)	Hedgerow height (m)	Hedgerow/grass strip length (m)	N° +	N° -
HR_small	3.75	5	3136	22	0
HR_medium	5	7.5	2352	16	6
HR_big	7.5	10	1568	11	11
GS_small	6	NR	2613	18	4
GS_medium	9	NR	2613	18	4
GS_big	12	NR	2613	18	4

Calculation of farm level ecosystem service delivery and biodiversity change was done by combining parcels. The calculation tool that we developed in chapter 5 was used to determine total farm production and farm income. To describe hedgerow impact on crop yield and thus farm income, a maximum and minimum effect were calculated by considering only a yield reduction (maximum effect) next to the hedgerow and both a yield reduction and a yield increase (a minimum effect). To consider only a yield reduction, the effect relationship describing relative crop yield influenced by a hedgerow (developed in chapter 2) was applied between H=0 and H=2.1, which is the negatively affected yield zone. To consider both a negative and a positive yield change, the full range of H-values was considered (Figure 6.1).

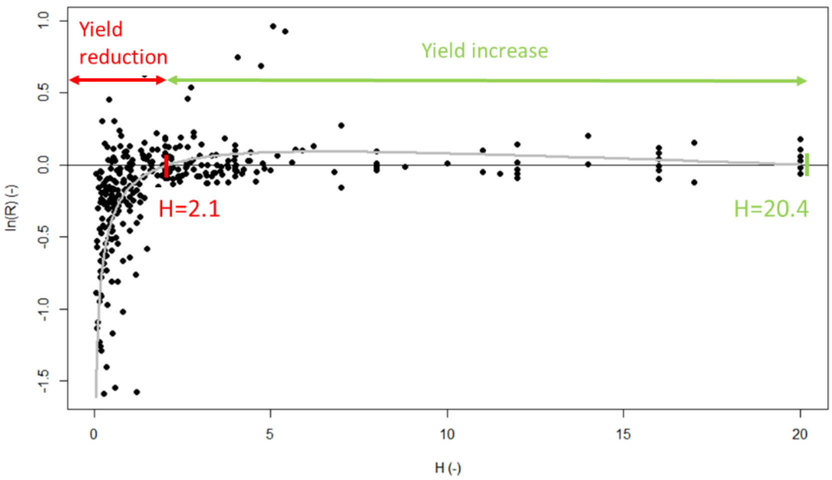


Figure 6.1: relative crop yield (R) as affected by relative distance (H) from the hedgerow. Near the hedgerow (until H=2.1), crop yield is reduced. Beyond H=2.1 until H=20.4, crop yield is increased.

In order to quantify the SOC stocks and N, P and erosion interception rates of every scenario, the following assumptions were made: i) initial arable SOC stock in the 0-30 cm soil layer was 50 ton ha^{-1} (Reubens et al., 2010), ii) total N and P losses via surface water from arable land were 26.19 and 1.86 kg ha^{-1} respectively (MIRA, 2015), iii) total N losses via subsurface water (0-90 cm) from arable land were 78 kg ha^{-1} (De Waele et al., 2017) and iv) loss of soil particles caused by erosion was 10 ton ha^{-1} (Vlaamse Overheid, 2011). Assumptions for SOC stock and N and P losses resulted from averaging of data collected in Flanders. The estimation of erosion, however, was based on the erosion level that was perceived as problematic to maintain soil quality. By consequence, actual erosion rates and thus absolute hedgerow and grass strip interception will often be lower.

Ecosystem service and biodiversity indicators of the hedgerow and grass strip scenarios applied on the virtual farm are compared with a business-as-usual (BAU) scenario, in which no hedgerows or grass strips are present on the farm. In the BAU scenario, crop yield is unaffected and the initial assumptions on SOC stock, N and P losses and erosion are applied on the total farm area.

When interpreting the results, it is important to take into account that the effect relationships were based on data from hedgerows and grass strips that have been there for several years. In terms of SOC stock, this means that the estimated effect is the result of multi-year accumulation. As age was not a significant predictor for hedgerow or grass strip impact on SOC stock, this factor was not taken into account in the effect relationships (see chapter 2). Also for N and P interception and erosion reduction, we expect that the impact immediately after the implementation of the measures will be lower compared to the predicted effect. More specifically, this means that 1) the predicted change in SOC stock results from SOC accumulation since the implementation of the hedgerow or grass strip while crop yield changes and N and P loss and erosion reductions are yearly trends and 2) during the first years after hedgerow or grass strip implementation, it is likely that the predicted effects will not be achieved.

When comparing the hedgerow scenarios (Figure 6.2), farm income is the lowest when medium-sized hedgerows are implemented as EFA. This is due to the combination of a lower required amount of parcels with a hedgerow (compared to the HR_small scenario) and to the share of yield increase. The biggest income loss (HR_big) amounts to 6014 euro per year or 9% of the total farm income. We did not include hedgerow wood as a revenue, because in Flanders, hedgerow maintenance costs are barely compensated by the wood revenues. When taking into account EFA payments (100 euro ha^{-1}, thus 4704 euro at farm level), the income loss is fully compensated when a minimum effect is considered. In the case of maximum effect, EFA payments cover the income losses in HR_small scenario but not in the HR_medium and HR_big scenario. Hedgerow implementation results in every scenario in a SOC stock accumulation of about 32 ton. The amount of N, P and erosion that was intercepted from the surface water flow was the highest in the HR_small scenario, due to a higher number of parcels bordered by a hedgerow. These results, which represent the most optimal situation, are based on the assumption that on the parcels with a hedgerow, all surface water flows are directed through the hedgerow. In the meta-analysis, we found no significant effect of hedgerow presence on the number of natural predators, and thus we conclude that there will be no effect on farm level either. As the meta-analysis indicated a higher number of natural predator species on parcels with a hedgerow, farm level calculations show a higher increase of predator species numbers in the scenario with the highest amount of hedgerow-bordered parcels (HR_small).

Figure 6.3 represents the ecosystem service and biodiversity indicators of the three grass strip scenarios on the virtual farm compared to the BAU scenario. Because the required length of the grass strip is fixed, farm income losses increase with increasing grass strip widths. The farm income loss varies between 2070 and 4139 euro per year and thus the EFA payments (4704 euro $year^{-1}$) towards the farm fully compensate the income reduction as a result of arable land loss. Because there is no trade-off between grass strip width and the number of parcels with a grass strip, wider grass strips results in increasing SOC stocks and more N, P and erosion interception. The number of parcels with a grass strip was the same in all scenarios and thus the farm level predator activity-density and diversity did not vary among the scenarios.

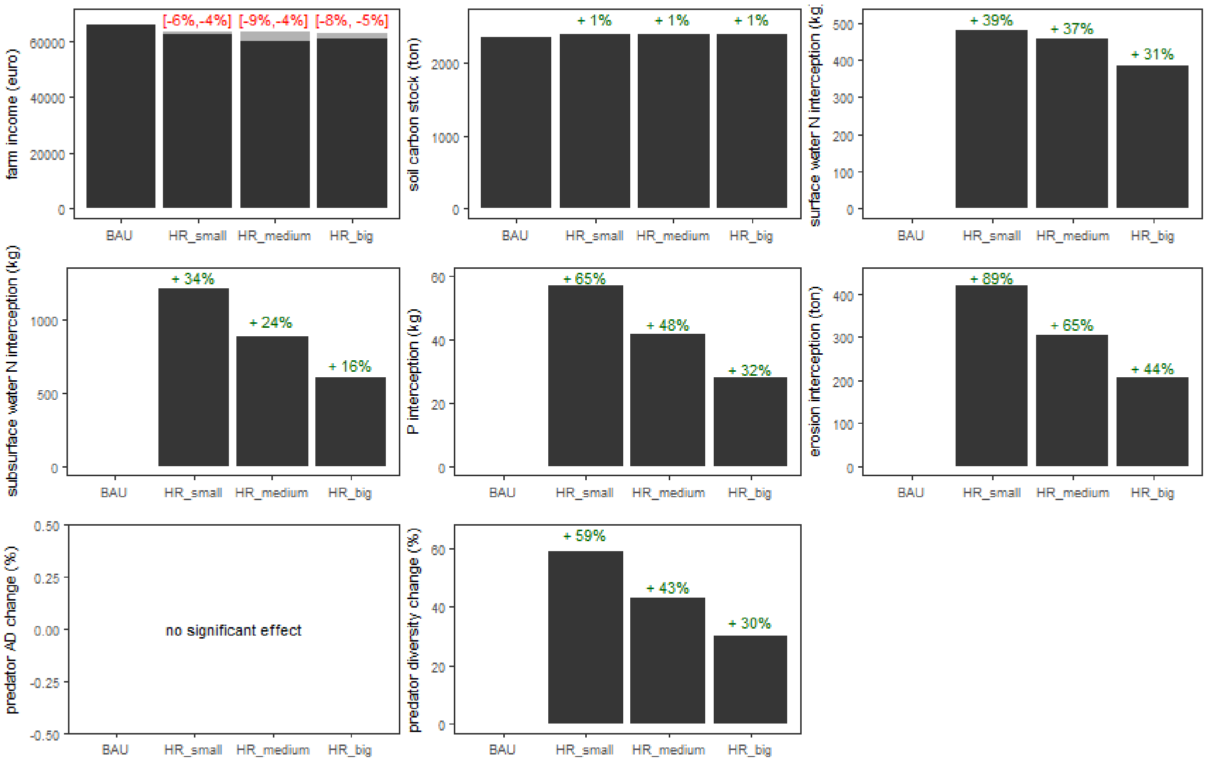


Figure 6.2: farm level ecosystem service and biodiversity indicators for three scenarios with hedgerows (HR) with varying widths (HR_small, HR_medium, HR_big) as EFAs. Response variables are compared to the BAU scenario, in which no hedgerows are implemented on the farm. Farm income estimated both on i) only a negative impact on crop yield and ii) both a negative and a positive impact on crop yield. AD stands for activity-density.

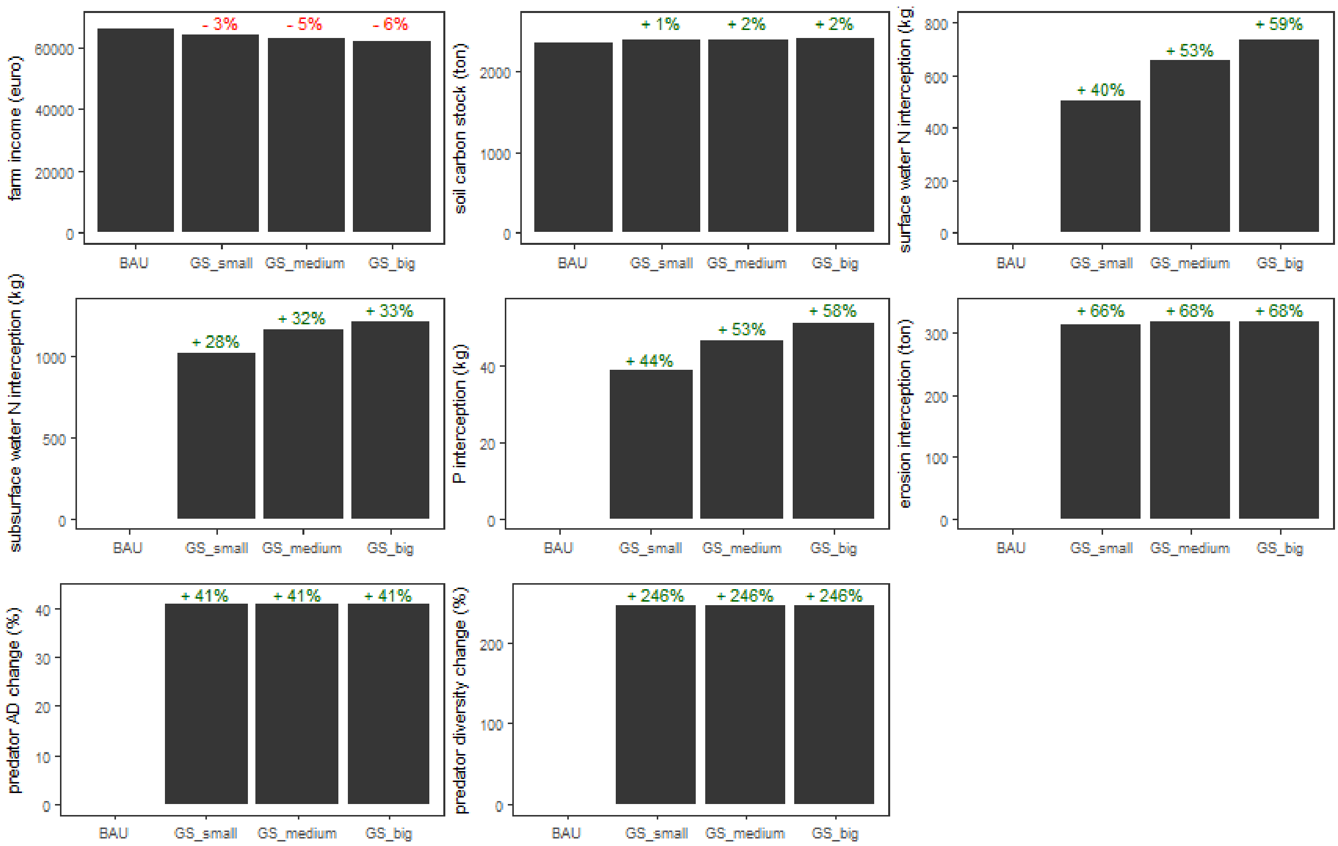


Figure 6.3: farm level ecosystem service and biodiversity indicators for three scenarios with grass strips (GS) with varying sizes (GS_small, GS_medium, GS_big) as EFAs. Response variables are compared to the BAU scenario, in which nog grass strips are implemented on the farm. AD stands for activity-density.

This case study illustrates that both hedgerows and grass strips have the potential to increase ecosystem service delivery and biodiversity at farm level, but at the same time reduce farm income when by-products such as wood chips or grassy biomass are not valorised. However, trade-offs can be minimized and depending on local priorities, implementation and management strategies can be optimized. For example, the implementation of more and smaller hedgerows has less impact on farm income and simultaneously enhances ecosystem services and biodiversity more than fewer and bigger hedgerows do. Also, increasing grass strip width equally increases relative farm income loss and SOC stock, but the contribution to N, P and erosion interception increases at a lesser rate. Therefore, when a high reduction of nutrients in the surface or subsurface water is needed, grass strip width will have to increase exponentially, resulting in high income losses. In this case, it may be preferable to adapt the parcel management in terms of fertilization and ploughing instead of widening the grass strips.

Due to the EFA requirements, the hedgerow and grass strip scenarios show opposite trends when width of the hedgerow/grass strip increases. For hedgerows, it seems that the implementation of smaller but more hedgerows increases both provisioning and regulating ecosystem services more at farm level compared to the implementation of bigger and less hedgerows. Because only grass strip length is taken into account for EFA calculation and thus grass strip length does not decrease when width increases, wider grass strips enhance ecosystem service delivery and biodiversity more than narrow grass strips.

Our calculations show that the potential of hedgerows and grass strips for multifunctionality at farm level is considerable and that income loss depends on hedgerow and grass strip configuration. At the same time, the development of the scenarios was strongly driven by EFA requirements and not by for example trade-off minimization. Also, our research can help to provide a framework for the exploration of a more effective payment system (both in the EFA and AES context) that takes into account the wide range of benefits that comes along with semi-natural elements in the agricultural landscape. However, we acknowledge that our research only provides a first step towards a holistic assessment of multifunctionality at farm level, and several ecosystem services as well as alternative valorisation pathways for by-products generated by the measures were not considered.

6.4 Relevance on regional and global level

Within the parcel, implementation of the hedgerows and grass strips significantly increased SOC stock, corresponding to 1.60 ton ha^{-1} year^{-1} of carbon sequestration in the grass strip and 0.29 ton ha^{-1} year^{-1} within 1 m from the hedgerow. Two other widely applied agricultural practices to increase SOC are the use of cover crops and reduced/no-tillage (Nelissen et al., 2016). Via a meta-analysis, Poeplau and Don (2015) found a positive contribution of 0.32 ton ha^{-1} year^{-1} of cover crops to SOC. The effect of reduced/no-tillage is variable. Studies have found SOC increases in the upper (0-10 cm) soil layer and a reverse effect in the deeper soil layers (10-30 cm), resulting in no net change in SOC (D'Hose et al., 2016). Similarly, the review of Merante et al. (2017) reported cases with no effect of reduced/no-tillage on SOC and cases with positive contributions up to 0.45 ton ha^{-1} year^{-1}. Compared to cover crops and reduced/no-tillage, the contribution of grass strips to SOC increase seems high. However, because grass strips are only implemented along field borders, whilst cover crops and reduced/no-tillage are applied over the whole field, parcel-level contribution of grass strips to SOC stock is rather limited. Consequently, the contribution of grass strips to global climate regulation will be very limited. Although carbon sequestration in both aboveground and belowground biomass was not considered, the same can be said for hedgerows and because of their limited area, their contribution to global climate regulation will be small.

In chapter 2, it was shown that both grass strips and hedgerows have a great potential to intercept N, P and TSS from the water flow, resulting in a significant contribution to the regulation of water quality and erosion. This conclusion is, however, based on the assumption that all water flows are directed through the grass strips and hedgerows and that there is no by-passing. Nevertheless, in reality, this will depend on, among other, the slope of the parcel and drainage. Therefore, in order to realise the nutrient and TSS interception potential of grass strips and hedgerows, catchment level hydrology and landscape need to be considered (Schoumans et al., 2014).

Increased activity-density and diversity of natural predators may increase pest control, reduce pest damage and thus increase crop yield. Whether the (increased) presence of natural predators actually results in enhanced pest control depends on several other factors such as insecticide use, timing of the pest outbreak and field colonization by the predators. Often, the use of insecticides minimizes both pest and predator populations, so the potential benefits of natural pest control are not exploited. Insecticide use safeguards crop yields, but it also entails additional production costs, a negative environmental impact and is a threat for biodiversity in the agricultural landscape. To enhance natural pest control, reduction of insecticide use on a regional level would be needed in order to avoid too severe damage to predator populations, requiring collaboration of various farmers. Current scientific knowledge on the linkages between the implementation of nature-oriented measures, presence of natural predators and actual pest control is very limited, but it is unlikely that natural pest control can fully replace insecticides. However, natural pest control may result in a lower insecticide use, when it is used in the context of Integrated Pest Management, for example by lowering the required dose, postponing the first application of insecticides or by raising the critical threshold level of pest species presence (Bruyn, 2014; European Commission, 2018).

6.5 Potential for further research

6.5.1 Widening the range of services and response variables

As described in chapter 1, a wide range of ecosystem services can be obtained from an agricultural landscape. This research has focused on the delivery of provisioning and regulating services and biodiversity enhancement by nature-oriented measures. We have described the impact of hedgerows, grass strips and grassland management on crop yield and quality, on SOC stocks, N and P interception, erosion regulation, potential for pest control and predator and plant diversity. However, nature-oriented measures have the potential to contribute to a broader set of provisioning ecosystem services than those considered in our study. For example, hedgerows also produce wood that can be used for bioenergy, timber, mulch, litter or composting (Viaene et al., 2016). Also, the set of relevant regulating services can be expanded. For instance, flowering species, both in hedgerows, grass strips and grasslands, may enhance the presence of pollinators (Marshall et al., 2006; Morandin et al., 2016) and grass strips can reduce insecticide runoff (Arora et al., 2003; Patty et al., 1997). Additionally, apart from increasing SOC, hedgerows also store carbon in their belowground and aboveground biomass. For example, Falloon et al. (2004) estimated the aboveground biomass carbon accumulation rate at 1 ton C ha^{-1} $year^{-1}$.

Although they directly affect human life and well-being, cultural ecosystem services were out of scope of this research. Cultural ecosystem services produce for example recreation, aesthetic and scientific benefits, cultural heritage and identity (see Table 1.1) (Díaz et al., 2015). As for almost all ecosystem services, the demand for cultural services is rising and this increase is even stronger than for regulating and provisioning services (Guo et al., 2010). In a study from the Netherlands, where the agricultural landscape has a lot of similarities with Flanders, the aesthetic beauty, potential for recreation and the cultural heritage were named as the most important cultural ecosystem services delivered by the agricultural landscape. Respondents designated a complex, natural mosaic landscape as the most preferred landscape type. Especially the combination of forest patches with tree lines and hedgerows was highly rated (van Berkel and Verburg, 2014).

In the IPBES framework, non-material contributions of ecosystems to humans go beyond the CICES definition of cultural services and are classified as instrumental, relational and intrinsic values. Instrumental values are the benefits that can be obtained from ecosystems, including both physical outputs and recreational, cultural and/or spiritual experiences. This type of value corresponds to the concept of ES as it was defined in CICES and as it was used throughout this study. Relational values are found in the relationships between humans and the ecosystem and among humans. The implementation of nature-oriented measures can produce relational values, for example by contributing to the farmers' identity (see also section 5.5.1). The last category encompasses intrinsic values. These are the values inherent to nature, independent from humans (Díaz et al., 2015; IPBES, 2015).

In order to better demonstrate the potential added value and trade-offs of nature-oriented measures, the set of considered variables should be expanded and more provisioning and regulating ecosystem services should be included, and cultural ecosystem services need to be introduced. Also, different value types and thus valuation methods need to be integrated in the assessment of potential of nature-oriented measures. (Jacobs et al., 2016). In chapter 5, it was shown that in almost all cases, financial compensations via subsidies covered the income losses resulting from the implementation of hedgerows or alley cropping. And still, adoption of these measures in Flanders remains low. Burton et al. (2008) explored the reasons behind the gap between economic incentives and farmers' response to AES and they concluded that the work related to the AES does not generate any social or cultural capital (and only economic capital). Social capital arises from networks with other people (in this case mostly farmers) and cultural capital is related to the pride and prestige that come with the agricultural practices.

To conclude, in this study we intended to develop a first stepping stone for the future development of a fully integrated valuation of the potential of multifunctionality in agriculture. It needs to be stressed that the current focus on a subset of provisioning and regulating ES is a first necessary step forward but should be completed with e.g. cultural ES and relational values in order to create a full assessment framework in line with the complex reality.

6.5.2 Ecosystem service monitoring

Despite the great potential of nature-oriented measures for ecosystem service delivery and biodiversity (chapter 2 and chapter 4), we have found that actual benefits can vary significantly across parcels (chapter 3). Therefore, it remains difficult to precisely predict the effects of the implementation of nature-oriented measures in every specific situation and monitoring in the field may be necessary. However, field monitoring is often resource and time consuming and thus simultaneously monitoring of multiple response variables is not straightforward. This imposes restrictions on the development of a generally applicable monitoring protocol. For this reason, it may be preferable to include the effects of local management and parcel characteristics in the ecosystem service and biodiversity prediction models, allowing an uncomplicated and relatively simple prediction of ecosystem service and biodiversity effects. This would be a fruitful area for further research: if we want to optimize the implementation of nature-oriented measures, a better understanding of the contribution of local management and parcel characteristics needs to be developed. We discussed the effect of ditch clearing and sludge deposit on grass strip performance, but for example mowing regime of the grass strips is expected to have an effect on various ecosystem services and biodiversity aspects as well (Badenhausser and Cordeau, 2012; Uusi-Kämppä and Jauhiainen, 2010b).

6.5.3 Upscaling

Next to parcel-level influences, the delivery of ecosystem services and biodiversity will be affected at the landscape scale. More concrete, agricultural landscape intensification generally has a negative impact on biodiversity and biodiversity-related ecosystem services like pest control and pollination. Landscape intensifications consists of, among others, a decline in the number of crops (species) and the reduction or removal of permanent field edges (Tscharntke et al., 2005). A lot of research has been performed on the link between landscape composition and pest control and pollination (F. J. J. a Bianchi et al., 2006; Cranmer et al., 2012; Rusch et al., 2013). Also, it has been shown that measures that are implemented to enhance pollination in the agricultural landscape will contribute to the delivery of other regulating ecosystem services such as an improved water quality and carbon accumulation (Wratten et al., 2012). More general, simplified landscapes will have less multifunctionality than more complex landscapes and to restore landscape functioning, management actions beyond farm-level scale are necessary (Landis, 2017). However, to date, little attention has been paid to the landscape-scale optimisation of multiple ecosystem service delivery and biodiversity and tools to assess and evaluate landscape multifunctionality are lacking (Landis, 2017). Finally, an efficient and effective landscape design requires the consideration of socio-economic factors and engagement of stakeholders is crucial. To do so, the development of network and participatory process guidelines is needed (Geertsema et al., 2016).

6.5.4 Potential for existing models

The concept of ES and integrated valuation is gaining importance for landscape planning, scenario development, conflict resolution etc. (Jacobs et al., 2016; Tscharntke et al., 2005). Very often, these projects require the quantification of ES delivery on a medium or large scale. In this context, several tools have been developed in Flanders. The ‘Natuurwaardeverkenner’ (Liekens et al., 2013) allows to value ES delivery of different types of land use on a qualitative, quantitative or monetary level (depending on the information available). This tool was, among others, used to calculate the benefits provided by the Natura 2000 network in Flanders (Broekx et al., 2013). The ECOPLAN project (Staes et al., 2017) consists of several tools allowing for the identification of relevant stakeholders, ES, for the development of various land use scenarios and corresponding impacts and trade-offs, etc. Both Natuurwaardeverkenner and ECOPLAN make use of land use change scenarios and do not take into account nuances or nature-oriented measures taken within the same land use type. However, an agricultural landscape can deliver a variable set of ES, depending on land management, for example the implementation of nature-oriented measures. Therefore, results of this research can be used to fine-tune and incorporate land management into existing models.

6.6 Management and policy recommendations

6.6.1 Trade-offs between regulating ES and biodiversity

Results of the meta-analyses (chapter 2) and literature review (chapter 4) suggest that hedgerows, grass strips and extensive grassland management have the potential to deliver multiple regulating ES and enhance biodiversity in the agricultural landscape. Field observations on Flemish fields confirmed the enhancement of several ES and biodiversity aspects, but also revealed that the simultaneous delivery of a broad range of ES is not always the case and that there are trade-offs in the extent to which some services can be provided simultaneously (chapter 3 and chapter 4). These trade-offs are strongly linked with management and context.

More specifically, ploughing next to the hedgerow enhances carbon mineralisation and hampers hedgerow root growth in the upper soil layers, impeding both an increase of SOC stock and nutrient removal from the water flow. Also, mowing of grass strips will encourage nutrient uptake and removal, contributing to the regulation of chemical water quality, but will simultaneously increase disturbance in the grass strip, which may have a negative effect on natural predator abundance and diversity. Further, input of nutrients into the grass strip through water transport from the arable field will limit botanic diversity in the grass strips and deposit of sludge may enhance SOC stock but simultaneously increase soil P concentration. In grasslands, the application of high levels of manure may increase SOC stock, but simultaneously increases soil N content, thus impeding the regulation of chemical water quality and botanical diversity.

Therefore, in order to increase effectiveness of nature-oriented measures the delivery of regulating ES and biodiversity, specific management schemes, targeting pre-set priorities, are required. Maximization of one (or a bundle of related) ES or biodiversity aspects might in some cases be at the expense of other benefits.

6.6.2 EFA compensations

In order to comply with the CAP greening requirements, Ecological Focus Areas (EFAs) have to be implemented on 5% of the total arable area of those farms having more than 15 ha of arable land. In 2015, 8 million ha of land was declared as EFA, corresponding with 13% of the European arable land falling under the greening requirements. Of this total EFA surface, 96% was composed of nitrogen-fixing crops (38%), catch crops (33%) and land lying fallow (26%), all temporally allocations of productive arable land (European Commision, 2017). In Flanders, 130 000 ha was declared as EFA, spread over 9 200 farms and 85% of all EFAs were nitrogen fixing crops (VILT, 2016). As mentioned in chapter 1, it is very likely that these temporally elements will have little or no contribution to the CAP's goal of putting an end to current environmental degradation and biodiversity loss caused by agricultural activities. Our research has shown that other, permanent EFA options (hedgerows and grass strips) have the potential to enhance both ecosystem service delivery and biodiversity. Given the low adoption rate of these measures, it seems that the EFA regulations need to be improved and more specifically, the weighting and conversion factors should be strengthened. Also, specifically for grass strips, the EFA requirements are not in line with the expected benefits: whereas we have shown that increasing grass strip width will improve N and P interception and erosion regulation and reduce parcel-level yield, this is not taken into account and the calculation of EFA area is only based on grass strip length. Finally, we showed positive effects of both hedgerows and grass strips on multiple ecosystem services, but both the data from the meta-analysis and our own monitoring were derived from hedgerows and grass strips that have been there for several years. During the first years after hedgerow/grass strip implementation, more limited effects may be expected. Therefore, EFA requirements or payments should be linked to the age of the hedgerows and grass strips, stimulating the long-term conservation of the measures.

6.6.3 Link with AES

The implementation or management of hedgerows, grass strips and species-rich (botanical) grassland are part of the current set of AES (VLM, 2018b). In Flanders, hedgerows can be planted on any field border and they are promoted as a habitat and corridor for various animal and plant species, because of their potential to improve microclimate, to offer shelter to cattle and to reduce erosion. Our research has shown that hedgerows may also increase SOC stock and intercept N and P from the water flow. However, the implementation of hedgerows does not guarantee a positive impact under all circumstances. For instance, both the reduction of erosion as the interception of nutrients strongly depends on the topography, direction of the water flow and local hydrology.

Grass strips are implemented in Flanders along vulnerable landscape elements (e.g. water courses), buffering them from the input of fertilizers, insecticides and from damage due to soil cultivation. Additionally, the application of adapted mowing is claimed to promote the development of a vegetation of high botanical value. Minimum width of grass strips in the AES is 5 meters. Based on the effect relationships developed in chapter 2, a grass strip of 5 m wide can intercept a considerable share of the N, P and erosion from the water flow. Similarly to hedgerows, this assumes that the water flow goes through the grass strip and that there is no bypassing. Trade-offs need to be considered. Because nutrient input impedes floral diversity (de Schrijver et al., 2011; Schelfhout et al., 2015), it seems unlikely that a grass strip intercepting N and P will develop a botanically valuable vegetation, despite of restricted mowing. Moreover, because intensive mowing will presumably increase nutrient uptake, it seems that there is a trade-off between the buffering function of grass strips and their potential to develop valuable botanical diversity. Also, grass strip mowing increases disturbance of the habitats of the natural predators and most multiannual grass strips lack the flowering plants that provide the pollen that are important for predators as an additional food resource. Also management nearby grass strips is important. For instance, on the grass strips that were monitored, sludge deposited on the strip was probably increasing soil P concentrations, thus hampering the positive contribution of grass strips to the regulation of water quality.
Optimization of hedgerow and grass strip measures thus involves selection of the required and/or prioritized ES, consideration of the landscape and context and finally development of an appropriate management scheme. More specifically, both hedgerows and grass strips may intercept nutrients and erosion from the water flow if there is no by-passing. In the case of hedgerows, this potential can be enhanced by the presence of an unfertilized, unsprayed grass strip next to the hedgerow. Additionally, the wider the hedgerows or the grass strips, the higher the nutrient and erosion interception. Both for hedgerows as for grass strips, it seems not feasible to combine biodiversity-related targets with the interception of nutrients.

Development of species-rich grassland (as an AES) is in Flanders only possible in certain areas and management of species-rich grassland is possible (as an AES) on grasslands that already have a high conservation value. In these AES, application of fertilizers and insecticides is not allowed and the mowing and grazing regime is restricted. The grasslands that were monitored in TVG were located in an area that was indicated as appropriate for the development of a species-rich grassland and the number of plant species was higher in the grasslands with no fertilizer or pesticide application. However, we also found that the botanical conservation value was low, indicating that the realization of a botanically valuable grassland may be hard to reach or may take a very long time.

6.6.4 Towards multifunctionality in agriculture

To meet the rising and broadening demands, both from policy as from society, and in order to maintain or enhance its viability, a change in agriculture is needed, and this reform should benefit both farmers and non-farmers. This reform will only be successful if multiple ecosystem services and values obtained by the various stakeholders on a local, regional and global level are considered (IPBES, 2015; Jacobs et al., 2016). The full range of potential ES delivery and trade-offs among various ES bundles and/or biodiversity needs to be known in order to assess the impact of land management or land use change scenarios. Once the desired scenario has been identified, according nature-oriented measures should be promoted. Land management and land use scenarios are preferably developed on a regional scale (see section 6.4 and section 6.5.3). Financial incentives should be sufficient to overcome economic drawbacks related to the implementation of nature-oriented measures, but this will not suffice and the role of the farmer needs to be altered on various levels. On a personal level, most farmers acknowledge their role as a landscape manager, but interpretations vary among individuals. While some value their contribution to the management of semi-natural elements or ecological and/or environmental improvement, others take pride in so-called tidy landscapes as a proof of skilled farming (Burton, 2012; Burton et al., 2008; Ryckebusch, 2017). On a regional level, interactions with other farmers and more generally other land users contribute to the perception by the farmer. In order to make nature-oriented management contribute to the farmers' professional identity, efforts of all stakeholders are needed, including farmers associations, agri-food chains, banks, and governments and cooperation between scientists, landscape planners and farmers should be strengthened. On a national and a European level, incentives should promote agricultural practices that enhance environmental quality and biodiversity in the agricultural landscape without restricting farmers' flexibility, independence and freedom. This can be achieved by introducing a shift from area-based measures with imposed management instructions towards result-based measures. This may enhance both the ecological benefits that are obtained via the measures and the contribution of the measure-related-work to the farmers' professional and personal identity, thus improving uptake of the measures.

Legal uncertainty is also one of the main drawbacks for the adoption of agroforestry (Borremans et al., 2018; Runhaar, 2017) and some of the farmers that participated in the field monitoring in this study expressed the concern that the hedgerows or grass strips they managed would be assigned a permanent character or that management prescriptions would be strengthened. In fact, some of the grass strips already were assigned the status of permanent grassland.

Finally, acknowledging multifunctionality in agriculture may contribute to an increased resilience of the sector, for example via the diversification of income (via subsidies or other financial compensations), certification, short-chain supply etc.

6.6.5 Land sharing vs land sparing

Our study contributes to the 'land sparing vs land sharing' discussion (Kremen, 2015; Phalan et al., 2011; Salles et al., 2017). Land sparing entails the spatial separation of intensive agriculture for food, forage and livestock production on the one hand and biodiversity conservation on the other hand. Land sharing involves the implementation of nature-oriented measures in the agricultural landscape, allowing the integration of agriculture and biodiversity conservation. It is striking that in this discussion on optimal landscape design, only provisioning ecosystem services and biodiversity are considered, while the agricultural landscape has the potential to deliver a wide range of regulating and cultural ecosystem services as well.

From our results, we conclude that the selected nature-oriented measures can increase both alpha and beta diversity but at the same time have little conservation value, as they particularly promote generalist species. This outcome strengthens the land sparing argument, which states that biodiversity conservation is hampered by most human influences or disturbance and thus that in order to have both agricultural production and biodiversity, both land use types should be separated. When, however, the discussion is extended to a more holistic approach and when regulating and cultural ecosystem services are taken into account, it is evident that land sharing becomes at least equally valuable. Moreover, it seems that land sparing has little chance of succeeding without land sharing: an attractive and complex agricultural landscape will offer recreational opportunities and at the same time buffer the negative impacts of agriculture on the environment. When the agricultural landscape is intensified, both the recreational as the environmental pressure on the remaining natural areas will increase considerably, immediately undermining the success of the land sparing approach. Finally, it may not be overlooked that both approaches act on different scales: while land sharing is mostly important on a local and landscape scale, land sparing is relevant at a regional scale. Thus, land sharing is embedded in land sparing (Ekroos et al., 2017). Therefore, we conclude that multifunctional and sustainable landscape planning entails both land sparing and land sharing and that decision-making and management should be a multi-level process.

6.6.6 Recommendations for policy

During the legislative process and debates in run-up to the CAP reform, the original greening ambitions were gradually weakened. For example, the required EFA area was screwed back from 7% to 5% of the arable land and the list of potential EFAs was supplemented with production-related elements such as nitrogen-fixing crops and catch crops. Already before the actual implementation of the reformed CAP, researchers expressed their doubts about its efficiency and the results that could be expected in terms of environmental and biodiversity benefits (Matthews, 2013; Westhoek et al., 2012). Whereas the European Commission has not yet evaluated the latest CAP reform, BirdLife Europe and the European Environmental Bureau have done so. They concluded that generally, the nature-oriented measures that are included in the CAP (both in Pillar I and in Pillar II) are ineffective, due to their low uptake and lack of appropriate design both at farm and at regional level. More concrete, the greening requirements, among which the EFAs, are expected to contribute very little to both the reverse of biodiversity loss and of environmental degradation, mainly as a result of exemptions, the inclusion of non-effective EFA options and poor design of the required measures (Pe’er et al., 2017).

Our research has shown that hedgerows and grass strips on arable field margins and extensive grassland management have the potential to contribute to the achievement of the biodiversity and environmental goals of the CAP. This requires both the stimulation of these measures by policy and guidelines for farmers and land managers for an appropriate and effective implementation and maintenance.

Current payments for the nature-oriented measures are only linked to expected income loss. Based on the current implementation rates, this seems inappropriate to stimulate the uptake of nature-oriented measures, indicating that the payment level should be increased. The way forward seems to be by the introduction of result-based payments, for example via payment for ecosystem services (Bellver-Domingo et al., 2016). A major drawback in this method is the estimation of actual results. We have shown that despite of the great potential for ecosystem service delivery and biodiversity, there is great variability in the parcel-level effects that we measured. To develop efficient and cost-effective result-based payments, a detailed and parcel-level estimation of the effects is needed. This requires a substantiate and well-thought monitoring campaign, which imposes restrictions on the general feasibility of simultaneously measuring multiple response variables. An alternative may be to develop insight into the causes of parcel-level variability, allowing an accurate prediction of ecosystem service delivery and biodiversity. We have shown that for hedgerows and grass strips, width, height (only for hedgerows), management of the adjacent arable parcel (for example ploughing) and management of the ditch affects the benefits of the measures. For grassland, management intensity, based on fertilization, mowing and grazing, allows prediction of ecosystem services and biodiversity indicators, but the first date of mowing and parcel characteristics such as soil P concentration should be taken into account as well. Incorporating these factors in ecosystem service and biodiversity modelling may improve accuracy of the predictions.

Additionally, EFA and other greening requirements should include measure characteristics that are linked to enhanced ecosystem service delivery and biodiversity. For example, wider grass strips contribute more to N and P interception and erosion reduction. However, when a grass strip is wider than 6 m, this additional area is not acknowledged as EFA. Where it is justifiable to set a minimum width for grass strips, a maximum width of 6 m seems not. We have shown that until a grass strip width of 20 m, nutrient interception and erosion reduction increase. Also, greater benefits may be expected if the timespan of the measure increases, suggesting that compensations should be higher for long-term management of the measures. In this context, our research can contribute to the development of a detailed insight into the causes of parcel-level variability, allowing an accurate prediction of ecosystem service delivery and biodiversity and to the establishment of guidelines for optimal implementation and maintenance of the nature-oriented measures.
More generally, if the biodiversity loss and environmental degradation caused by agricultural practices is to be halted, actions at various levels and by different actors are needed and the following recommendations can be formulated:

Scale	Actions needed	Actors
Measure	Optimal **design and maintenance** of the nature-oriented measures *Example: in order to increase SOC stock of grass strips and grasslands, ploughing should be avoided*	**Farmers** need clear design and management guidelines, provided by **scientists and regional landscape planners**. Additionally, feedback loops from **farmers** to landscape planners should be facilitated, as their on-field experience may enhance optimal design and management.
Parcel	Implementation and management of the measures should **minimize trade-offs** between provisioning ecosystem services, regulating ecosystem services and biodiversity. *Example: adjustment of hedgerow height can maximize the beneficial micro-climate effect and minimize negative shading impact on crop yield*	The measure implementation process should involve both **farmers, landscape planners and scientists**.
Farm	Management of nature-oriented measures should become a **proper part of farm work**. *Example: in most scenarios, farm income losses were compensated by EFA payments. This did not result in a considerable uptake of the measures, indicating that there are other (than financial) drawbacks, for example farmers' perception and attitude.*	**Policy makers, farmers associations, agri-food chains, NGOs and society** need to acknowledge and promote the multifunctionality of agriculture, so that **farmers** can link their professional identity to implementation of nature-oriented measures.

Regional	**Products and by-products** of the measures should be appropriately valorised *Example: if the silage of extensive grasslands does not contain any poisonous plants, it may be used to feed sheep or beef cattle instead of dairy cows*	**Farmers associations, individual farmers and regional landscape organizations** should setup a network to simplify collaboration and to support the use and processing of by-products. **Policy** should promote these networks and facilitate collaborations.
	The contribution of farmers to **landscape management** should be stronger acknowledged *Example: a complex agricultural landscape provides more cultural ecosystem services than a monoculture agricultural landscape*	The management of nature-oriented measures should be perceived as proper farm work, both in the attitude of individual farmers and in social networks among farmers, as in the viewpoint of other land users like tourists. This requires participation of **farmers associations, individual farmers, regional landscape organizations, the touristic sector, policy and citizens.**
	Regional landscape planning is necessary to maximize ecosystem service delivery and biodiversity by nature-oriented measures. *Example: grass strips contribute to the regulation of water quality if the dominant water flow is through the grass strip and if there is no by-passing*	**Farmers, regional landscape organizations, NGOs and all other stakeholders** should work together to develop an optimal landscape design and plans of various landscape organizations should be aligned. **Policy** should promote these networks and facilitate collaborations.
	Regional planning may allow a **minimization of trade-offs**. *Example: extensification of grasslands on less productive soils will have a limited impact on income losses compared to highly productive grasslands*	**Farmers, regional landscape organizations, NGOs and all other stakeholders** should work together to develop an optimal landscape design. **Policy** should promote these networks and facilitate collaborations.
National and supra-national	In the attempts of putting an end to biodiversity loss and environmental degradation, measures with a **true ecological benefit should be prioritized** *Example: permanent semi-natural elements should be promoted as EFAs*	There is a need for a strong and stimulating **policy** that is customized to the needs of the member states. This policy should focus on measures with a true ecological benefit. Regular evaluation of both policy and measures is needed.
	Acknowledge the trade-offs in the ES that can be obtained and develop and	The development of implementation and management guidelines for nature-oriented measures should take into

adapt nature-oriented measures in order to **maximize the prioritized ES** *Example: a grass strip intercepting nutrients from the water flow will probably not host a vegetation of botanical value. Increasing the mowing intensity will potentially increase nutrient interception but will simultaneously reduce botanical diversity*	account spatial context and prioritized ES. At a **policy** level, this requires a close collaboration with **scientists** and **farmers** in order to introduce more nuance into the management schemes. **Regional landscape planners** should assist the farmers in their choice of measure.
Develop appropriate financial **compensation schemes** for measures that enhance the delivery of ES *Example: via payment for ES, the uptake of nature-oriented measures may be promoted*	**Policy** should develop result-based, flexible compensation schemes that allow the consideration of local context, landscape and priorities.
Promote products that are delivered by nature-oriented-measure-including-farming. *Example: via certification or short chain supply*	**Policy** should develop a labelling practice that allows **consumers** to distinguish products that have been produced together with a wide range of other ES
Acknowledge **temporal and spatial variability** in the effects that can be expected *Example: immediately after hedgerow plantation, the impact on crop yield and on regulating ecosystem services will be limited.*	**Compensation schemes** should take into account measure characteristics that may influence the expected impact and promote long-term maintenance of certain measures.

7. Appendices

7.1 Systematic literature search and selection process for ecosystem service delivery by hedgerows and grass strips

Studies were searched on the Web of Science using following search terms (Table 7.1). All hits (ranging from 1955 to 2016) were considered. During our search for studies describing the effect of grass strips on carbon stock, the initial combination of search terms did not result in appropriate data. Therefore, we redefined our criteria and focussed on carbon storage after arable crop land conversion to grassland.

Table 7.1: search terms used for every literature search, the number of hits and the date of latest search. HR stands for hedgerow, GS stands for grass strip.

HR/GS – ES combination	Search terms	Hits	Search updated until
HR– crop yield	(hedge* OR tree row OR woody field margin OR woody edge) AND (crop yield OR crop product* OR crop OR tree crop interaction	951	22/06/2016
HR– carbon stock	(hedge* OR tree row OR woody field margin OR woody edge) AND soil carbon	245	23/06/2016
HR - N interception	(hedge* OR tree row OR woody field margin OR woody edge OR filter strip) AND (nitrogen OR nitrate) AND (runoff OR leach *)	221	24/06/2016
HR - P interception	(hedge* OR tree row OR woody field margin OR woody edge OR filter strip) AND (phosphorus OR phosphate) AND (runoff OR leach *)	229	24/06/2016
HR– soil sediment interception	(hedge* OR tree row OR woody field margin OR woody edge OR filter strip) AND (erosion OR suspended solids OR sediment)	960	30/06/2016
HR– (potential) natural pest control	(hedge OR tree row OR woody field margin OR woody edge) AND ("natural enemy" OR "natural pest control" OR "biological control" OR predat*)	382	30/06/2016
GS – carbon stock	("grass strip" OR "grassy field margin" OR "buffer strip") AND soil carbon grass AND ("soil carbon" OR "soil organic carbon")	27 1016	16/03/2016 02/07/2016
GS – N interception	("grass strip" OR "field margin" OR "buffer strip" OR filter strip) AND (nitrogen OR nitrate) AND (runoff OR leach *)	177	03/07/2016
GS – P interception	("grass strip" OR "field margin" OR "buffer strip" OR filter strip) AND (phosphorus OR phosphate) AND (runoff OR leach*)	223	03/07/2016
GS – soil sediment interception	("grass strip" OR "field margin" OR "buffer strip" OR filter strip) AND erosion	263	06/07/2016
GS – (potential) natural pest control	("grass strip" OR "field margin" OR "buffer strip") AND ("natural enemy" OR "natural pest control" OR "biological control" OR predat*)	117	08/07/2016

In the following overview, the search process is given. Additional researches consist of grey literature scanning and expert consultation for unpublished data.

Hedgerows – crop yield

Of the 951 studies identified through database searching, 28 were retained based on title and abstract. Of these 28 studies, the following were used in the analyses: Rivest & Vézina (2014), Burgess et al. (2005), Chirko et al. (1996), Gao et al. (2013), Reynolds et al. (2007), Senaviratne et al. (2012), Woodall & Ward (2002). After reference checking and author searching, Stamps et al. (2009) was added to the dataset. Finally, own data (Van Vooren et al. (2016) and unpublished data (Pardon et al.)) and data from Chaves (2001) were included.

Hedgerows – carbon stock

Of the 245 studies identified through database searching, 23 were retained based on title and abstract. Of these 23 studies, the following were used in the analysis: Paudel et al. (2012), Walter et al. (2003), Bambrick et al. (2010), Cardinali et al. (2014), D'Acunto et al. (2014). After reference checking and author searching, Wotherspoon et al. (2014), Oelbermann et al. (2006) and Sharrow & Ismail (2004) were added to the dataset. Finally, own data ((Van Vooren et al. (2016) and unpublished data (Pardon et al.)) were included.

Hedgerows – N interception

Of the 221 studies identified through database searching, 31 were retained based on title and abstract. Of these 31 studies, the following were used in the analysis: Schmitt et al. (1999), Duchemin and Hogue (2009), Wang et al. (2012), Borin et al. (2005), Schoonover et al. (2005), Borin et al. (2010), Salazar et al. (2015), Yang et al. (2015), Udawatta et al. (2011).

Hedgerows – P interception

Of the 229 studies identified through database searching, 31 were retained based on title and abstract. Of these 31 studies, the following were used in the analysis: Duchemin and Hogue (2009), Borin et al. (2005), Schoonover et al. (2005), Borin et al. (2010), Udawatta et al. (2011), Sheppard et al. (2006). After reference checking and author searching, Uusi-Kämppä and Jauhiainen (2010) was included. From the hedgerow – N interception search, Schmitt et al. (1999) and Yang et al. (2015) were retained.

Hedgerows – erosion reduction

Of the 960 studies identified through database searching, 53 were retained based on title and abstract. Of these 53 studies, the following were used in the analysis: Schmitt et al. (1999), Duchemin and Hogue (2009), Borin et al. (2005), Schoonover et al. (2005), Borin et al. (2010), Udawatta et al. (2011), Yang et al. (2015), Leguédois et al. (2008). After reference checking and author searching, Uusi-Kämppä and Jauhiainen (2010) was also included.

Hedgerows – natural pest control

Of the 382 studies identified through database searching, 22 were retained based on title and abstract. Of these 22 studies, the following were used in the analysis: Asteraki et al. (2004), Pfiffner & Luka (2000), Thomson & Hoffmann (2010). After reference checking and author searching, Nazzi et al. (1989) was also retained. After expert consultation, Griffiths et al. (2007) was also included.

Grass strips – carbon stock

Of the 1016 studies identified through database searching, 38 were retained based on title and abstract. Of these 38 studies, the following were used in the analysis: Kumar et al. (2010), Culman et al. (2010), Potter et al. (1999), Bowman & Anderson (2002), Al-Kaisi et al. (2005), Arshad et al. (2004), Römkens et al. (1999), Cardinali et al. (2014), Paudel et al. (2012), Purakayastha et al. (2008), Omonode & Vyn (2006).

Grass strips – N interception

Of the 177 studies identified through database searching, 20 were retained based on title and abstract. Of these 20 studies, the following were used in the analysis: Duchemin and Hogue (2009), Patty et al. (1997), van Beek et al. (2007), Wang et al., (2012), Schmitt et al. (1999), Blanco-Canqui et al. (2004), Hay et al. (2006), Udawatta et al. (2011), Yang et al. (2015), Lim et al. (1998), Lee et al. (1999). After reference checking and author searching, Magette et al. (1989) was also included. From the grass strip – erosion reduction search, Miller et al. (2015) and Mendez et al. (1999) were also retained.

Grass strips – P interception

Of the 223 studies identified through database searching, 26 were retained based on title and abstract. Of these 26 studies, the following were used in the analysis: Duchemin and Hogue (2009), Patty et al. (1997), Blanco-Canqui et al. (2004), Mankin et al. (2007), Hay et al. (2006), Udawatta et al. (2011), Yang et al. (2015), Lim et al. (1998), Lee et al. (1999), Abu-Zreig et al. (2003), Sheppard et al. (2006). After reference checking and author searching, Uusi-Kämppä and Jauhiainen (2010) and Magette et al. (1989) were included. From the hedgerow – N interception search, Schmitt et al. (1999) and Miller et al. (2015) were also retained.

Grass strips – erosion reduction

Of the 263 studies identified through database searching, 36 were retained based on title and abstract. Of these 36 studies, the following were used in the analysis: Blanco-Canqui et al. (2004), Mankin et al. (2007), Lee et al. (1999), Abu-Zreig et al. (2003), Miller et al. (2015), Mendez et al. (1999), Hay et al. (2006), Van Dijk et al. (1996), Le Bissonnais et al. (2004), Arora et al. (1996), Mickelson & Baker (2003). After reference checking and author searching, Uusi-Kämppä and Jauhiainen (2010) and Magette et al. (1989) were included. From the grass strip – N interception search, Duchemin and Hogue (2009), Patty et al. (1997), Udawatta et al. (2011), Yang et al. (2015), Lim et al. (1998) were also retained.

Grass strips – natural pest control

Of the 117 studies identified through database searching, 30 were retained based on title and abstract. After reference checking of these 30 retained studies, Thomas & Marshall (1999) was included. From the hedgerow – NPC search, Pfiffner & Luka (2000) was retained. Based on expert consultation, Powell et al. (2004), van Alebeek et al. (2003), van Alebeek et al. (2006) were also included.

7.2 Experimental setup of unpublished data (Pardon et al.)

Crop yield

In 2015, crop yield was measured on 13 conventional arable fields partially bordered by a tree row. Fields were spread throughout Flanders and soil types varied from sandy loam to loamy. Winter wheat was grown on six fields, maize on four fields and winter barley on two fields. Estimated tree heights varied between 9 m and 35 m. On the wheat and barley parcels, plots of 1.5 m x 6.5 m were harvested. On the maize parcels, the experimental plot had a length of 5 m and consisted of two maize rows. Plots were situated on transects perpendicular to the tree rows and measurements were done on approximate distances of 2 m, 5 m, 10 m, 20 m and 30 m. On the same parcel, at the part that is not bordered by trees, control measurements were done on the same distance from the field edge. On the alley cropping and hedgerow parcel, transects were replicated 3 times. On the control part of the parcel, transects were replicated two times. Tree species were *Populus* sp. and *Juglans regia*.

Soil organic carbon

In 2015, organic carbon content (%) was measured on 13 conventional arable fields partially bordered by a tree row. Fields were spread throughout Flanders and soil types varied from sandy loam to loamy. Winter wheat was grown on six fields, maize on four fields and winter barley on two fields. Estimated tree heights varied between 9 m and 35 m. Soil samples (0-23 cm) were taken on transects perpendicular to the tree rows and measurements were done on an approximate distance of 2 m, 5 m, 10 m, 20 m and 30 m. On the same parcel, at the part that is not bordered by trees, control measurements were done on the same distance from the field edge. On the alley cropping and hedgerow parcel, every distance was replicated three times. On the control part of the parcel, every distance was replicated two times. Tree species were *Populus* sp. and *Juglans regia*.

7.3 different N and P forms used in the analyses

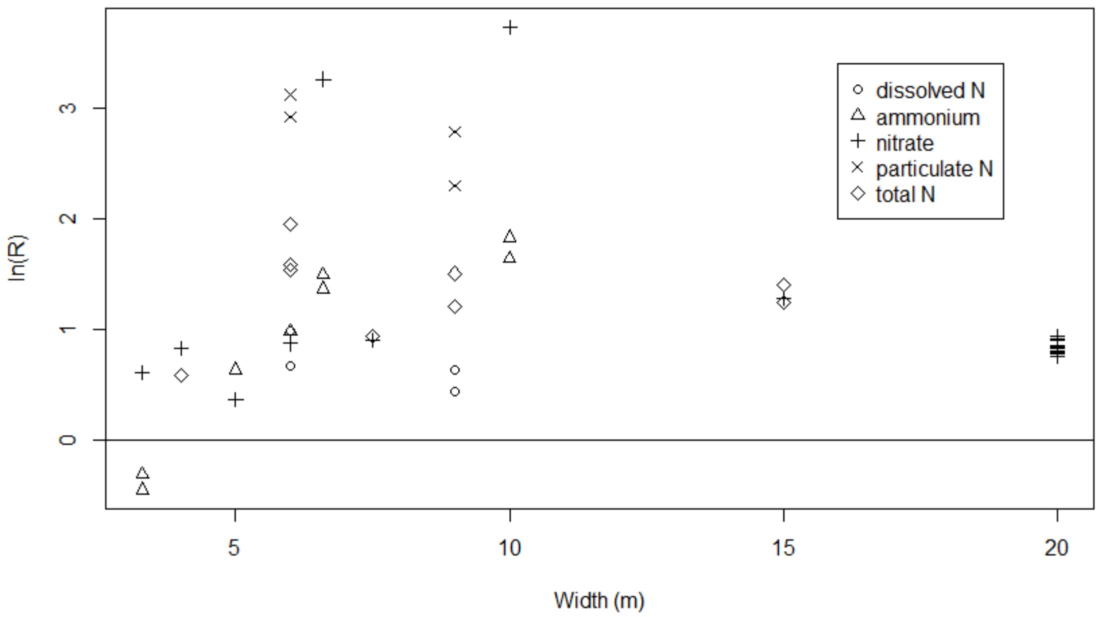


Figure 7.3.1: :surface flow nitrogen (N) interception for various hedgerow widths

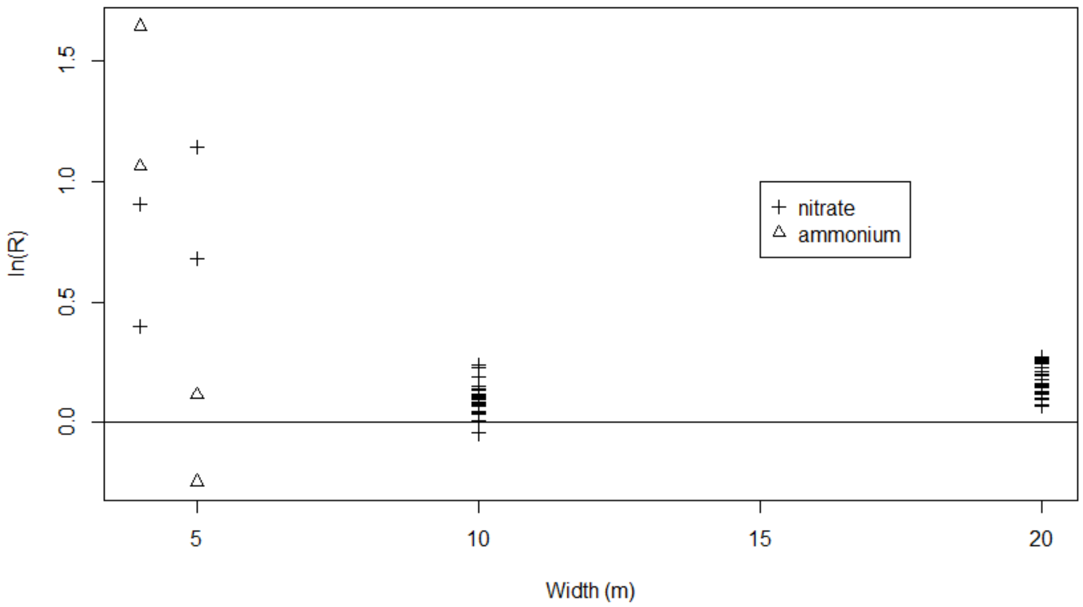


Figure 7.3.2: subsurface flow nitrogen (N) interception for various hedgerow widths

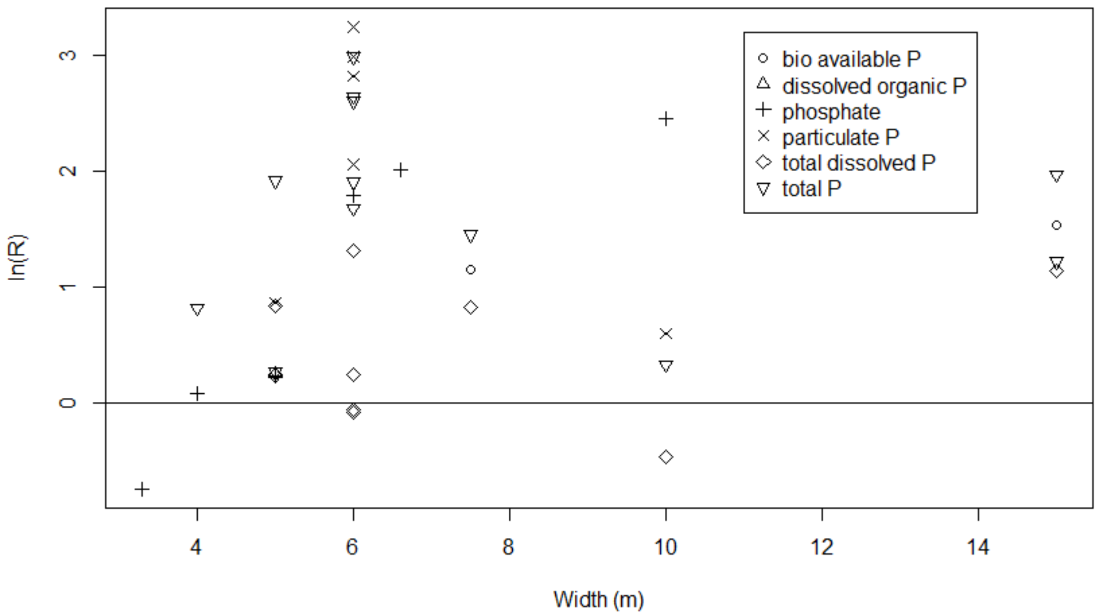


Figure 7.3.3: surface flow phosphorus (P) interception for various hedgerow widths

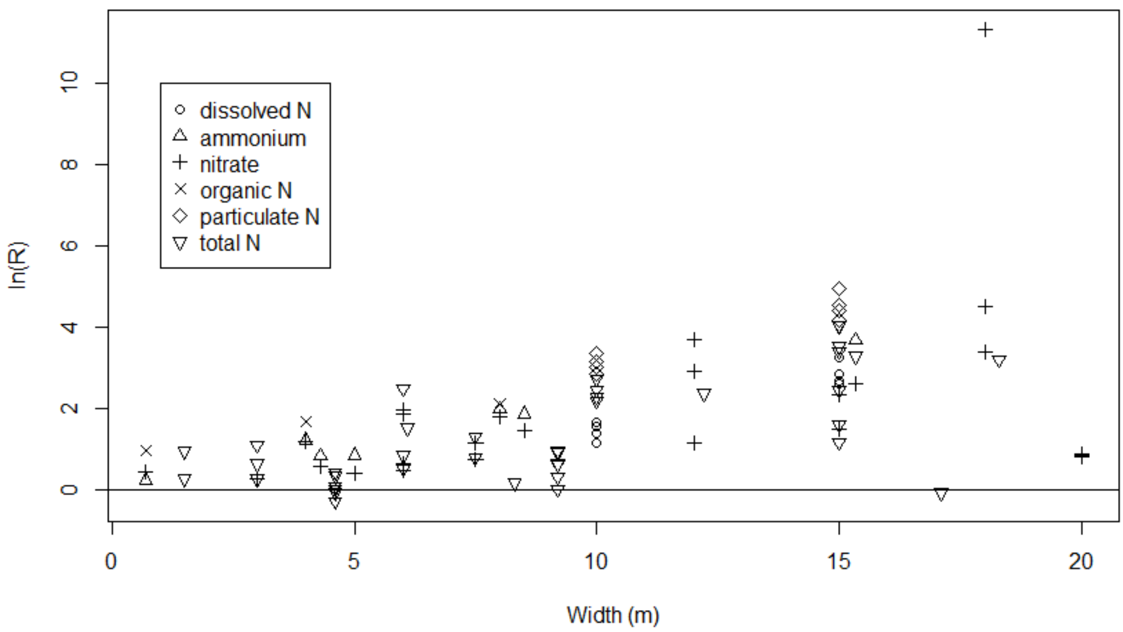


Figure 7.3.46: surface flow nitrogen (N) interception for various grass strips widths

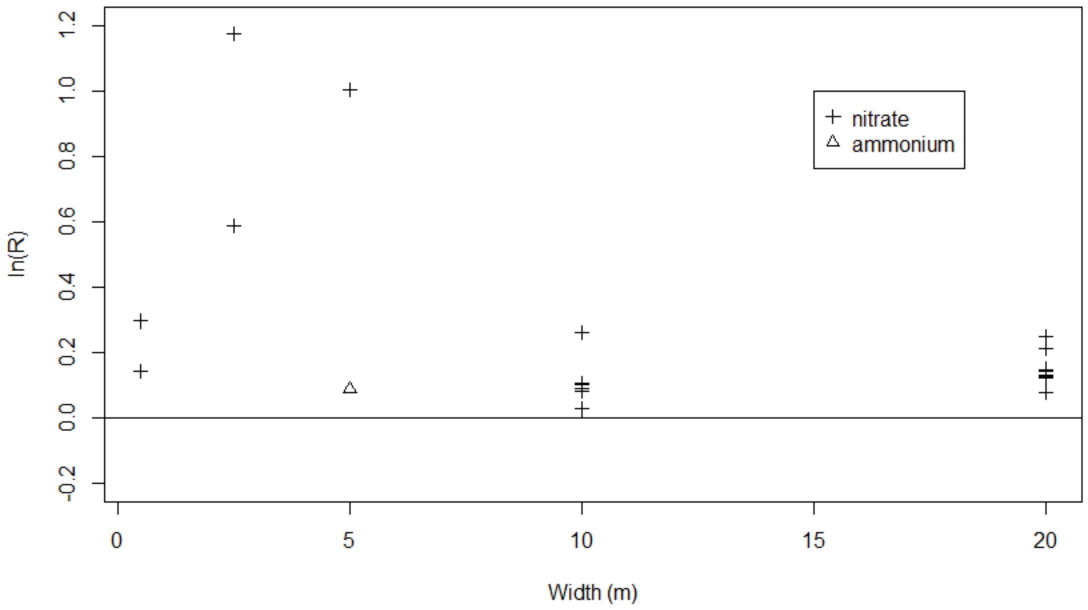


Figure 7.3.5:sub surface flow nitrogen (N) interception for various grass strip widths

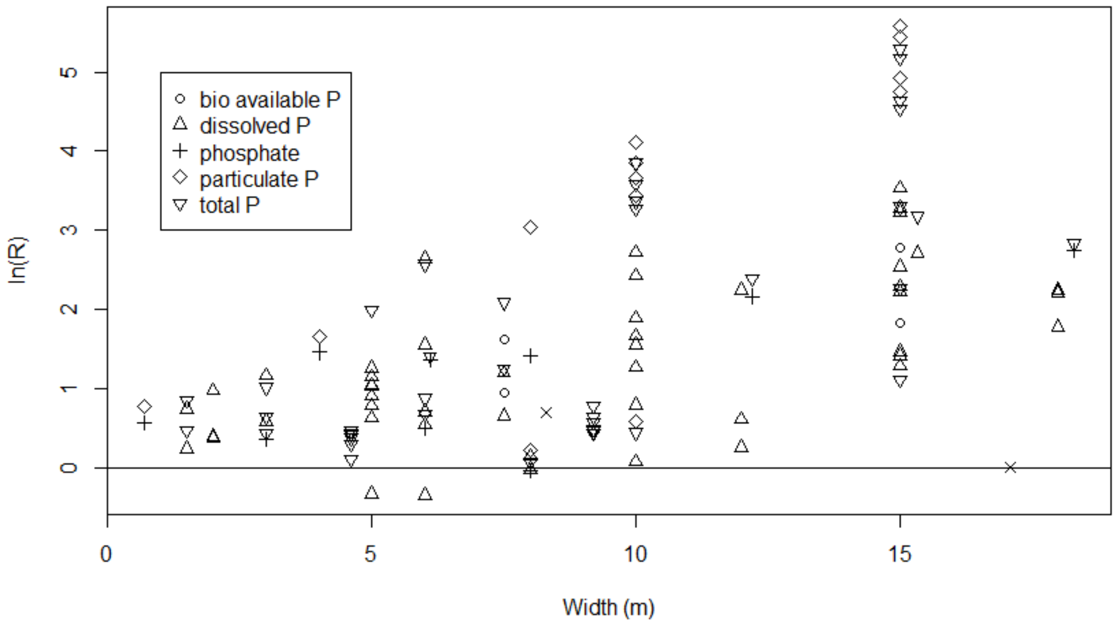


Figure 7.3.6: surface flow phosphorus (P) interception for various grass strip widths

7.4 Comparison of a mixed model and a traditional weighted meta-analysis

When analysing data from multiple studies, it is advisable to consider study statistics such as population size and variance of the data (Koricheva and Gurevitch, 2014). However, in most studies these statistics were not given. In order to include as much studies as possible, a mixed model was applied. Because neglecting study statistics could negatively affect the preciseness of the result, the intercept-only mixed model was compared with the traditional meta-analysis. For every analysis, both models were applied on the same subset of studies. The subset consisted of studies with the required statistics, i.e. population size and variance. In all cases, the results of the mixed model and meta-analysis were very similar (Figure.7 and Figure). We consider the mixed model to be appropriate for the analysis.

For the hedgerow – crop yield analysis, following studies were retained: Chirko et al. (1996), Gao et al. (2013), Rivest and Vézina (2014),Stamps et al. (2008),Van Vooren et al. (2016)

For the hedgerow – carbon stock analysis, following studies were retained: Cardinali et al. (2014), Sharrow and Ismail (2004), Walter et al. (2003). Additionally, own data were used.

For the hedgerow – N interception (surface flow) analysis, following studies were retained: Wang et al. (2012), Schoonover et al. (2005)

For the hedgerow – P interception analysis, following studies were retained: Schoonover et al. (2005), Sheppard et al. (2005)

For N interception from the subsurface flow by hedgerows, only one study (Wang et al., 2012) reported the required statistics and for erosion reduction by hedgerows, no studies reported the required statistics. Therefore, the mixed model could not be compared with a traditional meta-analysis.

For the grass strip – N interception (surface flow) analysis, following studies were retained: Mankin et al. (2007), Magette et al. (1989), Hay et al. (2006), Mendez et al. (1999).

For the grass strip – P interception analysis, following studies were retained: Mankin et al. (2007), Magette et al. (1989), Hay et al. (2006), Sheppard et al. (2005)

For the grass strip – erosion reduction analysis, following studies were retained: Mankin et al. (2007), Magette et al. (1989), Hay et al. (2006), Mendez et al. (1999), Mickelson and Baker (2003).

For soil carbon stock and N interception from the subsurface flow by grass strips, no studies reported the required statistics. Therefore, the mixed model could not be compared with a traditional meta-analysis.

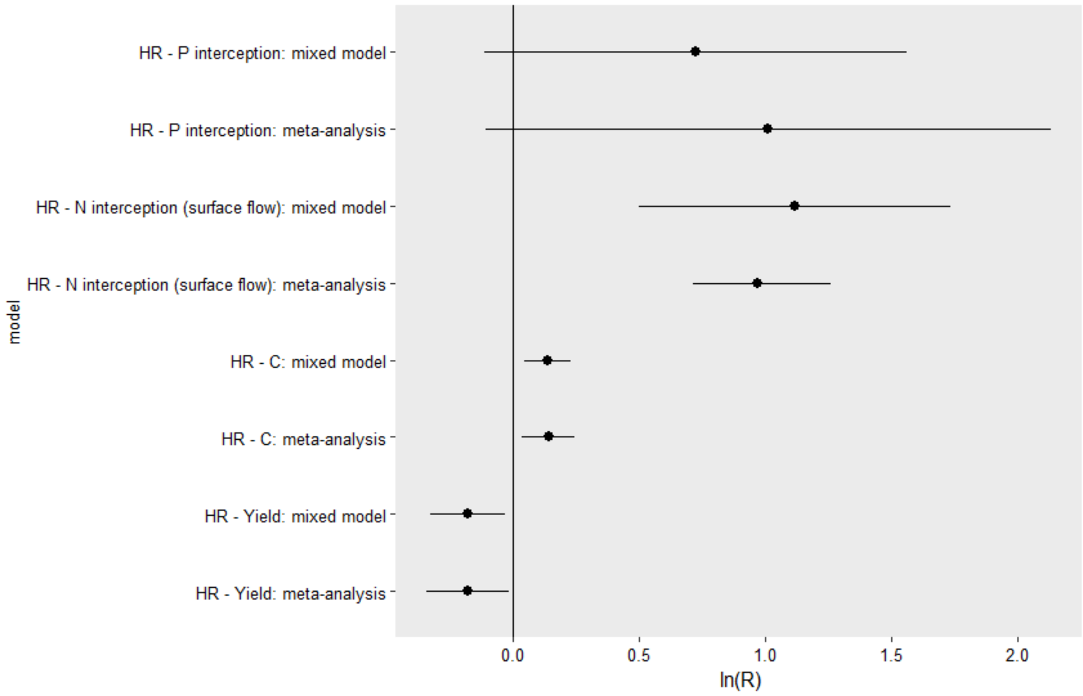


Figure 7.4.1: Comparison of the intercept-only model result according to the mixed model and the traditional meta-analysis for the hedgerows (HR). 95% confidence intervals are represented by the bars.

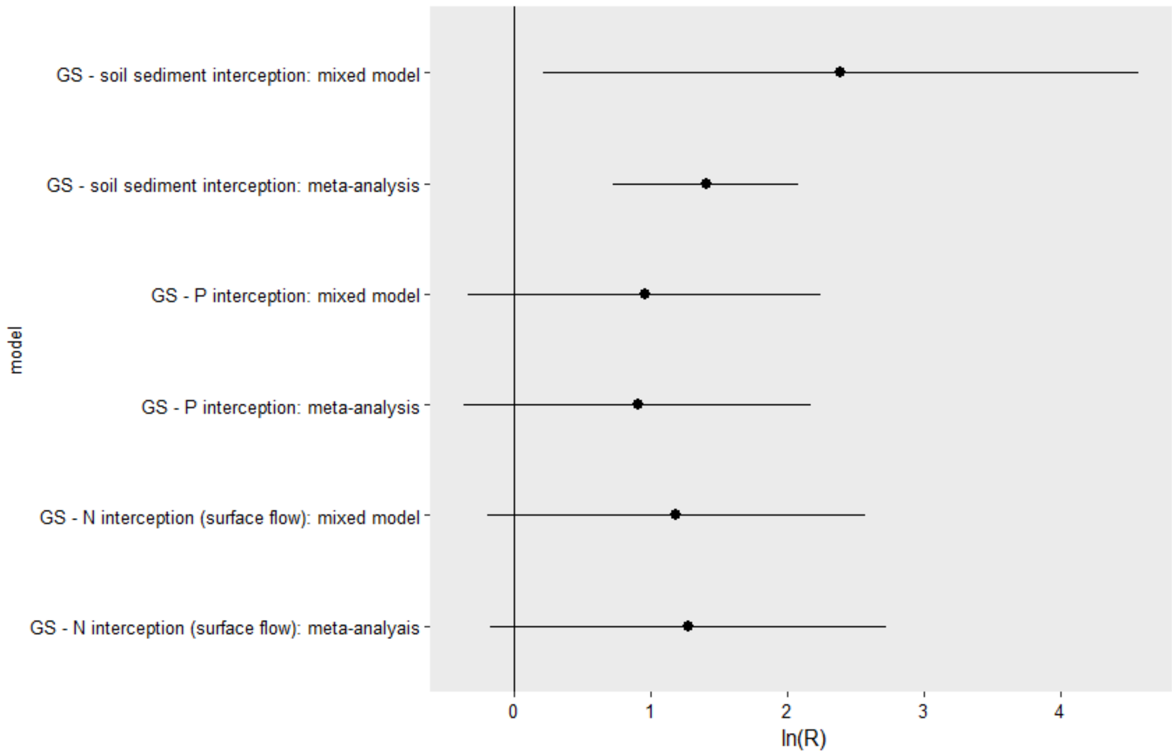


Figure7.4.2: Comparison of the intercept-only model result according to the mixed model and the traditional meta-analysis for the grass strips (GS). 95% confidence intervals are represented by the bars.

7.5 Model selection

Hedgerows

In Table 7.5.1, AIC, BIC and the log-likelihood of the tested spatial models are presented. Model selection was based on these criteria combined with biophysical relevance and applicability.

Table 7.5.1: Overview of the tested spatial models for hedgerows.

	Mixed model	Type	AIC	BIC	logLik
Crop yield	$\ln(R) = c$	Intercept-only	265.0968	276.6012	-129.5284
	$\ln(R) = a \times \log_{10}(H)^2 + \mathrm{b} \times \log_{10}(H) + \mathrm{c}$	Polynomial	101.5399	139.9172	-40.76996
	$\ln(R) = Asymp + R_0 * e^{(b*H)}$	Asymptotic	86.22218	124.5995	-33.11109
	$\ln(R) = R_0 * e^{(b*H)}$	Asymptotic	90.79623	113.8226	-39.39811
SOC	$\ln(R) = \mathrm{c}$	Intercept-only	-37.26934	-30.161	21.63467
	$\ln(R) = \mathrm{a} \times \mathrm{H} + \mathrm{c}$	Linear	-38.68733	-29.26049	23.34366
	$\ln(R) = \mathrm{a} \times \log_{10}(H) + \mathrm{c}$	Linear	-42.86501	-33.43817	25.4325
	$\ln(R) = Asymp + R_0 * e^{(b*H)}$	Asymptotic	-42.15654	-18.33627	31.07827
	$\ln(R) = R_0 * e^{(b*H)}$	Asymptotic	-52.68241	-38.39025	32.3412
N interception: surface	$\ln(R) = \mathrm{c}$	Intercept-only	126.1596	131.7733	-60.07982
	$\ln(R) = \mathrm{a} \times \mathrm{width} + \mathrm{c}$	Linear	132.9866	140.3872	-62.49328
	$\ln(R) = \mathrm{a} \times \log_{10}(width) + \mathrm{c}$	Linear	123.3886	130.7892	-57.69432
	$\ln(R) = Asymp + R_0 * e^{(width)}$	Asymptotic	127.4906	138.8416	-57.74532
N interception: subsurface	$\ln(R) = \mathrm{c}$	Intercept-only	-36.16331	-29.41782	21.08165
	$\ln(R) = \mathrm{a} \times \mathrm{width} + \mathrm{c}$	Linear	-27.29475	-18.35832	17.64738
	$\ln(R) = \mathrm{a} \times \log_{10}(width) + \mathrm{c}$	Linear	-33.41184	-24.47541	20.70592
	$\ln(R) = Asymp + R_0 * e^{(width)}$	Asymptotic	-35.5931	-22.01702	23.79655
P interception: surface	$\ln(R) = \mathrm{c}$	Intercept-only	110.1036	114.7697	-52.05181
	$\ln(R) = \mathrm{a} \times \mathrm{width} + \mathrm{c}$	Linear	114.7239	120.8293	-53.36194
	$\ln(R) = \mathrm{a} \times \log_{10}(width) + \mathrm{c}$	Linear	107.7673	113.8727	-49.88363
	$\ln(R) = Asymp + R_0 * e^{(width)}$	Asymptotic	114.6512	124.1524	-51.32562
Soil sediment interception	$\ln(R) = \mathrm{c}$	Intercept-only	86.85584	90.39001	-40.42792
	$\ln(R) = \mathrm{a} \times \mathrm{width} + \mathrm{c}$	Linear	91.47538	96.01736	-41.73769
	$\ln(R) = \mathrm{a} \times \log_{10}(width) + \mathrm{c}$	Linear	85.76205	90.30402	-38.88102
	$\ln(R) = Asymp + R_0 * e^{(width)}$	Asymptotic	93.53086	100.8441	-40.76543

For crop yield, the AIC, BIC and the log-likelihood indicate that both asymptotic models better describe the data compared to the polynomial model. However, the first asymptotic model returns an asymptotic value of 0.05, indicating that for (very) high H values, relative crop yield will be positive. In reality, it is expected that the effect of hedgerows on crop yield will be negligible for high H values, and thus from a biophysical point of view, the model does not meet the prerequisites. Because of its format, the second asymptotic model does not return positive ln(R) values. However, the lower AIC of the first asymptotic model and the average value of 0.05 of all ln(R) values beyond H=2.1 indicate that the observations do report a positive effect on crop yield. Thus, beyond H= 2.1, the model does not adequately describe the observations. Solely based on the statistical selection criteria, the polynomial model is not the preferred model. However, it describes a negatively affected impact zone and a positively affected impact zone, which is in accordance with previous reviews on this topic (Kort, 1988; Nuberg, 1998; Kowalchuk and de Jong, 1995). Based on the abovementioned arguments, we retained the polynomial model. Visualization (Figure) shows us that the models are very similar and the negatively affected impact zone extends in all cases up to H=2, confirming that all three models adequately describe the observations. Therefore, model selection based on biophysical criteria (instead of solely statistical criteria) is justifiable.

For SOC stock, the first asymptotic model is preferred based on statistical criteria. However, the first asymptotic model returns an asymptotic value of 0.17, indicating that for (very) high H values, relative SOC stock will be positive. In reality, it is expected that the effect of hedgerows on SOC stock will be negligible for high H values, and thus from a biophysical point of view, the model does not meet the prerequisites. The model with the second best set of statistical selection criteria was also biophysically relevant and thus it was retained. For N interception, P interception and soil sediment interception, statistical and biophysical selection criteria resulted in the same model. The asymptotic model approaching zero for increasing widths is not tested for N interception, P interception and soil sediment interception because it is biophysically not relevant. As mentioned in chapter 2, no spatial models were tested for pest control.

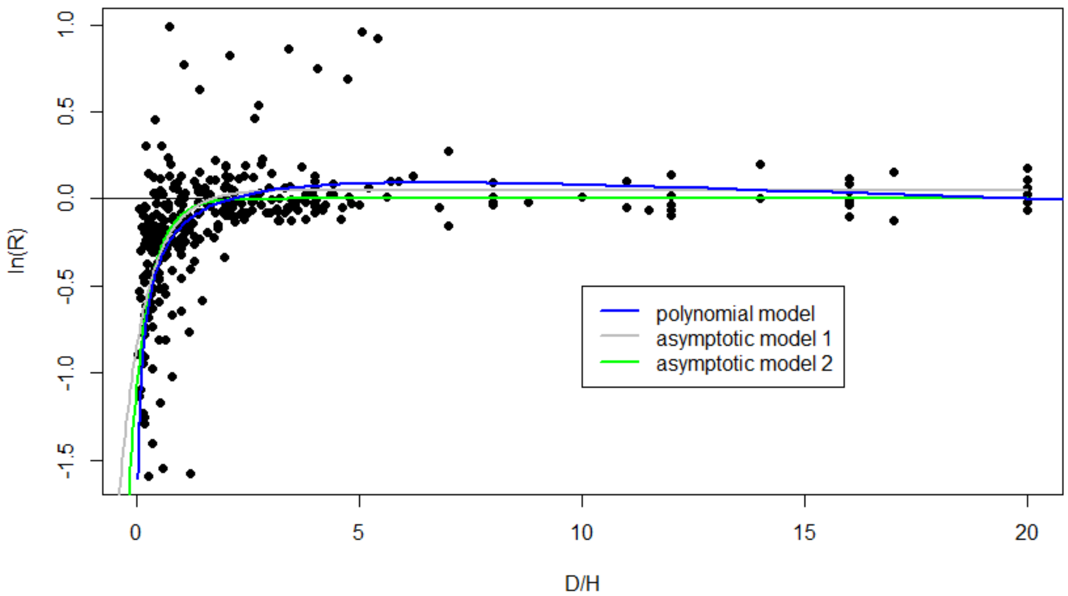


Figure 7.5.1: Comparison of tested models describing the effect of hedgerows on crop yield.

Grass strips

In Table 7.5.2, AIC, BIC and the log-likelihood of the tested spatial models are presented. Model selection was based on these criteria combined with biophysical relevance and applicability. For SOC stock, N interception, P interception and soil sediment interception, statistical and biophysical selection criteria resulted in the same (linear) model. The asymptotic model approaching zero for increasing widths is not tested for N interception, P interception and soil sediment interception because it is biophysically not relevant. As mentioned in chapter 2, no spatial models were tested for pest control.

Table 7.5.2: Overview of the tested spatial models for grass strips.

	Mixed model	Type	AIC	BIC	logLik
SOC	$\ln(R) = \mathrm{c}$	Intercept-only	40.75951	48.778	-17.37976
	$\ln(R) = \mathrm{a} \times \ depth + \mathrm{c}$	Linear	44.54367	55.19743	-18.27184
	$\ln(R) = \mathrm{a} \times \ \log_{10}(depth) + \mathrm{c}$	Linear	20.27552	30.92928	-6.137762
	$\ln(R) = Asymp + \ R_0 * e^{(b*depth)}$	Asymptotic	15.18164	42.00295	2.409179
	$\ln(R) = R_0 * e^{(b*depth)}$	Asymptotic	23.09824	39.19103	-5.54912
N interception: surface	$\ln(R) = \mathrm{c}$	Intercept-only	252.085	259.5509	-123.0425
	$\ln(R) = \mathrm{a} \times \ \mathrm{width} + \mathrm{c}$	Linear	212.6614	222.5707	-102.3307
	$\ln(R) = \mathrm{a} \times \ \log_{10}(width) + \mathrm{c}$	Linear	222.0356	231.9449	-107.0178
	$\ln(R) = Asymp + \ R_0 * e^{(width)}$	Asymptotic	297.6654	312.7306	-142.8327
N interception: subsurface	$\ln(R) = \mathrm{c}$	Intercept-only	11.94725	15.08082	-2.973627
	$\ln(R) = \mathrm{a} \times \ \mathrm{width} + \mathrm{c}$	Linear	21.1382	25.12113	-6.5691
	$\ln(R) = \mathrm{a} \times \ \log_{10}(width) + \mathrm{c}$	Linear	11.80826	15.79119	-1.904128
	$\ln(R) = Asymp + \ R_0 * e^{(width)}$	Asymptotic	15.2302	21.77646	-1.615102
P interception: surface	$\ln(R) = \mathrm{c}$	Intercept-only	322.6196	330.8544	-158.3098
	$\ln(R) = \mathrm{a} \times \ \mathrm{width} + \mathrm{c}$	Linear	290.5847	301.5295	-141.2924
	$\ln(R) = \mathrm{a} \times \ \log_{10}(width) + \mathrm{c}$	Linear	292.9185	303.8633	-142.4593
	$\ln(R) = Asymp + \ R_0 * e^{(width)}$	Asymptotic	306.6386	323.1601	-147.3193
Soil sediment interception	$\ln(R) = \mathrm{c}$	Intercept-only	388.5374	396.4123	-191.2687
	$\ln(R) = \mathrm{a} \times \ \mathrm{width} + \mathrm{c}$	Linear	371.9238	382.3842	-181.9619
	$\ln(R) = \mathrm{a} \times \ \log_{10}(width) + \mathrm{c}$	Linear	367.8547	378.3152	-179.9274
	$\ln(R) = Asymp + \ R_0 * e^{(width)}$	Asymptotic	390.9643	406.7727	-189.4822

7.6 Outlier analysis

- hedgerows – crop yield: removal of 5 outliers increased the model fit (AIC from 281 to 102)
- hedgerows – subsurface flow N interception : removal of 1 outlier increased the model fit (AIC from -8 to -36)
- grass strips – surface flow N interception: removal 1 outlier increased the model fit (AIC from 297 to 213)

7.7 Overview of the measured ES indicators on the grass strip parcels

Table 7.7.1: Mean SOC stocks for every GS parcel averaged over the transects. Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel.

SOC stock (ton ha^{-1}) (0-20 cm)	Position -10	Position -1	Position 30
GS 1	38.30±2.69	42.27±2.28	32.35±5.04
GS 2	44.96±2.06	38.07±1.32	26.81±1.74
GS 3	59.83±4.19	31.00±13.90	26.48±3.87
GS 4	54.22±8.10	32.29±3.48	26.96±1.54
GS 5	48.49±6.45	41.46±1.93	27.09±1.10
GS 6	40.56±3.94	36.83±2.43	23.32±1.27

Table 7.7.2: Mineral N content for every GS parcel. Different transects were pooled. Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel.

Mineral N content (kg ha^{-1}) (0-90 cm)	Position -10	Position -1	Position 30
GS 1	115.96	94.74	114.56
GS 2	59.10	95.05	140.84
GS 3	85.30	123.35	186.86
GS 4	60.48	116.14	79.11
GS 5	114.13	122.11	184.95
GS 6	123.58	87.66	160.12

Table 7.7.3: Mean P-Olsen content for every GS parcel averaged over the transects. Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel.

P-Olsen (mg kg^{-1}) (0-10 cm)	Position -10	Position -1	Position 30
GS 1	58.63±7.09	63.58±6.79	39.36±14.38
GS 2	89.47±5.73	52.71±6.95	43.34±12.04
GS 3	79.24±8.40	66.08±4.51	28.65±13.13
GS 4	97.81±7.91	43.44±5.16	20.83±9.87
GS 5	80.11±9.65	40.41±4.29	19.04±2.99
GS 6	96.17±3.41	69.46±4.27	41.34±6.71

Table 7.7.4: Mean carabid activity-density for every GS parcel averaged over the transects and collections. Position -5 is in the middle of the grass strip, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel.

Carabid activity-density	Position -5	Position 10	Position 30
GS 1	2.83±1.94	13.62±12.17	30.22±13.94
GS 2	14.33±17.35	59.33±55.51	114.62±72.48
GS 3	11.44±12.45	28.22±26.16	20.12±20.61
GS 4	3.63±3.66	18.22±21.18	17.28±16.35
GS 5	11.11±8.68	62.00±57.77	58.77±50.05
GS 6	3.11±2.26	75.55±80.98	34.75±35.91

Table 7.7.5: Mean spider activity-density for every GS parcel averaged over the transects and collections. Position -5 is in the middle of the grass strip, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel.

Spider activity-density	Position -5	Position 10	Position 30
GS 1	37.00±26.25	6.88±7.90	2.22±2.44
GS 2	28.00±21.39	12.78±9.98	8.25±6.52
GS 3	51.56±41.88	13.11±7.66	6.25±4.46
GS 4	34.25±37.39	4.22±4.18	7.00±3.78
GS 5	37.22±47.35	5.89±4.05	8.68±7.04
GS 6	15.44±20.31	3.67±3.81	5.25±4.71

Table 7.7.6: Mean rove beetle activity-density for every GS parcel averaged over the transects and collections. Position -5 is in the middle of the grass strip, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel.

Rove beetle activity-density	Position -5	Position 10	Position 30
GS 1	10.83±6.18	5.25±3.96	5.33±4.58

GS 2	6.78±4.20	4.00±4.42	2.38±2.38
GS 3	7.11±7.85	14.00±14.07	4.75±5.09
GS 4	4.25±2.55	2.33±3.08	1.14±1.07
GS 5	16.33±14.76	6.56±9.14	19.88±30.23
GS 6	4.44±3.24	5.22±3.38	3.00±4.57

7.8 Overview of the measures ES indicators on the hedgerow parcels

Table 7.8.1: Mean crop yield for every HR parcel averaged over the transects. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Crop yield (ton ha^{-1})	Distance 1	Distance 10	Distance 30
HR 1	5.85±0.54	9.21±0.69	9.83±0.46
HR 2	9.89±1.55	11.04±0.10	12.16±0.62
HR 3	5.26±1.60	8.03±0.97	7.31±0.61
HR 4	7.66±0.68	9.77±0.66	10.31±0.05

Table 7.8.2: Mean thousand kernel weight for every winter wheat HR parcel averaged over the transects. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Thousand kernel weight (g)	Distance 1	Distance 10	Distance 30
HR 1	59.31±1.39	54.79±4.24	51.24±3.13
HR 2	39.78±2.25	39.09±2.12	36.33±4.80
HR 4	48.82±2.88	45.66±2.84	45.46±3.14

Table 7.8.3: Mean SOC stock for every HR parcel averaged over the transects. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

SOC stock (ton ha^{-1}) (0-20 cm)	Distance 1	Distance 10	Distance 30
HR 1	39.84±2.36	34.86±0.32	40.59±5.65
HR 2	38.26±5.18	40.45±1.96	39.21±7.05
HR 3	45.46±0.91	36.08±1.05	32.28±0.35
HR 4	46.08±6.22	44.74±2.51	40.47±1.62

Table 7.8.4: Mineral N content for every HR parcel. Different transects were pooled. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Mineral N content (kg ha^{-1}) (0-90 cm)	Distance 1	Distance 10	Distance 30
HR 1	109.85	112.04	112.41
HR 2	137.05	124.09	101.98
HR 3	81.23	89.35	72.23
HR 4	63.48	64.41	75.91

Table 7.8.5: Mean P-Olsen for every HR parcel averaged over the transects. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

P-Olsen (mg kg^{-1}) (0-10 cm)	Distance 1	Distance 10	Distance 30
HR 1	87.73±9.27	69.69±4.86	73.17±4.90
HR 2	70.48±7.07	83.36±11.49	74.18±30.30
HR 3	62.11±6.47	57.65±9.80	36.44±5.18
HR 4	109.95±0.52	111.28±9.63	103.76±5.78

Table 7.8.6: Mean carabid activity-density for every HR parcel averaged over the transects and collections. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Carabid activity-density	Distance 1	Distance 10	Distance 30
HR 1	8.75±4.14	9.39±6.04	7.40±6.23
HR 2	3.94±5.57	8.32±11.62	8.43±9.39
HR 3	3.57±2.41	3.80±4.64	9.31±5.93
HR 4	15.90±4.48	9.34±4.08	6.12±4.25

Table 7.8.7: Mean spider activity-density for every HR parcel averaged over the transects and collections. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Spider activity-density	Distance 1	Distance 10	Distance 30
HR 1	12.38±12.63	7.76±5.77	6.34±4.08
HR 2	10.71±10.97	4.67±1.36	8.81±3.59
HR 3	16.91±13.85	14.35±6.42	13.10±9.87
HR 4	39.87±53.92	35.32±47.87	18.25±35.03

Table 7.8.8: Mean rove beetle activity-density for every HR parcel averaged over the transects and collections. Plots are situated on a distance of 1 m from the hedgerow, 10 m from the hedgerow and 30 m from the hedgerow.

Rove beetle activity-density	Distance 1	Distance 10	Distance 30
HR 1	10.97±11.95	5.72±6.79	6.31±5.73
HR 2	4.11±3.71	5.33±1.94	5.64±4.83
HR 3	19.22±34.21	12.04±6.91	23.41±19.33
HR 4	14.25±12.68	14.59±11.83	7.05±5.76

7.9 Monitored soil variables on arable parcels with grass strips and hedgerows

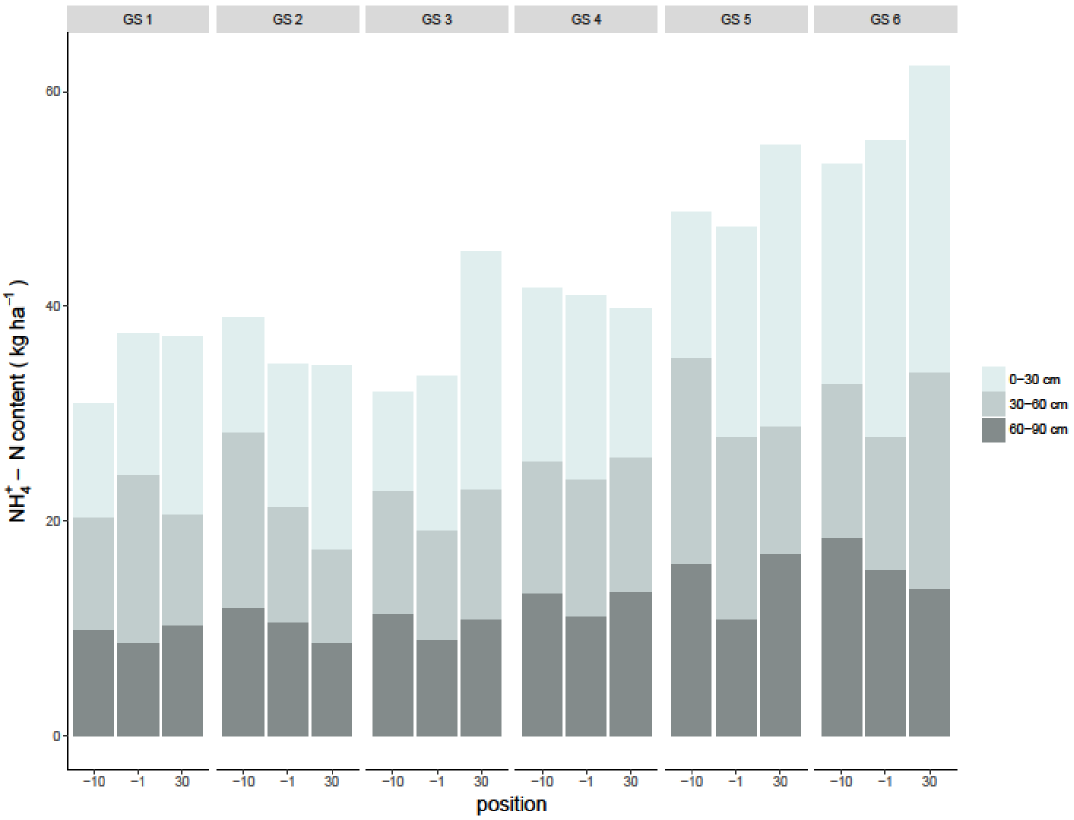


Figure 7.9.1: Soil ammonium (NH_4^+-N) (kg ha^{-1}) (0-90 cm) content measured on the grass strip parcels (GS 1 up to GS 6). Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel. The different sampling layers are represented.

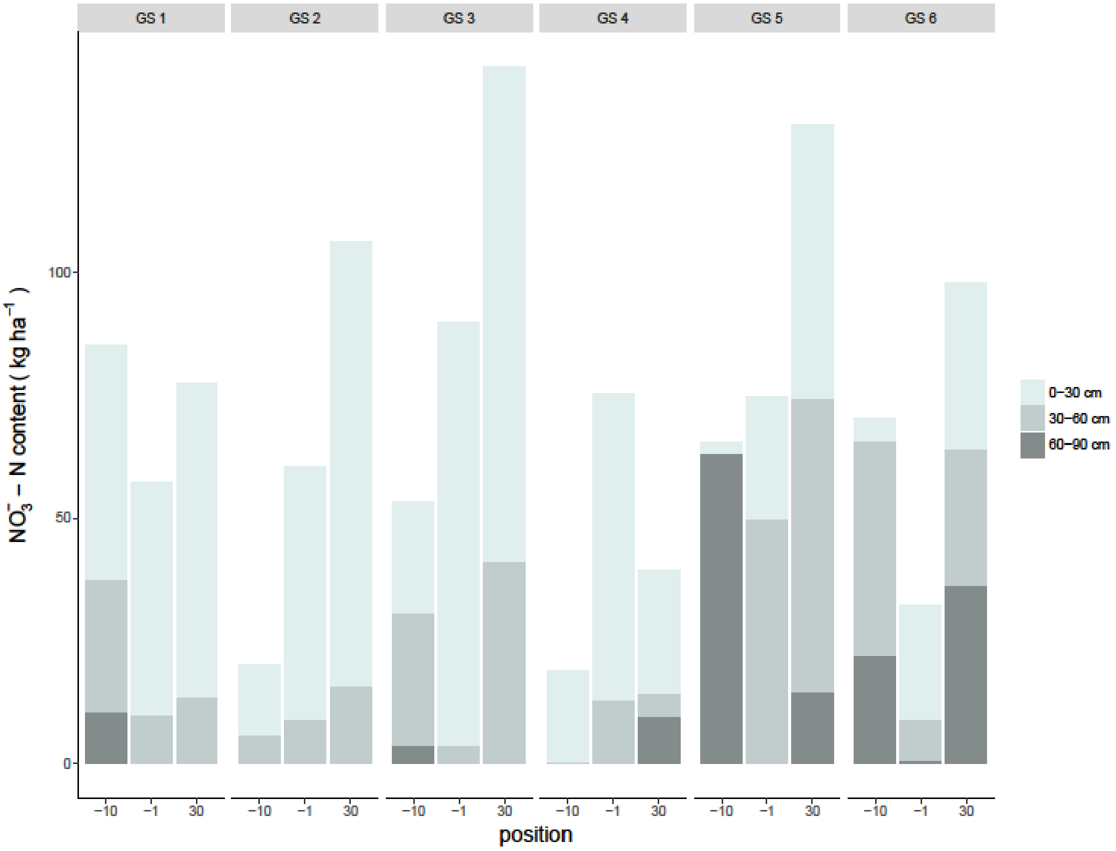


Figure 7.9.2: Soil nitrate (NO_3^--N) (kg ha^{-1}) (0-90 cm) content measured on the grass strip parcels (GS 1 up to GS 6). Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel. The different sampling layers are represented.

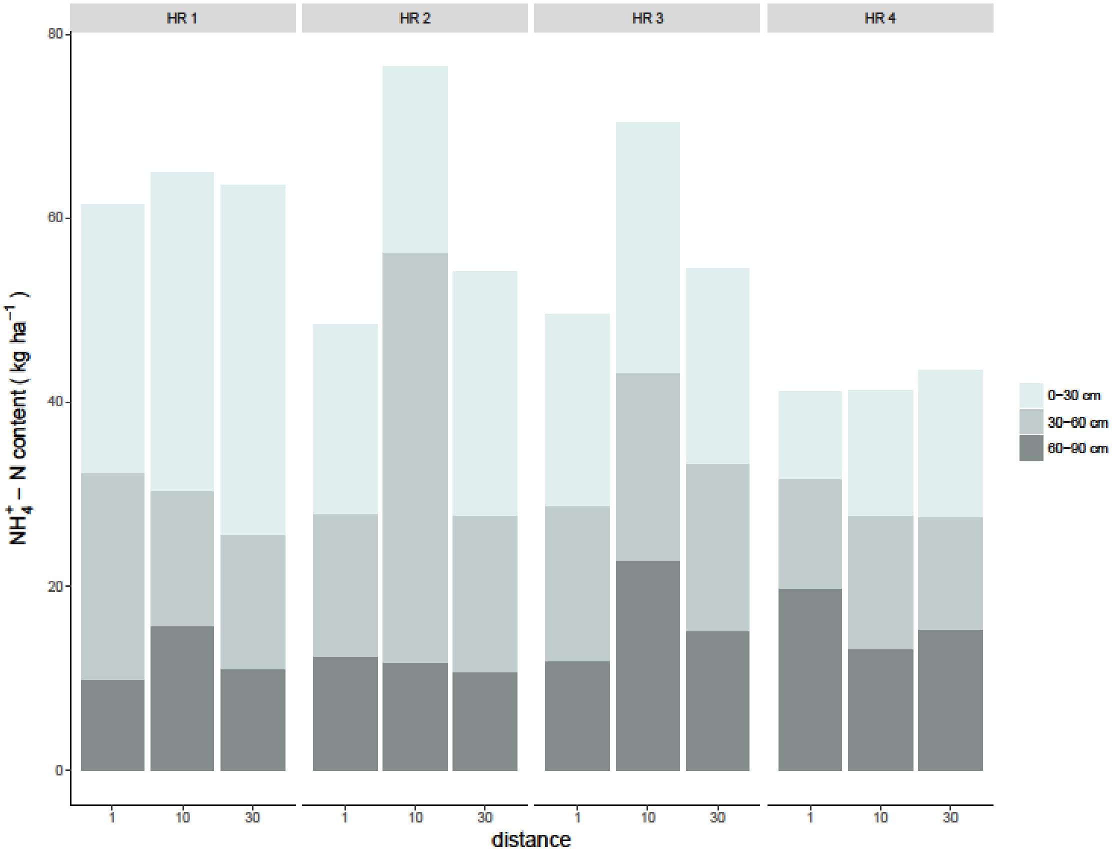


Figure 7.9.3: Soil ammonium (NH_4^+-N) (kg ha^{-1}) (0-90 cm) content measured on the hedgerow parcels (HR 1 up to HR 4). The soil is sampled at 1 m, 10 m and 30 m from the hedgerow. The different sampling layers are represented.

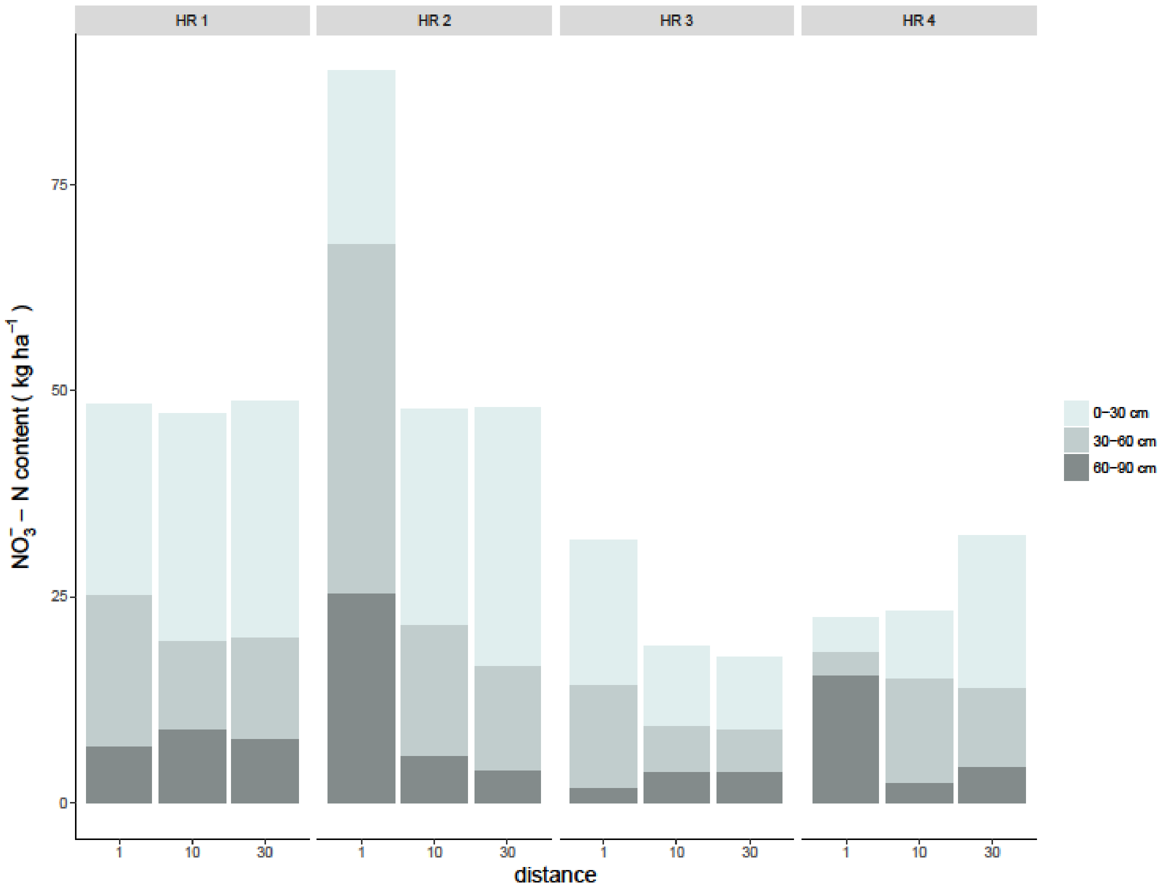


Figure 7.9.4: Soil nitrate (NO_3^--N) (kg ha^{-1}) (0-90 cm) content measured on the hedgerow parcels (HR 1 up to HR 4). The soil is sampled at 1 m, 10 m and 30 m from the hedgerow. The different sampling layers are represented.

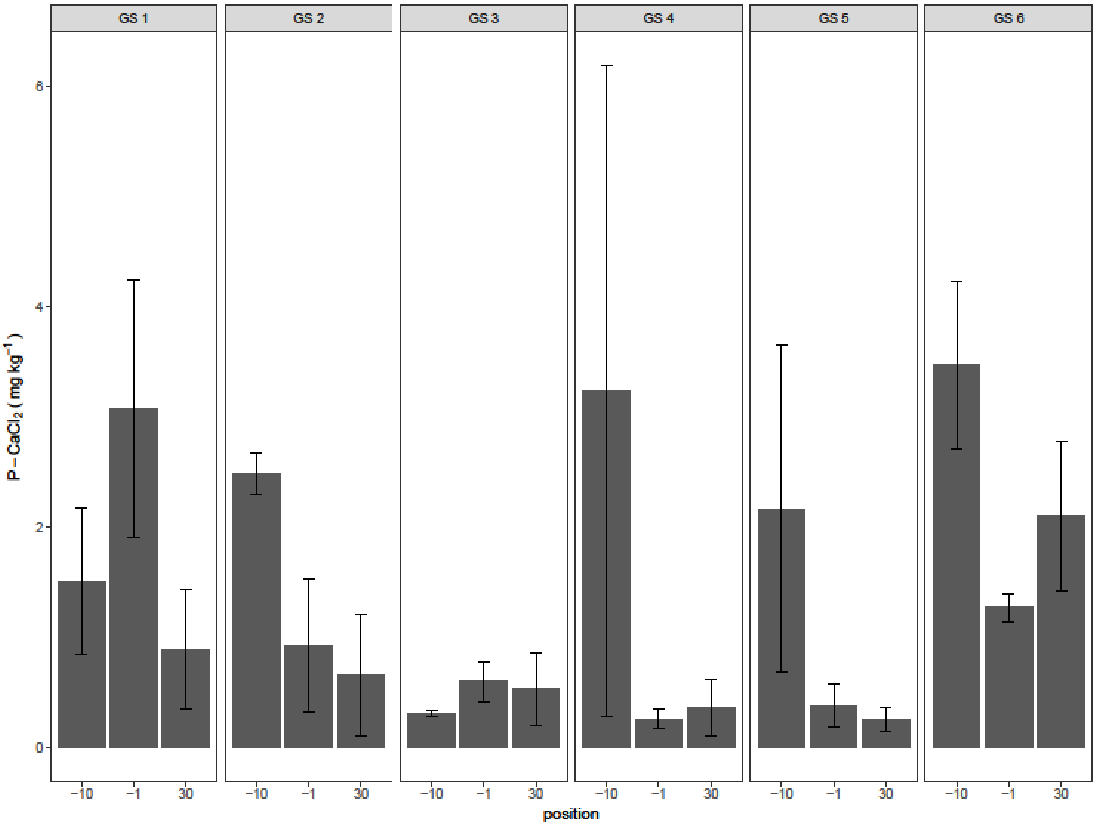


Figure 7.9.5: Soil P-CaCl$_2$ (mg kg^{-1}) (0-10 cm) concentration measured on the grass strip parcels (GS 1 up to GS 6). Position -10 is in the grass strip next to the ditch, position -1 is in the grass strip next to the parcel and position 30 is 30 m in the parcel. Error bars represent standard deviations among samples from the same row.

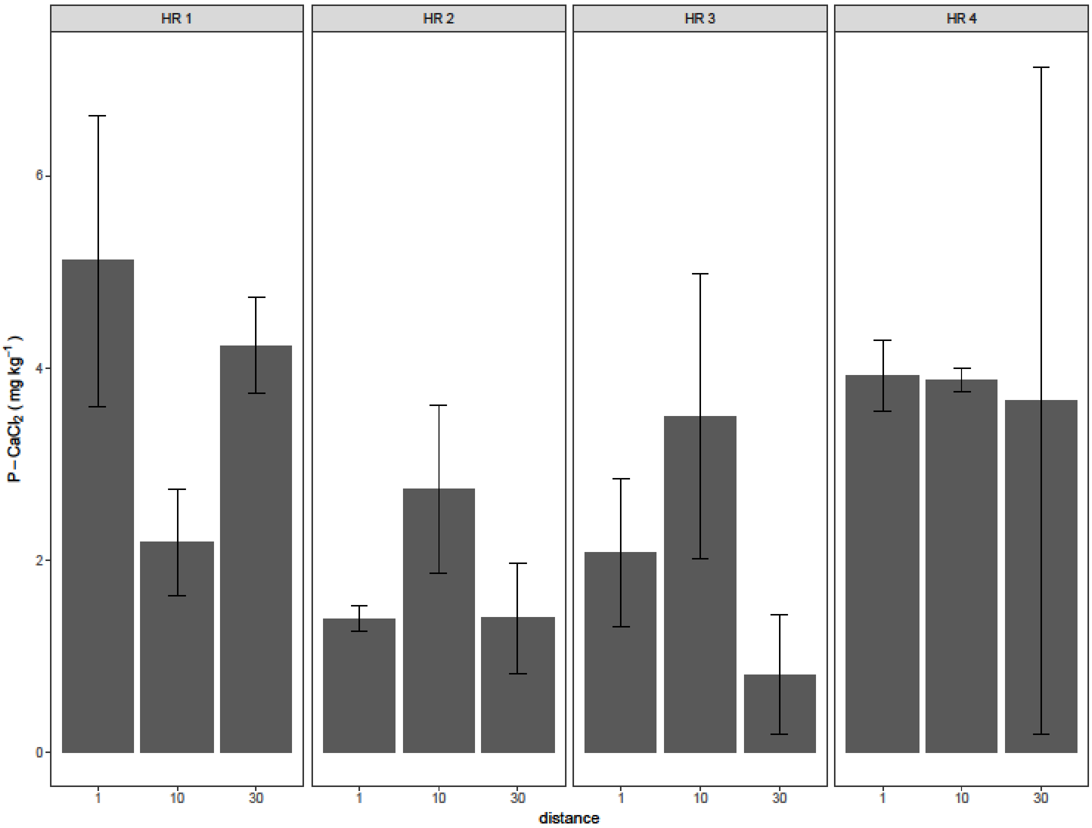


Figure 7.9.6: Soil P-CaCl$_2$ (mg kg^{-1}) (0-10 cm) concentration measured on the hedgerow parcels (HR 1 up to HR 4). The soil is sampled at 1 m, 10 m and 30 m from the hedgerow. Error bars represent standard deviations among samples from the same row.

7.10 Carabid species on arable parcels with grass strips and hedgerows

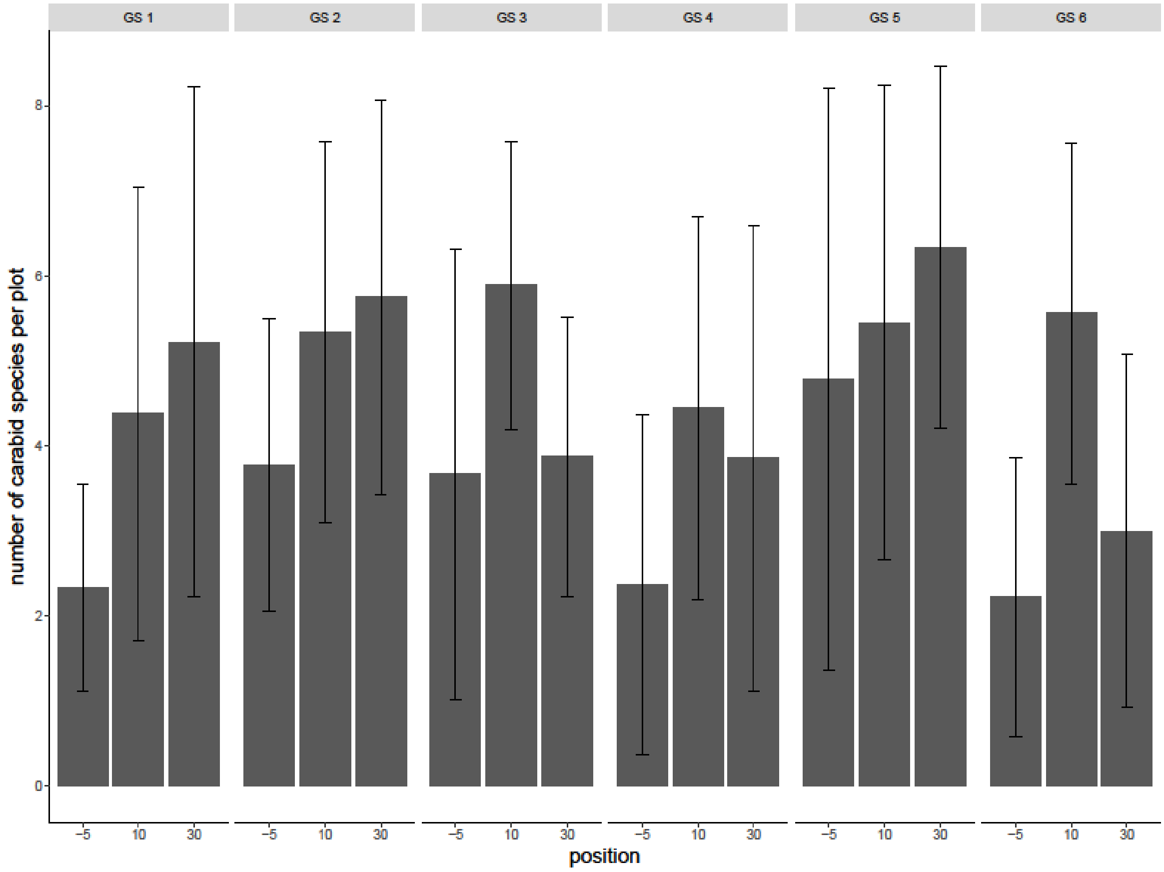


Figure 7.10.1: Average number of carabid species caught on the grass strip parcels (GS 1 up to GS 6). Position -5 is in the middle of the grass strip, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel. Error bars represent standard deviations among samples from the same row and collections.

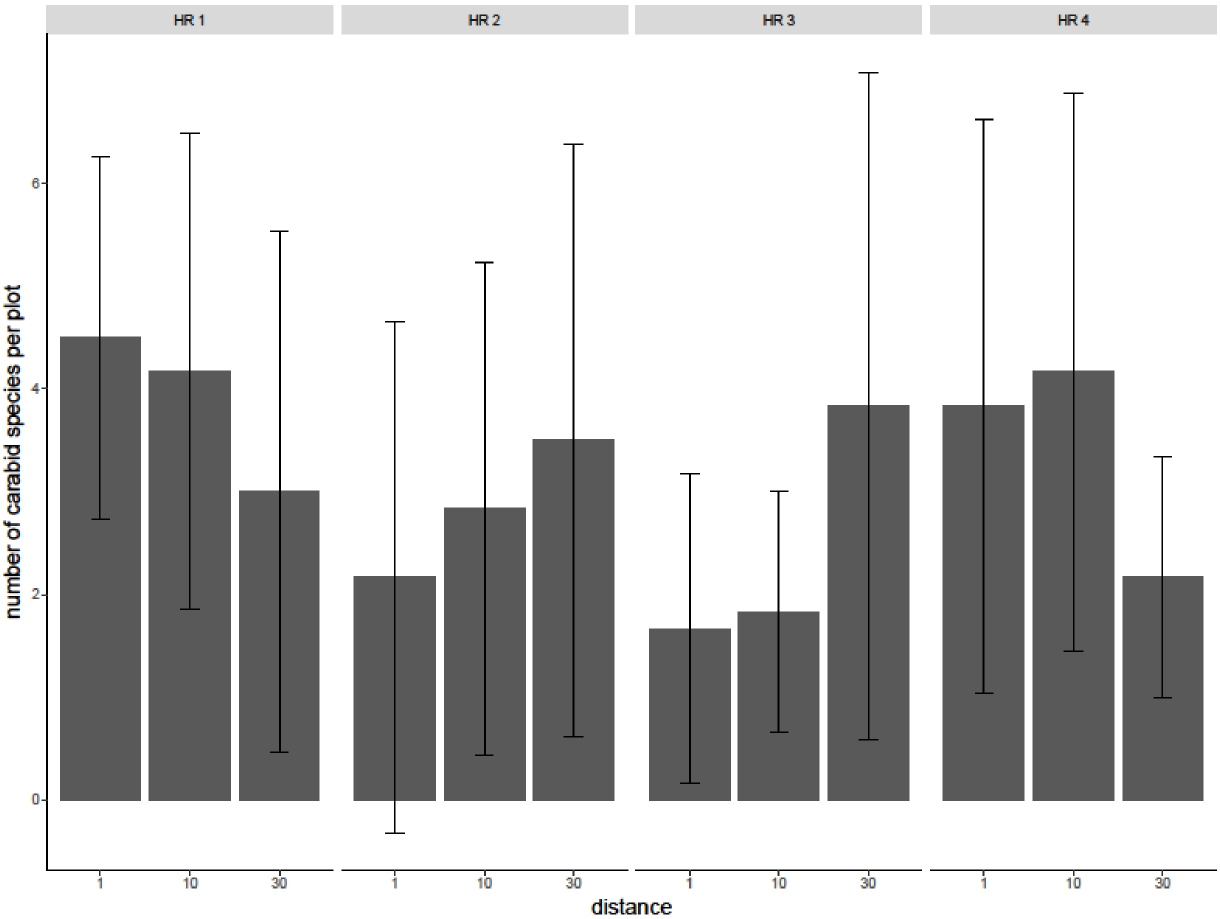


Figure 7.10.27: Average number of carabid species caught on the hedgerow parcels (HR 1 up to HR 4). Carabids were collected at 1 m, 10 m and 30 m from the hedgerow. Error bars represent standard deviations among samples from the same row and collections.

7.11 Carabid species compositions on arable parcels with grass strips

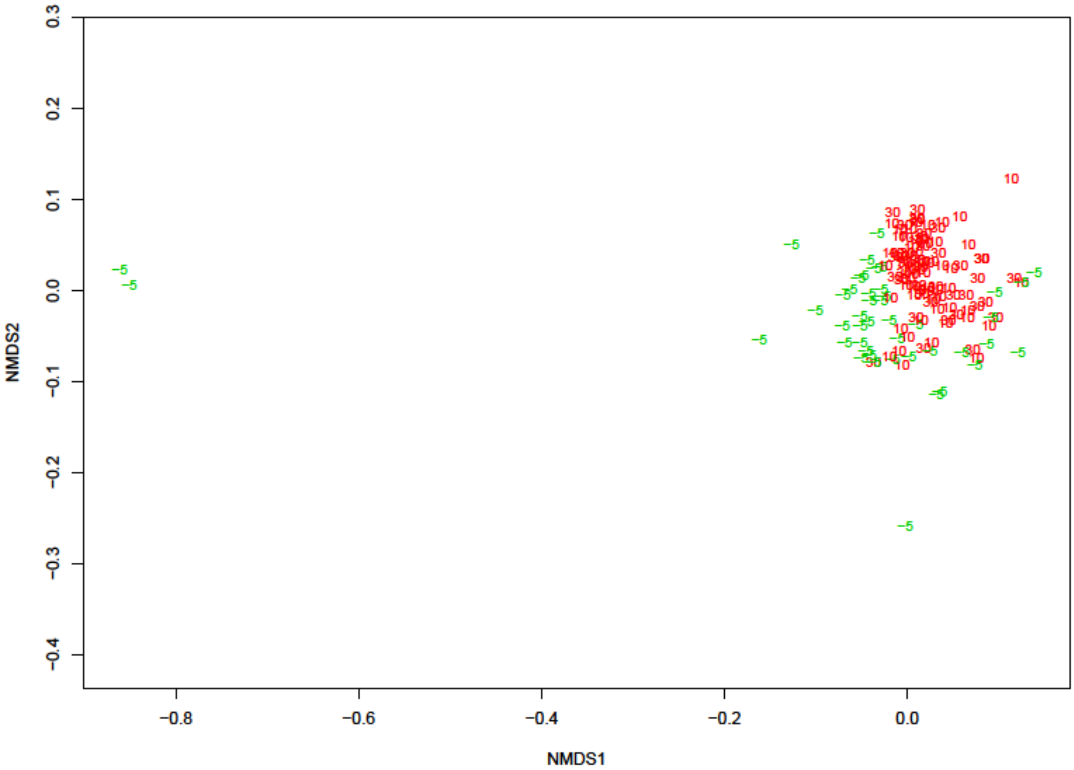


Figure 7.11.1: Non-metric multidimensional scaling (NMDS) plot representing dissimilarity in carabid species composition among different sampling positions on the grass strip parcels Position -5 is in the middle of the grass strip, position 10 is 10 m in the parcel and position 30 is 30 m in the parcel. Observations from the grass strip are in green, samples from the arable land are in red.

7.12 Overview of the measured ES indicators in TVG and BVA

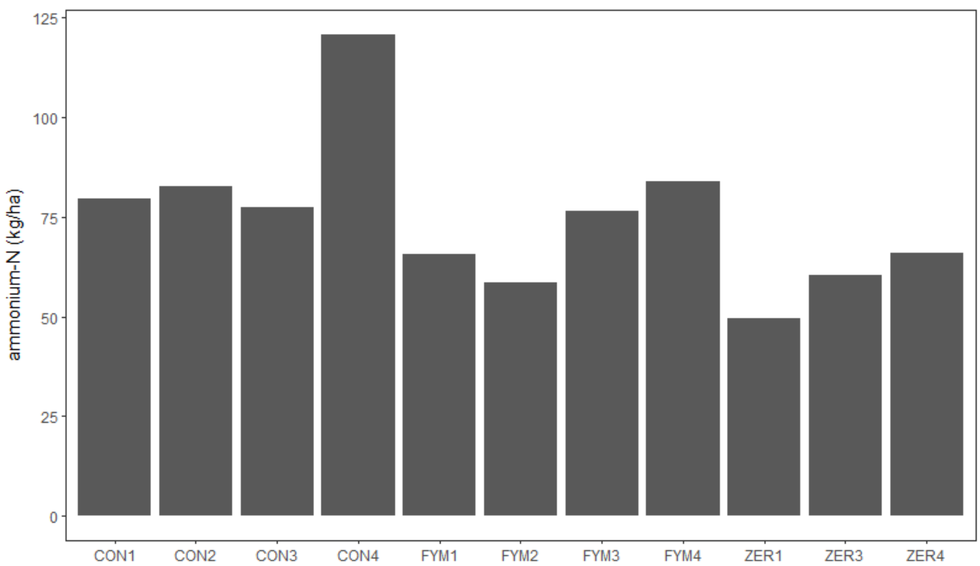
Table 7.12.1: Measured ES indicators for every parcel in TVG. CON grasslands were under a regular management. FYM grasslands received farmyard manure and had a delayed first cut and grazing. ZER grasslands received no fertilizers and had a delayed first cut and grazing. Crude protein concentration and organic matter digestibility of the first cut are given. Vegetation survey quadrats were 2 m x 2 m.

Parcel	Biomass yield (ton ha^{-1})	Crude protein concentration (%)	Organic matter digestibility (%)	SOC stock (ton ha^{-1})	Soil mineral N content (kg ha^{-1})	Number of carabid species	Number of plant species per quadrat
CON1	15.12±0.80	22.23±1.64	80.68±1.35	35.87±4.85	153.08	3.89±2.37	2.33±0.58
CON2	14.05±0.71	18.70±1.16	79.88±0.31	49.25±938	164.04	2.57±1.33	1.33±0.58
CON3	13.03±0.99	17.45±0.54	79.48±0.86	72.52±25.13	103.72	1.66±1.10	3.33±0.58
CON4	11.98±0.52	17.34±1.35	83.78±2.06	53.34±8.88	144.95	1.34±0.83	2.33±0.58
FYM1	12.62±1.23	15.47±1.13	80.26±1.48	51.37±1.44	73.89	3.73±1.81	8.33±1.53
FYM2	NM	NM	NM	55.86±11.68	61.94	1.77±1.70	4.33±2.08
FYM3	10.68±1.02	12.87±5.23	78.78±0.70	59.45±3.41	77.30	2.13±2.14	4.67±0.58
FYM4	7.65±0.45	11.61±0.37	77.82±2.34	43.71±9.21	84.55	3.21±2.25	4.67±1.15
ZER1	6.81±0.69	7.86±0.87	44.11±1.35	47.68±869	50.68	3.63±2.37	11.00±3.00
ZER2	7.33±0.79	9.57±0.50	61.18±4.40	NM	NM	2.40±0.88	10.33±2.51
ZER3	4.81±0.77	10.99±2.65	62.50±3.08	53.14±12.77	61.62	4.28±1.64	9.00±2.00
ZER4	5.98±092	10.46±0.47	63.71±2.85	55.21±1.79	66.89	3.00±1.27	7.67±1.53

Table 7.12.2: Measured ES indicators for every parcel in BVA. CON grasslands were under a regular management. ZER grasslands received no fertilizers and had a delayed first cut and grazing. Crude protein concentration and organic matter digestibility of the first cut are given. Vegetation survey quadrats were 2 m x 2 m.

Parcel	Biomass yield (ton ha^{-1})	Crude protein concentration (%)	Organic matter digestibility (%)	SOC stock (ton ha^{-1})	Soil mineral N content (kg ha^{-1})	Number of carabid species	Number of plant species per quadrat
CON1	10.34±2.19	15.47±0.49	77.31±7.20	75.95±20.30	56.03	1.56±01.01	4.67±1.15
CON2	14.03±0.71	16.02±1.29	73.69±3.01	89.17±8.88	77.64	2.00±1.07	4.33±0.58
CON3	11.47±1.48	16.91±0.45	76.27±1.34	84.20±3.88	152.52	1.89±1.27	3.33±0.58
ZER1	6.03±0.45	6.67±0.17	57.82±3.04	76.75±9.41	40.92	2.33±1.00	6.67±0.58
ZER2	4.44±0.79	9.10±2.55	64.74±5.48	70.69±11.74	40.90	0.43±0.53	12.00±1.73
ZER3	5.20±0.48	7.92±0.95	57.49±6.30	77.35±11.12	48.78	1.00±0.89	10.33±2.08

7.13 Ammonium-N and nitrate-N content monitored in the grasslands



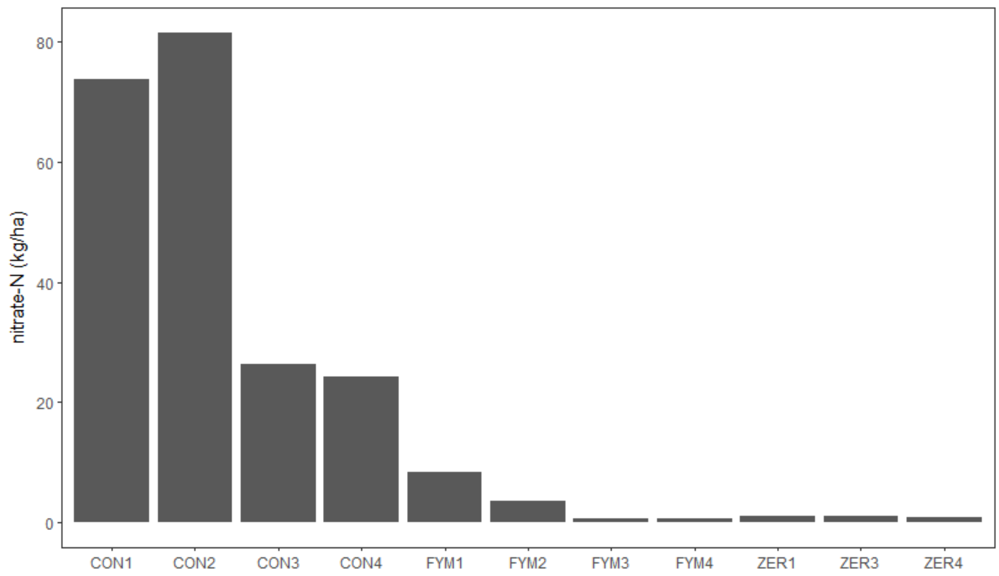
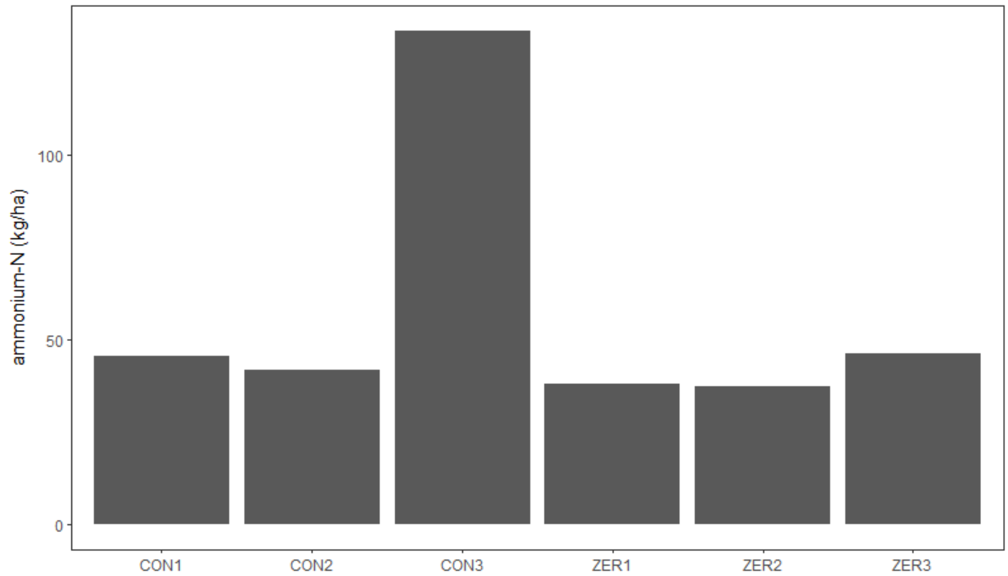


Figure 7.13.1: ammonium-N and nitrate-N content (kg ha-1) in the 0-90 cm soil layer in the monitored grasslands in TVG. CON grasslands were under a regular management. FYM grasslands received farmyard manure and had a delayed first cut and grazing. ZER grasslands received no fertilizers and had a delayed first cut and grazing.



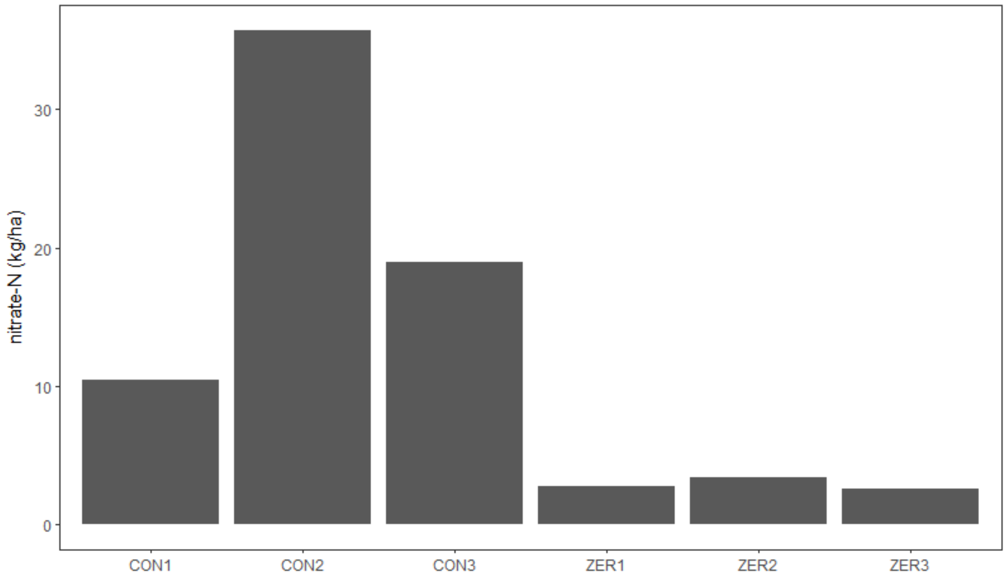


Figure 7.13.2: ammonium-N and nitrate-N content (kg ha-1) in the 0-90 cm soil layer in the monitored grasslands in BVA. CON grasslands were under a regular management. ZER grasslands received no fertilizers and had a delayed first cut and grazing.

7.14 Overview of the collected carabid species on the grasslands

Table 7.14.1: Overview of the collected carabid species, the abbreviation used in the NMDS plot and their relative activity-density in both study regions (TVG and BVA)

Species	Abbreviation	Relative activity-density (%)	
		TVG	BVA
Acupalpus brunnipes	AC.BRUN	0.13	0
Agonum muelleri	AG.MU	4.92	0
Amara aenea	AM.AE	1.94	0.62
Amara communis	AM.COM	0.26	3.11
Amara lunicollis	AM.LUN	1.55	0
Amara plebeja	AM.PLEB	0.52	1.24
Amara similata	AM.SIM	0.13	0
Amara tricuspidata	AM.TRI	0.13	0
Anchomenus dorsalis	ANCH.DO	0.13	1.86
Anisodactylus binotatus	AN.BI	2.59	1.24
Asaphidion flavipes	AS.FL	0.13	0
Bembidion guttula	BE.GU	0.13	0
Bembidion lampros	BE.LA	1.17	1.24
Bembidion properans	BE.PR	5.44	0.62
Bembidion tetracolum	BE.TET	0.13	1.24
Calathus fuscipes	CAL.FU	0.13	0
Carabus granulatus	CAR.GRA	0.91	7.45
Clivina fossor	CLI.FO	1.30	0.62
Harpalus affinis	HA.AF	0.13	0
Harpalus rufipes	HA.RU	0.78	0
Loricera pilicornis	LO.PI	0.52	0
Nebria brevicollis	NEB.BR	0.78	9.32
Nebria salina	NEB.SAL	1.04	1.86
Notiophilus spec.	NOT	0	0.62
Poecilus cupreus	POE.CU	3.76	4.35
Poecilus versicolor	POE.VE	23.19	60.87
Pterostichus melanarius	PTE.ME	42.75	0.62
Pterostichus niger	PTE.NI	0.52	0
Pterostichus strenuus	PTE.STR	0.39	0
Pterostichus vernalis	PTE.VE	4.40	3.11
Stenolophus teutonus	STE.TEU	0.13	0

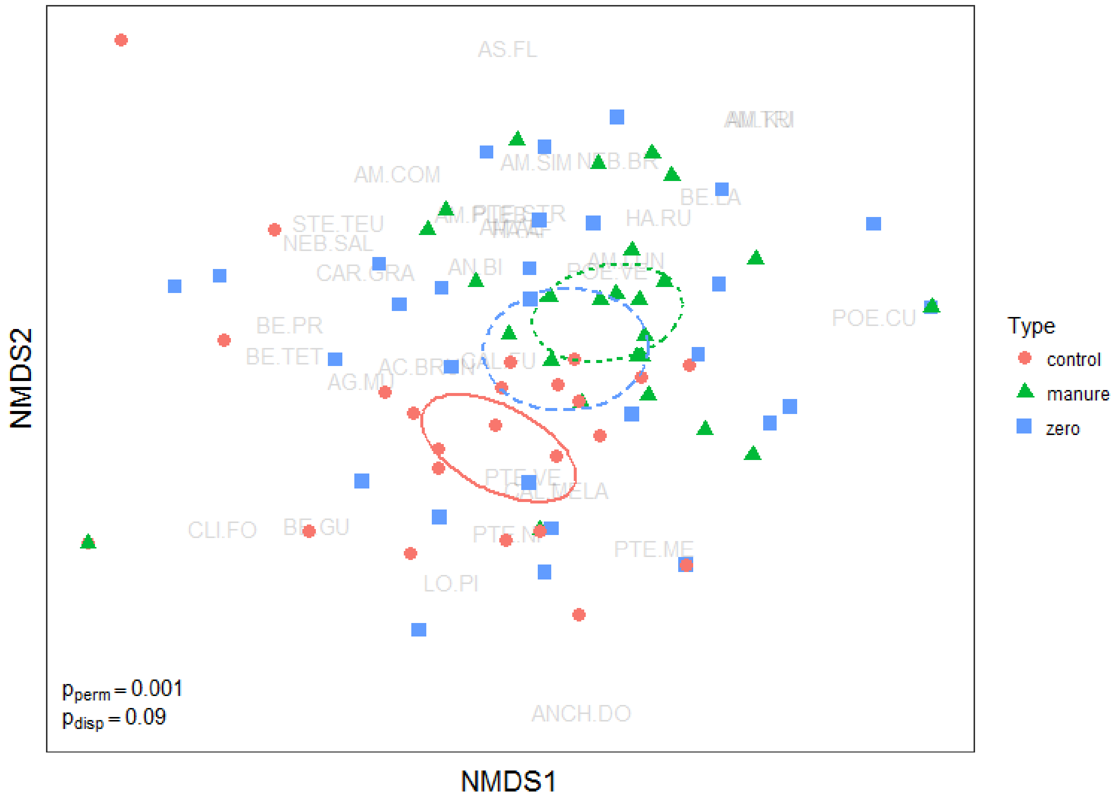


Figure 7.14.1: Nonmetric multidimensional scaling (NMDS) plot showing the carabid species and compositions found in the grasslands in TVG with varying management types species (k = 2, stress = 0.184). The p_{perm} value indicates the combined significance of the location and dispersion effect, based on PERMANOVA; the p_{disp} value indicates the significance of the dispersion effect, based on the function betadisper in R. Symbols represent the different management types (control, manure, zero). Lines show dispersion ellipses (1 standard deviation) around sample group centroids.

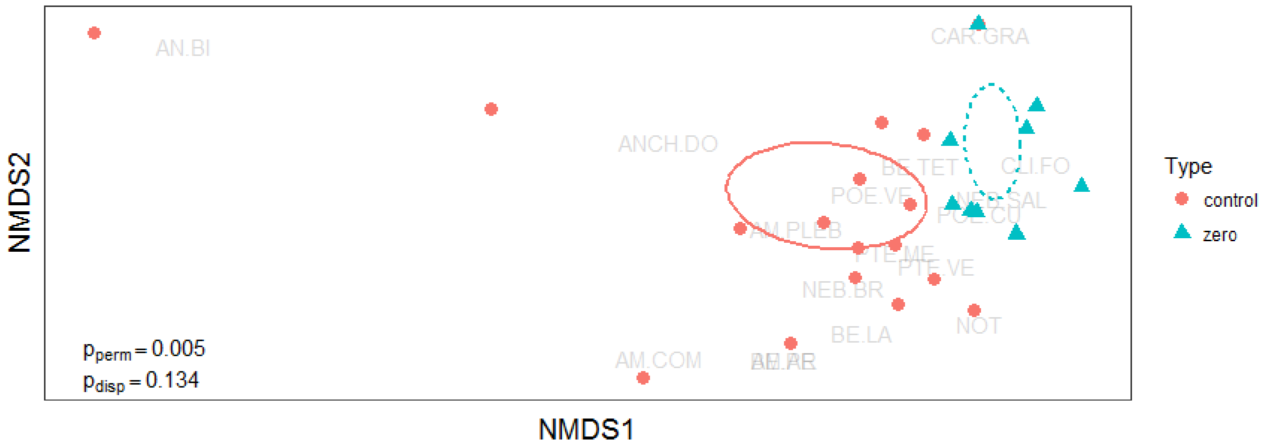


Figure 7.14.2: Nonmetric multidimensional scaling (NMDS) plot showing the carabid species and compositions found in the grasslands in BVA with varying management types species (k = 2, stress = 0.104). The p_{perm} value indicates the combined significance of the location and dispersion effect, based on PERMANOVA; the p_{disp} value indicates the significance of the dispersion effect, based on the function betadisper in R. Symbols represent the different management types (control and zero). Lines show dispersion ellipses (1 standard deviation) around sample group centroids.

7.15 Overview of the collected plant species on the grasslands

Table 7.15.1: Overview of the collected plant species, the abbreviation used in the NMDS plot, management types where they were found and their relative cover in both study regions (TVG and BVA)

Species	Abbreviation	Management type	Relative cover (%)	
			TVG	BVA
Agrostis capillaris	AG.TEN	control, manure, zero	10.46	24.00
Alopecurus geniculatus	ALO.GEN	zero	0.87	0
Anthoxanthum odoratum	ANTHO.ODO	zero	4.36	5.75
Bromus racemosus	BRO.RA	manure	0.05	0
Centaurea jacea	CENT.JA	zero	0.44	0.63
Centaurium erythraea	CENTAU.ERY	zero	0.05	0.36
Cerastium fontanum	CER.FO	manure, zero	0.44	0
Cerastium glomeratum	CER.VULG	zero	0.48	0
Cirsium arvense	CIR.ARV	manure, zero	0.61	0.18
Cynosurus cristatus	CYN.CRIS	manure, zero	1.77	0
Dactylis glomerata	DACT.GLO	control	0.39	4.04
Epilobium tetragonum	EPI.TET	zero	0.10	0.45
Eupatorium cannabinum	EUP.CAN	zero	0.02	0.04
Geranium dissectum	GER.DIS	manure	0.05	0
Holcus lanatus	HOL.LAN	control, manure, zero	34.29	14.34
Juncus compressus	JUNC.COM	zero	2.95	0.18
Juncus effusus	JUNC.EFF	zero	2.81	5.75
Lolium multiflorum	LOL.MUL	manure	1.16	0
Lolium perenne	LOL.PER	control, manure, zero	25.72	22.83
Lotus corniculatus	LOT.CORNICUL	zero	0.51	0.54
Lysimachia vulgaris	LYS.VULG	zero	0.02	0.18
Matricaria chamomilla	MATRIC.CH	manure	0.43	0
Persicaria bistorta	PER.BIST	zero	0	0.13
Phalaris arundinacea	PHALA.ARU	zero	0.02	0
Phleum pratense	PHLEU.PRAT	manure, zero	1.21	0
Plantago major	PLANT.MAJ	manure	0.02	0
Poa trivialis	POA.TRIVI	control	0	15.51
Ranunculus acris	RANUNC.AC	manure	0.05	0
Ranunculus repens	RANUNC.REP	manure, zero	3.54	0.67
Rumex acetosa	RUM.AC	manure, zero	1.26	0
Rumex obtusifolius	RUM.OBT	control, manure, zero	0.29	0.27
Silene flos-cuculi	SIL.CU	zero	0	1.35
Tanacetum vulgare	TAN.VULG	manure	0.05	0
Taraxacum campylodes	TARA.OFF	manure, zero	1.21	1.44
Trifolium dubium	TRIF.DU	manure, zero	0.53	0
Trifolium repens	TRIF.REP	control, manure, zero	3.80	0.72
Valeriana officinalis	VAL.OFF	zero	0	0.09
Vicia hirsuta	VICIA.HIRSU	manure, zero	0.02	0.54

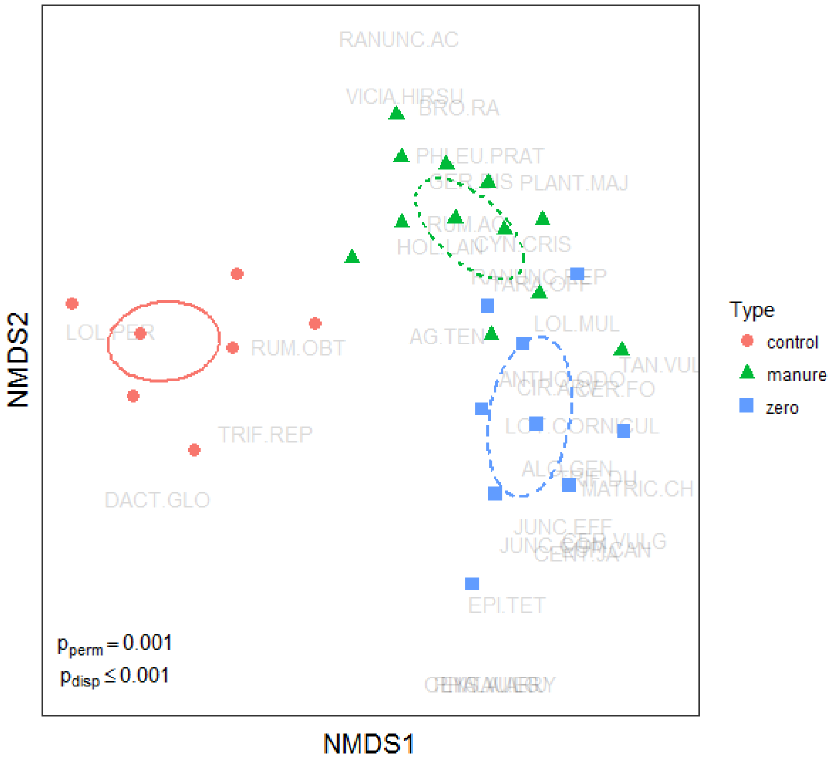


Figure 7.15.1: Nonmetric multidimensional scaling (NMDS) plot showing the plant species and compositions found in the grasslands in TVG with varying management types species (k = 2, stress = 0.145). The p_{perm} value indicates the combined significance of the location and dispersion effect, based on PERMANOVA; the p_{disp} value indicates the significance of the dispersion effect, based on the function betadisper in R. Symbols represent the different management types (control, manure and zero). Lines show dispersion ellipses (1 standard deviation) around sample group centroids.

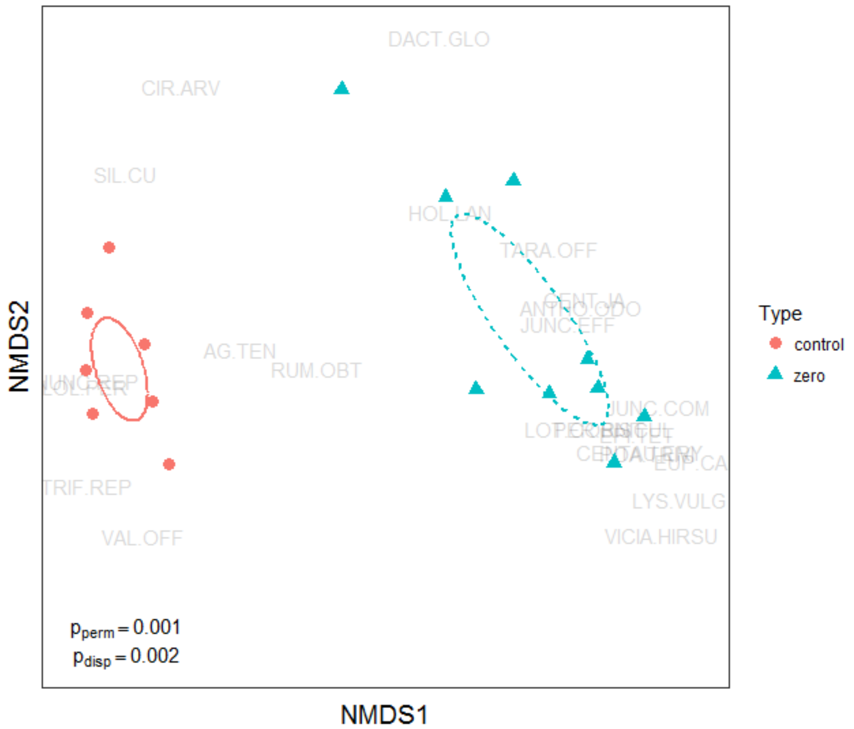


Figure 7.15.2: Nonmetric multidimensional scaling (NMDS) plot showing the plant species and compositions found in the grasslands in BVA with varying management types species (k = 2, stress = 0.074). The p_{perm} value indicates the combined significance of the location and dispersion effect, based on PERMANOVA; the p_{disp} value indicates the significance of the dispersion effect, based on the function betadisper in R. Symbols represent the different management types (control and zero). Lines show dispersion ellipses (1 standard deviation) around sample group centroids.

7.16 Experiments used to describe the effect of trees and hedgerows on crop yield

Table 7.16.1: Overview of the experiments that were used in the analysis. Own data are marked in grey. The various experiments are grouped per study. Per experiment, the authors, year of publication, monitoring location, type, tree species, tree height, tree row orientation and monitored crop are given. NG stands for not given. NS stands for north-south, NE-SW stands for northeast-southwest, EW for east-west and NW-SE for northwest-southeast.

Experiment	Authors	Year	Location	Type	Tree species	Tree height (m)	Tree row orientation	Crop
1	Rivest and Vézina	2014	Canada	tree row	*Picea glauca*	11	NS	maize
2	Rivest and Vézina	2014	Canada	tree row	*Picea glauca & Fraxinus pennsylvanica*	10	NE-SW	maize
3	Rivest and Vézina	2014	Canada	tree row	*Picea glauca*	11	NS	maize
4	Rivest and Vézina	2014	Canada	tree row	*Pinus sylvestris*	13	NE-SW	maize
5	Rivest and Vézina	2014	Canada	tree row	*Larix laricina*	8	NE-SW	maize
6	Rivest and Vézina	2014	Canada	tree row	*Picea glauca & Fraxinus pennsylvanica*	10	NE-SW	maize
7	Rivest and Vézina	2014	Canada	tree row	*Picea glauca*	11	NS	maize
8	Rivest and Vézina	2014	Canada	tree row	*Larix laricina*	8	NE-SW	maize
9	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	1.53	NS	barley
10	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	2.8	NS	barley
11	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	3.79	NS	barley
12	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	5.09	NS	beans
13	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	6.54	NS	wheat
14	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	8.35	NS	barley
15	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	1.49	NS	barley
16	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	2.54	NS	peas
17	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	3.49	NS	wheat
18	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	4.48	NS	wheat
19	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	5.83	NS	barley
20	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	7.52	NS	mustard
21	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	8.8	NS	wheat
22	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	5.61	NS	wheat
23	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	7.02	NS	wheat
24	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	8.43	NS	wheat
25	Burgess et al.	2004	Great Britain	tree row	*Populus sp.*	9.9	NS	beans

26	Chirko et al.	1996	China	tree row	*Paulownia*	12	NS	wheat
27	Gao et al.	2013	China	tree row	*Malus sp.*	2.4	EW	soybean
28	Gao et al.	2013	China	tree row	*Malus sp.*	2.5	EW	peanut
29	Reynolds et al.	2007	Canada	tree row	*Populus sp.*	12.1	NS	soybean
30	Reynolds et al.	2007	Canada	tree row	*Populus sp.*	11.1	NS	soybean
31	Reynolds et al.	2007	Canada	tree row	*Populus sp.*	12.3	NS	maize
32	Reynolds et al.	2007	Canada	tree row	*Populus sp.*	13.3	NS	maize
33	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	7.6	NS	soybean
34	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	8.5	NS	soybean
35	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	10.1	NS	maize
35	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	10.1	NS	maize
36	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	7.8	NS	maize
36	Reynolds et al.	2007	Canada	tree row	*Acer sp.*	7.8	NS	maize
37	Stamps et al.	2009	California	tree row	*Juglans nigra*	9.5	NG	alfalfa
38	Stamps et al.	2009	California	tree row	*Juglans nigra*	9.5	NG	alfalfa
39	Chaves	2001	Belgium	tree row	*Quercus robur*	19.6	NE-SW	maize
40	Chaves	2001	Belgium	tree row	*Quercus robur*	19.6	NE-SW	maize
41	Chaves	2001	Belgium	tree row	*Salix sp.*	17.4	NE-SW	maize
42	Chaves	2001	Belgium	tree row	*Salix sp.*	17.4	NE-SW	maize
43	Chaves	2001	Belgium	tree row	*Salix sp.*	9.1	NE-SW	maize
44	Chaves	2001	Belgium	tree row	*Salix sp.*	9.1	NE-SW	maize
45	Chaves	2001	Belgium	tree row	*Populus sp.*	28.6	NE-SW	maize
46	Chaves	2001	Belgium	tree row	*Populus sp.*	28.6	NE-SW	maize
47	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	3.1	NG	maize
48	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	3.6	NG	maize
49	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	4.3	NG	maize
49	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	4.3	NG	maize
50	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	3.5	NG	soy
51	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	3.9	NG	soy
52	Senaviratne et al.	2012	Missouri	tree row	*Quercus sp.*	4.6	NG	soy
53	Pardon	2015	Belgium	tree row	*Populus sp.*	22	NS	wheat
54	Pardon	2015	Belgium	tree row	*Populus sp.*	25	NS	wheat

55	Pardon	2015	Belgium	tree row	*Populus sp.*	20	NS	wheat
56	Pardon	2015	Belgium	tree row	*Populus sp.*	35	NS	barley
57	Pardon	2015	Belgium	tree row	*Populus sp.*	19	NS	maize
58	Pardon	2015	Belgium	tree row	*Populus sp.*	13	NS	maize
59	Pardon	2015	Belgium	tree row	*Populus sp.*	13	NE-SW	wheat
60	Pardon	2015	Belgium	tree row	*Populus sp.*	30	NS	maize
61	Pardon	2015	Belgium	tree row	*Populus sp.*	30	NS	maize
62	Pardon	2015	Belgium	tree row	*Populus sp.*	28	NS	barley
63	Pardon	2015	Belgium	tree row	*Juglans regia*	9	NS	wheat
64	Pardon	2015	Belgium	tree row	*Juglans regia*	9	NS	wheat
65	Esterka	2008	Czech Republic	hedgerow	*various*	13.2	NG	wheat
66	Chaves	2001	Belgium	hedgerow	*Alnus sp.*	9.2	NW-SE	maize
67	Chaves	2001	Belgium	hedgerow	*Alnus sp.*	9.2	NW-SE	maize
68	Chaves	2001	Belgium	hedgerow	*Crataegus sp.*	5.7	NS	maize
69	Woodall and Ward	2002	Australia	hedgerow	*Pinus radiata & Schinus areira*	10	NS	wheat
70	Woodall and Ward	2002	Australia	hedgerow	*Pinus radiata & Schinus areira*	10	NS	wheat
71	Woodall and Ward	2002	Australia	hedgerow	*Pinus radiata & Schinus areira*	10	NS	wheat
72	Van Vooren	2014	Belgium	hedgerow	*Crataegus monogyna, Sambucus nigra, Corylus avellana, Fraxinus excelsior*	5	NS	wheat
73	Van Vooren	2014	Belgium	hedgerow	*Carpinus betulus*	6	E-W	wheat
74	Van Vooren	2014	Belgium	hedgerow	*Crataegus monogyna*	2	NS	oats & peas
75	Van Vooren	2014	Belgium	hedgerow	*Sambucus nigra, Corylus avellana, Acer campestre, Fraxinus excelsior*	5	NE-SW	wheat

7.17 Comparison of the meta-regression and non-linear mixed model

Meta-regression

In Figure 7.17.1, data suitable for the meta-regression (with standard deviation given) are represented on the left. The size of each point is proportional with its weight, which is related to the standard deviation. We can see that the loess-line intersects zero at H=1.7. For H between 0 and 1.7, a meta-regression is performed. The result is shown on the right of Figure 7.17.1.

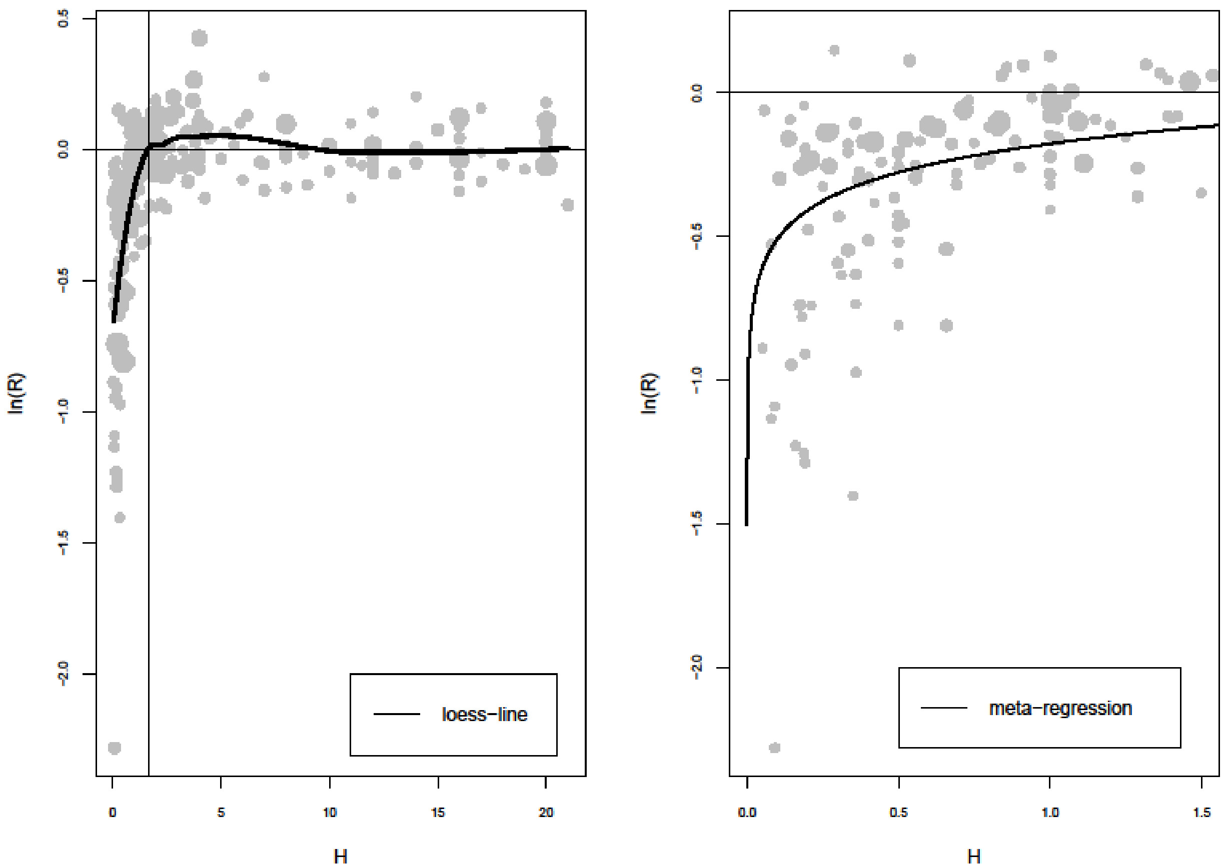


Figure 7.17.1: data used in the the meta-regression. The size of each point is proportional with its weight in the meta-regression. On the left, a loess-line is fitted to the data. On the right, a meta-regression is fitted.

Below, statistical information on the meta-regression is given:

```
Multivariate Meta-Analysis Model (k = 133; method: REML)

  logLik   Deviance       AIC        BIC       AICc
```

```
-465.4351   930.8702   936.8702   945.4958   937.0592

Variance Components:

          estim    sqrt  nlvls  fixed   factor
sigma^2    0.0164  0.1279     10     no  article

Test for Residual Heterogeneity:
QE(df = 131) = 1925.2363, p-val < .0001

Test of Moderators (coefficient(s) 2):
QM(df = 1) = 881.8560, p-val < .0001

Model Results:

         estimate      se     zval    pval    ci.lb    ci.ub
intrcpt   -0.1889  0.0415  -4.5513  <.0001  -0.2703  -0.1076  ***
mods       0.3314  0.0112  29.6961  <.0001   0.3095   0.3533  ***

---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

Non-linear mixed model

A non-linear mixed model is applied on the whole dataset (Figure 7.17.2)

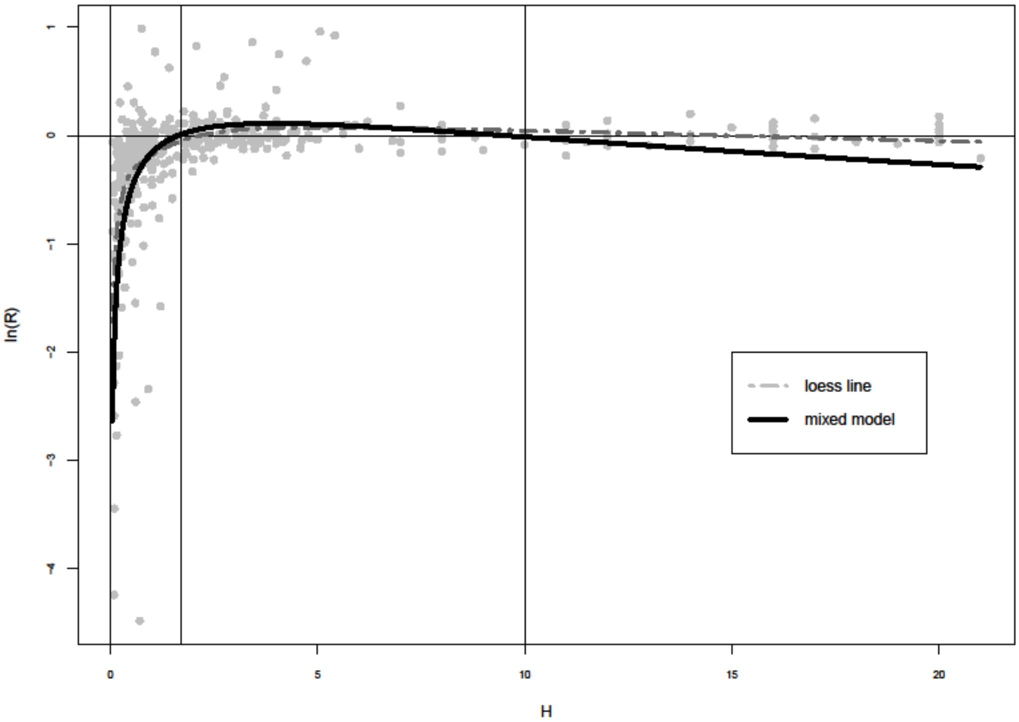


Figure 7.17.2: data that were used in the non-linear mixed model. A loess-line and the non-linear mixed model are fitted.

Below, statistical information on the non-linear mixed model is given:

```
Nonlinear mixed-effects model fit by maximum likelihood
  Model: lnR ~ a * logH^2 + b * logH + c
 Data: table
       AIC      BIC    logLik
  356.5799 396.9107 -168.2899

Random effects:
 Formula: list(a ~ 1, b ~ 1, c ~ 1)
 Level: article
 Structure: General positive-definite, Log-Cholesky parametrization
         StdDev    Corr
a        1.1538478 a      b
b        1.0678153 -1.000
c        0.2645301  0.831 -0.845
Residual 0.3288081

Fixed effects: a + b + c ~ 1
       Value  Std.Error  DF   t-value p-value
a -0.7632372 0.30213283 399 -2.526165  0.0119
b  0.9094616 0.27729901 399  3.279715  0.0011
c -0.1590143 0.07170714 399 -2.217551  0.0271
 Correlation:
  a      b
b -0.981
c  0.701 -0.749
```

```
Standardized Within-Group Residuals:
        Min          Q1         Med          Q3         Max
-7.99774435 -0.33267934 -0.04449248  0.32329235  3.78149620

Number of Observations: 417
Number of Groups: 16
```

In the meta-regression, all data are weighted by the inverse of the corresponding standard deviation. In the non-linear mixed model, this is not done. Therefore, it is necessary to investigate whether this affects the result of the analysis. To do this, we applied the non-linear mixed model to the meta-regression subset. Both models are shown in Figure 7.17.3. Between H=0 and H=1.7, the models are very similar. Because the non-linear models extends beyond H=1.7, this model will be used in the calculations.

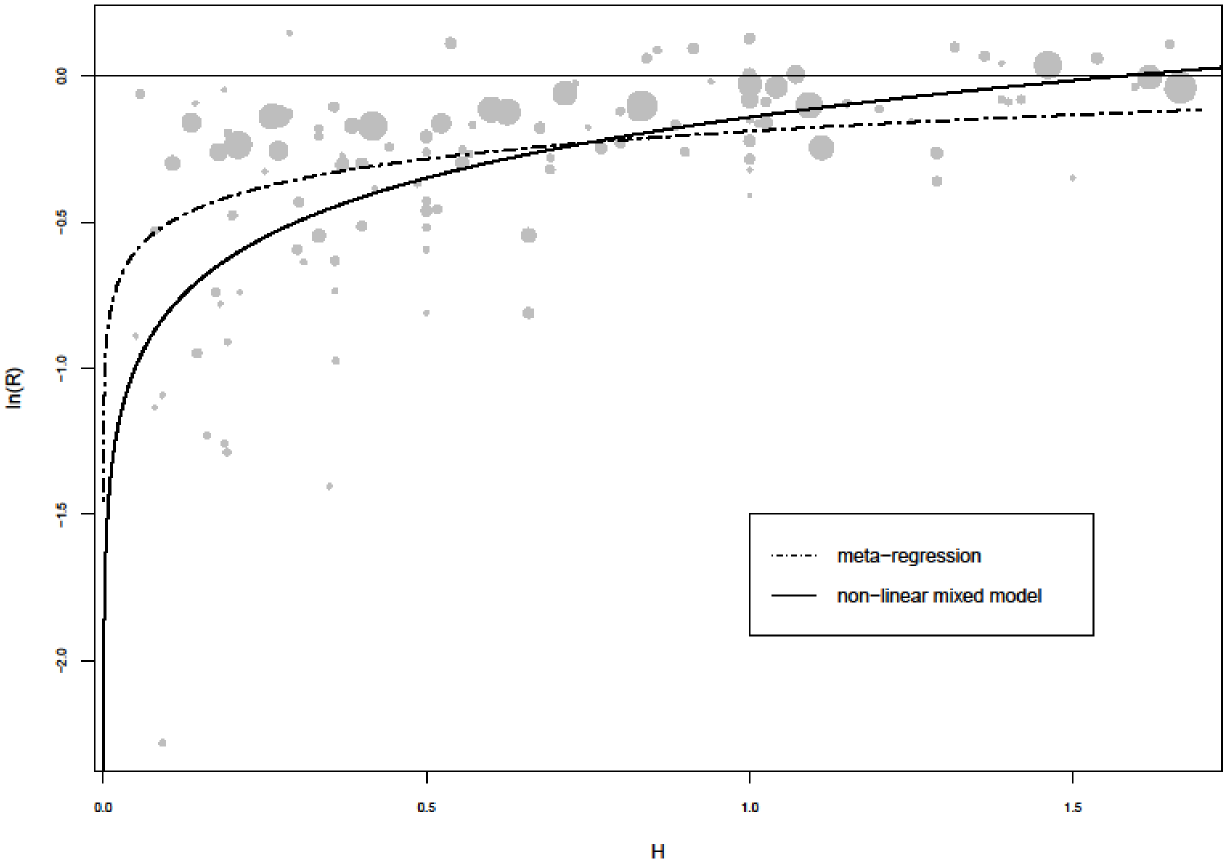


Figure 7.17.3: comparison of the meta-regression and the non-linear mixed model on the same dataset

7.18 Comparison of the effect of tree rows and hedgerows and of crop type on crop yield

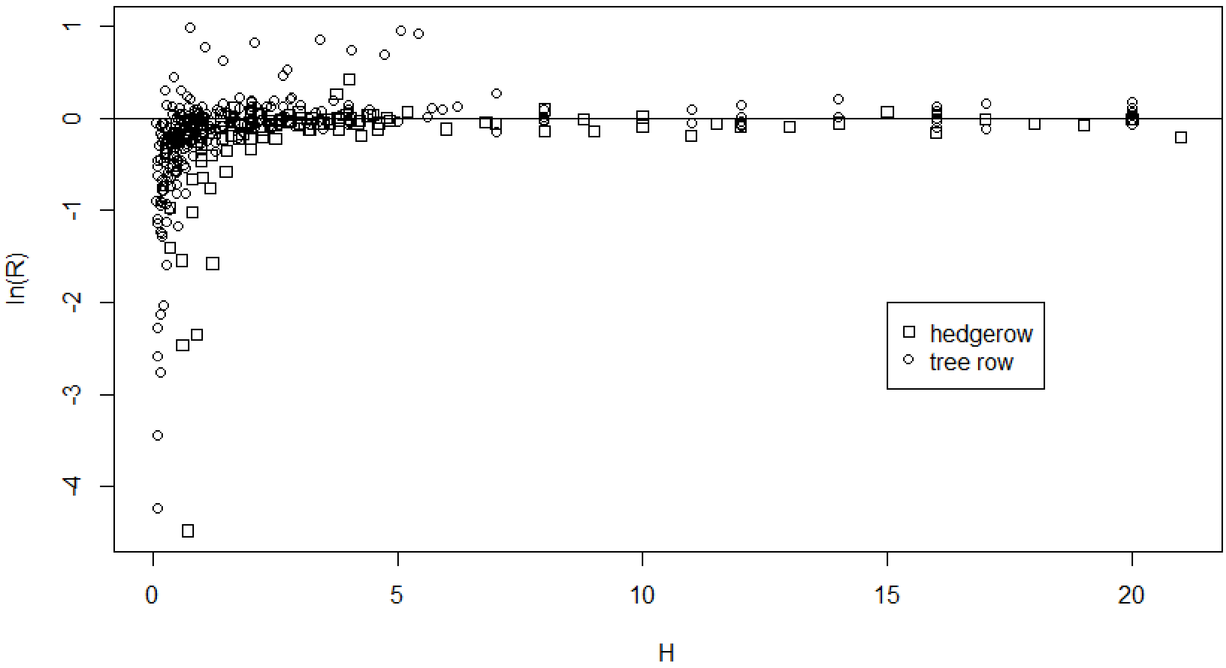


Figure 7.18.1: Relative crop yields (R) for varying relative distances (H) from the tree row and hedgerow. A distinction is made between studies based with tree rows and studies with hedgerows.

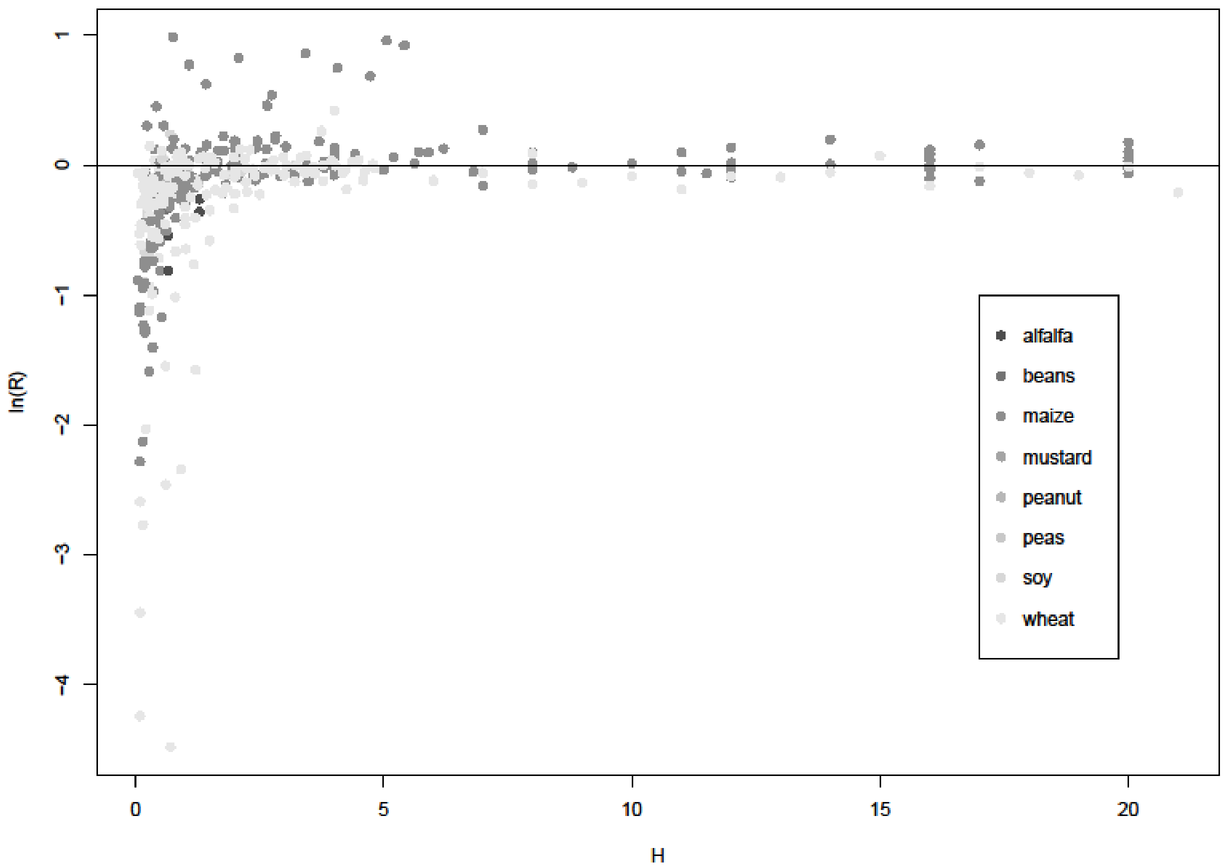


Figure 7.18.2: Relative crop yields (R) for varying relative distances (H) from the tree row and hedgerow. The various crops that were investigated are presented.

7.19 Description of the calculation tool

Required parameters to perform the calculations are: farm size, area per crop and layout of the hedgerow or alley cropping system. Maximum crop yield, crop revenues, costs etc. are given for every crop. Data are retrieved from Flemish Department of Monitoring and Study (2006-2012). Crop yield on parcels without hedgerows or tree rows equals the maximum crop yield. On parcels with hedgerows or tree rows, the area with these elements is taken out of rotation. On the cropped part of the parcel, a unidimensional grid of transects perpendicular to the hedgerows or tree rows is implemented. Space between grid points is 0.01 H. For every grid point, H to the nearest hedgerow or tree row is determined and this H is used to calculate relative crop yield based on the relation we derived from the non-linear mixed model. On every grid point, actual crop yield is relative yield multiplied by maximum crop yield. Yearly net parcel margins are the crop revenues minus the variable costs. Fixed costs and revenues are not included, they are supposed to be equal for the BAU and hedgerow and alley cropping option. Extra subsidies such as greening payments can be added to the yearly revenues. To combine the crop annual revenues with the initial tree plantation and maintenance costs and wood revenues at the end of the rotation, a discounted cash flow is calculated. The discounted cash flow is computed as the annual difference between income and costs, and for every year, this difference is discounted in accordance with the opportunity cost of an investment assuming risk adversity. General textbooks on agricultural business planning advise to use a discount factor between 5% and 20%, depending on the opportunity costs and risks of the investment. The higher the discount factor, the higher the opportunity costs and risk adversity and the more an immediate cash flow is preferred to a future cash flow. Choice of discount rate is thus context specific. James and Eberle (2000) state that long-term land investments do not have a discount rate as high as shorter-term machinery investments, so we initially chose the lower bound of the eligible discount rates. Discounting factor is thus set at 0.05. Finally, the discounted gross margins of the hedgerow and alley cropping systems are compared to the discounted gross margin of a BAU system.

7.20 Hedgerow and alley cropping scenario development

A concrete hedgerow scenario was developed to comply the EFA conditions and the Flemish situation. Although hedgerows can take many forms and dimensions, which most probably will have an impact on the final economic consequences, only one type is considered here. The example is based on following assumptions:

i) the hedgerow is coppiced every ten years

ii) after ten years, a height of 10 m is reached. This is the maximum height for most species that are typically used in hedgerows and is the average height of a hedgerow network that was surveyed in the north-east of Flanders (Deckers et al., 2004)

iii) a hedgerow width of 7.5 m (Deckers et al., 2004)

iv) no maintenance costs, because we assume the utilities of the wood compensate the maintenance work, which is either done by volunteers or by the farmer himself

v) no other edge effects are present.

We assume that the farmer, besides implementing the hedgerow option, also meets the other greening conditions and receives the greening payment, which is here supposed to be € 100 ha^{-1}. For a hedgerow width of 7.5 m, necessary length to meet the EFA requirements is 1500 m (7.5 m * 1500 m = 1.125 ha)

Management operations and costs that we included are given in Table 7.20.1

Table 17.20.1: management operations and costs for the hedgerow option. Data extracted from Normenboek Bos, Natuur en Landschap 2014

Management action	Cost	Timing
Plant purchase	0.12 € $tree^{-1}$	Year 1
Planting	1.52 € $tree^{-1}$	Year 1
Plant protection	0.94 € $tree^{-1}$	Year 1

In the alley cropping option, poplar trees (*Populus* sp.) are planted. Tree height during the rotation is based on Jansen et al. (1996). An average rotation length of 20 years is used in the case study. To allow compatibility with normal agricultural management, distance between the tree lines is set at 35.5 m (to allow for a fluent passage of machinery), distance between trees within the same line is set at 6 m. Tree row width is 2 m. This results in a tree density of 52 trees ha^{-1}, which satisfies the EU requirements for greening payments (Pillar I) as well as the requirements for the EU Rural Development submeasure 8.2 (Pillar II) as implemented in Flanders (Vlaamse Regering, 2014). Alley cropping is only a valid EFA option if it is planted in accordance with the EU Rural Development submeasure 8.2. We assumed that the trees were planted on an arable field, so no additional soil preparation was necessary. The included management operations and costs are given in Table 7.20.2.

Table 7.20.2: management operations and costs for the alley cropping option. Data extracted from Normenboek Bos, Natuur en Landschap 2014 and expert advice

Management action	Cost	Timing
Plant purchase	0.12 € tree^{-1}	Year 1
Planting	1.52 € tree^{-1}	Year 1
Plant protection	0.94 € tree^{-1}	Year 1
Weed removal around tree	12 trees per hour 30 € per hour	Year 1 – 5
Formation pruning	4 trees per hour 30 € per hour	Year 4, year 7, year 10
Tree row maintenance	100 m per hour 30 € per hour	Yearly

Tree row maintenance mainly consists of removal of undesired species from the tree row. In contrast to weed removal around the tree (years 1-5), where its main goal is to avoid competition for water and nutrients, tree row maintenance seeks to remove invasive and aggressive species. However, the soil can still be covered with herbaceous species.

From Graves et al. (2007), who modelled poplar growth in the Netherlands for a density of 50 trees ha^{-1} with Yield-sAFe, we deduct a wood production of 2 m^3 tree^{-1} after 20 years, resulting in 104 m^3 of wood production per hectare. In 2014, average price for standing poplar timber (pruned) was € 27.5 m^{-3} (De Mey et al., 2013).

7.21 Crop yields, costs and revenues used in the calculation tool

Table 7.21.1: crop yields, costs and revenues used in the analysis. Source: Flemish Department of Monitoring and Study, agricultural monitoring network (data 2006-2012)

Winter wheat	
Revenues	
Physical yield of the main product (kg ha^{-1})	8600
Price per unit of the main product (€ kg^{-1})	0.200
Physical yield of the straw (kg ha^{-1})	3000
Price per unit of the straw (€ kg^{-1})	0.063
Costs	
Seed and propagating material (€ ha^{-1})	96.60
Fertilizers (€ ha^{-1})	190.5
Crop protection (€ ha^{-1})	196.2
Contract work (€ ha^{-1})	83.1
Other costs (€ ha^{-1})	12.5
Winter barley	
Revenues	
Physical yield of the main product (kg ha^{-1})	8000
Price per unit of the main product (€ kg^{-1})	0.178
Physical yield of the straw (kg ha^{-1})	3100
Price per unit of the straw (€ kg^{-1})	0.0476
Costs	
Seed and propagating material (€ ha^{-1})	99.9
Fertilizers (€ ha^{-1})	153.3
Crop protection (€ ha^{-1})	138.5
Contract work (€ ha^{-1})	78.4
Other costs (€ ha^{-1})	11.2
Maize	
Revenues	
Physical yield of the main product (kg ha^{-1})	12000
Price per unit of the main product (€ kg^{-1})	0.0796
Costs	
Seed and propagating material (€ ha^{-1})	167.6
Fertilizers (€ ha^{-1})	125
Crop protection (€ ha^{-1})	101.2
Contract work (€ ha^{-1})	87.6
Other costs (€ ha^{-1})	19.6
Sugar beet	
Revenues	
Physical yield of the main product (kg ha^{-1})	87000
Price per unit of the main product (€ kg^{-1})	0.0311
Costs	
Seed and propagating material (€ ha^{-1})	234.4
Fertilizers (€ ha^{-1})	224.9
Crop protection (€ ha^{-1})	263

Contract work (€ ha^{-1})	89.3
Other costs (€ ha^{-1})	86.1
Potatoes	
Revenues	
Physical yield of the main product (kg ha^{-1})	36000
Price per unit of the main product (€ kg^{-1})	0.1190
Costs	
Seed and propagating material (€ ha^{-1})	621.1
Fertilizers (€ ha^{-1})	341.6
Crop protection (€ ha^{-1})	591.3
Contract work (€ ha^{-1})	168.5
Other costs (€ ha^{-1})	73.5

Table 7.21.2: Evolution factors of crop prices (between 2006 and 2012), used in $SR_{agricultural\ prices}$. Source: Flemish Department of Monitoring and Study, agricultural monitoring network.

Crop	Variable	Evolution factor
Winter wheat	Main product selling price	1.11
	By-product selling price	1.07
Winter barley	Main product selling price	1.13
	By-product selling price	1.07
Potatoes	Main product selling price	1.0
Sugar Beets	Main product selling price	1.06
Maize	Main product selling price	1.07

7.22 Experimental setup

Data collected by Pardon et al.:

In 2015, crop yield was measured on 13 conventional arable fields partially bordered by a tree row. Fields were spread throughout Flanders and soil types varied from sandy loam to loamy. Winter wheat was grown on six fields, maize on four fields and winter barley on two fields. Estimated tree heights varied between 9 m and 35 m. On the wheat and barley parcels, plots of 1.5 m x 6.5 m were harvested. On the maize parcels, the experimental plot had a length of 5 m and consisted of two maize rows. Plots were situated on transects perpendicular to the tree rows and measurements were done on at approximate distances of 2 m, 5 m, 10 m, 20 m and 30 m. On the same parcel, at the part that is not bordered by trees, control measurements were done on the same distance from the field edge. On the tree row part of the parcels, every distance was replicated 3 times. On the control part of the parcel, every distance was replicated two times. Tree species were *Populus* sp. and *Juglans regia.*

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9. Curriculum vitae

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Education

2011-2013	MSc. in Bioscience Engineering, Forest and Nature Management, Ghent University (Belgium). Including an Erasmus exchange in Cordoba (Spain) in 2011
2008-2011	Bsc. in Bioscience Engineering: Land and Forest Management, Ghent University (Belgium)
2002-2008	Secondary School, Latin-Sciences, O.L.V. Ten Doorn College, Eeklo

Scientific activities

Peer-review scientific articles included in Web of Science

Published or in press

D'Hose, T., Molendijk, L., **Van Vooren, L.,** van den Berg, W., Hoek, H., Runia, W., van Evert, F., ten Berge, H., Spiegel, A., Sandèn, T., Grignani, C., Ruysschaert, G. (accepted for publication in Pedobiologia). Responses of soil biota to non-inversion tillage and organic amendments: an analysis on European multiyear field experiments.

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Van Vooren, L., Reubens, B., Broekx, S., Reheul, D., Verheyen, K. Assessing the impact of grassland management extensification in temperate areas on multiple ecosystem services and biodiversity: monitoring and a literature review. Submitted to Agricuclture, Ecosystems & Environment, revisions requested.

Pardon, P., Reheul, D., Mertens, J., Reubens, B., De Frenne, P., De Smedt, P., Proesmans, W., **Van Vooren, L.,** Verheyen, K. Gradients in abundance and diversity of ground-dwelling arthropods in temperate silvoarable fields. Submitted to Agriculture, Ecosystems and Environment.

Scientific reports

Nelissen, V., Van Gossum, P., Reubens, B., Ruysschaert, G., D'Hose, T., Pardon, P., **Van Vooren, L**. 2016. Maatregelen om het ESD-aanbod van landbouw te verhogen. Natuurraport – Aan de slag met ecosysteemdiensten. Technisch Rapport. INBO, Brussels, Belgium.

Participation in congresses, symposia or workshops

Oral presentations

27.03.2015. Brussel. Starters in het natuur- en bosonderzoek

25.03.2016. Brussel. Starters in het natuur- en bosonderzoek

10.06.2016. Brussel. BEES young scientists day

23.06.2016. Montpellier. Greening and producing: an economic assessment framework for integrating trees in cropping systems. EURAF Congres.

20.09.2016. Gent. Ecosystem services of agri-environment schemes on arable land: a synthesis for hedgerows and grass strips. Belgian Agroecology Meeting

05.10.2017. Melle. Ecosysteemdiensten op akkers met houtkanten en grasstroken

Poster presentations

17 december 2014. Gembloux. BEES Christmas Market

27 maart 2015. Brussel. Starters in het natuur- en bosonderzoek

2 – 12 september 2015. Peyresq. ALTER-Net Summer School on biodiversity and ecosystem services

Scientific courses

2014	Statistics – Flames Summer School
2014	Introduction to R – doctoral schools
2015	ALTER-Net Summer School on biodiversity and ecosystem services
2015	Advanced Academic English Writing Skills – doctoral schools