EFFECTIVENESS OF CONSERVATION CROPPING PRACTICES IN MITIGATING RUNOFF, SOIL EROSION, AND PESTICIDE SURFACE LOSSES IN NORTHWESTERN EUROPE

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ABSTRACT

Soils of the Western European loess belt are intensively cultivated and prone to runoff and soil erosion. Besides threatening soil resources, overland flows cause muddy floods that damage infrastructure downstream and pollute water bodies by transferring nutrients and pesticides. Numerous conservation farming practices are known to mitigate surface flows, but the magnitude of their effectiveness remains uncertain in Northwestern Europe. Based on a systematic review and meta-analysis, we estimated the runoff and erosion mitigation effects of conservation (i.e., non-inversion) tillage, winter cover crops, and tied-ridging (i.e., micro-basin tillage in potato fields). These conservation practices decreased seasonal soil losses by 66%, 72%, and 92%, respectively, compared to conventional tillage. For conservation tillage, we further identified key factors explaining the variability of the flow mitigation effects, including the time since ploughing was stopped, the type of implemented conservation tillage scheme (e.g., number of operations), and the main crop type (winter or spring crop). Besides its use in potato crops, micro-basin tillage has also recently been developed in Belgium for maize crop, using an innovative agricultural roller designed to shape depressions in-between the maize rows. Based on past plot experiments, micro-basin tillage in maize was found to reduce runoff (-69%), soil erosion (-83%), and associated pesticide transfer (e.g., -65% for the active ingredient flufenacet). Lastly, two uncommon conservation practices in Northwestern Europe were investigated through original plot experiments in maize crops: undersowing and strip-tillage. Undersowing (simultaneous sowing) red fescue or white clover in maize was very challenging in terms of weed control and resulted in no mitigation effect on surface flows compared to a conventionally tilled sole maize crop. Strip-tillage (i.e., tine-tilling the maize row only) significantly decreased runoff (-31%) and soil erosion (-60%). A hydrological process-based model was further calibrated based on the measured flow data in strip-tilled and conventionally-tilled maize plots. Subsequently, a scenario analysis was conducted to assess the conservation potential of the strip-tillage technique under extended climate, soil and slope conditions. Simulations reveal that strip-tillage

cultivation would halve both the mean erosion rate as well as the flood risk (maximum daily runoff), as compared to conventional tillage. Unfortunately, available data did not allow to evaluate the impact of conservation practices on pesticide transfer in a conclusive way, neither from the quantitative review nor from our field trials. Meanwhile, the outcomes of this thesis can assist farmers, farm advisors or policy makers in (promoting) the implementation of effective conservation farming practices.

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ABBREVIATIONS

CI	Confidence interval
C_K	Ksat decay coefficient
(SCS-)CN	(Soil Conservation Service) Curve number
CREHDYS	Continuous Runoff and Erosion Hillslope Model with DYnamic soil Surface properties
d50	Median particle diameter
GLM	Generalized linear model
Ksat	Saturated hydraulic conductivity
NSE	Nash-Sutcliffe model efficiency coefficient
NW	Northwestern
RR	Response ratio
ST	Strip-tillage
USDA	United States Department of Agriculture
USLE	Universal soil loss equation

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CHAPTER 1: INTRODUCTION AND OBJECTIVES

Many scientific definitions exist for Soil, although they could all roughly agree on this parsimonious one: the uppermost layer of the earth's crust, made up of mineral and organic particles forming a coherent unit, providing habitat for plant and animal growth (Hartemink, 2016). Soil is indeed by far the primary substrate for our food production, providing more than 95% of the global supply (Panagos et al., 2022). More recently, as our understanding of soil-related processes progresses, other functions have been attributed to soils. The main recognized soil-related ecosystem services are carbon sequestration, nutrients cycling, the regulation of pests and diseases, and water purification and regulation (Evangelista et al., 2023; Kibblewhite et al., 2018; Panagos et al., 2022). The latter refers to the soil's buffer capacity to infiltrate and release water, particularly during heavy rainfall and drought periods respectively, through the maintenance of a good soil structure (Blanchy et al., 2023). Rainwater that does not infiltrate the soil runs off and can trigger flooding and erosion by water (Yin et al., 2018). In agricultural areas, it can carry nutrients and pesticides polluting surface water downstream (Wang & Li, 2019; Stehle & Schulz, 2015). The **water regulation function of soil** is all the more essential in view of climate change, which is (already) increasing the frequency and intensity of extreme weather events (IPCC, 2023).

However, soil degradation – whether physical, chemical, and/or biological – impairs the ability of soil to fulfil its functions (Jie et al., 2002). Besides salinization, acidification, nutrient imbalance, pollution and sealing, soil erosion is considered one of the most severe threats to soils worldwide (Kraamwinkel et al., 2021). It is broadly defined as the accelerated removal of topsoil from the land surface through water, wind or tillage (Amanullah et al., 2017). This loss of soil material jeopardizes soil productivity, but also causes side-effects such as the transfer of sediments downstream or the loss of soil organic carbon (Blanco-Canqui & Lal., 2010). These can in turn activate a positive feedback loop, whereby lower vegetation, reduced capacity of landscape features to retain water, and/or altered soil structure further increase the soil's vulnerability to erosion (Morgan, 2005). At the global scale, it is estimated that 6% of the land surface exceeds the generic tolerable soil erosion threshold of 10 t/ha/year, and a further 9% suffers

from erosion rates considered as moderate or high. Although cropland covers only about 11% of the area worldwide, it is responsible for 50% of the total soil loss (Borrelli et al., 2017). Besides overgrazing and deforestation, agricultural mismanagement is indeed a main anthropogenic pressure driving soil erosion (Jie et al., 2002). In light of the primary role of agriculture in this issue, this thesis focuses on cultivation practices that support the soil ecosystem service of controlling surface flows of water, sediments, and pesticides as pollutant, in the context of Northwestern (NW) Europe.

1.1 Overland flow in Northwestern European cropland

1.1.1 Pressures

Overland flow issues in NW Europe are closely related to soil vulnerability and agricultural management. During the Quaternary Glacial period, the last 2.6 million years, sediments were transported by wind from polar deserts bordering the ice caps, settling down between the English Channel and the river Rhine, forming a loess deposit called the "**(Western) European loess belt**" (Bakels, 2009). According to pedological classification, these loess soils typically evolved into Luvisol, or Cambisol when younger (Tóth et al., 2008). The loess layer is generally several meters thick, consisting mainly of silt particles (Haase et al., 2017). This textural content, which classifies them as **(silt) loam soils** according to the USDA, results in **low soil structural stability** which is a first factor of vulnerability to surface flows. Silt particles are the most susceptible to detachment and transport by the kinetic energy of raindrops or water flow, and therefore to soil erosion, because coarser-particle (sandy) soils have greater infiltration capacity, and finer particles (clay) provide better soil cohesion. Once dislodged from the soil aggregates, the silt particles clog the pores on the soil surface, forming a **surface crust** that drastically lowers infiltration rates, thereby enhancing the risk of runoff (Morgan, 2005).

Furthermore, loess soils feature good internal drainage and high plant-available water capacity, making them remarkably fertile (Catt, 2001). They are therefore **intensively cultivated**, which is a source of additional vulnerability to overland flow (Figure 1.3). **Annual crops**, to the

detriment of grassland, account for a large share of arable land (Ballot et al., 2023), restricting organic matter return to the soil, especially when crop residues are exported (Potter et al., 1997), which is the case for most spring crops. **High tillage** levels further dilute soil organic matter across the tillage depth, eventually weakening the topsoil structure (Meersmans et al., 2009; Jensen et al., 2019). Meersmans et al. (2009) calculated that, for cropland in Belgium, the mass of soil organic carbon in the top 30 cm of silt loam soils decreased by 22% on average between 1960 and 2006. Mechanised agriculture increases **soil compaction** through the use of heavy machinery, reducing soil porosity and therefore infiltration capacity (Hamza & Anderson, 2005; Hamlett et al., 1990). Topsoil compaction can easily be remedied by tillage which increases soil porosity in the short term, but this is more complicated in the case of subsoil. In tillage systems relying on the repeated use of a moldboard plough at the same depth, compaction appears as a 'plough pan' or 'hardpan' at the maximum tillage depth, a transition zone that is visible between the tilled horizon and the compacted subsoil. The traffic of heavy farm machinery on moist soil, typically during (late) harvests in autumn of spring crops such as sugar beet, maize or vegetables, is a critical situation leading to compaction (Batey, 2009).

1.1.2 Processes

But above that, **spring-sown crops** leave the soil bare or poorly covered between April and June, when the risk of erosive rainstorms is increasing in NW Europe regions (Ballabio et al., 2017). Without being intercepted by vegetation, raindrops possess maximum kinetic energy, causing rapid soil surface sealing (crusting) and high splash detachment. The latter is a first erosion process, that breaks up aggregates and disperses soil particles, leading to a net soil transfer downstream on sloping terrain. It also exposes soil particles to further transport by overland flow, which is then referred to as sheet or interrill erosion (Osman, 2014). In spring conditions, runoff is typically generated following rainstorms, the latter being characterized by a high rainfall volume over a short period of time, when the rainfall intensity exceeds the soil infiltration capacity. This is known as **Hortonian runoff**, in contrast to saturation-excess runoff occurring when rain falls on a soil whose water content is (already) at saturation (Morgan, 2005). The latter

type of overland flow can also trigger flooding, but following low-intensity, long-lasting events, typically during the winter period, in regions predominated by winter cereals such as the South Downs in the UK (Boardman et al., 1994). Hortonian runoff occurring in late spring and early summer clearly is the main source of **muddy floods** in most areas, and is responsible for the majority of annual soil losses in agricultural catchments (Vandaele & Poesen, 1995).

Once the overland flow velocity is high enough to reach a shear stress greater than the cohesive forces of the soil particles, the latter are detached and transported by the flow (Morgan, 2005). The flow shear stress and transport capacity (maximum sediment concentration), driving sediment detachment and redeposition respectively, are both proportional to flow velocity. When runoff shear velocity exceeds a critical value, surface runoff can be channeled into topographically defined concentrated flow paths called rills (Blanco-Canqui & Lal, 2010). When an eroded channel exceeds 900 cm^2, it is referred to as gully erosion. NW European landscapes are subject to ephemeral gullies, as they can be remedied by tillage, though frequently reoccurring in the same place (Vandaele & Poesen, 1995). Splash, interrill, rill, and gully erosion are increasingly localized but also increasingly severe erosion processes, respectively (Verheijen et al., 2009).

Interrill, rill, and gully erosion are grouped under the heading of "**water erosion**", and account for average soil loss rates of the order of a few to a few tens of t/ha/year in the European loess belt (Vandaele & Poesen, 1995; Auerswald et al., 2009; Panagos et al., 2015c). Other non-water-induced erosion processes also contribute to soil erosion overall. Erosion by wind amounts to no more than about one t/ha/year on average in Europe (Verheijen et al., 2009), and is more prevalent in the northern regions of NW Europe with light sandy soils (Borrelli et al., 2014). Erosion by tillage results from the combined effect of tillage dragging forces and gravity. It does not result in a net soil loss at the field scale, but spatially redistributes the soil within the field from convex to concave slopes. The degree of the process is proportional to the slope steepness and the intensity of the tillage operations, on the order of a few t/ha/year locally (Van Oost et al., 2009). Lastly, erosion due to harvesting arises from the export of soil stuck to harvested plant

products. It particularly occurs with root crops (sugar beet, chicory, vegetables, potato) and is strongly dependent on soil moisture conditions at harvest. Given the high incidence of these crops in NW Europe (Ballot et al., 2023), harvest erosion may account for a few additional t/ha/year on average (Boardman & Poesen, 2006; Saggau et al., 2024).

Unlike winter cereals, spring crops are therefore a major cause of soil vulnerability with regard to runoff and erosion. According to the FAO (2023), in Western Europe (including Austria, Belgium, France, Germany, Luxembourg, Netherlands and Switzerland), maize is the dominant spring crop (Figure 1.1a), accounting for almost half (49%) of the spring crop-cultivated area on average over the 2017-2021 period. It is followed by sugar beet (23%) and potato (17%). Interestingly, the share of the total cropland between winter and spring crops is approximately constant since 1960's, standing at 76/24%, respectively, on average between 1961 and 2021 (Figure 1.1b).

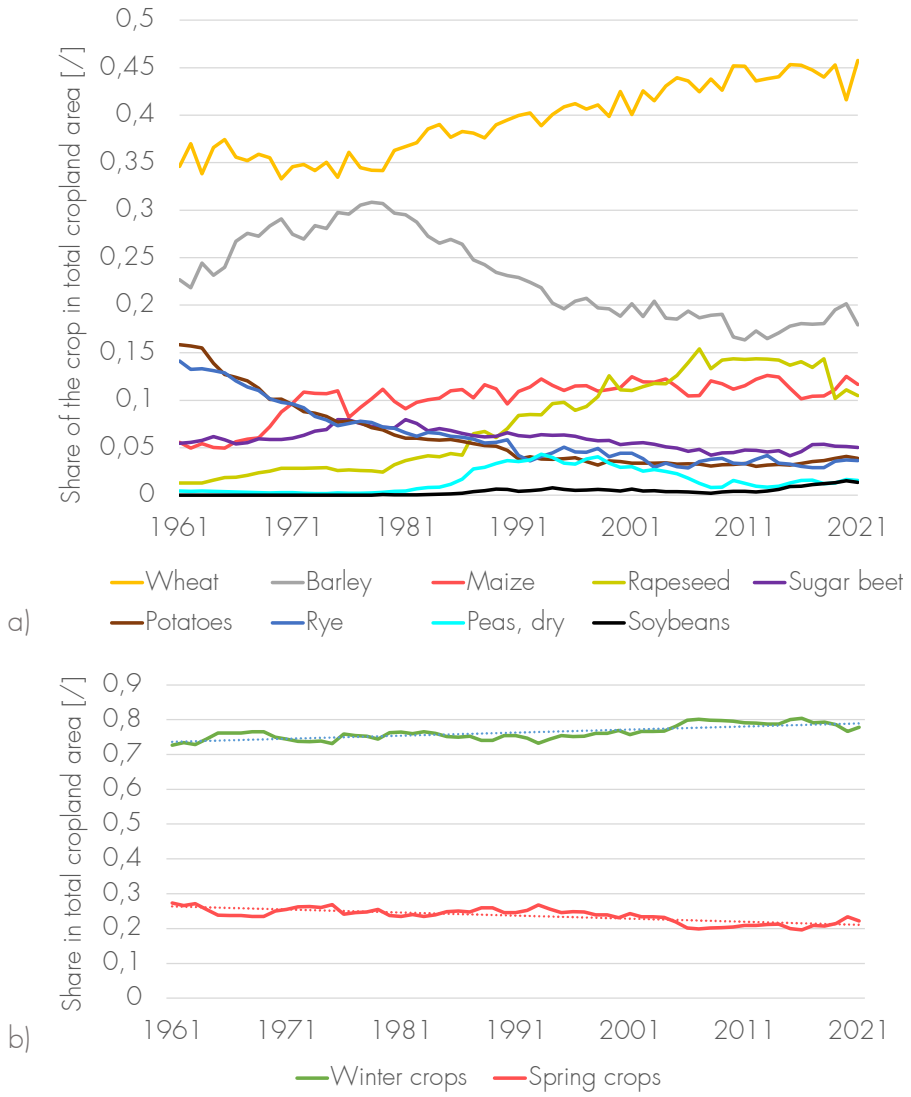


Figure 1.1 : Evolution of the share of crop types in the total cropland area in Western Europe between 1961 and 2021, adapted from FAO (2023). Minor crops such as fruits, vegetables, and rice were excluded. a) per individual crop type. b) per crop type family: winter (cereals and rapeseed) and spring (other) crops.

1.1.3 Consequences

Regardless of the crop type, summing up all the erosion processes outlined above, the overall soil loss rate in the European loess belt would average at least a few tons per hectare per year, but could in

some cases exceed 11 t/ha/year, a value historically considered as a generic threshold of **"tolerable" erosion rate** (Osman, 2014). However, the latter is derived from the USA context, assuming that tillage and incorporation of fertilizers and organic matter increase the formation rate of the A horizon, with the aim of maintaining soil productivity over a few decades, and given an initially thick soil layer (Morgan, 2005). More recently, this threshold value has been criticized as being too loose in view of a more comprehensive approach to sustainability. It was argued that the acceptable soil loss rate should be closer to soil formation rates by bedrock weathering, or maximum erosion rates under native vegetation, i.e. around 2 t/ha/year (Montgomery, 2007; Verheijen et al., 2009). According to Panagos et al. (2015c), multiple provinces in the Western European loess belt experience average erosion rates (water only) between 2 and 5 t/ha/year (Figure 1.2). In Belgium, model estimates for water and tillage erosion indicate that the proportion of agricultural land under erosion risk is 19% in Wallonia and 14% in Flanders (Gentile et al., 2009).

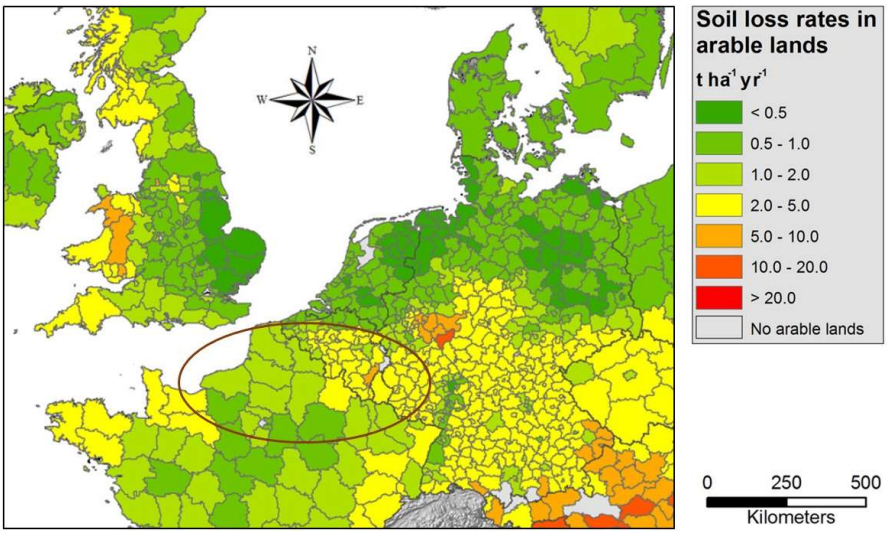


Figure 1.2 : Mean water erosion rate at province (NUTS3) level for arable lands in the EU, assessed using RUSLE model, adapted from Panagos et al. (2015c). The dark brown outline indicates the extent of the Western European loess belt according to Bakels (2009).

Behind the above-mentioned average erosion rates lie **significant temporal and spatial variabilities**. Beyond the random nature of rainfall events, there is an intra-annual variability, related to the seasonality of rainfall erosivity (Ballabio et al., 2017) in conjunction with that of the vegetation protecting the soil from erosion. Inter-annually, more (spring crops) or less (winter cereals) erosion-sensitive crops often alternate in crop rotations (Gabriels et al., 2003). Spatial variability is mainly due to topography (Panagos et al., 2015a). Steeper slopes increase net splash erosion as well as overland flow velocity, hence sheet or rill erosion. Longer slopes increase the contributing area, resulting in higher flow discharge, therefore greater soil detachment as well (Morgan, 2005). Soil erodibility is another key factor in the spatial variability in field erosion rates. As previously mentioned, high silt and low organic C content both favor surface crusting and detachment of soil particles, explaining the high erosion sensitivity of loess soils overall (Panagos et al., 2014).

Erosion threatens soil productivity in the long term, as the repeated removal of soil may ultimately reduce the explorable layer for crop roots. So far, this effect has been marginal in the Western European loess belt, because the thickness of the loess layer, several meters at least, allows a wide margin for soil loss (Bakker et al., 2007). Locally, however, erosion processes can affect soil profiles and therefore productivity, primarily on steep slopes (where the A horizon is reduced by long-term erosion) and in depressions (where sediment deposition can lead to slumping and crusting). Reyniers et al. (2009) found significant correlations between such landscape patterns and local wheat yields in a Belgian field. It is also argued that the effect of soil erosion on productivity is present but overshadowed by benefits from other agronomic factors on crop yields such as genetic improvements through varietal selection or better soil management (Morgan, 2005). Nonetheless, the main consequence of overland flow and soil erosion in NW Europe is definitely the occurrence of **muddy floods**, which harm both infrastructure and surface water downstream. According to a survey by Bielders et al. (2003) in the Walloon loess belt, 68% of the municipalities faced at least one flooding event attributable to agricultural runoff during a single decade. Evrard et al. (2007a) estimated that the total societal cost of muddy floods for central

Belgium ranges between 16 and 172 million euros each year, including cleaning and repairing of facilities (roads, houses, sewers, etc.) and fire brigade interventions. The methodology of this study may have led to an overestimation, as it appears to be based on an average cost per muddy flood event, without considering variations in severity. Most muddy flood events have a minor impact and, therefore, incur limited costs. More recently, Dogot et al. (2015) indeed downscaled the total cost estimate to around 600,000€ per year, for the whole Walloon region. Further including the building and maintenance of erosion control measures, for the Seine-Maritime area of Normandy, the mean societal cost of runoff and soil erosion was evaluated to 4 319 €/km²/year (Patault et al., 2021). Environmental externalities are not accounted for in these studies. Overland flow is indeed the primary pathway for the **diffuse transfer of nutrients and pesticides to surface waters** (Oenema et al., 2005; Huber et al., 2000). Nutrients from agricultural fertilizers are transported mainly in the dissolved phase in runoff for nitrogen (as nitrate) and adsorbed to sediment in the case of phosphorus (as phosphate), causing eutrophication in streams, rivers, lakes and seas (Chardon & Schoumans, 2007). Nutrient concentrations in EU surface waters are in excess of thresholds congruent with good ecological status in 57% of freshwater monitoring stations (Nikolaidis et al., 2022). Depending on the physico-chemical properties of the active substances, pesticides are transported preferentially in the dissolved (runoff) or particulate-bound phase (Rodríguez-Cruz & Sánchez-Martín, 2022). In Germany, 83% of agricultural streams do not meet the pesticide-related ecological targets (Liess et al., 2021).

The overall context of the intensively cultivated soils of the Western European loess belt thus combines multiple factors of vulnerability to overland flows (Figure 1.3). Densely populated regions are particularly exposed to muddy floods. This is the case, for instance, in the Pays de Caux in Normandy and South Limburg in the Netherlands, where numerous research efforts have been undertaken in the field (both figuratively and litterally) over the last decades (Richet et al., 2017; Winteraeken & Spaan, 2010). In Belgium, the Walloon region faces a similar situation, especially in the central silt loam area. Fortunately, a number of flow mitigation measures are now established (see Section 1.2). However, their effectiveness is uncertain and variable, and

predicting their (combined) impact within a watershed remains a huge challenge (Maetens et al., 2012a). Against this background, the **Intell'eau project** was initiated by Walloon universities and research organizations, and funded by the regional government. Running from 2020 to 2026, it aims to develop a decision support tool to assist and optimize the implementation of soil and water conservation techniques in the agricultural landscape. This tool is intended for use by farmers and agricultural advisors. In Wallonia, the primary advisors targeted are the dedicated staff from the “GISER” unit (part of the regional authority) and from the non-profit organization “Natagriwal”. They provide recommendations to tackle muddy floods issues, targeting municipalities and individuals in the former case, and farmers in the latter. This thesis was funded by the Intell’eau project and focuses on assessing the effectiveness of (in-field) conservation farming practices, with a view to supplying input for the future decision support tool.

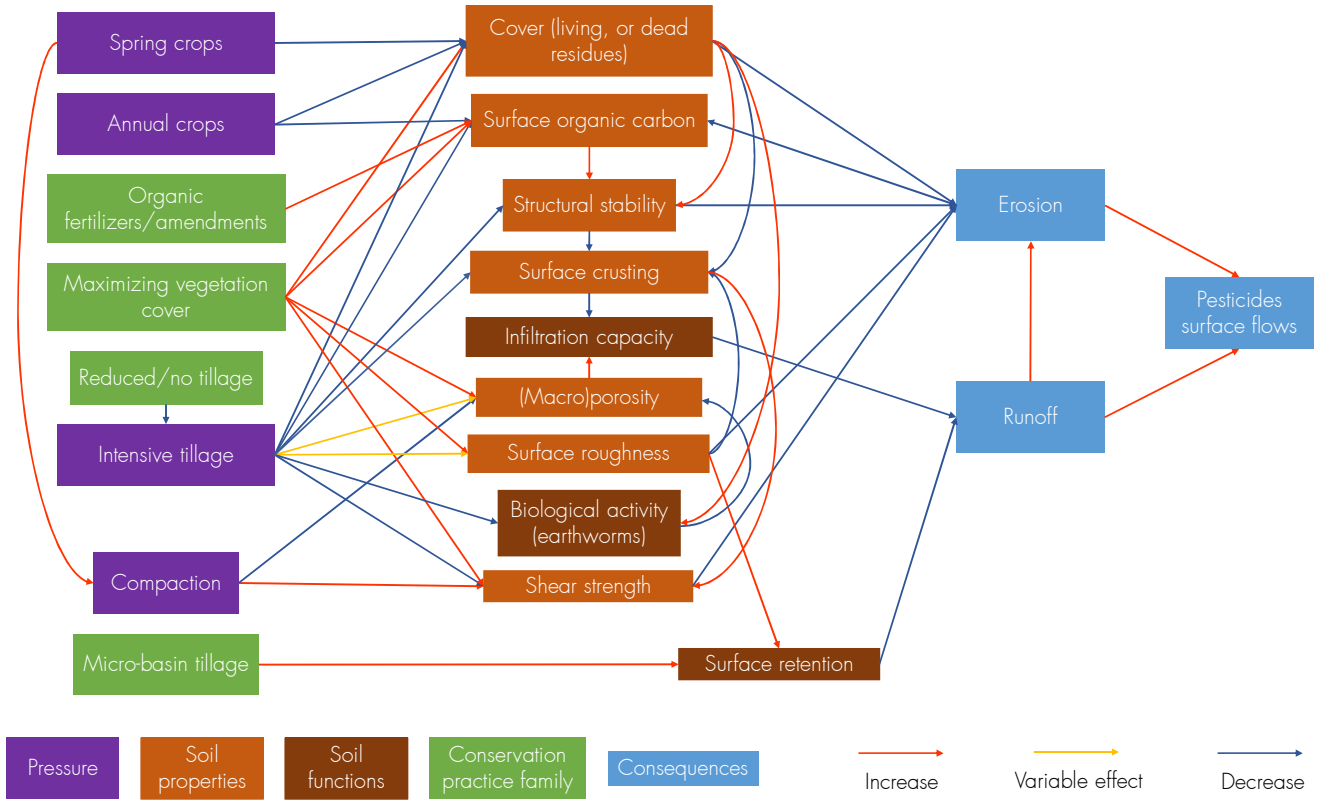


Figure 1.3 : Schematic diagram of how anthropogenic agricultural drivers (pressures from intensive agriculture, and soil conservation practices) affect soil physical properties and functions, and ultimately overland flow-related consequences (runoff, erosion, and surface flows of pesticides). Natural factors such as climate, topography, or soil texture are thus not represented.

1.2 Soil and water conservation

With the aim of tackling (adverse impacts of) overland flows and soil erosion, the principle of soil conservation has been introduced (Bennett, 1939). In line with the chronological recognition of soil functions, conservation was initially aimed merely at keeping the soil in place to maintain crop production, but then extended to other soil functions such as reducing water surface flows and pollution, hence the broader concept of **soil and water conservation**. For some, it even includes mitigating the concentration of greenhouse gases in the atmosphere (Blanco-Canqui & Lal, 2010).

In accordance with the objective of retaining water and soil in situ, any conservation approach should favor all in-field practices that limit runoff production and the detachment and transport of soil particles. Such **conservation farming practices** can be classified as follows (see Figure 1.4, based on the classification from Morgan (2005)).

1.2.1 Cropping practices

- **Vegetation**: This includes any practice consisting in diversifying and/or providing vegetation in addition to a main crop, in time or space. The cover provided by the additional vegetation intercepts raindrops, reducing its kinetic energy and hence splash erosion and surface crusting. A high density of plant material at the soil surface also provide hydraulic roughness reducing runoff velocity, hence sheet and rill erosion (Petryk & Bosmajian, 1975). The relationship between soil loss and the percentage cover is exponentially decreasing; compared to a bare soil, erosion can be halved with only 30% of vegetation cover (Morgan, 2005).

  Root growth-decay cycles create (macro)porosity in the soil, which is essential for water infiltration (Lu et al., 2020). Roots also strengthen the soil by adhering to its particles, increasing soil cohesion (Vannoppen et al., 2017). By diversifying the cultivated species, the services provided are also optimized; e.g. plants with taproot systems are beneficial for vertical macroporosity, while fibrous roots maximize the soil's stability.

Any plant biomass that is not harvested, in addition to leaving residues that also physically protect the soil (provided they are not buried by tillage), is a source of organic matter. As it decomposes, different kinds of organic binding agents are progressively released: transient (~ weeks), temporary (~ months), and persistent (~ years) binding agents, each ensuring the water-stability of the soil aggregates (Tisdall & Oades, 1982).

In the field, vegetation-based conservation practices can take the form of higher-density planting of the main crop, crop rotations that alternate more and less erosion-sensitive crops in time, or strip cropping that intersperse them in space (e.g. every 6, 12 or 24 meters, allowing independent cultivation of each strip). In NW Europe, high-density planting can be relevant when broadcast sowing cereals, under which it was proven to effectively mitigate winter erosion (Gyssels et al., 2002). This practice would, however, little address low cover under spring crops in May-June, and hence spring muddy floods, especially as they are sown in rows, thereby complicating the control of the seeding density. Crop rotations are almost always applied for many other reasons than soil conservation, and its effect on soil erosion is well established (Gabriels et al., 2003). In contrast, strip cropping is almost non-existent in NW Europe, as it considerably complicates labor organization (Morel et al., 2020), especially given the small plot sizes. For these reasons, these practices will not be further investigated in this thesis.

In the category of vegetation-based conservation cropping techniques, we will address in more detail **winter cover crops** and **intercropping** as part of this thesis. The former consists in covering the soil during the autumn-winter period with an unharvested (catch) crop, i.e. between (the harvest and sowing of) two main crops, typically after a winter cereal and before a spring crop (Martin, 1999). Intercropping refers to the mixed cultivation of two different crops simultaneously on the same plot, in mutual dependence, i.e. through joint cropping operations, as opposed to strip cropping (Wall et al., 1991).

- **Mulching**: This implies providing (several tens of %) cover by applying at the soil surface an external material. It can be either of natural origin such as straw, grass or wood from another field, for which the expected action on the soil is similar to that of a crop residue kept on the surface, or of synthetic origin such as a geotextile mulch. The technique is highly effective in fighting soil erosion, but its cost makes it of little relevance for most crops (Prosdocimi et al., 2016). Moreover, the potential for mulching is highly dependent on material availability, which is inherent to the farming system. Nevertheless, the recent extensive planting of hedgerows in the agricultural landscape of NW Europe (e.g. in Wallonia), could open up a new avenue for chipped wood mulching in the near future.

1.2.2 Soil management

- **Tillage:** This family of conservation practices covers tillage techniques intended to influence soil structure or roughness. This includes contour tillage, which consist in carrying out cropping operations along the contour lines, i.e. across the slope. The aim is to create a surface roughness oriented perpendicularly to the flow direction so as to limit its velocity and prevent concentrated flows, thus soil detachment (Blanco-Canqui & Lal, 2010). However, the technique should be restricted to relatively regular slopes (without pronounced thalwegs) with limited steepness (<10%), and it is challenging to apply in small-size fields, which are common in NW Europe.

  Historically intended for water-harvesting in the world's arid climates, **(micro)basin tillage** – also known as reservoir tillage or tied-ridging – consists in forming, by tillage operation, depressions on the soil surface to increase surface retention and temporarily store (part of) the water in infiltration excess during a rain event, to allow delayed infiltration (Jones & Stewart, 1990). Recently, the development of mechanized equipment designed for basin tillage in Europe has opened up promising possibilities for overland flow control, focusing on spring crops with wide inter-row spacing, i.e. potato and maize (Sittig et al., 2020).

Among all soil management techniques involving tillage, **conservation tillage** is by far the most prominent. This generic term encompasses a wide range of (sub)practices, listed here in decreasing order of restrictiveness. The USDA sets a criterion of at least 30% soil cover to be classified as conservation tillage (Morgan, 2005). Other authors emphasize a reduction in the tilled area or the number of tillage operations (up to direct drilling) (Osman, 2014). A common standard for conservation tillage is the absence of inversion tillage (i.e. ploughing) (Cannell, 1985; Holland, 2004), although some definitions even include occasional inversion tillage (Ferdinand & Baret, 2024) or autumn ploughing (Kwaad et al., 1998) as part of conservation tillage. Whatever the exact definition, the commonly accepted principle is to minimize soil disturbance and retain crop residues on the surface. In the short term, (cover) crop residues, if not buried by tillage, can fully serve for soil protection (Ranaivoson et al., 2017). Untilled soil also remains consolidated and therefore has greater shear strength (Knapen et al., 2007). From the medium term, reduced tillage better preserves earthworm populations (and other soil organisms), which supplement the role of vegetation in forming vertical macroporosity (Tebrügge & Düring, 1999). In the long term, the lack of (or limited) burial of plant residues results in a stratification of organic carbon in the topsoil. Although on average conservation tillage by itself does NOT increase the total stock of organic carbon in the soil, it indeed does modify its vertical distribution, especially following direct drilling (Dimassi et al., 2013; D'Haene et al., 2009). On a loess in Northern France, Dimassi et al. (2013) measured 53% more soil organic carbon in the uppermost layer (0-5cm) after 20 years of continuous direct drilling tillage (but 18% less in the deeper layer 10-28 cm), compared with full inversion. The higher organic matter content in the surface layer improves the structural stability of the topsoil, thereby increasing its resistance to erosive stresses to which it is most exposed, i.e. rain and overland flow.

- **Amendments and fertilizers**: Soil amendment can be another way of improving its structure. Organic soil conditioners (and fertilizers) like manure, digestate (from biomethanisation),

compost or straw are the most common and effective to this end (Morgan, 2005). Biochar, a source of highly stable carbon, is currently drawing attention but its soil conservation effect remains unclear (Zanutel & Bielders, 2023). As opposed to mulching, organic amendments are incorporated into the soil by tillage. Like residues, (decomposing) organic matter supports soil aggregate stability. The building up of soil organic carbon following regular application of fresh organic matter (whatever its origin) is, however, a slow process, and is reversible once the practice is stopped (Baveye et al., 2023). As for mulching, the potential for the amendment practice relies on the availability of the organic material, which is inherent to the surrounding agricultural system(s) (or the collection and treatment system of vegetation waste in the case of compost). For this reason, and since the relationship between organic matter content and soil erodibility is well established (Panagos et al., 2014), organic amendments will not be further considered in this thesis.

Mineral amendments were also studied for soil conservation worldwide, the best-known being (phospho)gypsum, which is a by-product of fertilizer production from phosphate rocks (Osman, 2014). It is used as a source of Ca^{++} cations to replace Na^{+} on soil colloids, in order to prevent clay dispersion, hence stabilize aggregates. However, the expected conservation effect of gypsum application on soils with low clay content such as silt loams appears largely offset by the risk of environmental pollution by radioactivity or heavy metals (Papastefanou et al., 2006; Akfas et al., 2024).

When looking at Figure 1.3, it is obvious that vegetation-based practices and conservation tillage have the most comprehensive effect on soil properties. Beyond the individual application of conservation practices, synergies between them are meaningful and form the fundamentals of **conservation agriculture**, which is the combination of tillage reduction, maximization of vegetation cover, and crop diversification (Landers et al., 2021). Specifically, vegetation improves soil structure and supplies residues on the one hand, while on the other, conservation tillage preserves it and retains them on the soil surface.

Except for winter cover crops, which are widely subject to conditionality rules of the Common Agricultural Policy, and organic amendments/fertilizers, especially manure which is an easily available by-product from cattle breeding, the adoption of in-field conservation practices by farmers remains limited in NW Europe. According to a survey by Heller et al. (2024), less than 16% of farmers in NW Europe are applying conservation agriculture. The reported barriers for adoption include the availability of knowledge and adequate machinery, financial risks, and farming traditions (Wauters et al., 2010). Moreover, among conservation tillage practitioners, reduced tillage is more widespread than no-tillage (Lahmar, 2010), likely because it results in lower yield losses, under NW European conditions at least (Van den Putte et al., 2010; Achankeng & Cornelis, 2023). Yet there is the drawback that reduced tillage appears less effective than direct drilling in mitigating overland flow and soil erosion (Maetens et al., 2012a).

1.2.3 Mechanical measures

Alongside conservation farming practices that involve cropping and soil management, so-called "mechanical" or **"physical" measures** have also been developed to mitigate overland flow. They aim to manage the landscape by reducing slope length and inter-plot connectivity (i.e. break up flow accumulation) and/or promoting runoff re-infiltration and sediment trapping through the use of structures, whether vegetated or not (Morgan, 2005). Terrace farming is a historic example of this approach, consisting in artificially shaping a "staircase" topographical pattern across the slope, i.e. whose levels follow the contour lines (Blanco-Canqui & Lal, 2010). Terracing greatly reduces slope continuity and steepness, thereby cutting down flow accumulation and velocity. It is beneficial on steep slopes that cannot be cultivated mechanically (e.g. mountainous regions) but requires laborious terrain work, making it unprofitable and irrelevant in NW Europe (Varotto et al., 2019). Another example of physical measures is the construction of dykes and retention ponds. These are designed to store water and sediment downstream of runoff-producing areas. Such structures are widespread, but require regular dredging of settled sediments (Verstraeten & Poesen, 1999). Back upstream, to convey runoff along a thalweg while reducing its velocity and thus concentrated

flows, the implementation of (perennial) grass(ed) waterways can be considered. Despite clear evidence of effectiveness (Fiener & Auerswald, 2006; Evrard et al., 2008b), the lack of subsidies for this measure within the EU agri-environmental program has hindered its implementation (Bielders et al., 2003), in contrast to the widespread installation of grassed buffer strips along field edges. Buffers strips are a family of vegetated strips – most commonly grasses, but also including miscanthus or willow more recently (Mazur & Kowalczyk-Juśko, 2021) – that are at least a few meters wide and placed at the field boundaries to re-infiltrate runoff and trap sediments lost from adjacent plot. The flow mitigation effect of buffer strips can, however, be limited by concentrated flows through the strip (Stehle et al., 2016). Hedgerows can also serve similar purpose, although they are originally intended to support biodiversity (Montgomery et al., 2020). Fascines (brushwood fences) are another example of physical structure designed to slow down water and trap sediments (Richet et al., 2017). Nevertheless, most of these physical structures consume productive area and require maintenance. Besides, they are curative techniques because they do not solve the problem of runoff and soil loss at the source in the field, unlike conservation farming practices. A proper soil conservation strategy should combine conservation farming practices and physical measures at the landscape scale (e.g., catchment) to prevent runoff and erosion. As part of this, physical structures are therefore intended to complement rather than substitute for conservation farming practices. However, the opposite has often occurred in NW Europe, as curative measures are easier to implement by public authorities and provide immediate effectiveness against overland flow (Verstraeten & Poesen, 1999; Evrard et al., 2007a). Within the framework of the Intell'eau project, physical structures are the subject of another work package (and Ph.D. thesis).

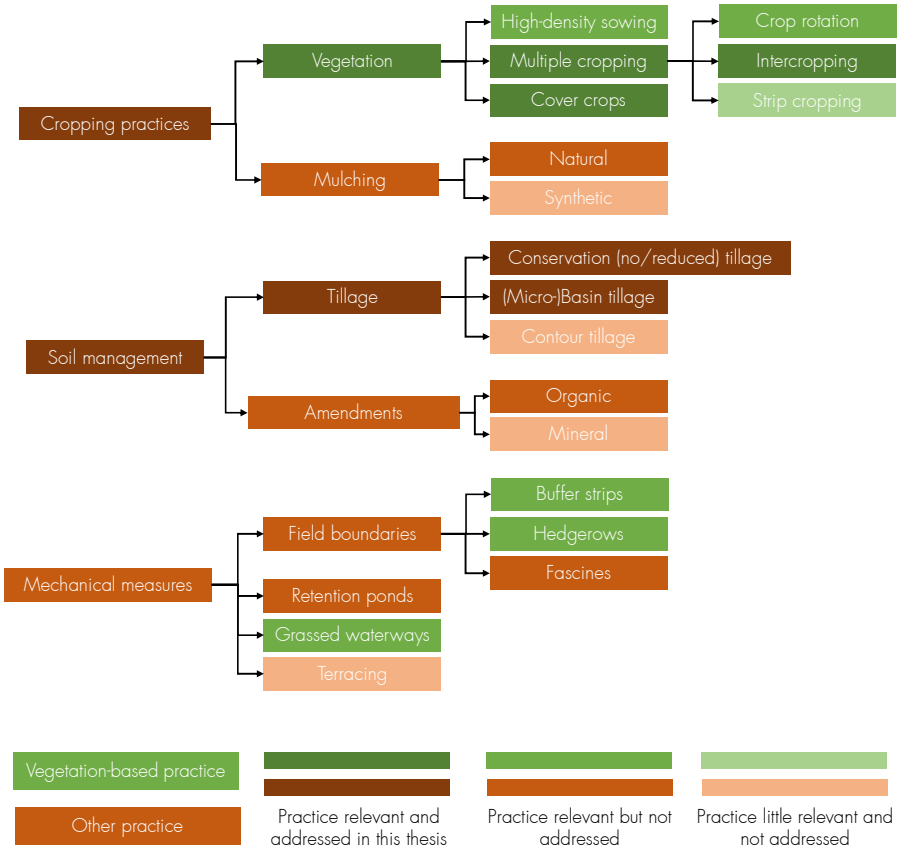


Figure 1.4 : Soil and water conservation practices for cropland, based on the classification from Morgan (2005). Vegetation-based practices appear in green boxes, others in brown. The color gradient distinguishes practices according to whether or not they are agronomically relevant to current Northwestern European agriculture, and whether or not they are addressed in this thesis.

1.2.4 Assessment of effectiveness

In this thesis, the focus is on the flow mitigation effect of conservation farming practices. As explained above and in Figure 1.3, the mechanisms involved in overland flow are highly complex. Multiple soil physical and chemical properties are directly or indirectly affected by conservation farming practices, and interact with one another. Undoubtedly, infiltration capacity (or saturated hydraulic conductivity) and structural (or aggregate) stability are two key soil physical properties. Consequently, they are regularly studied to assess the impact of agricultural practices on soil physical quality, particularly with

regard to runoff and erosion prevention (Basche et al., 2019; Benito & Diaz-Fierros, 1992), and plenty of methods exist for their measurement. For example, Low (1954) proposed a test simulating the action of raindrops on soil aggregates. Kemper & Rosenau (1986) introduced the widely used dry and wet sieving methods, which involve determining the size distribution of breakdown aggregates. A renowned and comprehensive method for assessing soil structural stability is the series of three tests on soil aggregates developed by Le Bissonnais (1996): fast wetting, slow wetting, and stirring after pre-wetting. Each test is designed to account for specific disaggregation mechanism(s). Other methods are derivatives adapted from these tests, such as the QuantiSlake test on bulk soil samples (Vanwindekens & Hardy, 2023). Devices such as torvanes and penetrometers focus more restrictively on the measurement of soil shear strength against a mechanical force (Madenford & Scholtes, 1973). Likewise, methods for measuring infiltration capacity or saturated hydraulic conductivity are also abundant. They include field measurements such as the Porchet method, Beerkan, single or double ring infiltrometers, whether constant- or falling-head, or disc permeameter (Lili et al., 2008). Constant- or falling-head permeameters can also be used with soil cores in the laboratory (Ghosh et al., 2019). From measured flow rates (transient or steady) and solutions of Darcy's law or Richards' equation, soil hydraulic properties can be inferred, of which the main one is saturated hydraulic conductivity, describing how easily water moves through soil pores (Angulo-Jaramillo et al., 2000). However, all these methods focus on a single, albeit crucial, soil property, but do not reflect the dynamics of the canopy protecting the soil, or the surface roughness storing water and slowing down runoff. The experimental setup allowing for the most comprehensive monitoring of overland flows from the whole soil-crop system is undoubtedly the **plot**. On the one hand, the plot needs to be big enough to average out part of the spatial variability in soil and crop attributes, and especially long enough to adequately reproduce significant overland flow processes, e.g. rill erosion (Smets et al., 2008; Langhans et al., 2019). On the other hand, its area should be kept small so as to facilitate practical and reliable measurement of surface flows. This includes hydrological isolation of the plots and full collection of sediment-laden runoff downstream using

reasonably sized experimental equipment (Stroosnijder, 2005). The standardized plots used by the USDA for developing the USLE empirical soil loss model serve as a reference for appropriate plot dimensions: 22 meters long, and between 2 and 8 meters wide (Renard et al., 1997). Plot experiments are therefore widely used to assess the impact of farming practices on runoff, erosion, and the transfer of associated pollutants (nutrients, pesticides) (Mhazo et al., 2016; Xiong et al., 2018).

The influence of climate on the nature of rainfall events, as well as of cropping systems and pedology on soil hydrological behavior, means that the findings from studies on conservation practices conducted outside NW Europe are not readily transposable to that region. Evidence supporting the effectiveness of some conservation farming practices in mitigating overland flow is sparse in NW Europe, despite occasional indications of their potential suitability. This is the case of **undersowing**, which is an application of intercropping whereby a secondary crop is sown simultaneously with a main crop growing over the former's canopy, or of **strip-tillage**, which is a specific implementation of conservation tillage for row crops seeking to exclusively (reduced-)till the seeding line. For other conservation farming practices, such as more traditional applications of **conservation tillage**, **micro-basin tillage**, or **winter cover crops**, a reasonable number of plot-scale studies have been conducted in NW Europe, but 1) surface flow measurements from plots (replicated within the same field) are inherently variable due to the heterogeneity of soil and crop characteristics (Nearing et al., 1999; Boix-Fayos, 2006); 2) Surface flow measurements from plots are variable between fields and between studies, due to differing environmental conditions (soil type, topography, events magnitude, etc.) and methodologies (Maetens et al., 2012b); 3) The effect of practices in mitigating surface flows is variable between fields and between studies, depending on environmental conditions and application modalities (Rickson, 2014; Xiong et al., 2018). All of this leads to a very high variability in (the effectiveness of conservation practices at mitigating) runoff and erosion as measured in such plot studies. **Meta-analysis**, which involves pooling data from multiple individual experimental studies, provides greater statistical power, allowing for more accurate quantification of the overall effect (in this case, the flow mitigation effect of a conservation practice) and

explanation of variability through factors such as those mentioned. Maetens et al. (2012a) reviewed the runoff and erosion mitigation effect provided by several conservation farming practices, through a meta-analysis for Europe and the Mediterranean (Figure 1.5). Their study provides interesting insights into the topic, but suffers from a number of drawbacks (see Section 1.3), starting with the broad geographical area and the limited number of factors investigated. Note also the wide variability in effect sizes for most conservation farming practices, as well as uncertain mitigation effects, e.g. for no-tillage or cover crops on runoff, or for reduced tillage on runoff and soil losses (Figure 1.5).

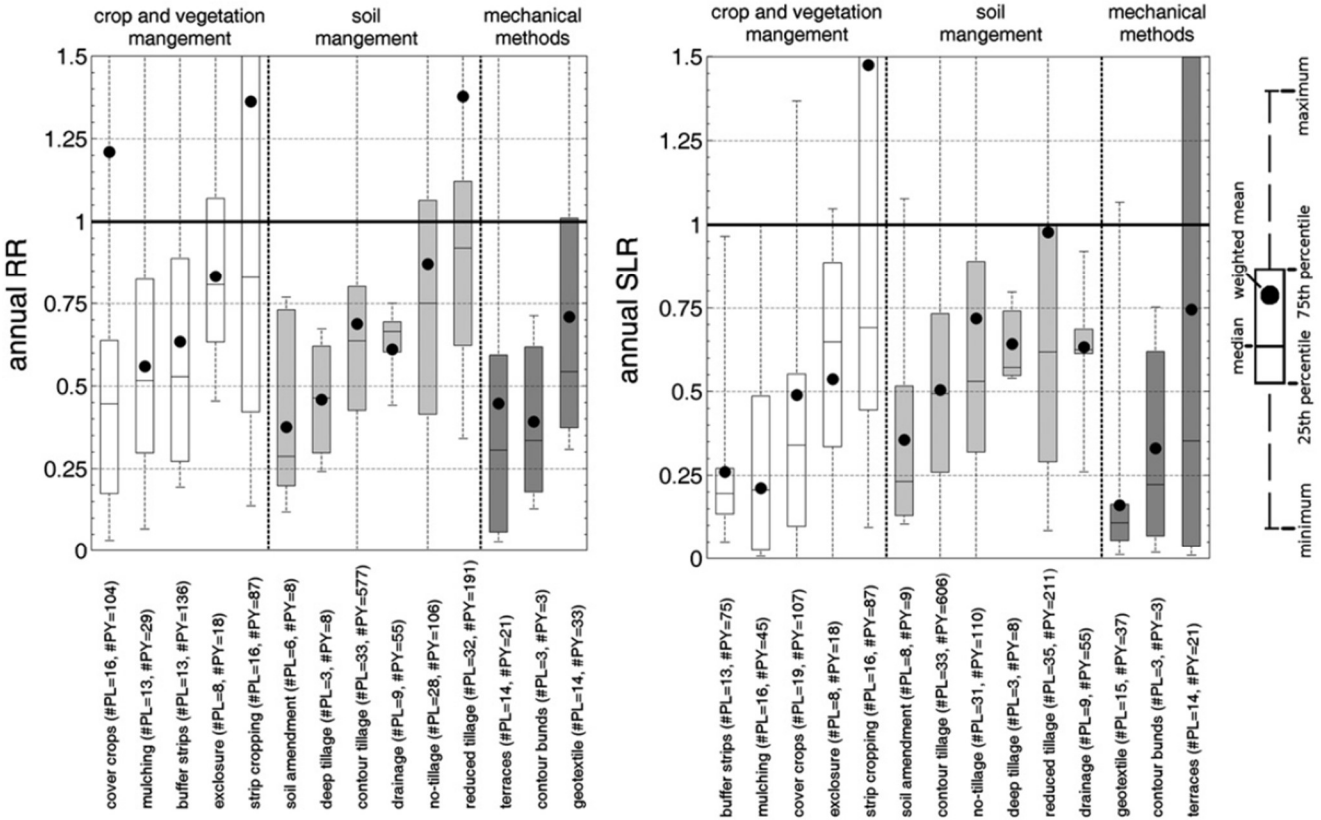


Figure 1.5 : Frequency distribution and mean annual runoff ratio ($\mathrm{RR} = \frac{\mathrm{Runoff_{conservation}}}{\mathrm{Runoff_{conventionnal}}}$) and annual soil loss ratio ($\mathrm{SLR} = \frac{\mathrm{Soil\ loss_{conservation}}}{\mathrm{Soil\ loss_{conventionnal}}}$) for different soil and water conservation techniques (Maetens et al., 2012a). #PL=number of plots, #PY=number of plot-year.

1.3 Objectives of the thesis

Based on the introduction presented in this dissertation, there is a need for effective conservation strategies suitable to the agricultural environment of NW Europe. The research undertaken in this thesis aims to support decisions in this area – whether taken by farmers, farm advisors or policy makers – through two general objectives:

1) **Quantify the (in)effectiveness of conservation farming practices in mitigating runoff, erosion, and pesticide surface losses, in the specific context of NW Europe.** Indeed, some practices might have potential but have been little evaluated in our regions or in comparable ones, e.g. undersowing or strip-tillage (Goeck & Geisler, 1988; Ryken et al., 2018). Other techniques have been studied but show uncertain effectiveness, e.g. reduced tillage (Maetens et al., 2012a). Furthermore, findings from studies carried out outside NW Europe are hardly transposable, because of the influence of cropping systems, soil type, or climate on overland flow processes.

2) **Determine how the flow mitigation effects of conservation farming practices are affected by specific conditions**. Indeed, the effectiveness of practices in controlling runoff and erosion is highly variable between different experiments, depending on environmental conditions (e.g. soil type, topography, climate) and application modalities (Rickson, 2014; Xiong et al., 2018). Some environmental and management factors have already been identified, but most of the variability remains unexplained (Maetens et al., 2012a). Better knowing these factors would help optimize the implementation of practices so as to maximize expected benefits.

These two general goals are applied to the following conservation farming practices (see Figure 1.6), which were considered relevant for the context of NW Europe (see Section 1.2) and selected as part of the "Intell'eau" project (see Section 1.1.3):

- **Well-documented practices:** They refer to conservation farming practices for which strong evidence already exists in the literature

about their effectiveness in mitigating surface flows, in NW European conditions.

- The quantitative review by Maetens et al. (2012a) is the current reference in this area, but it suffers from several drawbacks: 1) It also includes the Mediterranean, which covers very different climatic and cropping conditions from NW Europe; 2) The definition of some practices is wide: perennial cover in vineyard are considered as winter cover crop (e.g. Messer, 1980), autumn ploughing as conservation tillage (Kwaad et al., 1998), or the database includes experiments without any crop (bare soil only) (Jung & Brechtel, 1980); 3) The grey literature has been little surveyed; 4) Limited factors were investigated to explain the variability in the practices' effectiveness; 5) Statistical methods relevant to meta-analysis are lacking, i.e. overall effect quantification and comprehensive analysis of factors; 6) Micro-basin tillage was not addressed. We assumed that a more accurate and relevant (to the Belgian context) effect quantification could be achieved by focusing on NW European conditions as well as by being more restrictive in the practices' definitions, and that a greater part of the variability could be explained by retrieving additional factors. **Chapter 2 deals with a meta-analysis designed to quantify the mitigation effect against runoff and erosion as well as to explain their variability between individual experiments, for the following practices under NW European conditions: conservation tillage, winter cover crops, and micro-basin tillage in potato crop.**

- Recently, a farming equipment, an adapted roller within a seeding unit, has been designed and tested in Belgium by the CIPF (2012-2017; unpublished) to perform micro-basin tillage (soil pitting) in maize crop. Their study is not relevant for inclusion in the above-mentioned meta-analysis because, to our knowledge, no other study has evaluated the technique in Europe. However, extensive plot experiments were carried out by the CIPF, enabling a pooled analysis of the technique's effectiveness in mitigating overland flows. We hypothesized

that 1) the flow mitigation effect of the technique could depend on the magnitude of rainfall events, since it is based solely on surface retention; 2) The micro-basins could degrade following consecutive (intense) rain events, thereby limiting their effectiveness; 3) The storage capacity of the soil pits could decrease with the field slope steepness. Based on the experimental data collected by the CIPF (2012-2017) in central Belgium, **chapter 3 quantifies the effectiveness of micro-basin tillage in maize crop in mitigating surface flows of water, soil, and pesticides**, and further analyzes the influence of slope and rain event characteristics.

- **Innovative practices**: They involve promising conservation farming practices but poorly documented in the literature for NW European conditions, therefore requiring new plot experiments. Given the labor-intensive nature of these experimental set-ups, we chose to focus on the maize crop, which is the most widely cultivated spring crop (thus erosion-sensitive) in Western Europe (see Figure 1.1).

    - **Undersowing:** Intercropping has already been studied in maize, mainly in the form of relay cropping (=deferred sowing) of a secondary crop (e.g. ryegrass) in (e.g. 4-5 leaf stage of) maize (Goeck & Geisler, 1988; van Vliet et al., 2002). Delayed sowing of the intercrop limits competition with maize, but results in poor cover in spring, which is a critical situation triggering muddy floods (see Section 1.1.2). Sowing a moderately fast growing (so as to limit competition) intercrop species at the same time as the maize (=undersowing) appears as an interesting option for controlling overland flows from spring onwards, yet very few studies have assessed this potential solution.

    - **Strip-tillage:** This specific application of conservation tillage is very little featured in the database of the meta-analysis from Chapter 2. Specific zonal tillage in the row without disturbing the inter-row is expected to result in a spatially heterogeneous soil, potentially distinguishing strip-tillage from other conservation tillage practices. The latter indeed generally involve reducing the intensity or the frequency of tillage

operations, while still working the soil surface uniformly. To our knowledge, very few studies have investigated strip-tillage in maize crop for mitigating surface flows in NW Europe (Ryken et al., 2018; Geelen et al., 1996). Moreover, the specific practice of strip-tillage is much less adopted than reduced tillage in Europe (Heller et al., 2024), despite indications of success in terms of both soil conservation and crop yields (Achankeng & Cornelis, 2023).

Chapter 4 reports original plot experiments conducted in the Belgian loess belt assessing the potential of undersowing red fescue or white clover, and strip-tillage, in maize crop, at mitigating runoff, soil erosion and herbicide surface transfer. A plot-scale, process-based hydrological **model is further calibrated, and used for a scenario analysis** generalizing both the soil conservation and flood control effect expected under strip-tillage, for various environmental conditions.

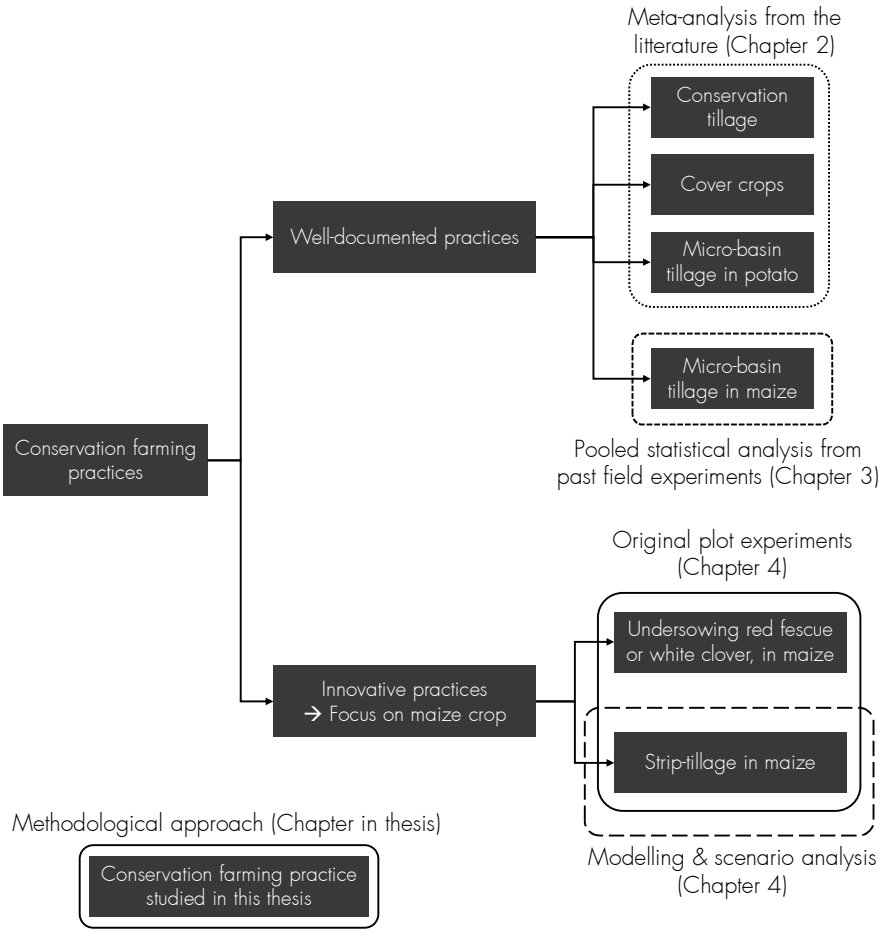


Figure 1.6 : Diagram of conservation farming practices studied in this thesis for their flow mitigation effect (runoff, erosion and/or pesticides). Boxes are conservation practices, and insets indicate the methodological approaches used per practice(s).

CHAPTER 2: HOW MUCH DO CONSERVATION CROPPING PRACTICES MITIGATE RUNOFF AND SOIL EROSION UNDER WESTERN EUROPEAN CONDITIONS: A FOCUS ON CONSERVATION TILLAGE, TIED RIDGING AND WINTER COVER CROPS[1]

[1] Adapted from Clement, T., Bielders, C. L., & Degré, A. (2024). How much do conservation cropping practices mitigate runoff and soil erosion under Western European conditions: A focus on conservation tillage, tied ridging and winter cover crops. *Soil Use and Management*, *40*(2), e13047. https://doi.org/10.1111/sum.13047

Abstract

Conservation farming practices are known for their capacity to mitigate runoff and erosion, but the magnitude of their effectiveness is highly variable across studies. In order to better understand the contribution of environmental and management factors to their effectiveness, up to 37 studies reporting 271 individual trials were collated for a quantitative review regarding 3 common conservation agriculture-related practices, at the plot scale and in a Western European context. Two different methods suitable for hierarchically structured datasets were used for the meta-analyses - hierarchical nonparametric bootstrapping and linear random effects models -, yielding nearly identical average outcomes but differing in terms of confidence intervals. We found that, on average, winter cover crops reduce cumulative seasonal (autumn-winter) runoff by 68% and soil losses by 72% compared to a bare soil. The occurrence and intensity of stubble tillage on the control plot is a key explanatory variable for the mitigation effect of winter cover crops. In potato crops, tied-ridging reduces cumulative seasonal (spring-summer) runoff by a mean of 70% and soil erosion by 92%. Conservation (non-inversion) tillage techniques alleviate cumulative seasonal overland flow by 27% and associated sediments losses by 66%, but strong evidence of publication bias was detected for this farming practice, probably leading to an overestimation of its effectiveness. These mitigation effects are shown to be much greater for spring crops than for winter crops, and to increase with time since ploughing was stopped. The type of conservation tillage scheme strongly affects the ability to attenuate surface flows. Intensive non-inversion tillage systems relying on repeated use of (powered) tillage operations appear to be the least effective for reducing both water and sediment losses. The best performing scheme against runoff would be a deep (non-inversion) tillage (-61%), while against erosion it would be a no-till system (-82%). Although several explanatory factors were identified, there remains a high (unexplained) variability between trials effect sizes, thus not attributable to pure sampling variability. Meanwhile, this review provides farm advisors or policy makers with guidance on the contexts

in which implementation of such conservation practices should be supported so as to maximize expected benefits.

2.1 Introduction

Throughout much of western Europe, erosive runoff from agricultural land constitutes a threat to flora and fauna through contamination of water bodies by pesticides, nutrients, and sediments (Bach et al., 2001; Stoate et al., 2001). Human communities are also impacted physically, economically and psychologically by muddy floods arising from stormy rains and the resulting damage caused to infrastructure (Evrard et al., 2007a; Boardman et al., 2003). In the longer-term, in some situations, cropping systems are no longer sustainable as soil loss rates exceed soil formation rates (Verheijen et al., 2009; Panagos et al., 2015c). More generally, soil health in intensive agricultural systems is at risk, particularly due to the depletion of soil organic matter (SOM) content to which soil erosion also contributes (Heikkinen et al., 2013; Sakrabani & Hollis, 2018). This decline in SOM further increases the sensitivity of soils to runoff and erosion (Berdeni et al., 2021; Jensen et al., 2019). In the context of climate change, the loss of water by runoff, together with a decrease in soil available water content due to the declining SOM, may also translate into more frequent plant drought stresses (Shiu et al., 2012; Fu et al., 2021; Dlapa et al., 2020; Falloon & Bets, 2010).

In view of these issues, conservation agriculture systems have been conceived based on 3 cultivation management pillars: tillage reduction, maximization of vegetation cover, and diversification of cultivated species (Morgan, 2005; Landers et al., 2021). The synergies between these three principles promote soil life and restore topsoil organic matter content, ultimately improving soil resistance against surface flows (Blanco-Canqui & Lal, 2010; Palm et al., 2014; Du et al., 2022). However, the extent to which conservation farming practices mitigate runoff and erosion remains quantitatively uncertain, especially with regard to runoff (Blanchy et al., 2023; Rickson, 2014; Maetens et al., 2012). The processes involved - rain interception by vegetation and crop residues, soil aggregate stabilization by roots, soil microbiota and binding agents released during biomass decomposition, changes in

surface roughness, porosity and structure by tillage and soil life, etc. - are indeed complex, dynamic, and condition-dependent (Morgan, 2005; Monnier, 1965; Tisdall & Oades, 1982; Takken et al., 2001; Le Bayon & Binet, 2001). This results in a wide and often unexplained variability in outcomes between studies measuring the mitigation effect of soil conservation practices on overland flow of water, sediments and associated pollutants at the plot scale (Rickson, 2014; Elias et al., 2018; Leys et al., 2007). As a result, the impact of conservation tillage on runoff is still being debated (Blanchy et al., 2023; Xiong et al., 2018; Maetens et al., 2012a; Mhazo et al., 2016). The factors behind this variability also remain unclear.

While local factors related to the ways and the contexts in which conservation farming practices are being implemented are likely to be strong determinants of their effectiveness in controlling runoff and soil losses (e.g., crop type, choice of tillage implements, climatic conditions; Ranaivoson et al., 2017; Ryken et al., 2018), most reviews on this subject have been conducted at a global scale or for regions outside Europe (Chen et al., 2020 for China; Wolka et al., 2018 for Africa; Rajbanshi et al., 2023 or Mhazo et al., 2016 at global scale). On the one hand, the validity of the outcomes of global studies for different regions of the world is questionable. On the other hand, the large amount of publications to be analysed in global reviews often does not allow for an in-depth search of publications and associated data (grey literature, contact with authors to retrieve missing data, etc.). At the European scale, only Maetens et al. (2012a) have, to our knowledge, conducted a quantitative review on the effect of soil and water conservation techniques on runoff and soil losses. However, these authors used a very broad definition of conservation practices which may ultimately bias their results, e.g., perennial grass cover in vineyard was considered as winter cover crop, or treatments involving autumn ploughing were in some cases considered as reduced tillage. Moreover, their study area also encompassed the Mediterranean, which is characterized by very different climatic and agronomic conditions from the rest of Europe. Recently, Blanchy et al. (2023) qualitatively reviewed existing meta-analyses on the impact of land management practices on soil properties, including runoff. However, effect quantification is missing, the meta-analyses they reviewed suffer from the same limitations mentioned above in terms of geographical

area of interest, some management practices were not considered (e.g. reservoir tillage techniques), and the effect of management techniques on soil losses was not targeted per se.

In this paper, we provide a quantitative review based on a systematic search and a meta-analysis on the effectiveness of three common conservation farming practices in mitigating runoff and soil erosion in the western European (or assimilated) context. These cultivation practices are winter cover crops, tied-ridging (= furrow diking = (micro)basin tillage) in potato crops, and conservation tillage techniques (defined as non-inversion tillage, from no tillage at all to reduced tillage). Besides data availability (sufficient number of trials), the choice of these three practices is based on their fairly widespread adoption by western European farmers, for legal (e.g. Water Framework Directive, cross compliance of the Common Agricultural Policy) and/or ideological reasons (Wauters et al., 2010). While additional techniques may also qualify as conservation agriculture (e.g., mulching, strip cropping, or crop associations), insufficient data is available for a reliable quantitative analysis of their effectiveness at the western European scale (Rickson, 2014).

2.2 Materials and methods

2.2.1 Systematic search

Table 2.1 outlines the main characteristics of the systematic search for each of the three conservation farming practices, including their definition and the associated conventional control practice, the research equations (keywords), the potential moderators (possible explanatory variables) that were surveyed, as well as the characteristics of the resulting meta-databases (data availability). Scopus was the main database explored, the first one hundred results of google scholar having also been investigated to browse through grey literature. We only selected studies which actually measured runoff and soil losses (by funnel and tanks or tipping bucket systems) at the plot scale, under natural rainfall, at a seasonal time scale, with control and conservation treatments on the same field, located in western Europe. Nonetheless, whenever a study located outside the target countries (see keywords in

Table 2.1) was deemed relevant (i.e., similar climatic and pedologic conditions, similar cropping systems), it was also selected. Accordingly, studies from Austria, Czech Republic, Poland, Croatia, and southern Canada were included in the meta-databases (Figure 2.1). Table A.1 in the Appendices exhaustively references all studies included in the meta-databases.

Corresponding authors were systematically contacted by e-mail to retrieve as many missing data as possible. When data (means, standard deviations, moderators) were not directly available in digital format in the primary studies, they were extracted from plots using "GetData Graph Digitizer" software v.2.26.0.20. When possible, missing standard deviations were calculated from confidence intervals, or estimated from standard error or from t or p values (Higgins et al., 2022).

Table 2.1 : Characteristics of the systematic searches and the resulting meta-databases for each conservation farming practice and for runoff or erosion outcome: Practices definitions, and keywords of the research equations.

	Winter cover crops	Tied-ridging potato furrows	Conservation tillage
Conservation practice definition	Post harvest (summer/autumn) sowing of a non-productive crop (typically mustard and/or grasses) intended to protect the soil during winter period, and destroyed before sowing of the next (spring) crop.	At planting time, formation of small dams in potato furrows to increase water surface retention.	Non-inversion tillage (no mouldboard plough) in order to retain crop residues at soil surface and to keep soil life and structure not or little disturbed. Possibly other primary tillage (subsoiling, loosening) and/or seedbed preparation (harrow, tines, discs, etc).
Corresponding control treatment	Soil left bare during winter, possibly with stubble tillage.	Classically hilled plot (no dams, smooth furrows).	Yearly mouldboard ploughing and "conventional" seedbed preparation (harrow and/or tines and/or discs).
General keywords	TITLE-ABS-KEY (erosion OR "soil loss*" OR "sediment yield" OR runoff OR "overland flow" OR "surface water") AND TITLE-ABS-KEY (plot OR field OR parcel) AND ALL (Belgium OR France OR Netherlands OR Holland OR Germany OR UK OR England OR Ireland OR Luxembourg OR Europe)		
Practice-specific keywords	TITLE-ABS-KEY (winter OR cover OR catch AND crop OR intercrop*)	TITLE-ABS-KEY (potato* AND (microdam* OR micro-dam* OR (furrow* AND (dik* OR ridg* OR dam*)) OR (tied AND ridg*) OR ((basin OR microbasin) AND (tillage OR listing)) OR barbutte OR cottard OR dycker OR grimme OR "Dema-construct" OR "Netagco-Rumpstad" OR "AgriMaas"))	TITLE-ABS-KEY ((conservation OR no OR reduced OR zero OR low OR minimum AND till*) OR (direct AND (seed* OR drill* OR sow*)))

Table 2.1 (continued) : Potential moderator variables (in bold, the moderators that were actually tested in the meta-regression, see Section 2.2.2.5).

	Winter cover crops	Tied-ridging potato furrows	Conservation tillage
General moderators	• Climate: total seasonal rainfall and erosivity • Crop: current and preceding crop type (spring or winter) • Soil: texture, organic matter or carbon content • Topography: slope steepness and length • **History: time since practice application**	• Climate: total seasonal rainfall and erosivity • Crop: preceding crop type (spring or winter) • Soil: texture, organic matter or carbon content • **Topography**: **slope steepness** and length • History: time since practice application	• Climate: total seasonal rainfall and erosivity • **Crop: current and preceding crop type (spring or winter)** • **Soil: texture (silty(-clay) loam or sandy loam)**, organic matter or carbon content • **Topography**: **slope steepness** and length • **History: time since practice application**
Practice-specific moderators	• Cover crop family (grasses, cruciferous, legumes) • Sowing, killing, and burying dates and modes (frost, mechanical, chemical) • **(squared) % max. vegetation cover** and biomass • **Stubble tillage on control plots (none, shallow, or deep stubble tillage)**	• Tool used (Barbutte, Dyker, etc) • **Max. daily rainfall in May-June**	• **Max. tillage depth for conservation tillage (direct drilling <5cm, intermediate 5-15cm, or deep tillage depth >15cm)** • **Intensity of the most intensive conservation tillage operation (none, passive or powered tool)** • **# of seasonal tillage passes in conservation tillage system (including sowing)** • Winter cover crop included in conservation tillage rotation • % residues cover at sowing on conservation tillage plots

Table 2.1 (continued) : Number of investigated sources, number of studies selected as relevant based on inclusion/exclusion criteria, total number of trials* reported by selected studies, proportion of trials from grey literature studies, proportion of trials with reported (and existing, i.e. n>1) sampling variances, structure of the linear random effects model considered for meta-analysis (Section 2.2.2.5), suspicion of publication bias for the meta-dataset, weighting of the individual effect sizes in the computation of the overall mean effect size.

	Winter cover crops	**Tied-ridging potato furrows**	**Conservation tillage**
# of investigated sources	578	22	1210
# of relevant studies selected, for runoff / erosion outcome	7 / 5	7 / 7	21 / 17
# of trials, for runoff / erosion	36 / 29	25 / 17	178 / 165
% of trials from grey literature, for runoff / erosion	36% / 21%	56% / 71%	9% / 6%
% of trials with existing or reported experimental variances, for runoff / erosion	61% / 76%	36% / 6%	34% / 27%
Structure of the random effect(s) model	Trials > plots**	Trials	Studies > trials
Publication bias suspicion	No	No	Yes
Weighting of trials effect sizes	No	No	Yes (Eq. 3)

*One individual trial corresponds to one pair (control vs. conservation practice) of cumulative seasonal runoff (or erosion) measurements (thus one single effect size measure, see Eq. 2.1), i.e. for one cropping season, on one trial site, with specific application modalities (e.g. specific tillage scheme) for the conservation practice and its corresponding control treatment.

** The symbol ">" separates successive hierarchical levels of the random effects model. In this case, "trials>plots" refers to a multi-level model with a "plots" random effect nested in a "trials" random effect (see Section 2.2.2.5).

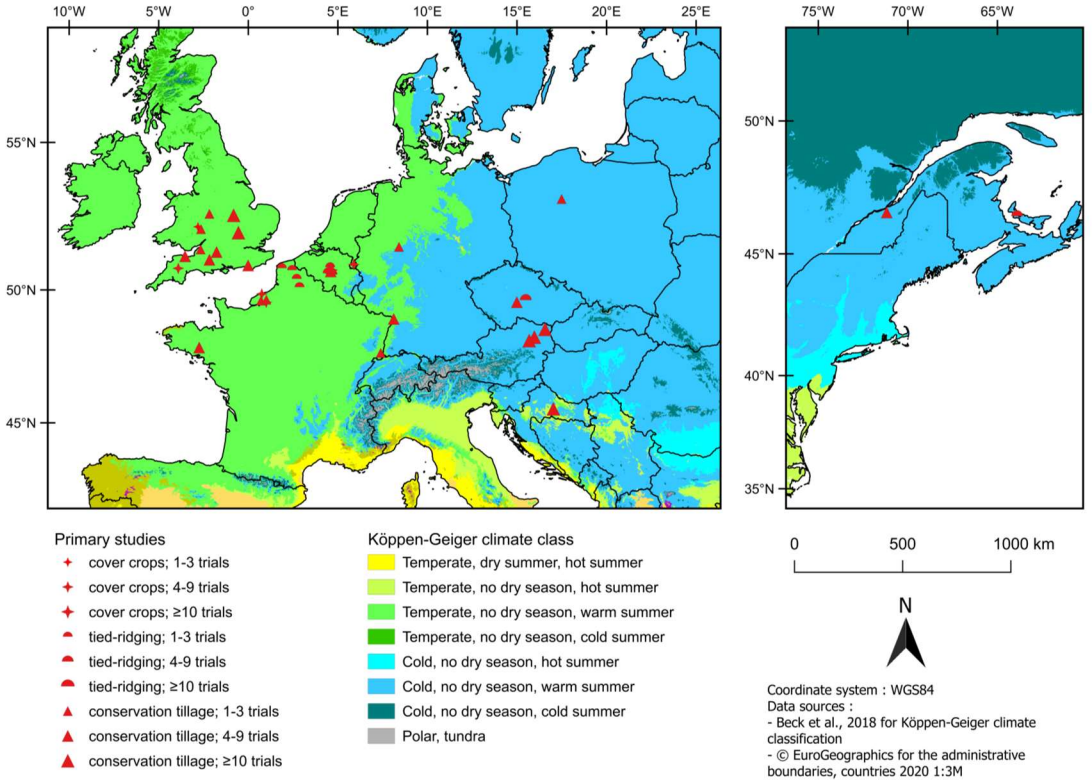


Figure 2.1 : Location of experimental sites of all primary studies selected for the present review in Europe (left) and Canada (right). A different symbol is used for each of the three conservation practices, while symbol size varies according to the number of trials at each location (see caption of Table 2.1 for trial definition). Background map corresponds to the Köppen-Geiger climate classes.

2.2.2 Data analysis

2.2.2.1 Effect size

The effect size measure chosen to quantify, at the trial scale (see 2.2.2.2 for meta-database structure), the effectiveness of the conservation practices in mitigating runoff (or soil losses) compared to the control was the natural logarithm of the response ratio (RR) (Hedges et al., 1999):

$$\ln(\mathrm{RR}_{\mathrm{runoff}})_{i,j} = \ln\left(\frac{\overline{\mathrm{runoff}_{\mathrm{CP}\ i,j}}}{\overline{\mathrm{runoff}_{\mathrm{C}\ i,j}}}\right) \qquad \text{(Eq. 2.1)}$$

$$\widehat{var}\left(\ln(\mathrm{RR}_{\mathrm{runoff}})_{i,j}\right) = \frac{s^2_{\mathrm{runoff\ CP}\ i,j}}{n_{\mathrm{CP}\ i,j} * \overline{\mathrm{runoff}_{\mathrm{CP}\ i,j}}^2} + \frac{s^2_{\mathrm{runoff\ C}\ i,j}}{n_{\mathrm{C}\ i,j} * \overline{\mathrm{runoff}_{\mathrm{C}\ i,j}}^2} \qquad \text{(Eq. 2.2)}$$

where $\overline{\mathrm{runoff}_{\mathrm{CP}\ i,j}}$ and $\overline{\mathrm{runoff}_{\mathrm{C}\ i,j}}$ are the mean (across replicates) cumulative seasonal runoff for trial *j* of study *i*, on conservation practice plots (subscript CP) and control plots (subscript C), respectively, $s^2_{\mathrm{runoff\ CP}\ i,j}$ and $s^2_{\mathrm{runoff\ C}\ i,j}$ are the corresponding experimental (sampling) variances for the mean cumulative runoff values, and $n_{\mathrm{CP}\ i,j}$ and $n_{\mathrm{C}\ i,j}$ are the sample sizes (number of plot replicates) for trial *j* of study *i* for each treatment, respectively. The same formulas apply to the erosion outcome.

Unlike standardized differences, experimental variances are not required to calculate the (log) response ratio, and interpretation of the response ratio in terms of mitigation efficiency is more straightforward. The use of logarithms allows normalizing the distribution of response ratios (Koricheva et al., 2013). The mitigation efficiency of the conservation practice is calculated as 1-RR.

2.2.2.2 Meta-databases structure

Each meta-database features a hierarchical, 3-level structure consisting of several studies, each study generally reporting several trials (one trial corresponds to one effect size measure), each trial being associated with an experimental (sampling) variability (Eq. 2.2) because of plot replicates within the trial (except if $n_{\mathrm{CP}\ i,j} = n_{\mathrm{C}\ i,j} = 1$, see Section 2.2.2.5). Such hierarchically-structured datasets can be analysed through linear random effect modelling (Stram, 1996) (Section 2.2.2.5) or

nonparametric hierarchical bootstrapping (Saravanan et al., 2020) (Section 2.2.2.6). These two very different methods were used in order to compare and strengthen the estimated overall effect sizes and associated uncertainties (confidence intervals).

2.2.2.3 Weighting of effect sizes

Regardless of the use of one or the other method (linear model or bootstrapping), the decision to weigh or not to weigh the effect sizes of individual trials in the calculation of the mean effect size differed according to the meta-database. For winter cover crops and potato tied-ridging, an unweighted analysis was performed because there was neither evidence that variability in effect size decreased with increasing sample size (Figure A.1 in Appendices), nor evidence of publication bias. For the conservation tillage meta-database, the suspicion of publication bias (see Section 2.2.2.4) as well as the large proportion in the meta-database of low sample size trials ($n_{\mathrm{CP}} = n_{\mathrm{C}} = 1$) from a single study (Klik & Rosner, 2020) motivated the choice for a weighted meta-analysis.

The weight formula considered in this article was based on the sample sizes rather than sampling variances (Koricheva et al., 2013):

$$w_{\mathrm{i,j}} = \frac{n_{\mathrm{CP}\,i,j} * n_{\mathrm{C}\,i,j}}{n_{\mathrm{CP}\,i,j} + n_{\mathrm{C}\,i,j}} \quad \text{(Eq. 2.3)}$$

where $w_{i,j}$ is the weight given to the (runoff or erosion) effect size from trial *j* of study *i*. Indeed, unlike sampling variances, sample sizes (number of plot replicates) were known for all trials of the three meta-databases.

2.2.2.4 Publication bias

The risk of publication bias was assessed for each meta-database by visually examining the funnel plots using trial weights (Eq. 2.3) as the measure of precision, and by testing for the significance of the slope of the regression between effect sizes and weights (Begg & Berlin, 1988) (Figure A.1 in Appendices). For the winter cover crops and potato tied-ridging meta-databases, there was no evidence of publication bias. For the conservation tillage meta-database, there was a strong indication of publication bias (slope of the linear regression between trial weights and

effect sizes significantly different from zero (*p*-value < 0.001), both for the runoff and erosion outcomes.

2.2.2.5 Random effects models

Random effects models are a category of general linear models (GLM) in which variance(s) of the so-called "random effect(s)" are estimated to account for a random deviation around the overall mean. This deviation is caused by the clustering of individuals within groups, thus sharing common conditions susceptible to impact the variable of interest. A normal (Gaussian) distribution is assumed for this deviation. In the case of our meta-databases, the factors (clusters) susceptible to cause this random variability in effect size measures are the studies and/or the trials within studies and/or the plot replicates (block effect) within the trials. When such random effects are nested, the random-effects GLM is referred to as a multi-level model (Quené & van den Bergh, 2004). The full multi-level model is written hereafter as "studies>trials>plots", which must be understood as a multi-level model with a "plots" random effect nested in a "trials" random effect nested in a "studies" random effect.

Data amount (i.e., number of studies, trials, and plot replicates) and reporting of sampling variability measures were very different for the three practices (Table 2.1). Consequently, the structure of the linear random effects model was specific to each practice (Table 2.1). For the winter cover crop meta-database, availability of sampling (experimental) variances for a sufficient number of trials allowed for a reasonable estimate of between-plots variability. However, because of the limited number of studies and of trials per study, both the "studies" and "trials (nested) within studies" random effects would be poorly estimated. It was therefore necessary to restrict the model to one of these two levels (study or trial). Since the AIC of the "trials>plots" random effects model was more than four times lower than the one of the "studies>plots" random effects model, the former model was selected. A similar reasoning was applied for the meta-database on potato tied-ridging, except that sampling variances were lacking for a large number of studies or non-existent ($n_{\mathrm{CP}} = n_{\mathrm{C}} = 1$). Between-plots variability was therefore neglected, i.e. considered null. For the conservation tillage

meta-database, since sampling variances were also often missing, the plots effect was also ignored. However, the much higher number of studies and trials allowed for a reliable estimation of both the "studies" and "trials within studies" random effects.

As previously mentioned, sampling variances for mean cumulative runoff or soil loss values were generally poorly reported (Table 2.1). In some cases, sampling variances were even non-existent since plots were not replicated within treatments ($n_{\mathrm{CP}\,i,j} = n_{\mathrm{C}\,i,j} = 1$). For the winter cover crop meta-database, since the values of the ratio between standard deviation and mean runoff (or erosion) cover a similar range for all trials and for both the control and the conservation practice treatments, missing standard deviations were imputed based on known standard deviations as follows (Koricheva et al., 2013):

$$\widetilde{s_{\mathrm{runoff\,CP}\,i',j'}} = \overline{\mathrm{runoff}_{\mathrm{CP}\,i',j'}} * \frac{\sum_{i,j}^{I,J} s_{\mathrm{runoff\,CP}\,i,j}}{\sum_{i,j}^{I,J} \overline{\mathrm{runoff}_{\mathrm{CP}\,i,j}}} \qquad \text{(Eq. 2.4)}$$

where $\overline{\mathrm{runoff}_{\mathrm{CP}\,i',j'}}$ is the cumulative runoff value from conservation practice plots for trial *j'* of study *i'*, associated with unknown (thus estimated) standard deviation $\widetilde{s_{\mathrm{runoff\,CP}\,i',j'}}$. $\overline{\mathrm{runoff}_{\mathrm{CP}\,i,j}}$ and $s_{\mathrm{runoff\,CP}\,i,j}$ are defined similarly, but for trial *j* from study *i* with known standard deviations. The same formulas were applied to runoff from control plots, and to the erosion outcome on control and conservation practice plots.

Note that in some cases, within a given study, a single measurement of cumulative runoff (or erosion) from a treatment was used to calculate different measures of effect size, i.e., a single treatment measure was associated to different trials because the same control treatment served as reference for several modalities of the conservation practice treatment, or the opposite. For simplicity, and since information to estimate the correlation between these effect sizes is scarce, we nevertheless considered independence between the effect sizes, i.e., a diagonal variance-covariance matrix, or in other words, a compound symmetric structure of the variance-covariance matrix with a null correlation coefficient rho.

For the multi-level models, heterogeneity at each level was calculated as the proportion of the total variance attributable to the corresponding level (Nakagawa & Santos, 2012; Harrer et al., 2022):

$$I_l^2 = \frac{\widehat{\sigma^2}_l}{\widehat{\sigma_\varepsilon} + \sum_l \widehat{\sigma^2}_l} \qquad \text{(Eq. 2.5)}$$

where I_l^2 is the heterogeneity (the proportion of variability) related to the level l. $\widehat{\sigma^2}_l$ is the estimated variance component of the level level l, and $\widehat{\sigma_\varepsilon}$ is the estimated sampling variance.

Moderators (explanatory variables) can be added as fixed effects to a random effect(s) model, resulting in a mixed effects model (Stram, 1996). Explanatory variables with too many missing values as well as variables highly correlated with other moderators (multicollinearity) were excluded from the analysis. The moderators that were ultimately tested (highlighted in bold in Table 2.1) were chosen by prioritizing those suspected to impact most the conservation practice efficiency, keeping in mind the rule of thumb that about ten data points (trial effect sizes) are required for one parameter estimation (Harrell et al., 1996). Missing data for moderators were mainly imputed by using the average value in other trials (with known value). Missing maximum daily rainfall in spring (tied-ridging meta-database) were estimated from closest public weather station data (CRA-W, 2022; Government of Canada, 2023).

For each meta-database, all possible combinations of models including the main (no interaction) fixed effects for the chosen moderators were automatically tested. The ultimate meta-regression model for each conservation practice is the one with the lowest AIC (Akaike, 1974). Note that for conservation tillage, inclusion in the model of interaction effects between moderators was attempted given the larger dataset. Although the final model resulted in a slightly better fit, the inclusion of interactions was not withheld because, compared to the best main effects only-model, it was mainly the inter-study variability that was reduced whereas these interactions are supposed to explain the variability at the trial level. Moreover, the interactions did not make sense from a physical point of view. Model overfitting was therefore suspected.

Runoff and soil loss mitigation effects predicted by the final meta-regression models are depicted as a function of each selected moderator individually (Figures 2.3 and 2.4, and Section 2.3). For this purpose, values of the other selected moderators were fixed at their (rounded) mean value in the original meta-dataset. For winter cover crops (Fig. 2.3), this corresponds to a shallow stubble tillage on the control plots, a max. vegetation cover of 70% by winter cover crops, and/or 1 year of winter cover crop practice. For conservation tillage, the reference scenario consists of a spring crop, with a winter crop as preceding crop, with 2 soil tillage operations, with passive tillage as the most intensive operation, at intermediate tillage depth, on a silt loam soil, on a field under conservation tillage for 4 years.

Underlying assumptions were visually validated for all final meta-models: good model convergence (optimal log likelihood), normality, linearity and homoscedasticity of residuals. Analyses were performed in R Studio 2022.07.2, using packages 'metafor' (Viechtbauer, 2010) for the random effects models and 'MuMin' (Bartoń, 2022) for moderator selection.

2.2.2.6 Hierarchical nonparametric bootstrapping

Nonparametric bootstrapping (Efron, 1979; Hesterberg, 2011) is a method consisting of resampling (with replacement) of an initial sample to generate many fictitious resamples (of the same size as the original one). For a statistical indicator (typically the mean) calculated on each resample, a distribution is derived that is expected to reproduce the one of the original population, from which a confidence interval can be drawn to conduct basic statistical inference. Nonparametric bootstrapping is widely used in meta-analysis (Bouma et al., 2016; Du et al., 2022; Jia et al., 2019; Ouedraogo et al., 2020; Daryanto et al., 2017; Elias et al., 2018). This method is especially relevant when experimental variances are missing, sample sizes (number of studies) are small, and/or distributional assumptions of parametric methods might be unfulfilled (Adams et al., 1997). However, as discussed by Laird & Louis (1987), Zhou et al. (1999) or Saravanan et al. (2020), hierarchical structure (thus dependence) in observations (i.e. effect sizes) should be accounted for in the resampling process, otherwise the variability depicted in the

posterior distribution is underestimated (increase of type-I error), potentially invalidating inferences. In this study, hierarchical bootstrapping was thus performed, i.e., the resampling was sequenced by successive hierarchical levels in order to mimic the dependencies between related observations (i.e. several trials from the same study). Specifically, the hierarchical bootstrapping consisted first in resampling the studies and then resampling the trials (effect sizes) within the previously resampled studies. This single hierarchical resampling operation was replicated 10 000 times, the mean (or weighted mean, see Section 2.2.2.3) effect size being calculated on each bootstrap sample. As originally intended by Efron (1979), the median and 95th percentile confidence interval were computed from the hierarchically bootstrapped posterior distribution of the means (Adams et al., 1997). Hierarchical nonparametric bootstrapping was performed in R Studio 2022.07.2, based on the 'resample' function written by Shotwell (2012).

2.3 Results

First, it should again be stressed that the response ratios reported in this study measure the runoff (or erosion) mitigation effectiveness on a seasonal (i.e., cropping season) time scale. For the winter cover crops, this corresponds to the reduction in runoff or erosion during the late summer-fall-winter period, while for tied-ridging it reflects the mitigation effect in spring and summer (between sowing and harvest of potatoes). For conservation tillage, the cropping season depends on the crop type: from autumn to summer for winter crops, or from spring to autumn for spring crops. Comparing these RR's is therefore not straightforward.

The overall mean (random effects model) or median (bootstrapping) runoff or erosion mitigation effects ($1 - e^{\widehat{\ln(RR)}}$) for the 3 investigated practices are reported in Figure 2.2. The overall effect sizes calculated by both methods returned (nearly) the same values. Overall, each of the three conservation practices yielded a significant mitigation effect on runoff and soil erosion, as compared to their control treatment, except for conservation tillage on runoff for both random effects modelling and bootstrapping, and for cover crops on runoff and erosion as computed by bootstrapping. As frequently reported in related

reviews (Du et al., 2022; Xiong et al., 2018), the mitigation effect is higher for soil losses than on runoff.

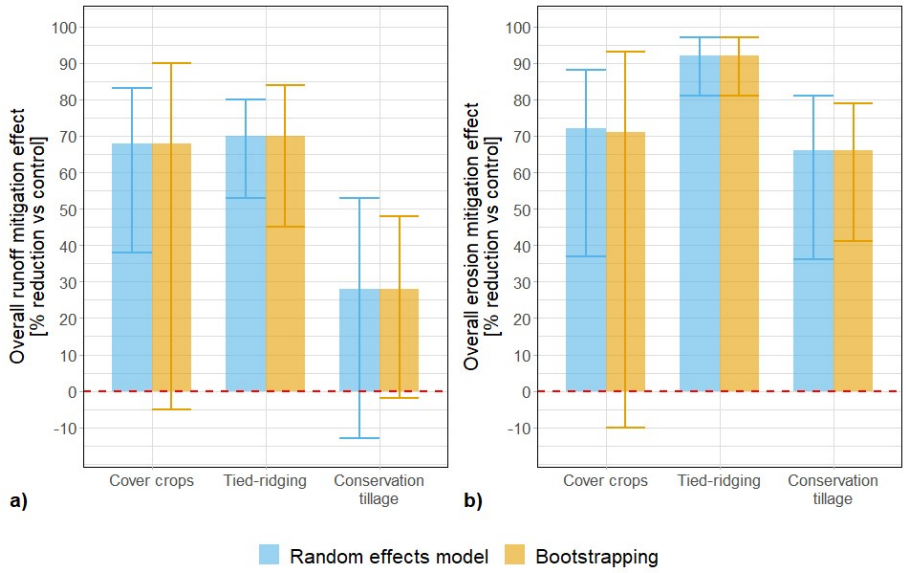


Figure 2.2 : Estimated overall (without moderators) average mitigation effect (= 1-RR) and associated 95% confidence intervals, for cover crops, tied-ridging and conservation tillage, derived from meta-analysis by both random effect(s) models and hierarchical nonparametric bootstrapping for a) runoff outcome and b) erosion outcome.

The forest plots of the meta-analyses of the 3 practices for the runoff and erosion outcomes are provided in Figure A.2 in Appendices, listing each study's individual contribution to the overall effect size, and reporting I^2 heterogeneity indicators (Eq. 2.5). Within the hierarchical structure, between-plots variability surprisingly appeared to be responsible for only a small proportion of the total variability in effect size. Inter-plot variabilities (= 1-I^2_{trials}) for the 2-level random effects (trials>plots) models on winter cover crops for runoff and erosion are indeed lower than 2.5%. Differences between trials accounted for the largest proportion of total variability. For the 2-level random effects (studies>trials) models on conservation tillage, heterogeneities across trials (I^2_{trials}) reached around 70%, whereas between-study variabilities (I^2_{studies}) were smaller but still meaningful (approx. 30%).

2.3.1 Winter cover crops

Compared to a bare soil, winter cover crops mitigate late summer-autumn-winter runoff by 68% and soil losses by 72% on average (Fig. 2.2). All initially tested moderators (Table 2.1) were included in the final meta-regression model (lowest AIC). As the number of years of winter cover crop implementation increases (Fig. 2.3e, f) and as the maximum cover percentage achieved by the cover crop increases (Fig. 2.3c, d), the seasonal mitigation effect on runoff and erosion becomes stronger. As an example, compared to a control plot undergoing shallow stubble tillage, the runoff mitigation effect of winter cover crops becomes statistically significant when the max. vegetation cover exceeds 55% (Fig. 2.3c) or starting from the second year of application (Fig. 2.3e). Furthermore, the more intensive the stubble tillage on the corresponding bare control plots, the lower the relative effectiveness of the catch crop (Fig. 2.3a, b). For instance, compared to a bare ploughed control treatment (deep stubble tillage), a winter cover crop does not significantly decrease surface flows (it could even increase runoff), whereas the effectiveness of cover crops is highest when compared with an untilled control. Alternatively, compared to a control plot that is left untouched after harvest of the main crop (i.e., no stubble tillage, which is a common practice), the runoff mitigation effect by winter cover crops would become statistically significant when the max. vegetation cover exceeds 42% (unshown). The equations and parameters values of the final meta-models are available in Appendices, Table A.2.

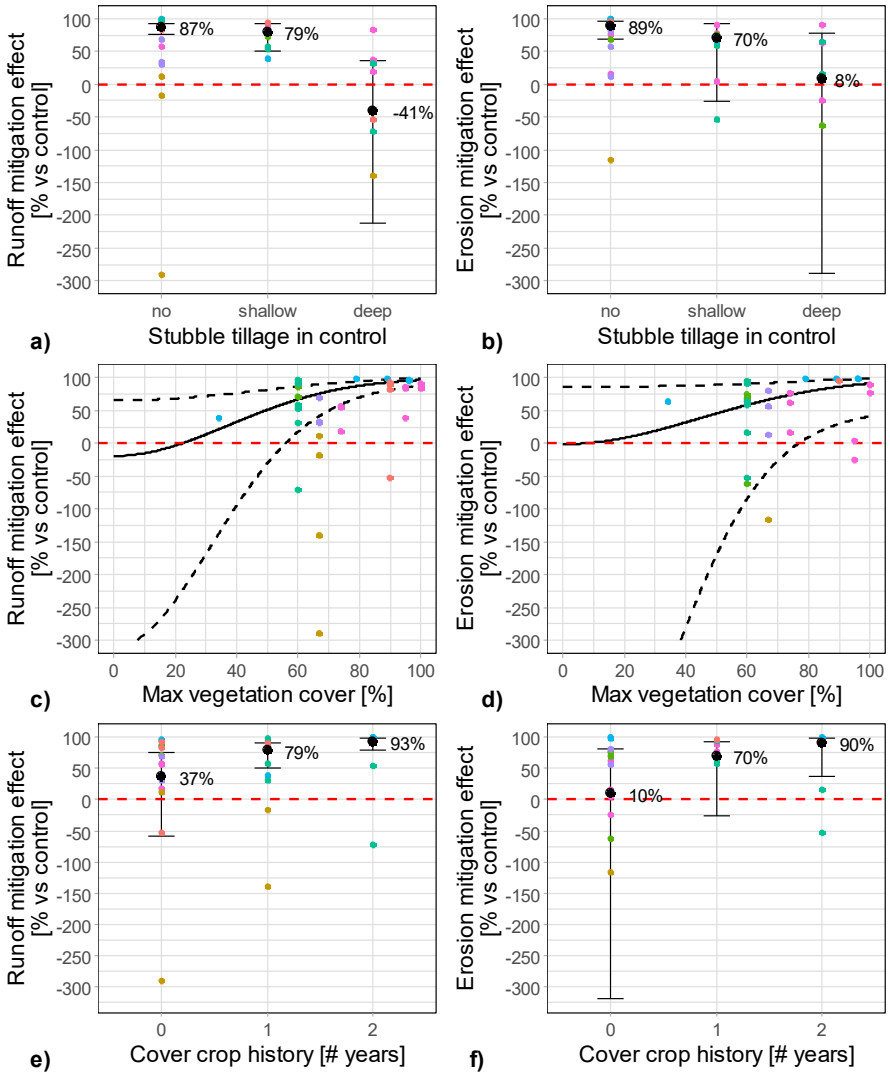


Figure 2.3 : Predicted runoff and soil loss mitigation effect (1-RR, i.e. % reduction relative to the control) by winter cover crops and associated 95% CI, as a function of relevant moderators (selected in the final meta-model) considered individually. Coloured points are measured trial effect sizes from the original meta-dataset, with points in the same colour coming from the same study. Note that several out of y-scale points with extreme negative effect sizes could not be plotted for readability reasons.

2.3.2 Potato tied-ridging

Compared to classically hilled and furrowed potato crops, tied-ridging mitigates seasonal runoff by a mean of 70% and soil erosion by 92% (Fig. 2.2). Based on AIC, neither slope (range: 3.0 to 8.8 %), nor maximum daily rainfall in spring (May and June) (range: 15 to 66 mm) were retained in the final meta-model (Table 2.1).

2.3.3 Conservation tillage

Overall, conservation tillage alleviates seasonal overland flow by 27% and associated sediments losses by 66% (Fig. 2.2). The moderators that were retained in the meta-regression for runoff outcome are the following: the number of years of conservation tillage application, the crop type, the number of primary and secondary tillage passes, soil texture, tillage depth, and tillage tool type. For erosion outcome, the same variables were retained based on AIC except that passive tillage type was not included while preceding crop type was. For tillage depth and tillage tool type moderators, dummy coding of categorical variables (Table 2.1) resulted in the selection in the final meta-model of only two groups out of the three initially coded classes: intermediate depth vs. other depths (i.e. deep and shallow) for the tillage depth moderator variable, and passive tools vs. other types (i.e. none and powered) for the tillage tool type variable. The equations and parameters values of the final meta-models are available in the Appendices, Table A.3. Cultivating spring crop as main crop, with a winter crop as preceding crop, on a sandy loam soil, on plots under conservation tillage for a long time, with as few tillage passes as possible, by using passive tools (i.e. neither direct drilling, nor powered tools), and not at intermediate tillage depth, are all factors independently positively correlated with the mitigation impact of conservation tillage on surface flows of water and/or sediment (thus negatively correlated with ln RR runoff and/or erosion). Based on Figure 2.4a-b, the predicted runoff and soil loss mitigation effect becomes statistically significant starting from the 2nd to 3rd successive year of conservation tillage. The impact of conservation tillage on runoff and erosion reduction is null or not significant for a winter crop, but is above 50% for a spring crop (Figure 2.4c, d). Regarding the preceding crop type effect, conservation tillage decreases soil losses by 66% after a winter

crop, but by 56% after a spring crop. Conservation tillage is predicted to cut runoff and erosion by respectively 51% and 63% on a silty(clay) loam soil, and by 66% and 83% on a sandy loam (Fig. 2.4e, f).

To improve understanding of the relationship between combinations of tillage variables and the response ratios, empirical predictions were made using the final meta-model and considering four fictitious typical tillage conservation systems: 1) “no-till”, one single shallow direct sowing operation; 2) “strip-tillage” like system, with 2 passes including one with a passive tool at intermediate depth; 3) "deep tillage" system, with 3 passes using passive tools, at least one of which at high depth; and 4) an "intensive" system with 4 passes at intermediate depth, using powered tools (not passive). The other variables were fixed at their average (rounded) value in the initial meta-dataset, i.e. a spring crop, succeeding to a winter crop, on a silty(-clay) loam soil, on a field under conservation tillage for 4 years. Although there was no statistically significant difference between the four systems (Fig. 2.4g, h), on average the intensive system would be the least effective at reducing both water and sediment losses. The best performing system against runoff would be the deep tillage (-61%), while against erosion it would be the no-till system (-82%).

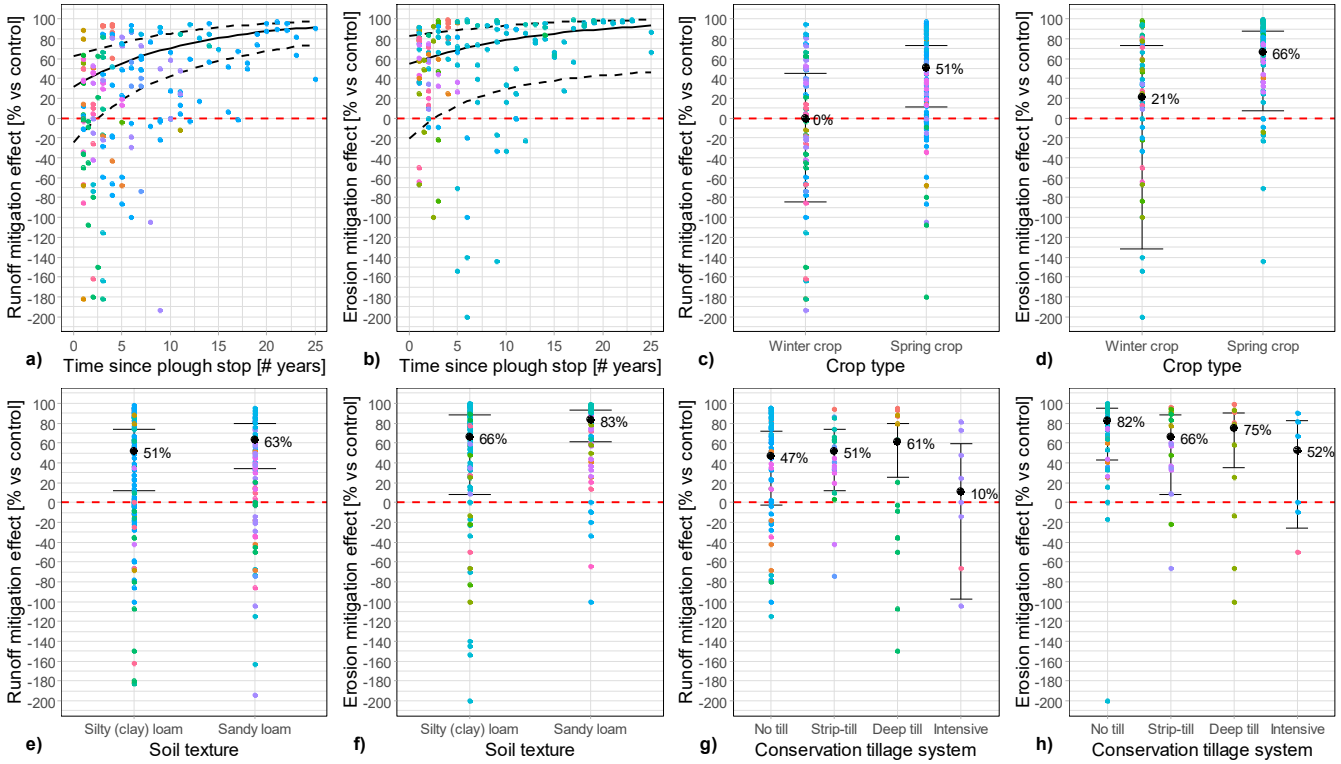


Figure 2.4 : Predicted runoff and soil losses mitigation effect (1-RR, i.e. % reduction vs control) by conservation tillage and associated 95% CI, as function of selected moderators. Coloured points are trials effect sizes from original meta-dataset, with points in same colour coming from the same study. Note that several points with extreme negative effect sizes could not be plotted for y-scale readability. g) and h) : see Section 2.3.3 for definition of tillage systems.

2.4 Discussion

2.4.1 Random effects model vs. hierarchical bootstrapping for meta-analysis

Overall, the average outcomes from the two statistical methods are consistent. However, as expected, the uncertainties calculated by the random effects models are smaller than those from the bootstrapping method when a decisive hierarchical level (in this case "studies") is neglected in the random effects models as is the case for cover crops and tied-ridging (Figure 2.2). For that reason, the most reliable estimates in these cases are probably those produced by hierarchical bootstrapping. For the conservation tillage meta-analysis, "studies" and "trials" levels were accounted for in the random effects models, but confidence intervals are slightly narrower for bootstrapping than for random effects models (Figure 2.2). In the latter case, as also observed by Saravanan et al. (2020) or Kulkarni et al. (2022), hierarchical bootstrapping may have performed better than the random effects model in terms of retaining statistical power without sacrificing Type-I error rate.

Random effects models are commonly used in meta-analyses. Inclusion of fixed effects in such models allows testing the impact of moderators on effect size. Yet, the availability of primary data constrained the parameter (random effects) estimations (Section 2.2.2.5), and hence the appropriate consideration of hierarchical dependencies and the accuracy of the resulting inferences. Hierarchical bootstrapping is presented here as an innovative adaptation of resampling methods whose outcomes (overall mitigation effects, Figure 2.2) reinforce those obtained by random effects models. This is further supported by its ability to reproduce the original hierarchical structure in the resampling process and the absence of distributional assumptions. After the methodological study of Zhou et al. (1999) introducing the "two-stage bootstrap" to analyse rates of success of a medical procedure from different studies, the present paper is the first actual attempt of using hierarchical (nonparametric) bootstrapping for meta-analysis. Indeed, to our knowledge, all published meta-analyses using nonparametric bootstrapping have been performing "simple"

bootstrapping, even on hierarchically-structured datasets (i.e. random resampling without accounting for dependencies between effect sizes). Potential consequences of this practice were described in Section 2.2.2.6 but mainly relate to poor inference. As expected, we found substantial differences between boundaries of confidence intervals calculated by simple and by hierarchical bootstrapping on the same hierarchical dataset (not shown in this paper).

2.4.2 Overall mitigation effects and moderators

The overall RR's of the present study for cover crops yield slightly lower estimates (i.e. higher mitigation efficiencies) than those from Maetens et al. (2012a) for Europe and the Mediterranean, with absolute differences around 13% for runoff and 6% for soil loss. A direct comparison with the latter study is, however, somewhat tricky since Maetens et al. (2012a) included perennial grass vegetation in vineyard in the cover crop category. Our outcomes match even more closely with those of Blanco-Canqui (2018) on cover crops in the USA, with absolute differences in runoff and erosion reduction percentage between both reviews below 3%.

The effects of moderators selected in the ultimate meta-regression model for cover crops appear to make sense from a physical perspective. Indeed, the effect of the maximum vegetation cover can typically be explained by greater protection of soil against sealing, lower detachment by splash, higher infiltration capacity and high hydraulic roughness (Blanco-Canqui & Ruis, 2020; Kervroëdan et al., 2019). The positive effect of the number of years of cover cropping can be attributed to soil structure improvement resulting from the recurrent supply of fresh biomass to the soil (Tisdall & Oades, 1982), although in our meta-analysis this variable could only be assessed for a limited number of successive years (from 1 to 3). Long term studies on the cumulative effects of cover crops on runoff and erosion are noticeably lacking. It should also be noted that, in practice, it is not so common to implement cover crops each year on the same plot. When spring crops alternate with winter crops, winter cover crops can only be implemented once every two years (in between the winter crop and spring crop). In such

cases, the cumulative benefits of cover cropping will be reduced, possibly close to single year trials.

The relative effectiveness of cover cropping strongly depends on the choice of control treatment. It is greatest if the control corresponds to post-harvest conditions. On the contrary, performing stubble tillage on control plots considerably increases infiltration capacity (by breaking up the sealing crust) and surface roughness (van Dijk et al., 1996b). The absolute magnitude of surface flows in the control plots is thereby strongly attenuated, as is the associated mitigation effect provided by the cover crop. Although the benefits of cover cropping for runoff and erosion reduction are less obvious when compared to stubble tillage, growing cover crops still remains a valuable practice given the numerous other benefits (prevention of nitrate leaching to groundwater ("catch crop"), addition of fresh organic matter to soil, etc.; Dabney et al., 2001; Thapa et al., 2018).

Regarding tied-ridging, Sittig and Sur (2023) also recently reviewed the potential of micro-dams to mitigate runoff and soil loss on agricultural fields, mainly in maize and potato. For the latter crop, our two databases broadly share the same primary studies. Unsurprisingly, the effectiveness computed in the present review (Fig. 2.2) thus fall very close to their geometric mean reduction of 81% for runoff and 89% for erosion for micro-dams in potato crop. Compared to reservoir tillage techniques applied to other crops (e.g. maize, barley), mitigation efficiencies are higher in potato crops (Sittig & Sur, 2023; Clement et al., 2023; Wolka et al., 2018), with absolute differences of 1-10% for runoff and 9-30% for soil losses, respectively. Indeed, because potatoes are cropped on ridges, the water storage capacity of the micro-basins is generally higher than for other crops grown in the absence of ridges, and a larger part of runoff water converges towards micro-dammed furrows (Clement et al., 2023).

For tied-ridging, surprisingly no significant effect of the moderator variables was found in our meta-regression, even though several authors reported a decrease in the surface storage capacity of depressions with slope (Martin, 2009) or micro-dam breakdown following intense spring storms (Vejchar et al., 2019; Olivier et al., 2014).

The small size of the meta-dataset may have restricted the ability to capture such effects, while the limited reliability of rainfall data from "closest" rain gauge stations to the trial sites (approx. 6 kilometers away) may also have contributed. Nevertheless, it should also be acknowledged that, considering the ranges of slopes and rainfall covered by our meta-database (see Section 2.3.2), these two factors could have a marginal impact on the overall effectiveness of tied-ridging in the Western European context. This is illustrated by the very low significance of these two moderators when included in the meta-model; computed p. values are 0.56 and 0.80, respectively, for the "slope" and "max. daily rainfall in spring" variables.

In light of current controversies regarding the effect of conservation tillage on runoff (Blanchy et al., 2023; Xiong et al., 2018; Maetens et al., 2012a; Mhazo et al., 2016), we show that in Western Europe, conservation tillage has at least an approximately null, but very probably an actual average reduction effect on runoff, albeit low (-27%). This outcome is closely aligned with the one calculated by Maetens et al. (2012a) for Europe and the Mediterranean. Regarding the erosion control effect of conservation tillage, the present study reveals a 19% greater median effectiveness than previously reported by Maetens et al. (2012a).

Moreover, Maetens et al. (2012a) reported a decrease in runoff mitigation effectiveness with practice history between 1 and 14 years from 154 plot-years data. Based on a larger meta-dataset (up to 25 years, 366 plot-years), our review reveals the opposite trend: an overall decrease in surface fluxes (runoff and soil losses) as the number of years of conservation tillage increases, consistent with the findings of Mhazo et al. (2016). In the longer term (> 5-10 years), runoff and soil loss mitigation prevail, most likely driven by improvements of the soil structure under conservation tillage. The concentration of organic carbon in the topsoil enhances the soil aggregate stability, and soil biological activity increases vertical macroporosity (Zhang et al., 2007; Tebrügge & Düring, 1999). In the first years after stopping inversion tillage (ploughing), however, more soil compaction and more surface crusting could occur, reducing soil infiltration capacity. As a result, runoff may actually be enhanced in the first years after stopping inversion tillage, thereby restricting the associated effectiveness of conservation

tillage in mitigating runoff as reported by a number of studies (Fig. 2.4a; Hartmann et al., 2017). Nevertheless, during the same period, soil erosion is already limited, thanks to the soil protection from the higher proportion of crop residues in the unploughed plots (Ranaivoson et al., 2017).

Our research also indicate that the seasonal mitigation effect of conservation tillage is higher for a spring crop than for a winter crop, a conclusion which was also reached by Mhazo et al. (2016) for soil losses even though they observed the opposite for runoff. This is probably related to the higher absolute magnitude of surface flows (i.e., the runoff or erosion rate) under spring crops than under winter crops (Panagos et al., 2015b). Indeed, in our meta-dataset, the mean erosion rate in control plots is 1.4 t/ha for winter crops while it is 20.3 t/ha for spring crops. This leads to a more pronounced mitigation potential for spring crops under conservation tillage, as it has also been reported for other factors controlling the absolute magnitude of surface fluxes such as soil type, slope steepness or length, climate erosivity (Mhazo et al., 2016; Du et al., 2022; Leys et al., 2007; Maetens et al., 2012a). In addition, because spring and winter crops often alternate in rotations (under European temperate climate, at least), the presence of non-harvested residues (stubble from cereals or residue from winter cover crops) at the soil surface in the spring when spring crops are being sown is expected to be greater than in winter crops preceded by a spring crop since the latter often leave less or less lignified residue behind. This may therefore also explain the greater mitigation efficiency of conservation tillage in spring crops. This difference in mitigation effect between spring and winter crops is particularly noteworthy given that conservation tillage is more commonly applied on winter crops than on spring crops (Lahmar, 2010). This distinction should therefore be considered when evaluating the expected benefits of conservation tillage adoption at larger spatial scales.

The meta-analysis revealed a greater mitigation effect on sandy loam than on silty(-clay) loam soils. This was unexpected because the latter are more erodible, something that is also confirmed by the measured erosion rates which are higher for the silty(-clay) loams than for the sandy loams. One hypothesis for this outcome, already put

forward by Du et al. (2022), could be a greater sensitivity to compaction for finer-textured soils (compared to coarser-textured soils) under conservation tillage. In other words, the benefits of conservation tillage on soil structure are partly offset by damage from increased soil compaction, and this offset would be more pronounced on silty(-clay) loam than on sandy(-loam) soils.

The predicted efficiencies for the 4 illustrative conservation tillage systems match outputs from other reviews as well. Maetens et al. (2012a) indeed showed that, on average, deep tillage is the most effective conservation tillage system to mitigate runoff, while reduced tillage is the least effective one. Against erosion, no tillage is the most effective and reduced tillage is the least effective. In the present study, splitting the reduced tillage category into lighter (strip-tillage-like) and more intensive systems appears to be key to explain part of the observed variability for such a practice covering a broad range of tillage schemes. These discrepancies in performance between tillage systems can be physically explained by the fact that on the one hand, deep tillage breaks both the surface seal and any deep plough pan (or “no-till” pan; Schlüter et al., 2018), greatly limiting runoff (Sojka et al., 1993). No-till on the other hand, by strongly concentrating soil organic matter near the soil surface and leaving the soil little or undisturbed (maintaining soil particle cohesion and adequate soil cover), has the greatest benefit on soil structure and thus sediment losses (Zhang et al., 2007; Tebrügge & Düring, 1999). The intensive system, by thoroughly mixing the topsoil and thereby burying residues and diluting the SOM content, by strongly fragmenting the soil aggregates, and/or by damaging macroporosity, probably acts similarly to a conventional ploughing system in terms of impact on soil structure and erodibility (Carter, 1991). Nevertheless, in Europe, reduced tillage causes lower crop yield loss and is much more widespread than no tillage (Van den Putte; Lahmar et al., 2006). When attempting to assess the impact of soil management practices on erosive runoff risks (Panagos et al., 2015c), it would be valuable to identify more closely the typologies of reduced tillage systems preferred by farmers within the broad range of possible intensities in tillage schemes. This is, however, a major challenge given the highly dynamic decision-making process by farmers over the time scale of a crop rotation: occasional ploughing and subsequent loss of beneficial effects (Blanco-

Canqui & Wortmann, 2020), adaptation of tillage strategy according to weather (and soil) conditions, crops choices depending on market prices, etc.

Despite the identification of some determinants for the mitigation effect of the three conservation practices considered in this study, there remains high (unexplained) variabilities between trials effect sizes, thus not attributable to pure sampling variability. There is therefore still room for an explanatory capacity that is not yet understood. As an illustration, the erosion meta-regression model for winter cover crops only allowed to lower the I^2_{trials} (between trials heterogeneity, Eq. 2.5) value to 96.9% compared to 97.8% for the empty (without moderators) model. The absolute magnitude of surface flows is known to be an important factor of this interannual variability (Maetens et al., 2012a); the higher the scale of runoff and soil losses (e.g. in control plots), the higher the mitigation effect of the conservation practice. This was also observed in our study. As an illustration, when introducing the natural logarithm of the mean erosion rate in control plots as a potential moderator in the conservation tillage meta-regression, this moderator variable is selected in the final meta-model with a *p*-value <0.0001. Yet we decided not to directly include the magnitude of surface flows as a potential explanatory variable, as it cannot be managed per se but is mainly weather-related, thus of random nature. Besides, some poorly reported factors could not be investigated in this study but might be decisive. For example, considering conservation tillage, one could mention the presence (or even the biomass) of winter cover crop in the rotation and the subsequent amount of residues at the soil surface (Ranaivoson et al., 2017), the soil (moisture) conditions at the time of tillage and the subsequent presence of wheel tracks (Withers et al., 2006), more differentiations in crop types, etc. Methodological aspects like plot size (Langhans et al., 2019) could also be involved in this persistent unexplained variability between trials, although the use of an effect size comparing two treatments under the same conditions (e.g. response ratio) at least partly corrects for this.

2.4.3 Data availability, limitations, and biases

The scarcity of primary experimental data quantifying the effects of winter cover crops, tied-ridging and, to a lesser extent, conservation tillage on runoff and erosion is striking, even though these practices are being widely promoted against surface flow-related risks (Soane et al., 2012; Winteraeken & Spaan, 2010; Sittig & Sur, 2023). While the present study confirms the overall effectiveness of these techniques, the data scarcity and high variability in results across trials does not allow to properly investigate the effectiveness of these measures under a wide range of agronomic management and/or environmental conditions (moderator variables) as would be needed for supporting site-specific policies. To better identify the impact of these variables and thus optimize the implementation of such techniques, a larger number of published studies (and long-term trials) are needed, together with experimental designs featuring replicated plots, as well as thorough reporting of experimental conditions and results. The lack of primary data also constrained the present study to focus on the mitigation effect of each conservation practice alone, independently from other cropping practices. This is an evident limitation, as several conservation techniques can obviously be combined on the fields.

Furthermore, as already pointed out in Section 2.3, the response ratios indicate the mitigation effect of a conservation practice for a given season, depending on the technique considered (and on the crop type for the case of conservation tillage). Direct comparison of RR's between different crops or practices, as well as direct extrapolation to a full year (e.g. use of erosion RR as USLE P factor) should therefore be avoided. However, a RR for a whole year or rotation could be estimated for a given crop, by weighting seasonal RR's by the (region-specific) average proportion of runoff or soil loss in the relevant season for the specified crop (Brychta et al., 2018), and by assuming that the practice is ineffective beyond the monitored seasons.

The use of ln(RR) as the effect size measure may have led to an underestimation bias of these mitigation efficiencies, by excluding some trials from the meta-databases. Indeed, for seasons featuring low runoff and soil loss measurements on the control plots, conservation practice

plots may not have produced any surface flow at all, which should correspond to a maximum mitigation efficiency. In these situations, the response ratio is, however, not computable because the denominator value is zero. No such case occurred in the meta-database for cover crops and tied-ridging but 14 trials could not be included in the meta-analysis on conservation tillage. The resulting underestimation for the conservation tillage meta-analysis is, however, probably offset to some degree by an overestimation of the effectiveness arising from the high risk of publication bias (Section 2.2.2.4) (Philibert et al., 2012; Chandler et al. 2021). The latter could be partly related to the smaller proportion of trial results derived from grey literature than in the other two meta-datasets (Table 2.1). Indeed, as opposed to white literature, many studies from the grey literature remain unpublished because they are characterized by small sample sizes, unexpected effect size and/or non-significant outcome (known as the "file drawer" issue). Exploring and including data from grey literature in meta-analyses helps to counteract the risk of publication bias and subsequent misestimation in effect size (Haddaway & Bayliss, 2015; Hopewell et al., 2010).

2.5 Conclusion

All three soil and water conservation practices analysed in this study, i.e. winter cover crops, tied-ridging in potato crop, and conservation tillage, are effective at mitigating runoff and soil erosion compared to current conventional practices under typical Western European conditions. From a methodological point of view, hierarchical nonparametric bootstrapping provided more precise effect size estimates than random effects models, positioning it as an appealing innovative method for meta-analyzes dealing with small datasets and/or non-normal effect size distributions. It is also shown that the time since the abandonment of ploughing, the crop type, or the specific conservation tillage scheme strongly impact the effectiveness of the conservation tillage practice. Nevertheless, the fairly low number of primary studies (for tied-ridging and cover crops) and of long-term studies (especially for cover crops), and the high unexplained variability (for all three techniques) indicate the need for additional studies in order to further unravel the conditions leading to the greatest effectiveness. Moreover, the strong suspicion of publication bias for the meta-dataset

on conservation tillage requires careful interpretation of the quantitative literature on this subject, and beyond that, a change of publishing and reviewing "mindset" for editors, referees and scientists involved in this field.

Meanwhile, the present review, by providing quantitative estimates of one of the main assets of conservation agriculture practices, can assist farm advisors or policy makers in deciding to what extent, in which situations, and/or by what mechanisms the adoption of these conservation practices should be promoted. A sensitive trade-off will have to be settled between surface water and soil conservation issues, and other societal requirements such as groundwater quality (Alletto et al., 2010), food security (Godfray et al., 2010; Van den Putte et al., 2010), reduction of greenhouse gas emissions (Garré et al., 2022; Palm et al., 2014; D'Haene et al., 2009), and reduction in the use and impacts of plant protection products (Wynn & Webb, 2022). Agronomic research on innovative conservation farming practices that are compatible with these pressures must also be pursued.

CHAPTER 3: SOIL PITTING MITIGATES RUNOFF, EROSION AND PESTICIDE SURFACE LOSSES IN MAIZE CROPS IN THE BELGIAN LOESS BELT[2]

[2] Adapted from Clement, T., Bielders, C. L., Degré, A., Manssens, G., & Foucart, G. (2023). Soil pitting mitigates runoff, erosion and pesticide surface losses in maize crops in the Belgian loess belt. *Soil and Tillage Research*, *234*, 105853. https://doi.org/10.1016/j.still.2023.105853

Abstract

Soils of the loess belt of Western Europe are intensively cropped and particularly prone to runoff and soil erosion, especially when planted with row crops such as maize or potatoes, characterized by a low soil cover during erosive spring storms. In this context, micro-basin tillage techniques could help mitigate these risks while being fairly easy to integrate into conventional cropping systems, yet very few studies have investigated the effectiveness of this technique in a maize crop. Soil pitting was therefore tested across 14 site*years of field trials in Belgium by a newly designed roller adapted to a seeding unit forming small depressions in-between maize rows. Runoff and surface losses of sediments and pesticides were measured on soil pitted plots under natural rainfall and compared with conventionally tilled plots. Seasonal runoff and erosion rates were reduced on average by 69% and 83%, respectively, following soil pitting. Median curve number (CN) values calibrated ($\lambda = 0.05$) on this dataset are 68 for the control, and 63 for the soil pitting treatment. Analysis of individual rainfall event data reveals that the mitigation effect remains consistent throughout the season, and is even slightly higher for highly erosive rainfall events than for light events. Whereas herbicide concentrations, and hence in-situ ecotoxicity risk indicators, were similar between control and soil pitted treatments at the experimental plot scale, at larger scales the environmental impact on water bodies would be mitigated by soil pitting thanks to the reduction in absolute runoff and soil loss. Future research should further investigate the impacts of soil pitting on crop yields under a broad range of rainfall conditions.

3.1 Introduction

Silt loam soils of the European loess belt are intensively used for industrial cropping but are known to be prone to surface crusting, runoff and erosion (Panagos et al., 2014; 2020). On sloping fields, soil erosion by water is a major cause of cropland degradation (Osman, 2014; Monnier and Boiffin, 1985; Virto et al., 2014), threatening the short-term and long-term land productivity as a result of the direct damage to crops and the loss of soil, organic matter and nutrients (Pisa et al., 1999; Pimentel and Kounang, 1998; Boardman and Poesen, 2006; Reyniers et al., 2006). The

transfer of sediment, organic matter, nutrients and pesticides also results in various off-site impacts such as muddy floods (Evrard et al., 2007a) and degradation of the quality of surface water bodies (Cooper, 1993). At subnational scale, the direct economic costs – mainly related to downstream (muddy) flood management infrastructure – are worth several to tens of millions of euros per year (Verstraeten and Poesen, 1999; Patault et al., 2021), not accounting for the environmental and social impacts. Along with fertilizers, pesticides – a mainstay of conventional agriculture since the Green Revolution – are well-known contaminants of surface waters, whose transfer by runoff and sediment flows during extreme rainfall events is the main source of non-point source pollution (Karlsson et al., 2020; Huber et al., 2000). This is especially true for herbicides (as compared to insecticides or fungicides), as they are generally applied during runoff-prone periods (i. e., bare soil conditions). In many intensively cultivated watersheds, concentrations of active ingredients in surface water bodies have been observed to temporarily (but sometimes several times a year) exceed acute toxicity thresholds for aquatic life (Verro et al., 2009; Hitzfeld et al., 2022). In the future, the sensitivity of European rural landscapes to surface flow hazards is expected to rise given an expected higher occurrence of extreme rainfall events due to climate change on the one hand (Panagos et al., 2017) and a persistent pressure for land artificialization on the other hand (Darly et al., 2020). Dealing with soil erosion by water and its off-site consequences therefore remains a high priority.

To mitigate these risks at the watershed scale, a frequent response is to design flow management structures like riparian buffers, retention ponds or wetlands (Richet et al., 2017; Momm et al., 2014). Although these measures have proven to be generally effective for this purpose (Zhang et al., 2010; Land et al., 2016), it has been pointed out that their mitigation effect may sometimes be overestimated and that they are often not sufficient on their own (Stehle et al., 2016; Dabney et al., 2006). Such curative solutions should be considered downstream of – or combined with – field-scale practices that have a preventive function (Morgan, 2005). In this category, reduced tillage systems in particular are well-known solutions to control soil losses and, to a lesser extent, runoff (Maetens et al., 2012a; Mhazo et al., 2016). They are being endorsed by farmers to some extent (Lahmar, 2010), though adoption is still limited

for technical, perception, or lack of knowledge issues (Soane et al., 2012; Wauters et al., 2010). There is therefore a strong interest in developing and promoting additional soil conservation techniques that can be integrated more easily into conventional cropping systems. Micro-basin tillage or reservoir tillage techniques such as soil pitting or tied ridging are one possible avenue. They consist in forming during secondary tillage a specific pattern of soil surface depressions, that increases surface water retention (Jones and Stewart, 1990). During a rainfall event, when the rainfall intensity exceeds the soil infiltration rate, water is first stored in the depressions, leaving more time for infiltration, thereby delaying runoff initiation and reducing the overland flow volume (Truman and Nuti, 2010). These techniques are considered to be particularly suitable on deep, well-drained soils with low to moderate slopes (Morgan, 2005; Liu et al., 2019).

In industrial cropping systems, reservoir tillage techniques have been applied to various crops in North America: corn, sunflower, sorghum in USA (Jones and Baumhardt, 2007), potato crops in Canada (Gordon et al., 2011). Jones and Stewart (1990) reported results from Jones and Clark (1987) from two years of trials with diked furrows in a sorghum crop in Texas. They reported a 72% decrease in total runoff but also a 143% increase in total sorghum yield compared to open furrows, suggesting additional benefits as a result of water harvesting under water-constrained conditions. In Europe, several tied-ridging devices have also been developed and tested in potato crops during the last decade, with 50–80% reduction in runoff and 70–90% reduction in soil losses compared to conventional tillage (Vejchar et al., 2017; Olivier et al., 2014). For maize (*Zea mays*), another spring crop that often induces high soil erosion rates on sloping land (Laloy and Bielders, 2010), little is known regarding the potential of reservoir tillage techniques in the western European agronomic context. The effectiveness of a reservoir tillage technique could indeed depend on (interactions between) the tool used to form the micro-basins and their dimensions (Sui et al., 2016), the rainfall regime (Wiyo et al., 2000), the cultivated crop and associated traffic sequence, thereby affecting the retention and infiltration potential into the micro-basins or possible pattern damage (Liu et al., 2019).

To our knowledge, no roller intended to shape micro-basins in maize crops is currently available from agricultural equipment manufacturers. In recent years, a roller was designed by CIPF (a non-profit agricultural experimentation and extension organization in Wallonia, Belgium; CIPF, 2023), which creates shallow pits in-between the maize rows at the time of sowing. Sittig et al. (2020; 2022) reported that this technique reduced runoff by 24–71% and erosion by 54–81%. However, their results are based on a fairly limited dataset comparing soil pitting to a control across four site*years. In the present paper, we quantify the effectiveness of soil pitting at reducing cumulative seasonal runoff and soil losses in a maize crop, based on a much larger dataset: 14 site*years of field trials under natural western European rainfall. In addition, we analyzed the intra-seasonal, runoff and soil losses mitigation dynamics (i.e. at rainfall event scale) in order to assess possible damage to depressions. For 6 out of the 14 site*years, we also report on the mitigation potential of soil pitting on herbicide concentrations and loads in surface flows, and we evaluate the consequences in terms of surface water ecotoxicity.

The vulnerability of an area to overland flows (and their associated consequences) is highly dependent on the landscape layout, including topography and land use. Evaluating scenarios for the implementation of conservation farming practices therefore often relies on hydrological models (Haith & Loehr, 1979). Many of them operate on an empirical basis, using the SCS-CN method to calculate runoff (Hawkins et al., 2009; Deliman et al., 1999). Adjusting the curve number parameter for a given management technique is a common methodology in the literature (Wang et al., 2008; Musyoka et al., 2023; Sittig et al., 2022; Kincl et al., 2021). Accordingly, an additional objective of this chapter was to provide a curve number value that best fits the runoff measured from our trials, for each of the two treatments: conventional and soil-pitted maize crop.

3.2 Materials and methods

3.2.1 Data

Data processed in this study were collected by CIPF (2012-2017; personal communication) in the framework of the regional "Eruistop" project from 2012 to 2017, or extracted from the publication of Sittig et al. (2022) for 2018 and 2019 experiments at Bayer "ForwardFarm" in Huldenberg (Belgium). The full dataset encompasses a total of 6 different trial sites on silt-loam and sandy-loam soils and 8 different years, accounting for 14 site*year pairs in total (Table 3.1; Fig. 3.2). The soil types at the trial sites are typical of agricultural land in central Belgium: silt loam to sandy-loam soils with low organic matter content (<2.5%) (Boardman and Poesen, 2006; Gillijns et al., 2005). The range of slopes at the trial sites were on the high end of slopes found on cultivated land in Central Belgium (Gentile et al., 2009) so as to favor runoff and erosion and test the technique for conditions where erosion control is most needed. In Wallonia (southern half of Belgium) in 2021, about 40% of the cultivated area under silage maize was located on slopes above 5%, and 8% of the cropped area had slopes steeper than 10% (SPW, 2022b; Farr et al., 2007).

The trials were based on a fully randomized block design with a control and a soil pitting treatment with 1–3 replicates (Table 3.1). Silage maize was grown on all plots, sown at a density of approximately 95 000 seedlings/ha with a between-row spacing of 75 cm. Seeding and harvest dates are provided in Table 3.1. The control treatment was conducted in a conventional way: spring moldboard ploughing, rotary harrow tillage (twice), classic roller, and sowing with a precision disc seeder. For the soil pitting treatment, the spring moldboard ploughing was followed by rotary harrow tillage (once) and sowing with a seeding unit (rotary harrow + roller + coulter drill) in which the roller is specifically designed to shape shallow pits (approx. 6 cm deep, 25 cm long, 40 cm wide) in-between maize rows, mostly by soil scraping rather than pressing (see also Sittig et al., 2020). There were no wheels on the seeder of the seeding unit to avoid damaging the pit pattern, the seeder being driven by the rotation of the roller Fig. 3.1.



Figure 3.1 : Adapted roller from the seeding unit used to form the pits between maize rows (left) and subsequent pattern at plot scale (right). Source: CIPF.

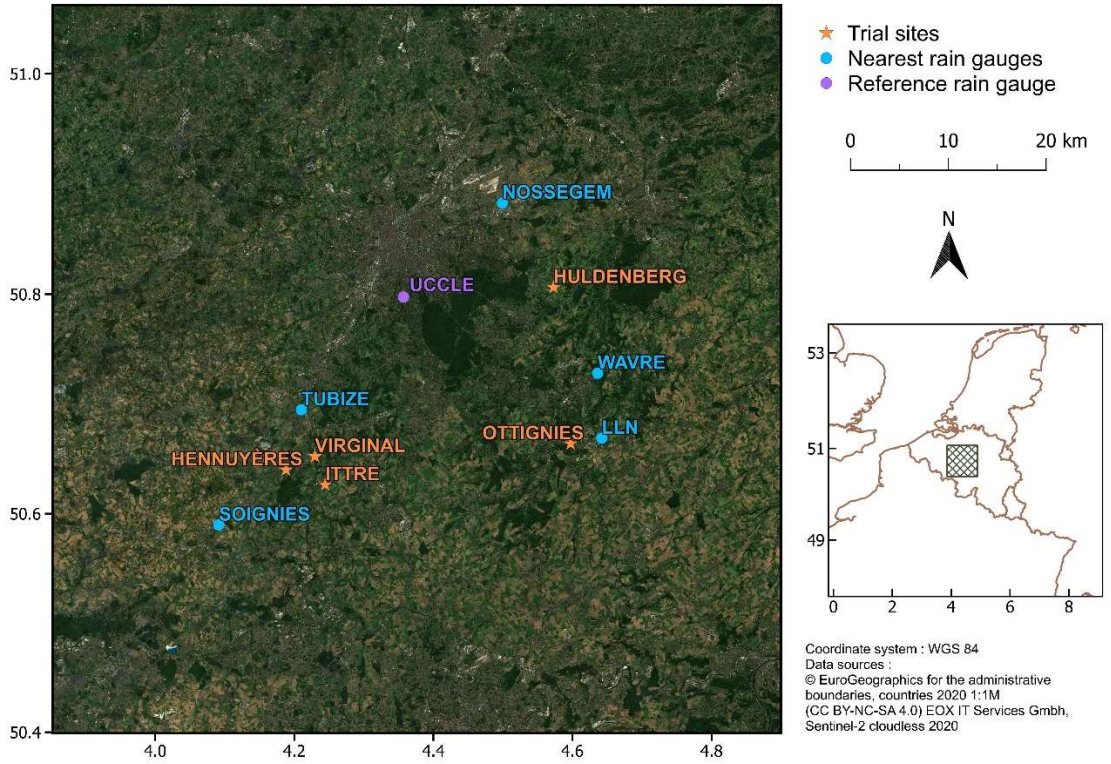


Figure 3.2 : Map of trial sites and rain gauges considered in the dataset. "Nearest rain gauges" are used for rainfall estimates at trial sites (see section 3.2.1 and Table 3.1), while the "reference rain gauge" is used for assessment of the long-term representativeness of the rainfall intensities recorded during the trials (see Section 3.2.1).

Erosion plots (54–75 m^2; Table 3.1) allowed to collect runoff and sediment (Wischmeier and Smith, 1978). Given that the 3-m plot width exactly matches one tractor pass width, two wheel tracks were found in each plot (pitted and control). These wheel tracks were, however, hardly or not apparent because the operations are generally carried out under dry soil conditions and the soil surface is reworked behind the tractor (tillage and/or seeder). Five-hundred-liter capacity tanks were placed at the downstream end of the plots to collect total runoff and sediment. Runoff volume was measured by a flow-meter pump or derived from the height of water in the tank. Total soil loss was determined based on total runoff volume and sediment concentration measurements. While thoroughly mixing the water and sediment in the tanks, 1-L samples were taken and oven dried at 45 °C during approx. 1 week. Note that one or more water collecting tanks overflowed for 6 events out of 88 reported by CIPF (3 records in Ottignies in 2014 and 3 records in Ittre in 2016). In these cases, for the affected plots, the runoff is underestimated (capped at the maximum tank capacity, i.e. 500 L), the sediment concentration is likely to be overestimated whereas the erosion rate is underestimated but to a lesser extent than runoff because the overflowing water was probably less loaded with sediment than the inflowing water as a result of sedimentation in the tank. Among the 54 runoff events reported by Sittig et al. (2022), overflow occurred for one record, but no measurement was reported so this event was not accounted for in the dataset. Note that in some cases, the water and sediment collected in the tanks resulted from more than one rainfall event (see further).

Pesticide concentrations were determined in 250 mL homogenized runoff water samples and/or decanted sediment samples by chromatography coupled with mass spectrometry at the Gembloux Office of Environment and Analysis (BEAGx). Herbicide concentrations in runoff and/or sediment samples were determined for some trial sites and years only (Table 3.1). Only average concentrations across all replicates of a given treatment could be retrieved from the existing reports.

Rainfall data were provided by tipping bucket rain gauges belonging either to the Walloon Mobility and Waterways Service (SPW, 2022a) or to the Walloon Agricultural Research Center (CRA-W, 2022), supplying hourly rainfall data in both cases. The spectrum of rainfall intensities encountered during the trials is representative of the local rainfall pattern. For the spring-summer period (May to September), the median, 75th quantile, and maximum hourly rainfall intensities at the Royal Meteorological Institute reference station in Uccle (2003–2023 data; Fig. 3.2) are 0.40, 1.10 and 27.60 mm/h, respectively. Considering all trials (sites and years), the same statistics for the rain gauges used in the present study (Fig. 3.2 and Table 3.1) are 0.30, 0.80 and 25.80 mm/h, respectively. For each trial site and year, the nearest rain gauge records were screened and all rainfall events that may have caused runoff ("potential runoff event") were identified. A rainfall event is defined as a period of rainfall preceded and followed by at least 6 h without rain (as in Bresson et al., 2017). A potential runoff-producing rainfall event was defined as a rainfall event with a cumulative rainfall > 5.0 mm, and a maximum hourly intensity > 1.5 mm/h. This rule is intended to exclude small rainfall events which most probably do not lead to runoff, when splitting a single measured cumulative runoff value over several potential events (see Section 3.2.3). The minimum values of cumulative rainfall (>5.0 mm) and maximum hourly intensity (>1.5 mm/h) were selected based on the characteristics of rainfall events producing no runoff when that information was available in the original CIPF reports, as well as on the basis of typical soil hydrological characteristics (minimum hydraulic conductivity and surface storage depth) under similar conditions (Laloy and Bielders, 2008). The following rain event characteristics were then calculated: event rainfall amount and duration, maximum hourly intensity, mean hourly intensity, kinetic energy estimated from hourly intensity based on van Dijk et al. (2002) with values of the coefficients for Belgium, rainfall erosivity EI_{30} based on EI_{60} and a conversion factor provided by Panagos et al. (2016). Cumulative time (days), cumulative rainfall height, and cumulative rainfall kinetic energy since sowing were also determined.

Table 3.1 : Characteristics of each trial site and year. Site name (municipality), mean slope steepness [%], soil texture class (grain sizes in % sand-silt-clay, respectively), organic carbon content [%], trial year(s), plot size [m length * m width = m² surface], number of experimental blocks (= replicates per treatment), sowing and harvest date, total seasonal rainfall (from sowing to harvest) [mm], total number of measured runoff and erosion events.

Site name	Slope [%]	Soil text. (Sa-Si-Cl)	OC cont. [%]	Trial year(s)	Plot size [m*m]	# exp. blocks	Sow. & harv. date	Tot. rain. [mm]	# meas.
Hennuyères[1] (Hen)	10	Sandy loam (31-55-14)	1.6	2012	25*3=75	2	28/6 – 24/10	240	10
Virginal[1] (Vir)	1	Silt loam (unknown)	Unknown	2013	25*3=75	3	6/5 – 4/10	306	5
Ottignies[1] (Ott)	6	Silt loam (12-70-18)	1.0	2013	25*3=75	3	26/4 – 16/10	272	5
				2014	24*3=72	3	30/4 – 16/10	403	10
				2015		3	16/4 – 8/10	254	3
				2016		3	4/5 – 10/10	238	3
				2017		1 (mean of 3)	25/4 – 2/10	219	2
Ittre[1] (Itt)	1	Sandy loam (32-52-16)	1.1	2015	24*3=72	3	28/4 – 3/10	233	6
				2016	25*3=75	3	5/5 – 3/10	289	7
				2017		1 (mean of 3)	28/4 – 21/9	215	4
Huldenberg (steep slope)[2] (Hul_SS)	16	Silt loam (11-74-15)	Unknown	2013	24*3=72	1	unknown	318	9
Huldenberg (medium slope)[2] (Hul_MS)	9			2013	24*3=72	1	unknown	318	9
				2018		1	unknown	214	13
				2019	18*3=54	1	unknown	385	10

[1,2] Reference (data source) for this trial. 1 = CIPF (2012-2017; personal communication) ; 2 = Sittig et al. (2022)

Table 3.1 (continued) : Herbicide application and monitoring (spraying date, dose and active compound, and medium analyzed: water and/or sediments), cumulative rainfall [mm] between sowing and spraying, rain depth of first runoff-producing rainfall event occurring after spraying (time between spraying and 1st event [# of days]), location of the nearest rain gauge providing hourly rainfall data and distance from trial site [km].

Site name	Trial year(s)	Herb. app. & monit. (medium)	Sow. – spray. cumul. rain [mm]	Rain [mm] of 1st runoff event after spray. (time [days])	Nearest rain gauge (km from site)
Hennuyères	2012	24/7 : 300 g/ha flufenacet (wat. & sed.)	91	8 (4)	Soignies (8.8)
Virginal	2013	No	N/A	N/A	Tubize (4.9)
Ottignies	2013	No	N/A	N/A	Louvain-la-Neuve (LLN) (3.2)
	2014	10/6 : 350g/ha flufenacet & 583 g/ha therbutylazine (wat.)	92	15 (17)	
	2015	9/6 : 350 g/ha flufenacet & 583 g/ha therbutylazine (wat.)	63	18 (13)	
	2016	7/6 : 350 g/ha flufenacet & 80 g/ha mesotrione (wat. & sed.)	86	22 (16)	
	2017	No	N/A	N/A	
Ittre	2015	9/6 : 350 g/ha flufenacet & 583 g/ha therbutylazine (wat.)	76	9 (9)	Tubize (8.0)
	2016	6/6 : 350 g/ha flufenacet & 75 g/ha mesotrione (wat. & sed.)	87	29 (1)	
	2017	No	N/A	N/A	
Huldenberg (steep slope)	2013	No	N/A	N/A	Wavre (9.8)
Huldenberg (medium slope)	2013	No	N/A	N/A	
	2018	No	N/A	N/A	
	2019	No	N/A	N/A	Nossegem (10.0)

3.2.2 Seasonal analysis

First, seasonal cumulative runoff as well as soil and pesticide losses were calculated for each trial site and year for each plot of both the control and soil pitting treatments. Herbicide concentrations below the limit of quantification were considered null. The overall mean seasonal amount of runoff and soil loss rate for each treatment was then statistically estimated by means of a 2-level hierarchical model (Makowski et al., 2019): site*year as random effect at level 2, experimental block (plot replicates) as random effect at level 1. The natural logarithm of the seasonal runoff or erosion rate was used as dependent variable. The logarithm allows to normalize the distribution of runoff and erosion rates, which is a required condition for the effect size measure in any linear (mixed) model (Koricheva et al., 2013). Because of the log-transformation of the dependent variables, these average seasonal runoff and erosion rates correspond to geometric – and not arithmetic – means (Martinez and Bartholomew, 2017; Feng et al., 2013). From the (exponential transformation of) intercepts of these models, the overall mean reduction in seasonal runoff and erosion rates by means of the soil pitting technique was estimated. This analysis could not be performed on the pesticide data because only mean pesticide concentrations across all replicates were available.

In order to interpret the herbicide concentrations in runoff water in terms of acute ecotoxicity risk for surface waters, the active substance concentrations were compared to Uniform Principle (UP) criteria, i.e. 0.01 * acute L(E)C50 values for *Daphnia* and fish and 0.1 * E(b)C50 values for algae and aquatic plants, according to standard European Tier I pesticide risk assessment (European Commission, 1997; Stehle et al., 2011). The total proportion of pesticide sampling events in which the concentration exceeded the UP criterion threshold was then calculated for pitted and control plots separately, for the four aquatic organism classes mentioned above and for the three herbicides monitored during the trials (Table 3.1).

3.2.3 Curve number fitting

The curve number (CN) method is a hydrological computation method developed by USDA-SCS (1954) which aims to calculate a direct runoff depth (Q [mm]) resulting from a rain event of total rainfall P [mm], with a dimensionless "curve number" (CN) representing hydrological response of a watershed based on soil characteristics (Eqs. 3.1 and 3.2):

$$Q = \frac{(P-\lambda S)^2}{P+(1-\lambda)S} \text{ if } P > I_a = \lambda S; \text{ else } Q = 0 \qquad \text{(Eq. 3.1)}$$

$$S = \frac{25\,400}{\text{CN}} - 254 \quad \text{(Eq. 3.2)}$$

There has been much discussion regarding the value of λ, the proportionality coefficient between initial abstraction I_a and potential maximum soil water retention S ($I_a = \lambda * S$). USDA-SCS (1954) originally proposed λ = 0.2, but most studies from the last decades have recommended λ = 0.05 (Ling and Yusop, 2014). It is now well accepted that it should be seen as a region-specific variable and chosen in the likely range of values 0.01 to 0.2 so as to maximize accuracy with local rainfall-runoff data (Ling and Yusop, 2014; Tan et al., 2018). For this reason, we tested lambda values of 0.2 and 0.05 and chose the one that minimized residual (difference between measured and CN-simulated runoff) sum of squares.

For both λ values (0.2 and 0.05), CN fitting was performed for the control and soil pitting treatments separately, by the median CN method as originally recommended by USDA NEH4 (Hawkins et al., 2009), on the 209 potential runoff-producing rainfall events of the dataset, using the mean runoff value of each treatment. The median-fitted CN is the CN value whose curve splits the rainfall-runoff pairs into 2 groups of the same size (same number of events). The median CN method was selected for its robustness over the least-squares calibration method, which minimizes the sum of squared differences between CN-simulated and observed runoff. From our dataset, least-squares fitting yielded unrealistically low CN values due to the influence of events characterized by very high rainfall but very low runoff (i.e., low-intensity but long-lasting rain events).

Because in some cases the collected runoff could have been generated by more than one prior rainfall event, the measured cumulative runoff was split up among all potential runoff-producing rainfall events by weighting the cumulative runoff by the maximum hourly intensity of each event (I_{max}) (Eq. 3.3).

$$\text{Runoff}_j \approx \frac{\left(\sum_{j=1}^{n} \text{Runoff}_j\right) * I\text{max}_j}{\sum_{j=1}^{n} I\text{max}_j} \qquad \text{(Eq. 3.3)}$$

where j correspond to one single potential runoff-producing rain event, and n is the total number of potential runoff-producing rainfall events included in a period between two actual runoff measurements. $\sum_{j=1}^{n} Runoff_j$ corresponds to the collected runoff.

3.2.4 Event-scale analysis

An event-based statistical analysis was conducted in order to study the impact of event characteristics on the ability of the soil pitting practice to mitigate runoff and soil losses. The natural logarithm of the response ratio (RR) was selected as the effect size measure (Eq. 3.4):

$$\ln \text{RR}_{\text{runoff}_{i,j,k}} = \ln\left(\frac{\text{Runoff}_{\text{soil pitting}_{i,j,k}}}{\text{Runoff}_{\text{control}_{i,j,k}}}\right) \qquad \text{(Eq. 3.4)}$$

where $\text{Runoff}_{\text{treatment}_{i,j,k}}$ is the runoff amount [mm] for a given treatment (soil pitting or control) for the *k*-th experimental block, the *j*-th rain event, and the *i*-th site*year pair.

In this event-scale analysis, when a runoff (or soil loss) measurement was available for a period including several potential runoff-producing rainfall events, only the rainfall event with the highest kinetic energy during the period was taken into account as dependent variable since the focus is on the response ratio and not on the absolute runoff (or soil loss) values. This was done so as not to artificially decrease the between-events variability. Indeed, the response ratios of each potential runoff-producing rainfall event from the same runoff (or soil loss) measurement take the exact same value since the weighting (Eq. 3.3) equally affects the runoff (or soil loss) in the control and in the pitted treatment. Events with zero runoff in the control treatment (while some runoff was measured in pitted treatment) are not included in this

analysis (= 7 events) because the response ratios cannot be calculated. The opposite (runoff measurements from pitted but not from control plots) did not occur. This dataset includes 84 runoff events. Consistent with the hierarchical structure of the database, statistical analyses of event response ratios for runoff or soil losses were performed using a mixed model with 3 nested-levels (Makowski et al., 2019), with site*year pairs as random effect at level 3, rain events as random effect at level 2, and experimental blocks (replicates) as random effect at level 1. The "site" and "year" sources of variability were combined into a single "site*year pair" random effect (corresponding to the trial-related source of variability, accounting for 14 clusters) because there were not enough clusters in the sites (6) or years (8) individually to consider them as crossed random effects (Harrison et al., 2018). Rain event characteristics as described above (cfr Section 3.2.1) were included individually as fixed effect in the model in order to investigate a possible effect on ln(RR). We used a random intercept model only and not a random slope model, i.e. we assumed that the effect of explanatory variables, if any, was only a fixed effect of the same magnitude for all site*year trials. The number of site*year clusters is indeed too low to estimate additional variance components, and we are more interested in the overall effect of the rain event characteristics on runoff or erosion mitigation across all trials than in the effect variability between trials (Oberpriller et al., 2022). Software used for the event-scale as well as the seasonal-scale statistical analyses was IBM SPSS Statistic v.27.

3.3 Results

3.3.1 Seasonal runoff, soil and pesticide losses

Mean measured seasonal runoff, erosion rates, and pesticide surface losses are shown for each site*year in Figs. 3.3, 3.4 and 3.5, respectively. Overall geometric mean seasonal runoff estimated by the 2-level random effect model is 5.49 mm (95% confidence interval (CI) is [2.85, 10.56 mm]) for the control and 1.72 mm (95% CI is [0.69, 4.28 mm]) for the soil pitting treatment, corresponding to a mean runoff reduction rate of 69% by the soil pitting practice. Overall geometric mean seasonal erosion rate is 1.99 t/ha (95% CI is [0.55, 7.13 t/ha]) for the control and 0.33 t/ha (95% CI is [0.09, 1.20 t/ha]) for the soil-pitted plots, giving a mean soil

loss reduction rate of 83% by means of soil-pitting. Flufenacet, therbutylazine, and mesotrione surface losses are lowered by on average 65%, 66% and 57%, respectively, although these results should be considered carefully given the limited number of monitored site*years. Note that pesticide losses in Ottignies 2014 were very similar in the pitted and control treatments. This is because average concentrations of flufenacet and terbuthylazine in runoff water after the first event following the spraying date were higher for pitted plots (124 µg flufenacet/L and 144 µg terbuthylazine/L) than for control plots (77 µg flufenacet/L and 104 µg terbuthylazine/L), offsetting the observed reduction in runoff in the soil pitting treatment. Across all sampling events, the average (across sampled events) ratios of mean (across replicates) herbicide concentrations in runoff water for pitted and control plots are 1.14 ± 0.97, 1.50 ± 1.62, and 0.55 ± 0.38 (mean ± standard deviation) for flufenacet, terbuthylazine, and mesotrione, respectively. There is thus no evidence of any significant differences in herbicide concentrations in runoff water between control and pitted plots.

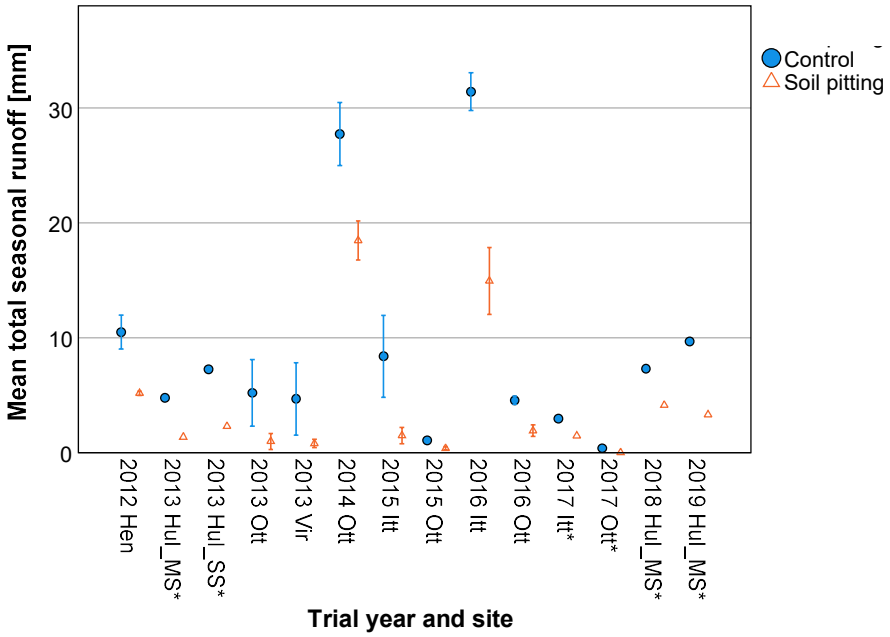


Figure 3.3 : Mean (across plot replicates) total seasonal runoff for all soil pitting trial years and sites. Error bars correspond to standard deviation. Site*years marked with an asterisk (*) correspond to trials without replicates (only 1 plot per treatment) thus without error bars.

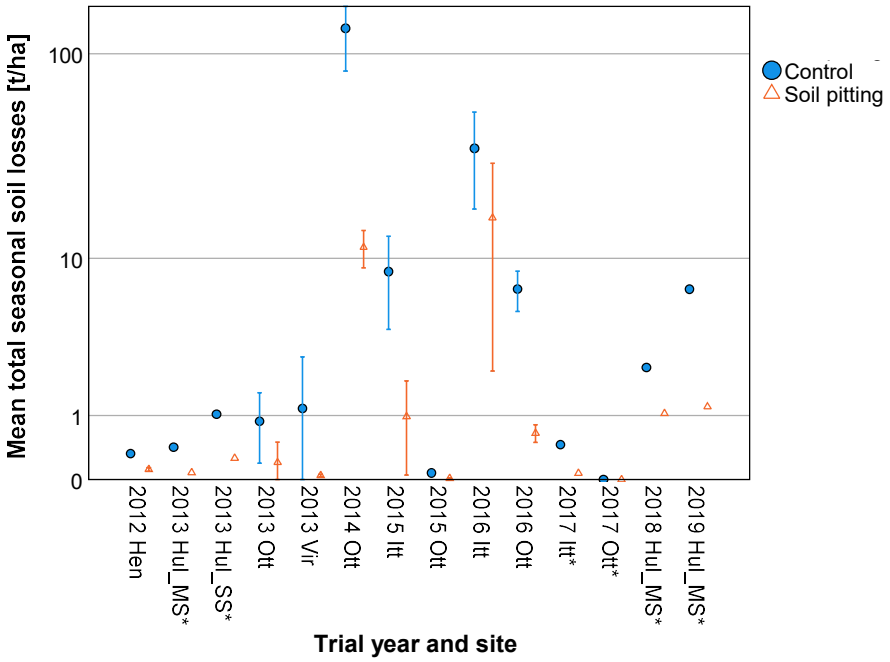


Figure 3.4 : Mean (across plot replicates) total seasonal soil losses for all soil pitting trial years and sites. Logarithmic scale for y-axis. Error bars correspond to standard deviation. Site*years marked with an asterisk (*) correspond to trials without replicates (only 1 plot per treatment), thus without error bars.

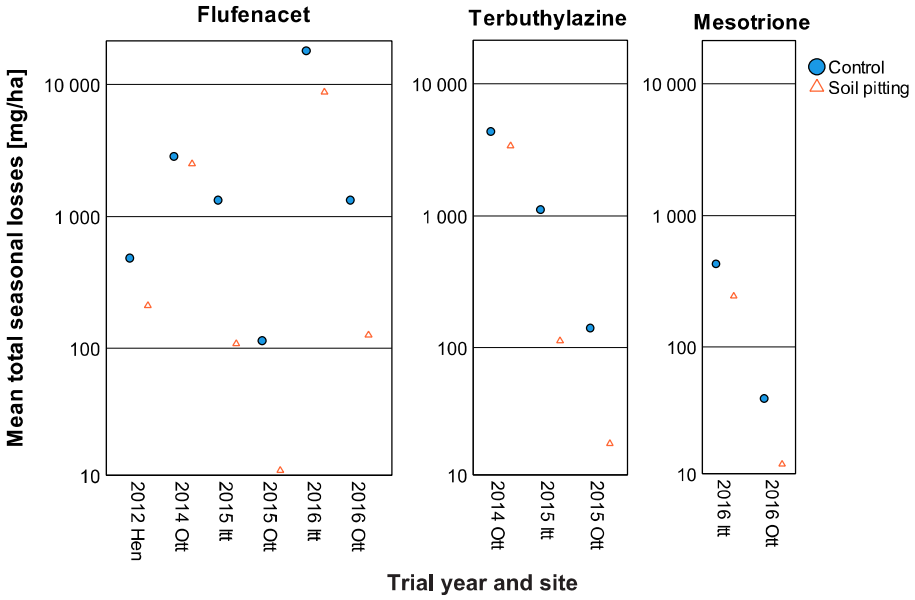


Figure 3.5 : Mean (across plot replicates) total seasonal pesticide surface losses for all soil pitting trial years and sites for which pesticides were monitored. Logarithmic scale for y-axis. The figure does not feature any error bars because only means of replicates for each treatment were available.

Table 3.2 lists the proportion of sampling events for which mean herbicide concentrations in runoff water exceed the UP criterion for the considered herbicide and type of aquatic organism. The values taken by this indicator vary from 0% to 100% of sampling events depending on the active substance and the organisms considered. Depending on the active substance, algae and/or aquatic plants were most frequently exposed to concentrations that are toxic for those organisms. However, the exposure is identical or very similar between the control and pitted treatments.

Two different soil types and two different slope steepness classes can be identified from the trial site characteristics (Table 3.1): silt loam and sandy loam, and moderate slope (9–10%) and steep slope (15–16%). Unfortunately, it is not possible to examine the effect of soil type and slope steepness independently because of a significant correlation between these two variables in the data set (Spearman's correlation coefficient = 0.65).

Table 3.2 : Monitored active substances, total number of pesticide sampling events (all site*years considered), reference (source) for toxicity data, target organism class, lowest value of toxicity concentration: acute L(E)C50 for fish and aquatic invertebrates (Daphnia) and E(b)C50 for algae and aquatic ("higher") plants, and proportion of sampling events where herbicide concentration exceeds UP criterion threshold (see Section 3.2.2), for control and pitted plots.

Active substance	Total # of pesticide sampling events	Reference for toxicity data	Target aquatic organism	Lowest value of toxicity concentration	% samplings > UP criterion in control plots	% samplings > UP criterion in pitted plots
Flufenacet	22	European Commission (2003)	Fish	2.13 mg/L	23%	23%
			Aquatic invertebrates	30.9 mg/L	0%	0%
			Algae	2.04 µg/L	100%	95%
			Aquatic plants	2.43 µg/L	100%	95%
Terbuthylazine	16	EFSA (2011)	Fish	2.2 mg/L	31%	31%
			Aquatic invertebrates	N/A *	N/A	N/A
			Algae	12 µg/L	88%	88%
			Aquatic plants	12.8 µg/L	88%	88%
Mesotrione	5	EFSA (2016)	Fish	71 mg/L	0%	0%
			Aquatic invertebrates	49 mg/L	0%	0%
			Algae	3.5 mg/L	0%	0%
			Aquatic plants	7.7 µg/L	60%	60%

* N/A: According to EFSA (2011), "No definitive acute toxicity endpoint was derived from the submitted aquatic invertebrate studies as neither of the submitted studies used a suitable method to determine the amount of terbuthylazine in solution. However, the studies were considered to be of adequate quality to clearly demonstrate that terbuthylazine is of less toxicity to aquatic invertebrates than other aquatic species and therefore the risk assessment for fish is deemed to cover the aquatic invertebrate risk assessment

3.3.2 CN fitting

The median-fitted CN values are 68 for the control and 63 for the soil pitting treatment, with respective root mean square errors (RMSE) of 1.7 mm and 1.3 mm (Fig. 3.6). A λ value of 0.05 suits better to our field data; the residual sum of squares (RSS) with λ = 0.05 are more than 3 times lower compared to RSS values with λ = 0.2. Considering both treatments, mean absolute error (MAE) between observed and CN-estimated runoff values is 0.67 mm. For 65% of the events, the error is smaller than the measured runoff by at least a factor of 2.

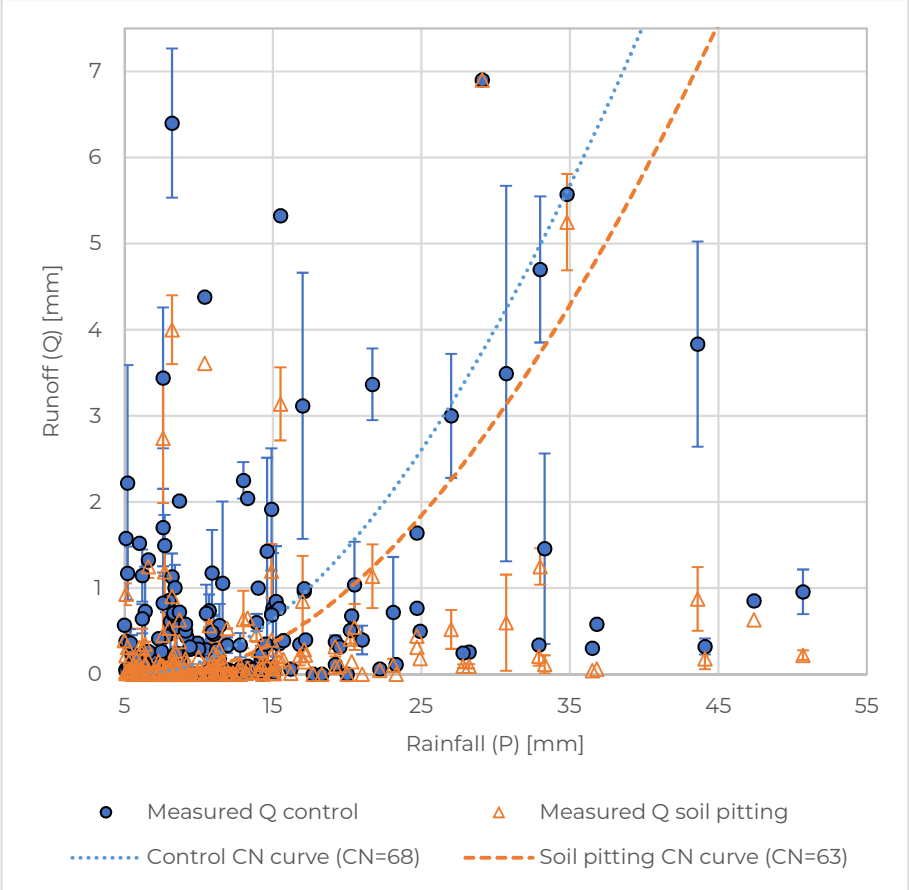


Figure 3.6 : Measured rainfall-runoff data and curves for median-fitted curve numbers (λ = 0.05), for control and soil pitting treatments in maize crop. Measured runoff values are means across plot replicates. Error bars correspond to standard deviations. Points without error bars are single plot (no replicates) experiments. Only rainfall events > 5 mm were considered (see Section 3.2.1).

3.3.3 Event scale analyses

We anticipated that the surface retention capacity of the soil pits could decrease over time because of progressive filling by sediment and/ or breakage during erosive rainfall events. However, neither cumulative time, nor cumulative rainfall, nor cumulative rainfall kinetic energy since sowing show a statistically significant effect on the runoff or erosion response ratios.

Similarly, rainfall event indicators (total rainfall, maximum hourly intensity, average intensity, kinetic energy or erosivity) also do not show a statistically significant effect on the runoff or soil loss response ratios. Differences in rainfall event characteristics therefore do not explain differences in soil pitting efficiency (ln(runoff RR) or ln(erosion RR)). Nevertheless, there is a significant effect of the ln(mean runoff in control plots) on the ln(runoff RR) (p-value = 0.036) and, similarly, a significant effect of the ln(mean erosion rate in control plots) on the ln(erosion RR) (p-val. = 0.042). This means that for events that actually produce more runoff or are more erosive (high runoff or erosion values in control plots), the soil pitting efficiency is greater than for events that produce less runoff or are less erosive. Based on the coefficients of these mixed models, the estimated runoff RR for light runoff events (mean runoff of 0.5 mm in control plots) is around 0.34 while it drops to 0.26 for major runoff events (mean runoff of 5 mm in control plots). Regarding the soil loss RR, the estimated value for little erosive events (mean erosion rate of 0.1 t/ha in control) is around 0.13, while for a severe event (mean erosion rate of 10 t/ha in control), it drops to 0.08.

3.4 Discussion

Overall, the mean seasonal runoff and erosion rates in control plots of 5.49 mm and 1.99 t/ha in the control plots may appear fairly low. However, because of the log transformation, these values correspond to geometric means and not arithmetic ones (Martinez and Bartholomew, 2017; Feng et al., 2013) and must therefore be interpreted accordingly. As an illustration, the arithmetic mean seasonal runoff in the control plots in Ottignies from 2012 to 2017 was 7.8 mm, i.e., 2.8% of the average seasonal rainfall. For the same site and years, the arithmetic mean

seasonal erosion rate was 28.0 t/ha, which is far beyond the well accepted threshold of about 10 t/ha considered as tolerable in the long term (Morgan, 2005). There is thus a strong need to adapt intensive maize farming practices for sustainable soil and water management.

Across all sites and years, soil pitting proved to be highly effective at reducing runoff (- 69%) and erosion (- 83%) rates. Although soil pitting is sometimes discouraged on steep slopes (Liu et al., 2019) or poorly draining soils (Morgan, 2005), runoff and erosion mitigation effects observed at trial sites in the present study are significant at all locations despite the steep slopes (10–16%) and soils that are highly susceptible to slaking given their texture (sandy loam and silt loam) and generally low soil OC content (Table 3.1). Unfortunately, slope and soil type effects could not be studied independently because of strong confounding between these two variables in the dataset. Nevertheless, it is expected that the surface storage capacity of the soil pits will decrease with increasing slopes. Based on a single site, Belis (2018) estimated by close range photogrammetry that the field-scale surface storage of soil pitting in maize is of the order of 7.4, 5.5, 4.5, and 3.8 mm for 0%, 5%, 10% and 15% slopes, respectively. Thus, the rate of change in surface storage appears to diminish with slope steepness. Based on this, given that our study sites feature moderate to steep slopes (10–15%), the differences in surface storage across sites are expected to be small. Even though we could not strictly study the effect of slope independently of other site factors, this may explain why the between site(-year) variability in mitigation effect remains relatively low, at least compared to the inter-event variability.

In addition to its applicability across a wide range of plot conditions, a major interest of soil pitting also lies in its convenient integration into a conventional cropping system. In order not to damage the depressions, pitting and seeding must be carried out in a single cultivation operation (typically with a seeding unit), and weeding should not be performed mechanically. The latter condition could, however, be overridden if soil pitting could be integrated to mechanical weeding, allowing for a reshaping of the depressions. Management of other cropping operations (spraying, harvesting, stubble plowing) is not impacted by soil pitting, except for a slight pitching of the machinery. This technique is not expected to exacerbate compaction problems, as

pits shaping by the adapted roller is mostly done by soil scraping rather than pressing. Compared to reduced tillage, the seasonal runoff and erosion reduction potential of soil pitting appears more than twice as high (Maetens et al., 2012a). However, no improvement in soil structure is expected from this technique in the longer term since it is based solely on the physical effect of depressions (water storage and possibly reduction in flow velocity). It is therefore not expected to prevent other soil degradation issues encountered in many agricultural fields, such as compaction and organic matter depletion (Jones et al., 2003; Heikkinen et al., 2013). Further studies should consider integrating this practice into cropping systems that also improve soil structure, such as reduced tillage (e.g., Sittig et al., 2022) and intercropping. Since it may take several years to observe the consequences of a conservation tillage system on soil properties (Rhoton, 2000), the soil pitting practice could also be considered as a way to mitigate the erosive risk on a field during the transition period between a conventional and a fully established conservation tillage system. However, the technical suitability of soil pitting in combination with no tillage or in the presence of significant levels of plant residues should first be evaluated, as the depression shaping could be hindered by crop residues and a more compact topsoil (Casamitjana et al., 2009). Given that proper pit shaping by the roller requires a fine and loose soil, similar concerns would apply for soils with a higher clay or stone content than those tested in our study.

In addition to limiting runoff volumes and soil losses, reservoir tillage techniques are expected to increase soil water content (Sui et al., 2016). In currently water-stressed regions, a positive impact on crop yields is sometimes reported (Jones and Clark, 1987). In western European areas, the impact of micro-basin tillage on soil moisture and crop yields is not well documented but should be further studied in terms of water stress reduction in a context of increasingly dry spring/summer months (Trnka et al., 2011) or in terms of (fungal) disease susceptibility, especially regarding late blight risk (*Phytophtora infestans*) in potato crops.

Based on measured seasonal pesticide losses, one can expect reduction in pesticide losses with the soil pitting technique to be of the same order of magnitude as for runoff and erosion. Unfortunately, it was

not possible to perform a reliable statistical analysis due to poor reporting of active substance concentration replicates. The calculated herbicide concentration ratios between control and pitted treatments and their confidence intervals (Section 3.3.1) seem nevertheless to indicate that herbicide concentrations from pitted and control plots are broadly similar, implying that the observed pesticide loss mitigation effect enabled by soil pitting stems predominantly from the reduction in runoff (and soil losses) rather than from any reduction in pesticides concentrations in these media. This is applicable regardless of whether the pesticide is mostly transported as solute in runoff or in association with sediment, at least in the range of Koc values of the pesticides used in the present study, which range from 52 mL/g for mesotrione (EFSA, 2016) to 202 mL/g for flufenacet (European Commission, 2003). Nevertheless, since the pesticide concentrations are similar in both treatments, similar ecotoxicity risk indicator values (i.e. proportion of sampling events in which the concentration exceeds the UP criterion) were obtained for both treatments (Table 3.2). This analysis does not, however, consider watershed scale processes. Indeed, the total runoff volume delivered downstream would be lower from a pitted soil as compared to a conventionally tilled field, leading to lower pesticide loads (=concentration*volume) transferred into water bodies (streams, ponds, ditches). Consequently, at the landscape (watershed) scale, lower herbicide concentrations in surface water bodies would be expected if the soil pitting practice were applied on (part of) the upstream area, as modeled by Sittig et al. (2022). Besides, some values of the ecotoxicity risk indicator are very high for some active substances and classes of surface water organisms, in particular algae and aquatic plants. These values must be interpreted with caution because the considered concentrations were measured in direct runoff water collected at the field level, whereas the UP criteria are intended for environmental surface water toxicity assessment. At the landscape scale, in addition to the dilution effect in water bodies discussed above, additional regulations regarding spraying (e.g., buffer strips along water bodies) are often applicable, mitigating their out-of-field impact.

The calibrated curve number values found in this study fall broadly within the same range as those computed by Sittig and et al., (2020, 2022). Depending on the treatment, the trial site and year, and the

calibration method (event-based or inverse modeling), these authors found CN values ranging from 63 to 81 (with λ = 0.2). Using inverse modeling, Sittig et al. (2022) calculated an average decrease of 4 CN units in soil pitting compared to conventional cropping practices (from mean CN=78.5 for control to mean CN=74.5 for soil pitting). Although a strict comparison of our results with those of the study of Sittig et al. (2020) is difficult because of the non-linearity of the rainfall-runoff relationship and because both studies used different values of λ, this decrease seems consistent with our own calibration results. Based on the larger data set used in the present study, we calculated a reduction by about 5 CN units, from CN= 68 for control to CN= 63 for soil pitting. Nevertheless, curve number values reported by Sittig et al. (2020) for potato tied-ridging trials show a much higher reduction potential, by 10–36 CN units between control and diked furrow plots (CN values in control potato plots ranging from 75 to 95). Two factors may explain this greater effectiveness of micro-basin tillage in potato crops compared to pitting in maize. First, the storage capacity in-between the dams is higher in dyked potato crops (12 mm for 10-cm high, 2-m apart microdams on a 5% slope; Martin, 2009) compared to pitted maize (5.5 mm on a 5% slope; Belis, 2018). In addition, all water converges towards the furrows in a potato crop, whereas in maize crops part of the runoff also flows along the seeding row and is therefore not captured by the soil pits (Liu et al., 2019). We would like to emphasize that the CN values provided in this study (Section 3.3.2) are intrinsically related to the λ value used for calibration, i.e., 0.05. Using other λ values would require conversion of our CN's; see for example the concept of "conjugate CN" in Hawkins et al. (2009). More generally, the CN values calibrated in this chapter are intended to assess the overall benefit of adopting soil pitting. However, the CN method, which relies on an empirical relationship between event rainfall and runoff, does not account for critical factors such as rainfall intensity or the soil surface state at the time of the event (Zhang et al., 2019; Martin et al., 2010). More

Contrary to Liu et al. (2019) who documented a negative correlation between rainfall intensity and the water or soil conservation efficiencies (1 – RR) of furrow diking, we did not find that the runoff and erosion mitigation effectiveness of soil pitting depends on the characteristics of the rainfall events at the trial sites. Based on the

dataset studied here (i.e., with approx. 6-cm deep depressions formed by an adapted seeding roller on maize crop and under typical western European natural rainfall regime), there is thus no evidence that the ability to mitigate runoff and soil losses by pitting maize inter-rows decreases over time at a seasonal scale. The discrepancy between the two studies may partly result from the much lower rainfall intensities in the present study (0–22 mm/h) compared to the study of Liu et al. (2019) (60–120 mm/h). In addition, the way the micro-basins are formed may also play a role. Indeed, furrow dykes are made of loose soil and are thus susceptible to collapse - especially if water overflows the dykes, whereas soil pits are dug into the soil and may therefore be more stable. The narrow range of plot lengths (from 18 to 24 m long) in the present study should also be remembered. A future avenue would be to investigate the role of plot length and subsequent overland flow accumulation on the stability of the depressions, thus on the mitigation effect of soil pitting.

Despite the lack of statistical correlation between all tested rain event characteristics (see Sections 3.2.1 and 3.3.3) and the runoff (or erosion) mitigation potential of soil pitting, a statistically higher runoff and erosion reduction efficiency by soil pitting was identified in our dataset for, respectively, higher measured runoff and soil loss in the control plots. This somewhat contradictory result may partly be caused by the fact that rainfall indicators are only proxies for the capacity of rain to induce erosion. The processes affecting runoff and soil loss are complex and nonlinear, involving multiple interacting environmental factors (crop, weather, soil life, etc). These processes cannot be fully encompassed by rainfall-derived indicators. Ultimately, the runoff and erosion measured in the control plots may be better indicators of the runoff / soil loss potential associated with a rainfall event at a given moment in time and in space. Besides, the fact that the rainfall gauges were sometimes distant by several kilometers from the trial sites introduces additional uncertainty in rainfall data (especially for convective storms in spring), thereby decreasing the correlation between rainfall data and observed plots runoff and soil losses.

Finally, the variance component estimates of the random effects allowed us to calculate the proportion of total variability accounted for

by each of the three levels included in the mixed models used for the event scale analysis. For the empty model (without explanatory variable) with ln(RR runoff) as dependent variable, the between site*year variability accounts for 16% and the inter-event variability is 25%, implying that the major part of total variability (59%) is related to the blocks (plot replicates). The same orders of magnitude are found with the ln(RR erosion). This suggests that a large part of the total variability is purely experimental variability, and therefore inherently difficult to explain, which is a well-known feature of erosion plot research (Nearing et al., 1999).

3.5 Conclusion

In intensively farmed lands, the magnitude and impacts of runoff and soil erosion require adopting cropping practices that mitigate these processes. Based on the large dataset analyzed in the present study, soil pitting appears as a little-constraining, highly effective soil and water conservation practice. Unlike some reported observations in tied-ridging, the effectiveness of depressions formed by soil pitting at mitigating runoff and erosion seems to be persistent throughout the whole growing season and for a wide range of rainfall events, at least under western European conditions. The technique can furthermore be easily inserted in the conventional tillage practices. Nevertheless, given the investment and the lack of proven additional direct benefits – besides soil conservation – policy analysis studies should be conducted to assess by which drivers and incentives, to what extent, and in which contexts adoption of the soil pitting practice should be supported. The curve number values calibrated in this paper can be used in hydrological models (e.g. SWAT, PRZM) to assist with such purpose.

Given its exclusively mechanical effect, association of soil pitting with other soil conservation practices would be meaningful to address physical, chemical and biological soil quality issues. Impact of soil pitting on soil water content and crop yields should also be further studied, in view of the more frequent drought stress periods that are expected in some parts of Europe as part of climate change.

CHAPTER 4: ABILITY OF STRIP-TILLAGE AND UNDERSOWING IN MAIZE CROPS TO PREVENT RUNOFF, SOIL EROSION, AND PESTICIDE SURFACE LOSSES: EXPERIMENTS AND MODELING

Abstract

In Western Europe, the silt loam belt is particularly at stake of erosion and muddy floods because of very erodible soils, intensive tillage and a high proportion of annual spring crops that leave the soil bare in spring coinciding with high intensity convective rainstorms. It is therefore essential to pursue the search for conservation practices that best reconcile agricultural production with soil and water protection. To this end, we assessed the potential of two practices to limit runoff, soil erosion and associated herbicide surface losses in maize during field trials under natural rainfall compared to a sole-maize control: undersowing conventionally tilled maize with red fescue or white clover and strip-tillage. The treatments were implemented in triplicate, on 6 trial sites, split over 3 cropping seasons from 2021 to 2023. Because of constraints on the choice of herbicides, weed control proved particularly challenging in the undersowing technique. Furthermore, no mitigation effect was observed on surface flows, and maize yields were reduced by 12% compared to the control treatment. Strip-tillage significantly decreased runoff (-31%) and soil erosion (-60%), but also reduced maize yields (-11%). Based on measured runoff and soil losses from our field experiments for the control and strip-tillage treatments, we further parsimoniously calibrated the process-based model CREHDYS, in order to conduct a scenario analysis for the silt loam belt context across a range of soil, rainfall and slope conditions. Following this, the average seasonal erosion rate simulated for the conventional tillage scenario (6.2 t/ha) was almost double that with strip-tillage (3.4 t/ha). The flood hazard would also be approximately halved. Future research could focus on other environmental benefits to be expected from undersowing, or on where and how to promote adoption by farmers of strip-tillage as a step stone to no till systems.

4.1 Introduction

On cultivated farmlands in Western Europe, bare (non-vegetated) soils, typically around sowing time, are vulnerable to surface sealing and splash detachment, especially those with high silt and low organic carbon content (Morgan, 2005; Blanco-Canqui & Lal, 2010). Soil compaction by heavy machinery used in mechanized agriculture

further reduces their infiltration capacity (Hamlett et al., 1990; Hamza & Anderson, 2005). High-intensity rainstorms are then likely to trigger sediment-laden Hortonian runoff, leading to flash floods and muddy floods in downstream areas (Van Dijk & Kwaad, 1996a; Boardman et al., 1994). Especially where rural areas are densely inhabited, such flooding damages public and private infrastructure (Evrard et al., 2007a). Moreover, fertilizers and pesticides commonly used in intensive agricultural systems are transported by runoff towards surface water bodies, causing diffuse pollution and threatening aquatic flora and fauna (Bach et al., 2001; Liess et al., 2021). Over the long-term, the imbalance between soil erosion and soil formation rates jeopardises the sustainable use of the soil resource, primarily on steep-sloping fields (Montgomery, 2007; Panagos et al., 2015c). The central Belgium area crossed by the silt loam belt meets all these aggravating factors, and is therefore prone to erosion and flash (muddy) floods (Gentile et al., 2009; Boardman & Poesen, 2006). In this region, spring crops such as maize, potato or sugar beet are of particular concern because the sowing period around April-May coincides with an increased occurrence of stormy rainfall events (Ballabio et al., 2017). The risk of soil erosion is expected to worsen in the future, as climate change is forecast to generate more intense and prevalent extreme rainfall events (Panagos et al., 2017).

In order to prevent surface flows and their adverse impacts, remedial measures such as retention ponds, riparian buffers, or shrub hedges can be effective to a certain extent, but they take up productive area, require maintenance, can be costly and, most importantly, they do not address the source of the problem, i.e., runoff generation and sediment detachment in the fields (Zhang et al., 2010; Verstraeten & Poesen, 1999; Evrard et al., 2007b). Accordingly, preventive practices should always be preferred. Conservation agriculture has been developed to this end, based on the three following pillars: minimization of tillage, maximization of vegetation cover, and crop diversification (FAO, 2008). These basic principles have been translated into farming practices of various forms, e.g., strip cropping, cover crops, mulching, and conservation tillage (Morgan, 2005; Osman, 2014). There is strong evidence that these techniques provide ecosystem services, especially soil structure improvements and control of overland flow of water and

sediments (Chabert & Sarthou, 2020; Du et al., 2022; Palm et al., 2014). Some of these cultivation techniques have been adopted on farmlands, although context-specific obstacles still hinder their implementation. In the context of intensive agriculture in Western Europe, the following limitations were identified: difficulties with pest and weed management, lower and/or fluctuating crop yields, the need for specific equipment, or the farmer's attitude (Heller et al., 2024; Van den Putte et al., 2010; Lahmar, 2010; Wauters et al., 2010; Casagrande et al., 2016). Furthermore, there is a wide variability across conservation agriculture techniques in terms of effectiveness in mitigating surface flows (Maetens et al., 2012a; Clement et al., 2024). It is therefore essential to pursue the search for conservation practices that best reconcile agricultural production with soil and water protection. To this end, we evaluated through field experiments the potential of two uncommon conservation cultivation practices in maize crop – undersowing and strip-tillage – to limit runoff, soil erosion and associated pesticide surface losses, as compared to conventional tillage practices.

Undersowing is a type of intercropping technique. Intercropping is the association of several (at least two) species cultivated simultaneously on the same field. Given that maize (*Zea mays*) is cultivated in widely-spaced rows, an intercrop can be grown in the inter-row area, possibly providing better soil protection by increased vegetation cover during critical erosion periods (Seran & Brintha, 2010). When an associated crop is not intended to be harvested and grows under the canopy of a main crop, the intercropping can be referred to specifically as "undersowing" or "underseeding" (Ramseier & Crismaru, 2014; Conrad & Fohrer, 2016). From an agronomic point of view, the intercropping technique in general is known to improve water and nutrient use, reduce pest, weed and disease pressure, and stabilize crop yields (Brooker et al., 2015; Bedoussac et al., 2015). However, difficulties with sowing, weeding, and harvesting operations are frequently reported barriers for implementation of this cropping practice, especially in mechanized agriculture (Blessing et al., 2022; Brannan et al., 2023). In Europe, intercropping maize with legumes (soybean, faba bean, etc.) was initially studied as a way of supplying the livestock ration with protein (Anil et al., 1998; Fischer et al., 2020) but the practice did not spread, probably because of aforementioned barriers, limited increase

in protein yield at high N fertilizer input (Li et al., 2023), uncertain silage quality as a result of different maturation stages (non-optimal dry matter contents) between the two crops at harvest (Anil et al., 1998), and limited knowledge by farmers (Ha et al., 2023).

Besides the agronomic advantages, intercropping is also promoted for environmental purposes, such as nitrate leaching mitigation (Manevski et al., 2015), carbon sequestration in soils (Alonso-Ayuso et al., 2020) and mitigation of overland flows of water, sediment and associated potentially harmful molecules (e.g., pesticides). Regarding the latter, different intercropping strategies have been investigated in maize. With fast-growing species such as ryegrass or red clover, delayed sowing of the undersown crop is recommended (between the 3- and 9-leaves stages of maize) in order to limit competition with maize and hence yield losses (Anil et al., 1998; Conrad & Fohrer, 2016; Wall et al., 1991). This delayed undersowing practices has also been referred to as relay-cropping. In Germany, Goeck & Geisler (1988) report considerable reductions in surface runoff under relay intercropping of white clover in maize, particularly when rainfall erosivity in May and June is relatively low. In a study in the Czech Republic, relay intercropping maize with cereals, clovers, grasses or mixtures decreased soil losses by 18 to 80% depending on the species and the rainfall simulation timing (Kincl et al., 2022). In Canada, relay intercropping of Italian ryegrass in maize reduced the mean annual runoff and suspended solid load by 53 and 74%, respectively (van Vliet et al., 2002). In these studies, runoff and erosion were mitigated mainly during late-season (summer, or post maize harvest) rainfall events and therefore had little effect in the early season which is a critical period for muddy floods in the Western European context (see above). To prevent overland flows during the early stages of maize growth, one can consider sowing the intercrop simultaneously with maize, but using slow-growing species to limit competition with maize. In New York state (USA), maize simultaneously undersown with perennial ryegrass, alfalfa, or red clover mitigated suspended solid loads by 77% on average, but the effect was only statistically significant for one out of the two monitored spring seasons (Kleinman et al., 2005). Of these three undersown species, only red clover did not cause a significant loss in maize yield compared to the control sole maize crop. In Germany, Goeck & Geisler (1988) reported 15

to 50% maize yield losses due to white clover simultaneously undersown with maize, while in Denmark, undersowing red fescue was achieved without maize yield loss (Manevski et al., 2015). Overall, there is, however, very little research available about the mitigation effect of (simultaneous) undersowing in maize crops on runoff and erosion – and none on pesticides – under temperate climatic conditions. Specifically, the mitigation potential during spring storm events and penalties on maize yields remain unclear with regard to undersown species.

Strip-tillage, also referred to as "zone-tillage", is another soil conservation practice, suitable for preparing the seedbed of row crops like sunflower, sugar beet, or maize (Morrison, 2002). It consists in tilling only the crop row, by means of a knife, a chisel, or a coulter, leaving the inter-row area untilled. Strip-tillage is a valuable trade-off between the benefits from both conventional and reduced tillage systems. On the one hand, soil working in the row ensures successful crop emergence through crop residue clearing, proper soil-seed contact and soil warming. On the other hand, the soil in the inter-row remains undisturbed, preserving its structure and retaining crop residues on the surface (Morrison, 2002; Laufer & Koch, 2017; Morris et al., 2007). The latter can be beneficial for soil erosion control. Under simulated rainfall in Belgium, Ryken et al. (2018) measured a soil loss of 2.4 t/ha under conventional tillage but only of 0.02 t/ha under strip-tillage in a maize crop. In Poland, erosion monitoring during one maize cropping season by Jaskulska et al. (2020a) showed a cut by 82% in soil loss thanks to strip-tillage compared to a conventional (plough) tillage producing 3.14 t sediments/ha. Besides, Achankeng and Cornelis (2023) reported in a meta-analysis an average 5% increase in crop yields in Europe following strip-tillage as compared to conventional tillage. Strip-tillage in maize crop thus has a great potential, but published studies assessing the mitigation effect of the technique on surface flows are also still scarce.

Considering the above, our study therefore evaluated the potential of 1) undersowing red fescue or white clover and 2) strip tillage, at mitigating runoff, soil and pesticide losses in maize crop based on field experiments. However, water and sediment flows measured from field experiments are expected to be highly specific to trial conditions, given the occurrence or the nature (intensity-duration) of rain events,

soil properties or crop vegetation cover development (Morgan, 2005). Given this limitation of field experiments, modeling has frequently been used for generalizing the outcomes of crop management practices on runoff and erosion across a broader range of situations, within the limits of the calibration dataset (Wang et al., 2008; Mikołaj et al., 2023; Sittig et al., 2022; Laloy and Bielders, 2009b). An additional objective was therefore to calibrate a physically-based model in order to assess the potential of strip tillage at reducing surface flows across a broader range of conditions. For reasons explained in Section 4.2.3, only the control and strip tillage treatments were considered for the modeling.

4.2 Materials and methods

4.2.1 Field trials and measurements

4.2.1.1 Experimental design and treatments

Trials were conducted in farmer fields. The field trials were set up during the maize cropping seasons 2021, 2022, and 2023, in central Belgium (Figure 4.1 and Table 4.1). Maize was grown for silage, which is the dominant use of maize in this region (Taube et al., 2020). In total, field trials were conducted at 5 sites for both undersowing and trip tillage. At 4 sites the erosion monitoring was conducted for a single year only, as maize was part of a rotation involving other crops. In Ottignies, the monitoring could be performed on the same field for two successive years. The trial sites were selected so as to cover a range of textures: sandy-loam, loam and silt-loam. Slopes ranged from 4 to 18%. Details of the field experiments are reported in Table 4.1.

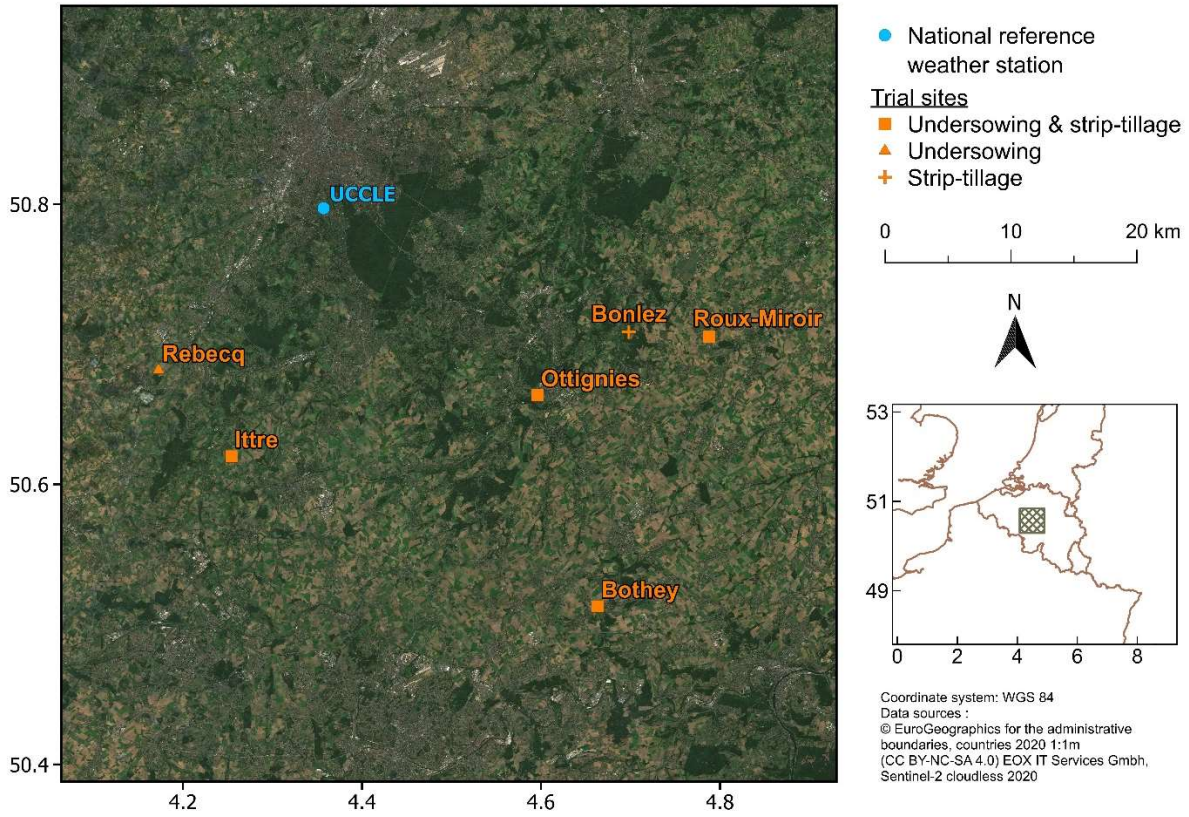


Figure 4.1 : Location of trial sites and national reference weather station used for scenario analyses (see 4.2.3.4)

The four treatments are as follows:

1) **Control**: Sole silage maize crop, sown at spacing of 75 cm between rows and 14 cm within rows. Primary tillage (decompaction or moldboard plough) if needed, followed by secondary tillage for seedbed preparation.

2) **Undersowing red fescue**: Same as control, but with red fescue (*Festuca rubra*) undersown in the maize inter-row at a 10 kg/ha seeding rate, simultaneous to maize sowing. This operation is enabled by an air seeder mounted on a seeding unit, broadcasting red fescue seeds in the inter-row area (see Figure B.1f in the Appendices). Harrows mounted at the rear of the seeding unit slightly cover these seeds with soil for better germination.

3) **Undersowing white clover**: Same as undersowing red fescue, but with white clover (*Trifolium repens*) as undersown crop, at a seeding rate of 5 kg/ha.

4) **Strip-tillage (ST)**: No primary tillage. One pass of strip-tiller (Zebra® strip-tiller from Maschio Gaspardo, tillage depth around 15cm) in the residues of the previous winter cover crop, followed by maize sowing with a classic disc seed drill (no seeding unit).

At each trial site, the treatments were implemented on 30-m long x 6-m wide plots cultivated along the slope, arranged following a randomized complete block design with 3 blocks.

4.2.1.2 Trial management

For soil conservation needs, weed control was performed chemically on all trials, as a mechanical weeding would have destroyed the undersown crops or disturbed the inter-row topsoil in strip-tilled plots. In previous undersowing field trials in Belgium, CIPF (n.d.) compared multiple herbicide schemes (different active compounds and mixtures) and identified sulcotrione as the active ingredient with the lowest impact on the growth of both red fescue and white clover undersowing. For this reason, sulcotrione was initially sprayed at a rate of 300 g/ha at the Rebecq 2021 trial, i.e. slightly below the recommended

dose for maize crop of 450 g/ha. However, soil and climatic conditions favouring high transpiration fluxes in the period following spraying promoted herbicide uptake by plants (weeds and undersown crops), leading to the total destruction of the undersown clover. For this reason, the application rate of sulcotrione was further decreased for the other trial sites (Table 4.1), but consequently a high weed pressure was regularly observed as some weeds remained alive even after herbicide spraying (see Figure B.1g in the Appendices). The addition of the active compound "pyridate" – also selective to the undersown species – to the herbicide mixture did not solve the weed control problem (Table 4.1). At Bonlez (2021) and Ottignies (2022), the control and strip-tillage treatments were additionally specifically sprayed with broader-spectrum active compounds for more effective weed control (Table 4.1). This was possible in Bonlez because no undersowing treatment was implemented at this site, and in Ottignies because a narrow-width sprayer could be used, thereby avoiding destruction of the adjacent undersown plots. At Ottignies, a manual weeding was carried out at the end of June 2021 in order to remove the remaining weeds.

On every trial, treatments were equally fertilized, applying NPK compound or liquid mineral fertilizer in spring, at a rate of around 140 kg N/ha. On the 2023 trials, farmers also spread 20 t/ha manure in winter. Additional information regarding trial management can be found in Table 4.1.

4.2.1.3 Monitoring of overland flows

Within each plot, a 20 m x 3 m sub-plot was bounded with rigid plastic sheets buried into the soil for surface hydrological isolation ("erosion plot"), and equipped downstream with a steel funnel and a 500-L capacity tank collecting runoff and sediment (see Figure B.1a in the Appendices). Consequently, every erosion plot is interspersed with a 3-meter wide "duplicated sub-plot" of the same treatment used for destructive interventions: tractor tramlines for herbicide spraying and destructive measurements (see below).

On-site rainfall was measured continuously by WatchDog 1120 tipping bucket rain gauges with a 10-minute time step. After substantial rain events (>5 mm rainfall) during the maize cropping season, the

runoff volume was determined by measuring the mean water height in the tank (average water height at the four corners) for large runoff volumes or using beakers for small runoff volumes. Water samples were taken after thorough stirring using a closed-circuit pump in the tank. A 1L-sample in a plastic bottle was taken to measure sediment concentration after decanting, removing the supernatant, and oven drying at 45 °C for >3 days. We found that the abovementioned sampling method is much more accurate than when the sample is taken by manually homogenizing the tank content, because of quick sedimentation in the tank. Indeed, based on 4 samples taken in a tank from a control treatment following an event on 8 June at Ott21, the between-sample variability in measured sediment concentrations was approximately 10 times lower following homogenization using the closed-circuit pump (standard deviation = 9 g/L) than by manual homogenization (standard deviation = 98 g/L), while average values were close (134 g/L and 119 g/L, respectively).

For the first two "meaningful" runoff events – i.e., mean runoff in any treatment > 0.5mm – occurring after herbicide spraying, an additional 1L-sample in an amber glass bottle was also taken to determine sulcotrione concentration in runoff water. These samples were stored in a fridge at 4 °C, then analyzed by the "Protection, control products and residues Unit" from the Walloon Agricultural Research Centre (CRA-W) in Gembloux, Belgium. There, samples were centrifuged and filtered at 0.2 µm to remove sediments, while the solution was extracted with acetonitrile and analyzed by liquid chromatography coupled to tandem mass spectrometry (LC-QToF-MS/MS). Not all runoff events of the monitoring period were considered for herbicides analyses for reasons of cost but also because extensive analyses in 2021 showed that sulcotrione concentrations in runoff dropped rapidly, from a few tens of µg/L for the first "meaningful" runoff event following spraying to a few µg/L after two additional events occurring within 2-3 weeks. In addition, only the dissolved sulcotrione was measured for reasons of cost but also because analyses of sediment-adsorbed sulcotrione concentrations from several consecutive runoff events in 2021 as well as scenario simulations using the PRZM model (Young & Fy, 2020) suggested that, on average, > 90% of the total seasonal surface flows of sulcotrione (dissolved + adsorbed)

is transported as a dissolved phase in runoff. The rapid decay of sulcotrione concentration over time as well as its predominance in the dissolved phase is attributable to the low mean Freundlich adsorption coefficient (K_{oc}= 36 mL/g), high water solubility (> 130 mg/L; ECHA, 2011) and fast degradation rate (geometric mean of the normalized soil half-life in field = 4.3 days; European Food Safety Authority, 2008).

Percentage vegetation cover was measured every 2-3 weeks during the maize growth phase based on orthogonal pictures taken from a height of 3 m. The proportion of green pixels was determined using the "Color Threshold" tool from imageJ software version 1.53e. Except for Bonlez and Rebecq in 2021, the total aboveground dry biomass produced by the winter cover crops were measured at the end of the winter, shortly before chemical destruction. Aboveground biomass was harvested in a 1 m² representative area for each plot, then the sample was weighed after drying at 80 °C for 2 days. For Rebecq and Bonlez in 2021, aboveground biomass was estimated based on sowing dates and temperature data using the cover crop growth function from Laloy & Bielders (2009a). Maize dry biomass yields (after drying at 80 °C for 2 days) were also estimated by harvesting the two central maize rows for each plot for a length of 1.5 m at the end of the cropping season.

Infiltration rates were measured on site in the duplicated plots interspersed between erosion plots (see above) using an automated constant-head, single ring infiltrometer (Di Prima, 2015; see Figure B.1b in the Appendices). The Wooding equation (Eq. 4.1) was used to estimate the saturated hydraulic conductivity (Ksat) from the steady-state infiltration rate, considering the parameter of the Gardner conductivity equation $\alpha^* = 12\ \mathrm{m}^{-1}$, which is deemed a suitable value for most structured soils (Elrick et al., 1989; van Es et al., 1999).

$$\mathrm{Ksat} = \frac{\mathrm{Q}*\mathrm{G}}{\frac{\mathrm{r}}{\alpha^*}+\mathrm{r}*\mathrm{H}+\mathrm{G}*\pi*\mathrm{r}^2} \qquad \text{(Eq. 4.1)}$$

Where Q is the measured steady-state infiltration rate [cm³/min], r is the ring inner radius (= 7.5 cm), H is the hydraulic head of water in the ring (= 10 cm), and G is a dimensionless shape parameter ($= 0.316 * \frac{\text{ring insertion depth [cm]}}{\mathrm{r}} + 0.184 = 0.48$).

To best highlight the effects of the conservation practices on soil structure, 4 Ksat measurements were taken in the central maize inter-row for undersowing treatment, while for strip-tillage 2 measurements were taken in the central inter-row and 2 in the row. In control plots, 4 measurements were taken in the central inter-row and 2 in the row to enable comparison with both conservation treatments (undersowing and strip-tillage). Measurements were performed twice, once during the early part of the maize cropping season (May-June) and once at the end (July-September).

Soil physico-chemical analyses (Table 4.1) were performed on one composite sample (0-20 cm) per experimental block, by the "Centre provincial de l'agriculture et de la ruralité" lab from the Brabant-Wallon province, Belgium. Soil texture (fraction < 2 mm) was determined by sieving (fraction > 50 µm) and sedimentation (fraction < 50 µm), and organic carbon content was determined by dry combustion.

All previously described measurements were performed during the maize cropping season. Monitoring of runoff, erosion, and vegetation cover was also performed during the winter period in Ottignies during the winter 2021-2022 (4/11/21 – 11/4/22) and 2022-2023 (6/10/22 – 29/3/23) for control and undersowing treatments. The erosion plots were again set up but were restricted to the width of a single inter-row, i.e. 0.75 m wide x 20 m long. After the maize harvest 2021 (19/10), the control plots were stubble tilled with a vibrating tine cultivator, and a cover crop was sown on the strip-till plots on 25/10/2021, while the undersown plots were left undisturbed. The same operations were applied to control and undersown plots following the maize harvest 2022. Total vegetation cover was monitored over the winter, and biomass of undersown crops (and cover crop) was measured at the end of the winter (end of March), using the same methods as previously described for the cropping season monitoring.

Table 4.1 : Characteristics of the field trials : Year, site name (municipality), GPS coordinates (WGS 84), average slope steepness [%], texture class according to the USDA soil texture triangle and fractions of sand-silt-clay respectively [%], average soil organic carbon content [%], treatments implemented (in triplicates), preceding main crop.

Year	**2021**			**2022**	**2023**	
Site	Rebecq	Bonlez	Ottignies	Ittre	Bothey	Roux-Miroir
Coordinates	50.682 N, 4.171 E	50.709 N, 4.700 E	50.664 N, 4.597 E	50.620 N, 4.253 E	50.513 N, 4.664 E	50.705 N, 4.791 E
Abbreviation	Reb21	Bonl21	Ott21 / Ott22	Itt22	Both23	RM23
Slope [%]	7.4	17.6	16.5	13.9	4.2	10.3
Texture (sand-silt-clay fraction [%])	Silt loam (14 – 70 – 16)	Silt loam (31 – 54 – 15)	Silt loam (8 -72 – 20)	Sandy loam (55 – 36 – 9)	Silt loam (4 – 74 – 22)	Loam (42 – 43 – 15)
OC content [%]	1.0	1.2	1.1	1.0	1.3	1.1
Treatments	Control Undersowing	Control Strip-tillage	Control Undersowing Strip-tillage	Control Undersowing Strip-tillage	Control Undersowing Strip-tillage	Control Undersowing Strip-tillage
Preceding crop	Winter wheat	Winter wheat	Maize	Winter wheat	Winter wheat	Winter barley

Table 4.1 (continued) : Preceding winter cover crop and total aboveground dry biomass produced at the end of the winter [t/ha], destruction mode of the winter cover crop, primary tillage operations performed on control and undersowing plots*.

Trial (site and year)	Rebecq 2021	Bonlez 2021	Ottignies 2021	Ottignies 2022	Ittre 2022	Bothey 2023	Roux-Miroir 2023
Preceding winter cover crop (aboveground dry biomass)	Mustard-clover (~ 1.7 t/ha)	Mustard-phacelia-clover (~ 1.7 t/ha)	No winter cover crop	Control: no winter cover crop. Undersowing fescue and clover: preceding undersown fescue (1.0 t/ha) and clover (0.3 t/ha) respectively. ST: rye-oats-mustard (0.5 t/ha).	Oats-phacelia (1.7 t/ha)	Control: no cover crop sown but widespread regrowth of preceding wheat (1.4 t/ha). Undersowing fescue and clover: fescue (1.3 t/ha) and clover (1.0 t/ha) respectively. ST: mustard-pea-sunflower-phacelia (2.0 t/ha).	Control: oats-pea-clover (mowed on 26/10) then bare soil. Undersowing fescue and clover: fescue (1.5 t/ha) and clover (1.7 t/ha) respectively. ST: oats-pea-clover (3.4 t/ha).
Destruction of the winter cover crop	Winter mouldboard plough	Winter frost	/	29/3 : glyphosate 1440 g/ha	29/3 : glyphosate 1440 g/ha	5/4 : glyphosate 1080 g/ha	31/10 : rototiller on control 5/4 : glyphosate 1080 g/ha
Primary tillage (control & undersowing)*	Winter mouldboard plough	Fall ripping	Spring ripping	Spring ripping	Spring mouldboard plough	Control : spring mouldboard plough. Undersowing : no primary tillage	No primary tillage

* Strip-tillage plots were prepared by 1 pass of strip-tiller followed by maize sowing with a disc seed drill, in all trials.

Table 4.1 (continued) : Secondary tillage operations performed on control and undersowing plots*, maize and undersowing sowing date, erosion plots monitoring period, cumulative rainfall [mm] during the monitoring period, maximum rainfall intensity measured at 10-min. resolution [mm/h], number of "meaningful" (mean runoff in any treatment > 0.5mm) runoff events.

Trial (site and year)	Rebecq 2021	Bonlez 2021	Ottignies 2021	Ottignies 2022	Ittre 2022	Bothey 2023	Roux-Miroir 2023
Secondary tillage (control & undersowing)*	2 x rotary harrow + 1 x seeding unit**	2 x vibrating cultivator 1 x disc seed drill	2 x rotary harrow + 1 x seeding unit**	1 x rotary harrow + 1 x seeding unit	1 x rotary harrow + 1 x seeding unit**	1 x vibrating tine cultivator + 2 x rotary harrow + 1 x seeding unit**	1 x vibrating tine cultivator + 2 x rotary harrow + 1 x seeding unit**
Sowing date	20/4	28/4	28/4	15/4	21/4	20/4	21/4
Monitoring period	23/4 – 20/9	4/5 – 15/9	10/5 – 15/9	22/4 – 25/8	27/4 – 24/8	27/4 - 11/9	26/4 - 8/9
Rainfall [mm]	443	546	538	171	156	315	282
Max. rainfall intensity [mm/h]	73.2	52.9	62.4	56.4	50.4	38.4	47.4
# meaningful runoff events	11	5	8	2	3	0	1

* Strip-tillage plots were prepared by 1 pass of strip-tiller followed by maize sowing with a disc seed drill, in all trials.

** Seeding unit: rotary harrow + roller + disc seed drill

Table 4.1 (continued) : Herbicide sprayed for weed control in cropping season (application date and rate), additional remarks and observations specific to the trial.

Trial (site and year)	Rebecq 2021	Bonlez 2021	Ottignies 2021	Ottignies 2022	Ittre 2022	Bothey 2023	Roux-Miroir 2023
Herbicide for weed control	2/6 : sulcotrione 300 g/ha	7/6 : sulcotrione 225g/ha (+ prosulfuron, flufenacet, terbuthylazine)	8/6 : sulcotrione 150 g/ha	9/6 : sulcotrione 150 g/ha (+ pyridate)	16/6 : sulcotrione 225 g/ha (+ pyridate)	5/6 : sulcotrione 150 g/ha (+ pyridate) 28/6 : sulcotrione 300 g/ha	5/6 : sulcotrione 150 g/ha (+ pyridate) 28/6 : sulcotrione 300 g/ha
Remark	Undersown clover destroyed by herbicide		High weed pressure, manual weed control. Rill erosion observed in strip-tilled maize rows. Winter monitoring of all treatments.	High weed pressure. Specific herbicide application on control & strip-tilled plots (3/6 : tembotrione, isoxadifen-ethyl, flufenacet, terbuthylazine, nicosulfuron). Winter monitoring of control and undersowing treatments.		High weed pressure	High weed pressure

4.2.2 Experimental data analysis

Prior to data analysis, data gaps in surface flows measurements were managed as follows. Some measurements were invalidated, typically when a (sub)surface flow path delivered water into the hole containing the tank which then started to float, disabling the water collection device. Considering all trials, such cases occurred for a total of 7 events out of 30 meaningful events, resulting in 18 missing event-plot data out of 288. In these cases, missing data were replaced by the (mean of the) measurement(s) of the remaining valid plot(s) for the same treatment. In addition, 4 tank overflows occurred during 2 events in 2021. In these cases, the runoff volume was capped at the maximum tank capacity. At Ottignies in 2022, one replicate of the control treatment was fully invalidated for the whole season due to a sowing failure that was not noticed in time, resulting in a much delayed soil cover and an outlier in measured runoff and soil losses.

To ensure a balanced dataset given that not all treatments were tested at all sites (Table 4.1), the statistical analysis was performed separately for each of the three conservation treatments. Cumulative measured surface flows – i.e. total seasonal runoff, soil losses, and sulcotrione losses – as well as maize biomass yield were analyzed by fitting a generalized linear model (GLM) with two factors, considering a "trial" (site-year pair) effect, a "treatment" effect (control vs. either undersowing of clover, or undersowing of fescue or strip tillage), and the "trial x treatment" interaction. To ensure normality of the residuals, the natural logarithms of cumulative flows were used as the response variables. Log-transformed measured Ksat values were analyzed by means of three-factor GLM, considering a "trial", "treatment", and "period" (early or late season) effect as well as all 2-way interactions. A square transformation was applied to the yield data to ensure normality and homoscedasticity of residuals, as the yields were slightly skewed towards low values (see Figure 4.2, fescue treatment at Ott22 or clover treatment at Both23). For each GLM, normality of residuals was visually checked on the Q-Qplot, and homoscedasticity of residuals was visually assessed by plotting residuals against fitted response. For each linear model (i.e. each outcome and each conservation treatment), a relative difference was calculated between the (back-transformed) estimated

marginal mean outcome from the conservation treatment and the one from the control (Eq. 4.2):

$$\text{relative difference [\%]} = \left(\frac{\widehat{Y_{\text{cons}}} - \widehat{Y_{\text{cont}}}}{\widehat{Y_{\text{cont}}}}\right) * 100 \quad \text{(Eq. 4.2)}$$

where $\widehat{Y_{\text{cons}}}$ and $\widehat{Y_{\text{cont}}}$ are the (back-transformed) estimated marginal mean outcome from the GLM for the conservation practice and the control treatment, respectively. Note that because of the logarithmic transformation of the response variable and the subsequent back-transformation of estimated marginal means from the linear model, the relative differences in cumulative flows or in Ksat values calculated by Eq. 4.2 correspond to relative differences between geometric (and not arithmetic) means (Feng et al., 2013).

Significant differences between estimated marginal mean outcomes from each conservation treatment and the control were assessed by marginal contrasts. When the *p*-value of the marginal contrast on a treatment was statistically significant (*p*-value < 0.05), we also tested for the "treatment x trial" interaction contrasts. Statistical analyses were performed with RStudio 2022.12.0 using the "emmeans" package (Lenth et al., 2024). Within each outcome (runoff, erosion, sulcotrione, Ksat, and yield), the "raw" *p*-values derived from all tested contrasts were ultimately adjusted using the Benjamini-Hochberg method (Benjamini & Hochberg, 1995; using a 5% false discovery rate), in order to assess effect significance of conservation treatments on the outcome while correcting for the multiple comparisons.

4.2.3 Modeling and scenario analyses

Given the lack of effect of the undersowing technique on surface flows and the difficulties encountered with weed control (see 3.1), only the control and strip-tillage treatments were considered for modeling.

4.2.3.1 Model description

The results from erosion plot trials remain strongly dependent on the particular experimental conditions (weather, including the occurrence and nature of rain events, soil properties and topography) of each trial. With a view to quantify the effectiveness of strip-tillage at

mitigating runoff and erosion for a broader range of conditions, a hydrological model was calibrated based on the field data. Given the complex, interacting processes driving surface flows and the additional requirement of representing the effect of different management techniques – including soil conservation practices – on the simulated processes, physically-based models seem better suited for this purpose.

Here, we used the CREHDYS model (Laloy & Bielders, 2009a). This is a spatially-distributed, continuous, seasonal time-scale runoff and sediment transport model applicable to plot-scale data. Furthermore, it explicitly accounts for intra-seasonal changes in soil properties, such as decreases in hydraulic conductivity and roughness due to surface crusting (Cerdan et al., 2002), as well as spatial heterogeneity from zone tillage. It has been successfully used to model the effect of winter cover crops on runoff and erosion in maize cropping systems under similar conditions to the present study (Laloy & Bielders, 2009b). The model includes a simple, temperature-driven crop growth module. Rainfall interception, depressional storage, evapotranspiration and percolation are calculated through empirical equations. Infiltration is computed using the Green-Ampt method modified for unsteady rainfall (Chu, 1978). The erosion module simulates splash detachment as a function of rainfall kinetic energy, flow detachment according to flow shear stress, and transport and deposition determined by flow transport capacity. The hydraulic module solves the continuity equation and Manning's equation using a kinematic wave approach. Crop growth and soil water balance (evapotranspiration and percolation) are calculated at a daily time step, while overland flow processes (interception, infiltration, soil detachment and deposition, overland flow) are calculated at a smaller time step, 1 minute in the present study. The cell size was set at 25 cm. For the soil water balance, CREHDYS works as a simple bucket model considering a homogeneous topsoil layer, assumed to be 20-cm thick. Consistent with our trial sites, we modelled plots with a uniform slope. More details regarding the hydrological and sediment modelling components of CREHDYS can be found in Laloy & Bielders (2008 and 2009a).

4.2.3.2 Model calibration and parameterization

Except for Ottignies, the trial sites changed each year due to crop rotation. Since (almost) every trial was conducted on a unique site, each site requires specific parameter values for soil physical properties. Considering the limited number of meaningful runoff events in some trials due to rainfall conditions unfavorable for runoff and erosion during the early part of the maize growing season, the number of site-specific parameters that could reliably be calibrated is low given the need to avoid equifinality. For the same reason, modeling was restricted to water and sediment. Indeed, modelling of pesticide losses would have required additional parameters which could probably not have been properly calibrated and parameterized given the available dataset and limited literature background in this area (Joyce et al., 2008; Lefrancq, 2014).

Based on a sensitivity analysis, the final saturated hydraulic conductivity ($Ksat_{\text{final}}$) and the saturated hydraulic conductivity decay coefficient (C_K) parameters were selected for optimization for the runoff outcome. The C_K coefficient reflects soil structural stability, controlling how fast Ksat decreases along the season as a function of cumulative rainfall kinetic energy. For the erosion outcome, Manning's *n* coefficient was selected for optimization as it strongly controls detachment by overland flow. Note that the median particle size *d50* is also a very sensitive parameter with regard to erosion output, but it can be estimated from soil texture as opposed to Manning's *n*. Besides their high sensitivity, we also chose to calibrate these three parameters because it makes sense physically to assume that strip-tillage impacts hydraulic conductivity (increased macroporosity from preceding crop roots and biological activity; Wahl et al., 2004) and hydraulic roughness (increased from crop residues; Panachuki et al., 2015). All other inputs and parameters were measured or estimated from pedotransfer functions or from the literature, as reported in Table 4.2.

Calibration was performed separately for each trial. As a first step, the final *Ksat* of the control treatment ($Ksat_{\text{final}_{\text{cal,control},i}}$) and the Ksat decay coefficient ($C_{K_{\text{cal,control},i}}$) were calibrated for each trial i by inverse

modelling minimizing the Nash-Sutcliffe model efficiency coefficient (NSE, Eq. 4.3; Jain & Sudheer, 2008) for the runoff outcome from the control treatment ($\mathrm{NSE}_{\mathrm{runoff,control},i}$) and considering Manning's *n* roughness coefficient = $0.03\ \mathrm{s.m^{-1/3}}$ in a first step (Laloy & Bielders, 2008). Secondly, we calibrated Manning's *n* for the control treatment ($\mathrm{Manning's}\ n_{\mathrm{cal,control},i}$) by minimizing the NSE for the erosion outcome ($\mathrm{NSE}_{\mathrm{erosion,control},i}$). For strip-tillage modelling (and again separately for each trial), we attempted to calibrate the parameters specifically for the (untilled) inter-row area, assuming the parameters values of the 40-cm wide tilled row to be the same as the control. This assumption was made to avoid equifinality when seeking to calibrate 6 parameter values on a small number of events for each trial (three for the tilled row and three for the untilled interrow).

For each calibration step (control runoff, control erosion, ST runoff, ST erosion), the parameter(s) space was first explored by a global screening ($Ksat_{\mathrm{final}}$ from 0.1 to 10 mm/h, Ksat decay coefficient from 10^{-5} to $10^{-1}\ \mathrm{m^2/J}$, Manning's *n* from 0.01 to 0.1 $\mathrm{s\,m^{-1/3}}$), then the Nelder-Mead algorithm was used to locally converge to the optimal parameter value, using the python package scipy.optimize (Virtanen et al., 2020).

$$\mathrm{NSE} = 1 - \frac{\sum_{j=1}^{n}(\widehat{y_j} - y_j)^2}{(y_j - \overline{y_j})^2} \qquad \text{(Eq. 4.3)}$$

where NSE is the Nash-Sutcliffe model efficiency coefficient, $\widehat{y_j}$ is the simulated event flow (runoff or erosion) at event j, y_j is the measured event flow at event j, $\overline{y_j}$ is the mean measured event flow across all events j. Note that because of the periodic flow collection methodology (see 4.2.1.3), j may occasionally correspond to a period with several runoff events. Strictly defined, y_j and $\widehat{y_j}$ therefore correspond to the cumulative flows (measured or simulated, respectively) produced in the period in-between two visits ($j-1$ and j) at a given site. Besides the trial-specific NSE values, an overall NSE (for all trials) was also calculated as a global indicator of model fit across all sites.

CREHDYS did not simulate any flow detachment for the Both23 trial as the simulated flow shear stress did not reach the critical value triggering flow detachment. Hence, no Manning's *n* could be calibrated

for this trial. Furthermore, the simulations for the Ott21 trial were restricted to the period prior to June 28 when a manual weeding was performed on the plots which significantly altered soil surface conditions (see Section 4.2.1). In addition, a particularly extreme rainfall event – with a return period of up to 200 years in some places – occurred in Belgium in mid-July 2021 (Journée et al., 2023). On 15/7 at Bonl21, 80 mm of rain fell in 24 hours, following 41 mm fallen in the previous two days. Within a reasonable range of *Ksat* values, the simulated runoff for this event exceeded by far the measured runoff. We deemed such a high-volume and long-lasting event to fall outside the limits of the Green-Ampt infiltration model as implemented in CREHYDS. The latter indeed neglects percolation to the subsoil occurring during the rain event, simulating a topsoil layer that quickly reaches (and remains at) saturation under such extreme conditions, hence likely underestimating infiltration. For this reason, we excluded this event from the model calibration. This issue did not arise for the Reb21 trial. Indeed, rainfall over the previous two days was much lower (4 mm), so the simulated soil water content at the start of the extreme rain event of 15/7 was well below saturation. Lastly, failures with on-site rain gauges occurred on some trials, requiring filling of data gaps with hourly rainfall records from nearby weather stations (CRA-W, 2024). Such a case covers the first 42, 10, and 8 days of the monitoring period at Reb21, Bonl21, and Ott21, respectively. Fortunately, no “meaningful” runoff event was recorded during those periods (see Table 4.1).

Table 4.2 Parameterization of the CREHDYS model for calibration and (cross-)validation. Inputs and parameters, process(es) directly involved, value setting (measured, or estimated from the literature), and values. Ranges of values indicate that parameterization is trial-specific.

Variable	Process(es)	Setting	Value(s)
INPUTS			
Daily solar radiation and average temperature	Crop growth, evapotranspiration	Measured from weather stations (CRA-W, 2024)	/
Event rainfall intensities (10 min. time step)	Interception, infiltration	Measured by rain gauges on site	/
Soil textural fractions (% sand-silt-clay)	Evapotranspiration	Measured	Sand: $4-55\%$ Silt: $36-74\%$ Clay: $9-22\%$
Slope	Hydraulic, surface retention	DTM	$4-18\,\%$
θ_i initial volumetric soil moisture content $[\frac{\mathrm{m^3 water}}{\mathrm{m^3 soil}}]$	Infiltration, percolation, evapotranspiration	Estimated	0.20
PARAMETERS			
r growth rate of maize cover	Crop growth	Adjusted and validated by (Laloy & Bielders, 2009a)	0.005
Extinction coefficient for maize LAI=f(cover)	Evapotranspiration	Literature (Andrieu et al., 1997)	0.34

Table 4.2 (continued) : Parameterization of the CREHDYS model for calibration and (cross-)validation. Parameters, process(es) directly involved, value setting (measured, estimated from the literature or from a pedotransfer function (PTF), or optimized by inverse modelling on measured runoff or soil losses), and values. Ranges of values indicate that parameterization is trial-specific.

Variable	Process(es)	Setting	Value(s)
PARAMETERS			
RR random roughness	Surface retention	Literature (Laloy & Bielders, 2008; van Dijk et al., 1996b)	$9\ \mathrm{mm}$
$Ksat$ subsoil	Percolation	Literature (Laloy & Bielders, 2008)	$20\ \mathrm{mm/h}$
$Ksat_{\mathrm{initial}}$ (surface soil layer)	Infiltration	Measured and literature (Laloy & Bielders, 2008)	$72\ \mathrm{mm/h}$
C_K Ksat decay coefficient	Infiltration	Optimized on runoff	$> 10^{-2}\ \mathrm{m^2/J}$
$Ksat_{\mathrm{final}}$ (surface soil layer)	Infiltration	Optimized on runoff	$\mathrm{Control:}\ 0.9 - 3.9\ \mathrm{mm/h}$ $\mathrm{ST:}\ 1.1 - 4.2\ \mathrm{mm/h}$
θ_{sat} volumetric soil moisture content at saturation $[\frac{\mathrm{m^3 water}}{\mathrm{m^3 soil}}]$	Infiltration, percolation	PTF* (Weynants et al., 2009) assuming bulk densities between 1.35 (silt loam) and 1.45 (sandy loam) g/cm³	$0.41 - 0.44$
θ_{FC} volumetric soil moisture content at field capacity	Percolation		$0.25 - 0.33$
θ_{WP} volumetric soil moisture content at wilting point	Percolation		$0.11 - 0.17$
Ψ soil matric potential at wetting front	Infiltration	PTF (Rawls & Brakensiek, 1989)	$147 - 625\ \mathrm{mm}$
Manning's n roughness coefficient	Hydraulic	Optimized on soil losses	$\mathrm{Control:}\ 0.016 - 0.082\ \mathrm{s.m^{-\frac{1}{3}}}$ $\mathrm{ST:}\ 0.016 - 0.100\ \mathrm{s.m^{-\frac{1}{3}}}$
As Soil aggregate stability (# of drops from the test of Low (1954)	Splash detachment	Literature (van Dijk et al., 1996b; Liu et al., 2003)	10
$d50$ median particle diameter	Transport capacity	Estimated from texture (Abeshu et al., 2022)	$20 - 57\ \mu\mathrm{m}$
Coh Soil cohesion	Flow detachment	Literature (Liu et al., 2003)	$10\ \mathrm{kPa}$

* Pedotransfer function

4.2.3.3 Model validation

Given the limited runoff and sediment loss data available for any given site and year (see 4.2.3.2), we undertook a leave-one-out cross-validation of the model (Wan Jaafar et al., 2011; Arsenault & Brisette, 2014). As is well established in the literature (Jarvis et al., 2002; Weynants et al., 2009), we found a positive relationship between the optimized *Ksat* values and sand fraction of the trial sites. For each treatment (control or strip-tillage) and for each cross-validation subset i (all n trials except trial i), we therefore computed the linear regression between the calibrated *Ksat* and the sand content for the $n - i$ trials. The $Ksat_{\text{cross-val,treatment},i}$ of the trial i was then predicted using this regression equation. For Manning's *n*, as it was not predictable from static soil properties, we simply calculated the $\text{Manning's } n_{\text{cross-val,treatment},i}$ of each trial i as the average of the calibrated Manning's *n* values from the other ($n - i$) trials. Each trial was simulated with their cross-validation parameters values, and overall NSE values were again calculated for runoff and erosion as indicators of model fit.

4.2.3.4 Scenario analysis

The scenario analysis aims to estimate the mitigation effect of strip-tillage on runoff and soil losses in the loess belt of central Belgium. Our scenario analysis consists of a comprehensive set of simulations encompassing all levels of different factors as outlined in Table 4.3, including the tillage treatment through specific values for the *Ksat* and Manning's *n* parameters. We simulated 30 cropping seasons from the 15th of April to the 1st of October, using weather data (daily solar radiation and temperature, 10 min. rainfall intensity) recorded at Uccle's national reference weather station between 1991 and 2020 (Figure 4.1; Royal Meteorological Institute, 2024). Two levels were assessed for slope steepness (5% and 10 %) and slope length (50 m and 100 m). In Wallonia in 2021, 32% of the cultivated area under silage maize was located on slopes exceeding 5%, and 6% of the maize cropped area had slopes steeper than 10% (SPW, 2022b; Farr et al., 2007). The median size of a silage maize field is 1.7 ha (SPW, 2022b), equivalent to 130 m in length assuming square plots. However, as the model has been calibrated on much smaller slope lengths (20 m), and since maize fields are unlikely to

feature such high slope gradient (5 or 10%) continuously over a long distance, slope lengths of 50 m and 100 m were chosen for the scenario analysis. Regarding soil texture, three textural classes were considered in order to cover the spectrum of textures found in the topsoil of soils from the Belgian loess belt (D'Or & Ephesia Consult, 2018; Table 4.3). Finally, inter-site variability in calibrated parameters was accounted for by sampling multiple values within their range found during the calibration process. For each level of soil texture and for each treatment, 20 values of *Ksat* were thus randomly sampled according to the normal distribution within the confidence interval resulting from the regression $Ksat_{\mathrm{cal,treatment}} = \mathrm{f(sand)}$ (see Figure 4.4). For Manning's *n*, 5 values were considered for each treatment, evenly distributed between the minimum and maximum value of the $\mathrm{Manning}^{'}\mathrm{s}\, n_{\mathrm{cal,treatment}}$. All other parameters values were set according to the scenario, i.e. $\theta_{\mathrm{sat}}, \theta_{FC}, \theta_{WP}, \psi, d50$ were estimated from pedotransfer functions (PTF from Table 4.2; estimated values in Table 4.3), and remaining parameters were set at the same value as during calibration.

Table 4.3 : Cross-simulated factors for the scenario analysis, and (number of) levels

Factor	Levels
Year (weather and rainfall sequence)	30 cropping seasons (15/4 – 1/10), from 1991 to 2020
Slope steepness	5% and 10%
Slope length	50 m and 100 m
Soil texture of the topsoil (0-20 cm)	Silt loam 1: 10-75-15 % sand-silt-clay ($\psi = 523$ mm ; $\theta_{\text{sat}} = 0.435$; $d50 = 22$ µm) Silt loam 2: 27.5-60-12.5 % sand-silt-clay ($\psi = 338$ mm ; $\theta_{\text{sat}} = 0.427$; $d50 = 30$ µm) Loam: 45-45-10 % sand-silt-clay ($\psi = 199$ mm ; $\theta_{\text{sat}} = 0.420$; $d50 = 43$ µm)
Treatment	Control or strip-tillage
Ksat	20 values sampled according to the normal distribution around the confidence interval resulting from the regression $Ksat_{\text{cal,treatment}} = \text{f(sand)}$. The sampling is therefore specific to each treatment and soil texture.
Manning's *n*	5 values evenly distributed between the minimum and maximum value of the Manning's $n_{\text{cal,treatment}}$, i.e.: Control: $0.016 - 0.082$ s. m$^{-1/3}$ ST: $0.016 - 0.100$ s. m$^{-1/3}$
TOTAL	30 x 2 x 2 x 3 x 2 x 20 x 5 = 72 000 simulations

4.3 Results

4.3.1 Maize yield, biomass of undersown crops and total vegetation cover

Measured maize dry biomass yield strongly depends on the trial (Fig. 4.2; *p*-value $< 10^{-3}$), but the GLM revealed no significant trial by treatment interaction. The three conservation practices significantly reduced mean maize yields by 8 to 15% depending on the treatment compared to the control (Table 4.4).

Overall, the total vegetation cover showed similar temporal dynamics in all treatments over the cropping season (see Figure B.2a in the Appendices), although cover in strip-tilled plots tended to slightly lag behind the other treatments in the first part of the season. Undersown clover exhibits a marginally higher total cover than the control on average, the difference between both treatments peaking around 60-75 days after maize sowing (84% mean cover in clover vs. 70% in control). This difference was not observed with undersown fescue. Consistent with this observation, undersown clover developed a higher biomass than fescue during the cropping season (see Figure B.1c and d in the Appendices).

During the winter-period monitoring at Ottignies in 2021-2022 and 2022-2023, the mean cover in undersown clover plots dropped from 37% after maize harvest to 19% at the end of winter (see Figure B.2b), while undersown fescue maintained around 65% cover during the whole winter. End-of-winter biomass averaged across the 2 winter seasons reached 0.4 t/ha for clover and 1.6 t/ha for fescue.

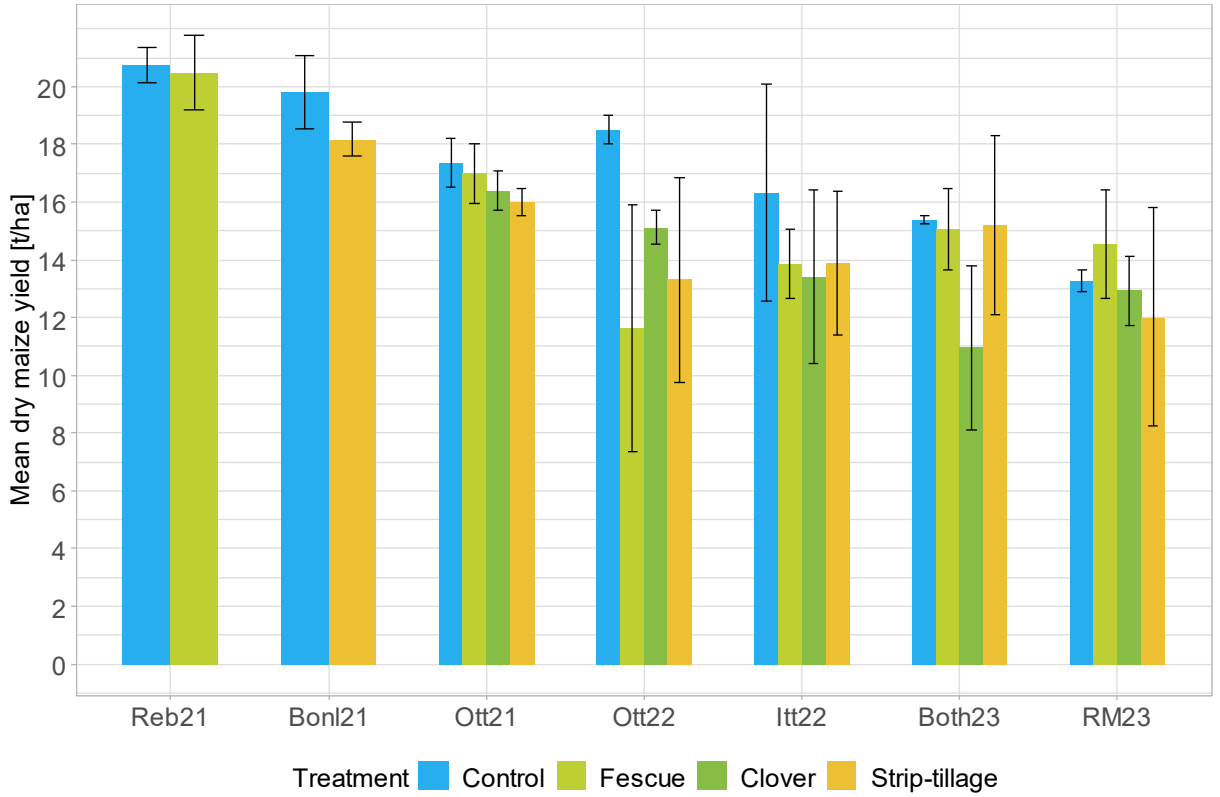


Figure 4.2 : Mean above-ground dry biomass maize yield for each treatment in each trial. Error bars are standard deviations. Not all treatments were implemented at each site-year. See Table 4.1 for trial abbreviations.

4.3.2 Runoff, sediment and pesticide losses

Figure 4.3 shows measured cumulative surface flows for each trial and treatment. The 2021 cropping season was particularly prone to runoff and erosion. On the contrary, almost no runoff and erosion was observed in 2023, while in 2022 the situation was intermediate. At Itt22, the three "meaningful" (>0.5 mm mean runoff) runoff events occurred before herbicide spraying, while at Both23 no meaningful event occurred at all, explaining the absence of measured sulcotrione losses on these two trials (Figure 4.3c). Unsurprisingly, the "trial" effect was highly significant (*p*-values < 10^{-3}), indicating that the magnitude of measured cumulative flows strongly depends on the trial.

Undersowing maize with red fescue or white clover did not result in any statistically significant mitigation effect on runoff, soil and sulcotrione losses (Table 4.4). Note that the flows measured at Reb21, which were considerably lower in fescue undersowing than in the control (see Figure 4.3), are likely due to weed hotspots and not to fescue; fortuitously, two of the three fescue plots were densely covered with weeds, while only one control plot was strongly affected.

Compared with conventional tillage (control), strip-tillage significantly reduced runoff and soil erosion (Table 4.4). Contrasts were applied to investigate ST vs. control differences by trial (i.e. "treatment x trial" interaction). Based on these interaction contrasts, the difference between ST and control was only statistically significant for the Itt22 trial, both for runoff (corrected *p*-value = $2 * 10^{-7}$) and erosion (corrected *p*-value = $4 * 10^{-8}$).

The surface flows measured during the two winter seasons at Ottignies were rather low for all treatments, especially regarding soil losses. On average across the winter seasons 2021-2022 (total rainfall of 332 mm) and 2022-2023 (405 mm rainfall), we measured only 1.4 mm, 2.2 mm and 4.6 mm of cumulative winter runoff in the control, undersown fescue and undersown clover treatments, respectively (not shown). Mean soil losses amounted to only 0.06, 0.05, and 0.11 t/ha, respectively, for these treatments.

Table 4.4 : Relative difference in (back-transformed) marginal mean estimated from linear models between each conservation treatment and the control (Eq. 4.2). Response variables are maize dry yields, seasonal cumulative flows (runoff, soil losses and sulcotrione losses), and saturated hydraulic conductivities Ksat. Adjusted *p*-values of marginal contrasts (see Section 4.2.2) assess the significance of the differences between (log- or square-transformed) estimated marginal mean from each conservation and the control treatment, bolded when *p*-value < 0.05.

		Treatment		
Response variable	Effect measure	Undersowing fescue	Undersowing clover	Strip-tillage
Maize yield	Relative difference [%] vs. control	**- 8%**	**- 15%**	**- 11%**
	Adjusted *p*-value	**0.017**	**0.0036**	**0.011**
Runoff	Relative difference in geometric* mean [%] vs. control	+ 3%	+ 21%	**- 31%**
	Adjusted *p*-value	0.75	0.31	**0.041**
Erosion	Relative difference in geometric* mean [%] vs. control	- 19%	- 11%	**- 60%**
	Adjusted *p*-value	0.47	1	**0.0009**
Sulcotrione	Relative difference in geometric* mean [%] vs. control	- 2%	- 12%	- 3%
	Adjusted *p*-value	1	1	1
Ksat	Relative difference in geometric* mean [%] vs. control	- 7%	- 8%	+ 6%
	Adjusted *p*-value	1	1	1

* Geometric mean, because of the log-transformation of the response variable and the subsequent back-transformation (see 4.2.2).

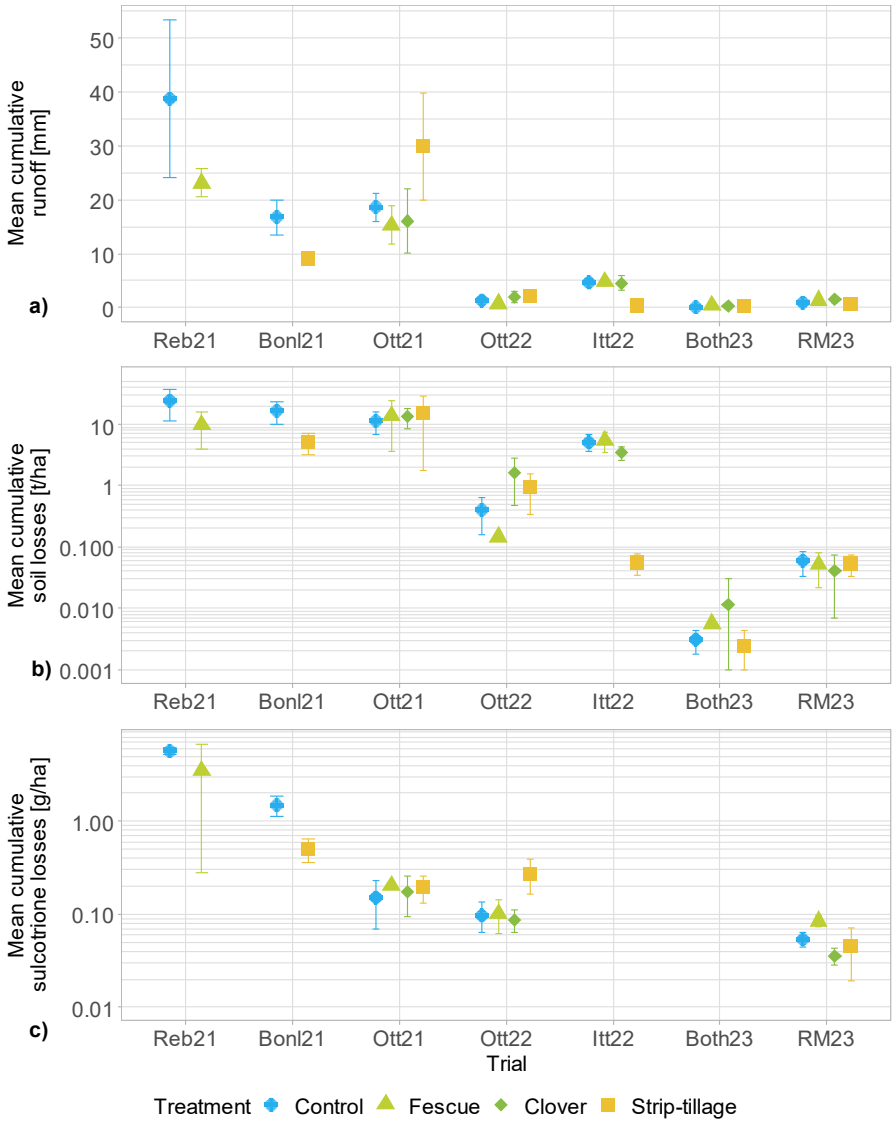


Figure 4.3 : Mean cumulative seasonal runoff (a), soil losses (b), and sulcotrione surface losses (c) for each treatment on each trial. Error bars are standard deviations. See Table 4.1 for trial abbreviations. Not all treatments were implemented at each site-year. No sulcotrione losses were measured at Itt22 and Both23.

4.3.3 Saturated hydraulic conductivity measurements

For measured Ksat, the full GLM revealed no significant treatment by period interaction. Consequently, this interaction was not included in the final model. Table 4.4 provides the results of GLM and contrasts on measured Ksat. Unsurprisingly, the "trial" effects were highly significant (*p*-values < 10^{-8}), indicating that the measured Ksat strongly depends on the trial. There were also significant differences between early and late season measurements (*p*-value < 10^{-2}) with a higher Ksat at the beginning of the season (back-transformed estimated mean in control = 38 mm/h) than at the end (20 mm/h). However, the treatment effect was not significant for any of the three conservation practices (see Table 4.4). Surprisingly, the back-transformed estimated marginal mean Ksat from the linear model is almost exactly the same for the control (26.3 mm/h) and strip-tillage (27.9 mm/h) treatments. Furthermore, the range of values for measured Ksat was highly inconsistent with that of calibrated Ksat (see Section 4.3.4), the former being higher by about one order of magnitude (Figure 4.4a). We therefore strongly question the ability of our Ksat measurements to reflect the actual soil surface state, and for this reason we do not further elaborate on these data (see Section 4.4.1).

4.3.4 Modeling and scenario analysis

Given the lack of significant impact of undersowing on runoff and erosion (Table 4.4) and the difficulties with weed control in these treatments (see Section 4.2.1.2), modeling of undersowing treatments was not attempted. This section therefore exclusively deals with the control and strip tillage treatments.

While optimizing the final saturated hydraulic conductivity $Ksat_{\text{final}}$ and the Ksat decay coefficient based on the runoff measurements of the control plots, it was found that the optimal Ksat decay coefficient was very high (> 10^{-2} m^2/J) for the large majority of the trials. This corresponds to situations in which surface crusting rapidly occurs following a few small rainfall events, such that Ksat reaches its final value very early in the season. In other words, our field-measured

runoff data are best modelled by simple situations where the soil Ksat is constant (and low) over the entire season. Consequently, we will hereafter refer to it as *Ksat* rather than $Ksat_{\text{final}}$. For the control treatment, we first calibrated for each trial i a constant $Ksat_{\text{cal,control},i}$. Subsequently, Manning's $n_{\text{cal,control},i}$ was calibrated using the erosion data. Response functions showing the model's quality of fit (based on the NSE) according to the values of calibrated parameters are provided in Figure B.3 in the Appendices. Figure 4.4 shows the calibrated values of *Ksat* and Manning's *n* for the control treatment. A linear relationship was found between $Ksat$ and sand content. Fortunately, no correlation was found between the two calibrated parameters (not shown). The CREDHYS model yielded good fits (Moriasi et al., 2007) for both runoff and erosion on the control treatment following calibration, with $\text{NSE}_{\text{cal,control}} > 0.65$ (Figure 4.5a and b). When the values of the parameters (*Ksat* and Manning's *n*) were estimated for each trial by cross-validation from the other trials, the model still produced fits that can be considered as "satisfactory" for event-scale surface flows ($\text{NSE}_{\text{cross-val,control}}$ between 0.29 and 0.54; Figure 4.5c-f; Yilmaz & Onoz, 2020) notwithstanding the high sensitivity to these two parameters.

For the strip-tillage treatment, calibrating *Ksat* and Manning's *n* specifically for the (untilled) inter-row area while considering the properties (*Ksat* and Manning's *n*) of the (tilled) row to be the same as the control did not yield satisfactory results. Indeed, trials where strip-tillage strongly mitigated surface flows (e.g., Itt22, Fig. 4.3) were poorly modelled using this strategy because too much flow was simulated from the row alone. This indicates that $Ksat$ of the rows in the strip-tillage treatment cannot be equated to the $Ksat$ of the control treatment. Because attempting to calibrate separate $Ksat$ and Manning's *n* values for the row and interrow for each trial would have led to equifinality issues, an effective $Ksat$ value was calibrated, representing a 'spatial average' across the entire surface of the strip-tillage plots. Overall model fits (NSE values) were slightly better when considering a uniform surface for the strip-tilled plots. Therefore, for each trial i, a single "effective" $Ksat_{\text{cal,ST},i}$ was calibrated based on runoff measurements, and a single effective Manning's $n_{\text{cal,ST},i}$ was calibrated using the erosion data. Note that this representation implies that the calibrated parameters are specifically valid for the proportion of the surface that is

tilled/undisturbed by the strip-tiller used in our study (approx. 55%/45%) and may not be applied to model other strip-tillage widths. Figure 4.4 shows the calibrated values of *Ksat* and Manning's *n* parameters for the strip-tillage treatment, as well as the linear regression $Ksat = \mathrm{f}(\mathrm{sand})$. For Bonl21 and Ott22 trials, the $\mathrm{Manning's}\ n_{\mathrm{cal,ST}}$ reached the upper limit of the range initially assumed, so it was constrained to this maximum value, i.e. $0.1\ \mathrm{s.m}^{-\frac{1}{3}}$. As for the control, the simulated event runoff matched well the observed event runoff under strip-tillage following calibration (Figure 4.5a; $\mathrm{NSE} = 0.56$), and the fit was still acceptable following cross-validation (Figure 4.5c and e; $\mathrm{NSE} \geq 0.36$) (Yilmaz & Onoz, 2020). While the modelling of soil erosion was nearly satisfactory using calibrated parameters values (Figure 4.5b; $\mathrm{NSE}_{\mathrm{cal,ST}} = 0.29$), it was unsatisfactory following the cross-validation (Figure 4.5d and f; $\mathrm{NSE} = 0.07\ \mathrm{and}\ 0.11$). Note that even in the latter case, the NSE coefficients take positive values, indicating that the model still better predicts event soil loss than the measured average event soil loss. Unsurprisingly, when the cross-validation is performed on parameters individually ($Ksat$ for runoff and n for soil loss; Figure 4.5c and d), slightly better fits are obtained overall compared to when cross-validated values of both *Ksat* and Manning's *n* are considered together (Figure 4.5e and f).

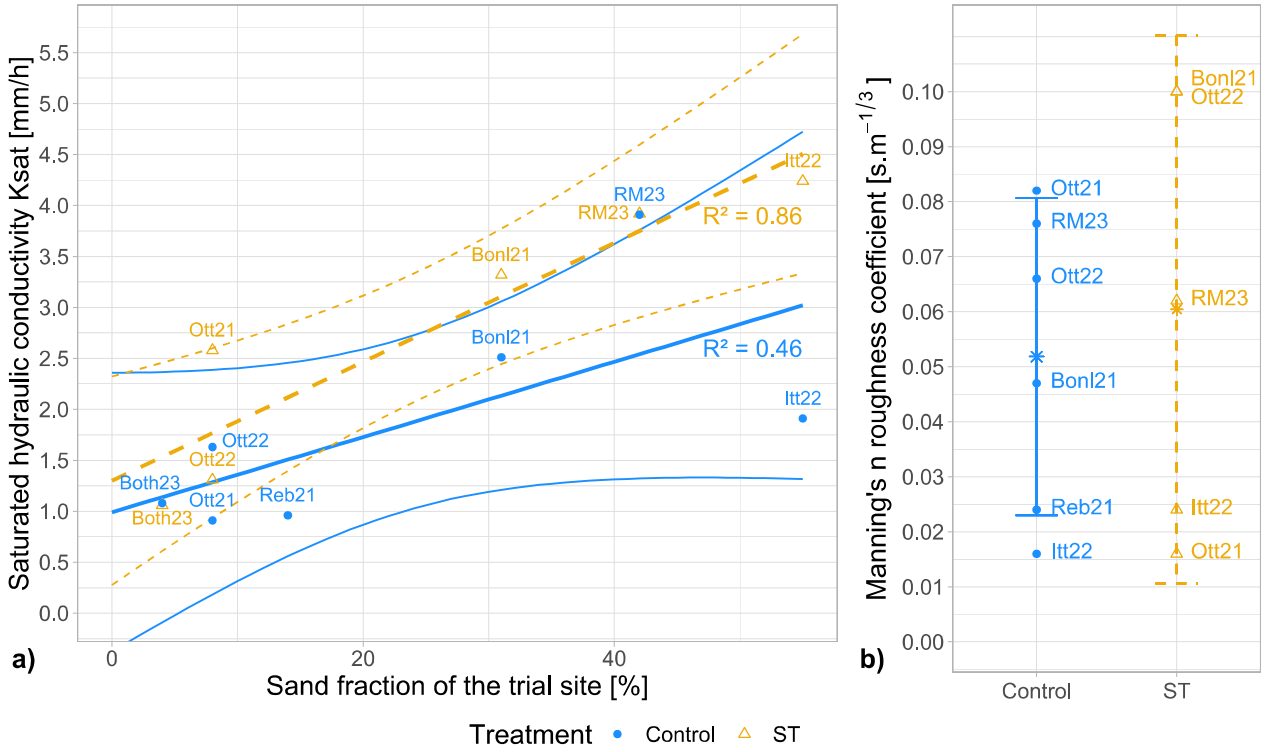


Figure 4.4 : Optimized parameters values following calibration of the CREHDYS model, per trial and treatment (control or strip-tillage ST). a) for *Ksat*, with linear regression against trial sand content, and corresponding confidence intervals. b) for Manning's *n*, with the mean value and confidence intervals.

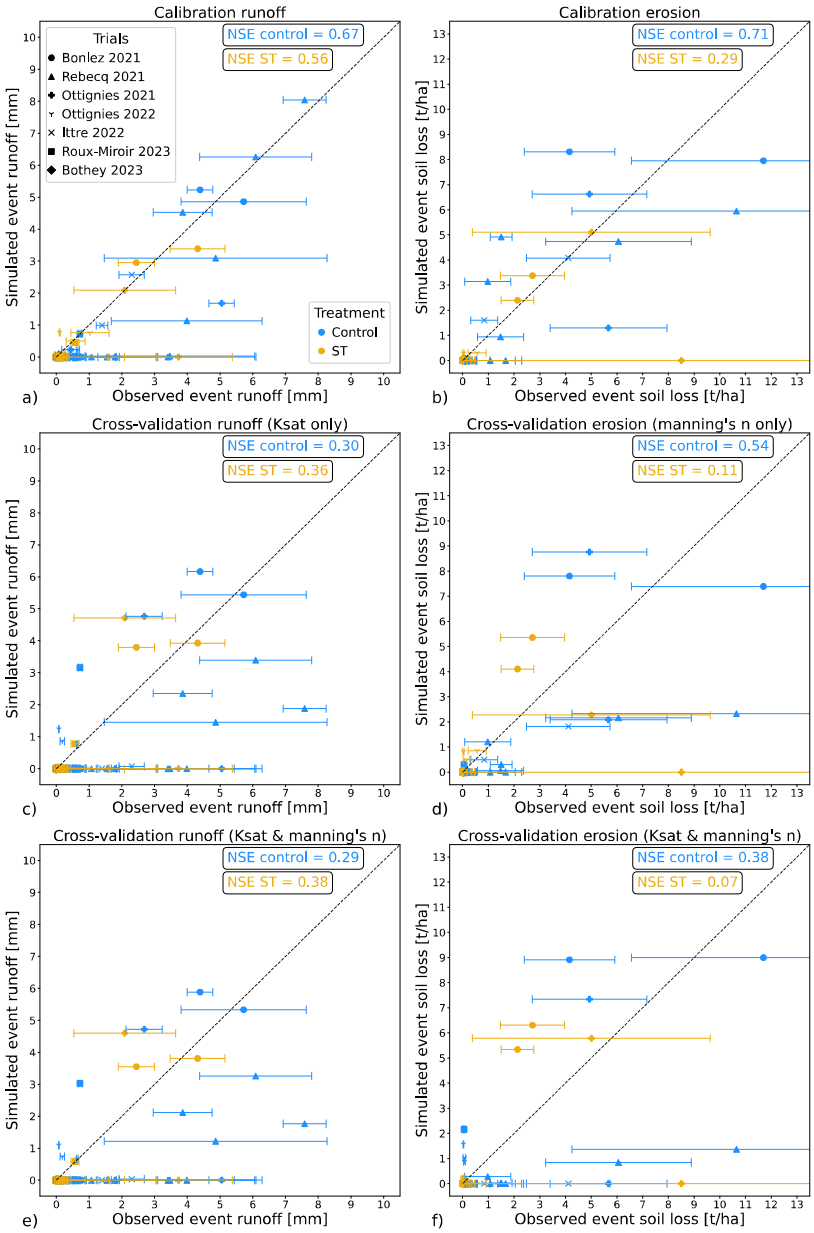


Figure 4.5 : Simulated vs. measured event flows (runoff and soil losses), per trial and treatment (control or strip-tillage ST). Overall (all trials together) Nash-Sutcliffe efficiency coefficients NSE are reported as indicators of model fit. a) Runoff fit, with calibrated *Ksat* (and nominal Manning's *n*=$0.03\ \mathrm{s.m^{-1/3}}$, see 2.3.2). b) Soil loss fit, with calibrated *Ksat* and Manning's *n*. c) Runoff fit, with cross-validated *Ksat* (see 4.2.3.3) (and nominal Manning's *n*=$0.03\ \mathrm{s.m^{-1/3}}$). d) Soil loss fit, with cross-validated Manning's *n* (and calibrated *Ksat*). e) Runoff fit, with cross-validated *Ksat* and Manning's *n*. f) Soil loss fit, with cross-validated *Ksat* and Manning's *n*.

At Reb21 or Bonl21, the occurrence of numerous major runoff events enables a look at the seasonal dynamics of the fits between simulated and observed flows (not shown). There is no discernible seasonal trend in the quality of runoff predictions. However, considering only events for which the runoff is correctly modelled, soil losses are systematically underestimated at the start of the season, and overestimated at the end, by around a factor of 2. This suggests a flaw in the erosion modelling process (see 4.4.2).

The outcomes from the scenario analysis are shown in Figure 4.6 as cumulative probabilities of flows for all combined factors, and Figure B.4 in the Appendices as boxplots of flow distributions by scenario factor. Soil texture and slope length actually had little influence on total seasonal or maximum daily runoff and erosion (Figure B.4i-p). In descending order, inter-annual variability in climatic (rain) conditions (Figure B.4a-d), variability in *Ksat* values, and treatment had a much greater impact on all surface flows. As expected, variability in Manning's *n* values and slope steepness had additional influence on soil losses, but very little on runoff (see Figure B.4e-h).

The cumulative seasonal runoff across all strip-tillage scenarios averages 3.7 mm (i.e., over all factors except treatment), which is 43% lower than the mean seasonal runoff across all control simulations (6.5 mm). In most cases, the total seasonal erosion of a given simulation stems, to a large extent, from a single event, meaning that the maximum daily soil loss provides little additional information compared with the total seasonal soil loss. From a soil conservation point of view, the (mean) total seasonal soil loss is the most relevant flow outcome. Under conventional tillage (control treatment), it amounts to 6.2 t/ha on average across all other factors, while it drops to 3.4 t/ha in the strip-tillage treatment. 19.2% of control treatment simulations yield a seasonal erosion rate above the threshold of 10 t/ha for a whole year (Morgan, 2005; Bielders et al., 2013). Under strip-tillage, there is only a 9.2% chance of exceeding this threshold. Note, however, that this 10 t/ha/year threshold is rather high, as a critical tolerable value of 2 t/ha/year has been recommended for Europe more recently, based on soil formation rates (Panagos et al., 2020). It is moreover relevant to consider this

conservative threshold, knowing that only water erosion is modelled in this study, whereas other erosion processes may also occur, e.g. tillage, harvest, or wind erosion. In addition, the simulated time frame does not cover a full year but only the cropping season, although this period is clearly associated with the highest erosion risk (Vandaele & Poesen, 1995). The probability of exceeding this threshold of 2 t/ha/year is 48.5% and 31.1% respectively under control and strip-tillage. Obviously, the situation is even more critical when focusing specifically on the more vulnerable 10% slope steepness scenario (Figure 4.6b). From the point of view of mitigating the risk of extreme (muddy) flood events, maximum daily runoff is probably the most relevant flow outcome to look at. We are preferring it to maximum daily soil loss since, for our case, runoff is better modelled than erosion (Figure 4.5). It is more difficult to define a threshold value of maximum daily runoff above which serious flood risk occur, since it depends on event duration and terrain topography like contributing area upstream and flow concentration downstream (Evrard et al., 2007a). In the control treatment, half of the simulations remains below 3.2 mm of maximum daily runoff, while in strip-tillage 50% of simulated max. daily runoff values are < 1.1 mm (Figure 4.6a). One year out of 5 (i.e. for 6 out of the 30 simulated years), the average (across other factors than years) maximum daily runoff from a maize field exceeds 7 mm under conventional tillage (control treatment), while this same runoff amount is only reached one year out of 10 (3/30 years) when applying strip-tillage (see Figure B.4c).

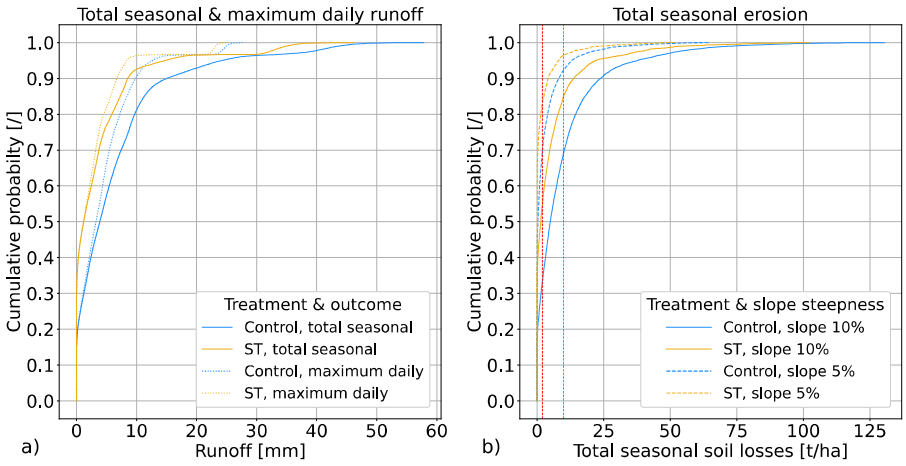


Figure 4.6 : Cumulative probability distribution for all simulations of the scenario analysis for a) total seasonal and maximum daily runoff by treatment, and b) total seasonal erosion by treatment and by level of slope steepness. The two vertical red dashed lines in (b) refer to the critical thresholds for "sustainable" erosion rate of 2 t/ha/year (Panagos et al., 2020) and 10 t/ha/year (Morgan, 2005).

4.4 Discussion

4.4.1 Field experiments

Overall, there is a slight, almost systematic reduction in maize yield under the conservation treatments as compared to the control (Figure 4.2). At Ott22, however, the gap between control and conservation maize yields was larger than average. This discrepancy is unlikely to be related to climatic conditions, as similar rainfall levels were experienced at Itt22 (see Table 4.1). Nor can it be explained by increased weed pressure, as both control and strip-tillage plots received broad-spectrum weed control at Ott22. Consequently, the presence of a winter cover crop on the conservation (undersowing or strip-tillage) plots, as opposed to the control, is the most likely cause of this discrepancy. Specifically, the bacterial decomposition of winter cover crops residues in spring may have led to nitrogen deficiency in the conservation treatments plots in the early cropping season (Crandall et al., 2005).

Ksat measurements were highly inconsistent with the collected runoff from the experimental plots. The range of *Ksat* values derived from model calibration on runoff measurements (1 to 4 mm/h

depending on the trial; Figure 4.4) is radically different from the spectrum of the measured Ksat values (back-transformed estimated mean per trial between 12 and 44 mm/h). In fact, when attempting to simulate the trial plots using the CREHDYS model parameterized with the measured Ksat values, no runoff was modeled for any of the trials. In the course of our infiltrometer measurements, we sometimes observed cracks in the surface crust following ring insertion into the soil, especially under dry conditions, or gaps at the interface between the ring and the soil, despite the use of grease on the ring's inner surface to limit preferential flows. All this suggests that our Ksat from infiltrometer measurements do not accurately reflect the actual hydraulic conductivity of the whole topsoil. This issue was actually already underlined by Vandervaere et al. (1997) or Alagna et al. (2019) who proposed alternative protocols or methods for Ksat measurements. Measurements from runoff plots are definitely much more representative and reliable from the standpoint of studying surface flows, considering the large plot size and given that this method causes very little disturbance to the soil; only plot edges are affected by the insertion of plastic sheets into the soil. Nevertheless, the Ksat values calibrated on plot runoff do not represent "true" values for soil saturated hydraulic conductivity either, as they depend on model structure and assumptions.

4.4.1.1 Undersowing

In contrast with the few previous studies from the literature reporting successful control of runoff and erosion by means of crop associations in maize, our undersowing trials showed no mitigation effect on surface flows, in addition to serious difficulties with weed control and reduced maize yields. We attribute the weed control issue to the low spectrum of action of the active ingredient(s) used, sulcotrione (and pyridate), which is nevertheless required to control weeds selectively from the undersown crops (i.e. without killing both fescue and clover), as well as to the highly variable response of weeds (and undersown crops) to the herbicides depending on moisture conditions (at constant application rate). As these herbicides do not kill grasses, *Echinochloa crus-galli* was a commonly spotted weed (see Figure B.1g in the Appendices), though *Chenopodium* was also

frequently observed. In previous undersowing field trials in Belgium, CIPF (n.d.) compared multiple herbicide schemes including pendimethalin, bromoxynil butyrate, dicamba, nicosulfuron, sulcotrione, pyridate, mesotrione, tembotrione, and mixtures. Only the last four active compounds resulted in a loss of red fescue and white clover biomass < 20% compared to the same undersown crops grown without any herbicide spraying. Research should therefore be pursued so as to develop undersowing techniques that are operational from an agronomic point of view.

The main explanation for the lack of mitigation effect on surface flows by undersowing is the slow growth of fescue and clover, resulting in similar total vegetation cover dynamics on control and undersown plots (see 4.3.1 and Figure B.2a). Indeed, red fescue and white clover did not substantially cover the soil until around mid-June, when maize plants also already provide extensive cover over the inter-row area. Although proper weed control was a significant issue in many of the trials, this does not seem to be the main reason for observed lack of effect. Indeed, even on the Itt22 trial where weeds were well controlled and where the three “meaningful” runoff events of the season occurred between May 20th and June 8th (i.e. 29 to 49 days after sowing, during the peak of vegetation cover in clover undersowing; see Figure B.2a and Section 4.3.1), neither runoff nor erosion was significantly mitigated under undersowing compared to the control treatment (Figure 4.3). Contrary to the present study, most of the few other studies from the literature conducted under similar agro-climatic conditions report significant reductions in runoff and erosion achieved by undersowing practices, whether sown simultaneously or with some delay (relay intercrop) relative to maize sowing. For instance, maize simultaneously undersown with ryegrass, alfalfa, or red clover mitigated suspended solid loads by 77% following Kleinman et al. (2005) in New York state (USA), although the fall season was included and the mitigation effect was only statistically significant for one out of the two monitored spring seasons. In the Czech Republic, Kincl et al. (2022) sowed relay intercrops in maize at the 3-4 leaf stage. From rainfall simulations performed 30 days later, they measured decreases in soil losses from 21% (*Phacelia tanacetifolia*) to 62% (*Trifolium incarnatum*) as compared to sole maize. Given the low number of published studies, it cannot be excluded that

many other studies have been left unpublished because of inconclusive results.

The maize yield loss of around 12% due to undersowing (Table 4.4) is more in line with the literature. The study from Kleinman et al. (2005) found no difference in maize yield when undersown with red clover as compared to sole maize, but observed a 20% lower maize yield with undersown ryegrass, and even 37% lower with alfalfa. Goeck & Geisler (1988) reported maize yield losses of 15 to 50% due to white clover undersown simultaneously with maize in Germany. By contrast, studies from Wall et al. (1991), Conrad & Fohrer (2016) or Alonso-Ayuso (2020) report equivalent maize yields with grass or legume intercrops. The more pronounced maize yield losses for clover than for fescue undersowing (Table 4.4) is consistent with the higher vegetation cover and biomass of undersown clover than fescue during the maize cropping season (Figure B.2a in the Appendices), assuming it reflect resources availability for maize as affected by competition with undersown crops.

In contrast, during the winter season (Figure B.2b), the fescue maintained its biomass and vegetation cover, while clover declined. The latter resulted from frost damage as well as consumption by overwintering Canada geese. Furthermore, the maize harvest operations in 2021 and 2022 damaged both undersown crops due to machine traffic under wet conditions, leaving prominent wheel tracks and resulting in heterogeneous vegetation cover (see Figure B.1h and i). Considering that the monitoring of vegetation cover and biomass was carried out in the inter-rows that were as free as possible of wheel tracks, and assessing that around half of the total area in the undersown plots was an almost bare soil resulting from wheel tracks (see Figure B.1h and i), the overall biomasses for the whole area would still not exceed 0.8 t/ha and 0.2 t/ha for clover and fescue, respectively, averaged across the end (March) of the 2022 and 2023 winters. Comparatively, on two sites monitored over three winter seasons in Belgium, Laloy & Bielders (2010) measured an average end-of-winter aboveground biomass of 1.2 t/ha developed by “standard” rye or ryegrass winter cover crops sown after maize harvest.

The monitoring of surface flows during the winter periods were also carried out in the inter-rows that were as free as possible of wheel tracks, thus representing a "best case" situation for the undersown crops. In spite of this, the runoff measured during these winter seasons, although small on all treatments, were lowest on the control plots. Indeed, even though the soil was bare (Figure B.2b), the stubble tillage carried out on the control treatment created a high surface roughness that greatly limited overland flow (Clement et al., 2024). Consequently, very low and similar winter soil losses values were obtained across all treatments.

Note that in our trials, the control treatment involved the same herbicide application as the undersown treatment and, therefore, experienced similar weed pressure. However, the strictly “conventional” practice carried out by a farmer would entail broader-spectrum weed control, resulting in soil that is (virtually) free of weeds. On the one hand, such conditions in the control plots would have been more favorable for maize growth due to minimal competition. On the other hand, weakly covered soil would have been more susceptible to runoff and erosion. The respective implications are: 1) the yield loss provided in this chapter for the undersowing technique (-12%; Table 4.4) underestimates the actual gap caused by the technique as a whole, including increased weed pressure, and 2) a greater flow mitigation effect of the undersowing technique might have been observed due to the soil protection provided by weeds.

In view of the two main issues encountered with the undersowing technique in the present study, i.e. poor weed control and the lack of a flow mitigation effect, the undersowing practice as tested cannot be recommended for erosion control. In the end, from a surface flows mitigation perspective, it may little matter whether the preceding winter cover crop was undersown (sown simultaneously with the preceding maize), relay cropped (deferred sowing), or a “standard” winter cover crop (sowing post maize harvest). When aiming to mitigate soil erosion during the critical period that immediately follows sowing of a spring crop, the key is probably to minimize soil disturbance by tillage and to retain residues on the surface (Clement et al., 2024). Beyond the strict definition of undersowing, sowing the main crop into an already

established living mulch presents promising opportunities for soil conservation, as it ensures substantial soil cover from spring onwards. The challenge, however, lies in minimizing competition by "controlling" the growth of the living mulch without completely eradicating it, using chemical and/or mechanical methods (e.g., rolling, strip-tilling, mowing, shredding/cutting) (Hartwig & Ammon, 2002; Siller et al., 2016). Nevertheless, research could still continue into evaluating undersowing strategies for targets other than surface flow mitigation: mitigation of nitrate leaching after maize harvest (Manevski et al., 2015), carbon storage (Alonso-Ayuso et al., 2020), insect biodiversity (Hüber et al., 2022), etc.

4.4.1.2 Strip-tillage

We demonstrated that strip-tillage in maize crop significantly mitigates runoff and soil erosion, with a relative difference in geometric mean compared with conventional tillage amounting to -31% and -61%, respectively (Table 4.4). These outcomes compare well to the meta-analysis by Clement et al. (2024), stating that a conservation tillage practice including a single tillage operation before sowing (i.e. strip-tillage (like) system), implemented for the first time, in a spring crop (e.g. maize), in a silt loam soil, reduces runoff by an average of 37% and erosion by 58% compared with conventional tillage (ploughing and secondary soil preparation). Yet, other specific studies on strip-tillage in maize crop have reported even greater control on runoff and erosion compared to our findings: from 2.4 t/ha (conventional tillage) to 0.02 t/ha (strip-tillage) in Belgium (Ryken et al., 2018), or from 3.14 t/ha to 0.57 t/ha in Poland (Jaskulska et al., 2020a). One explanation for these differences could lie in the level of soil protection against crusting and detachment afforded through cover by residues from the preceding crop. Ryken et al. (2018) indeed measured a 14 % soil cover by crop residues from preceding grain maize crop in strip-tillage but only of 2.3% in conventional tillage. In the experiment of Jaskulska et al. (2020a) testing different preceding crops (winter and spring cereals, soybean, sunflower, rapeseed, maize), crop residues covered as much as 80% of the area in strip-tillage (between 4 to 13 t/ha of crop residues on the surface), but only 4% in conventional tillage (between 0.2 to 0.8 t/ha of crop residues on the surface). In our trials, such levels of cover by

residues were never achieved. At Itt22, residue cover estimated from our orthogonal pictures was the highest, approximately 10% in strip-tillage plots but almost 0% in control plots. In contrast, at Bonl21, although the biomass of the winter cover crop developed well in autumn, residue cover in spring (after tillage and sowing) was interestingly similar in control and strip-tillage plots, around 1-2% only. This may to a large extent have resulted from the early destruction of the cover crop by frost and subsequent decay. Nevertheless, even under such conditions, strip-tillage significantly reduced runoff and soil erosion (Figure 4.3).

The highest mitigation effects of strip-tillage were observed for trials where preceding winter cover crops had developed the greatest biomass, i.e. Itt22 and Bonl21 (as opposed to Ott21 and Ott22, and excluding trials with very low measured surface flows; see Figure 4.3 and Table 4.1). In the case of Itt22, this may result in part from the higher residue cover as explained above, but other mechanisms must be invoked for Bonl21 where residue cover was very low in all treatments. This suggests that the mitigation effect of strip-tillage is not only attributable to soil protection by residues but also to improvements in soil structure, even at the timescale of a few months. The enhanced structure is likely related to the macroporosity formed by the preceding (cover) crop's root system and by biological activity (earthworms most notably), which is not or less disturbed by strip-tillage as opposed to conventional tillage (Heisler et al., 1998; Fraser et al., 2003). An important implication of all this would be the varying flow mitigation potential of strip-tillage depending on the crop rotation. In NW Europe at least, the efficiency of strip-tillage would be optimal when implemented after a winter crop (e.g., a winter cereal), provided that the latter is succeeded by an early-sown (in summer, hence rapidly developping) winter cover crop. The technique would be less beneficial after a spring crop, which requires later harvesting and, consequently, later sowing (in autumn) of the cover crop. On a compacted, bare soil (e.g. following a sugar beet crop), light surface tillage might even be preferable to strip-tillage. Nevertheless, in our study, the effect of the preceding cover crop biomass on flow mitigation by strip-tillage cannot be statistically disentangled from the "soil texture" effect, especially given that a coarser texture is also known to increase the efficiency of conservation tillage practices (Clement et al., 2024). Indeed, the two sites where the

mitigation effect was most pronounced (Bonl21 and Itt22) also feature a higher sand content, compared to Ottignies (2021 and 2022) where strip-tillage increased surface flows.

During the field trials at Ott21, a specific rill erosion process in the strip-tilled maize rows was observed (Figure B.1l in the Appendices), while not observed in the control plots, resulting in increased runoff and soil losses for this treatment (Figure 4.3). This phenomenon could be due to the particular combination of repeated erosive events, erodible silt loam soil, lack of preceding winter cover crop (i.e. no soil-protecting residues), and a slightly elevated inter-row relative to the row, thereby concentrating runoff flow towards the latter. Although the strip-tiller used in this study is equipped with two discs surrounding the tine (Figure B.1j) so as to limit the transfer of soil from the row to the inter-rows, a slightly elevated inter-row relative to the row was frequently (but not always; Ott22 and Itt22) observed.

Despite the rather large reduction in runoff and soil loss, we measured a relative difference in geometric mean of only 5% in sulcotrione losses between strip-tillage and control treatment. This discrepancy seems mainly related to the lack of pesticide flow data for the Itt22 trial where the mitigation effect of strip-tillage was outstanding on runoff and erosion, but where the two meaningful runoff events took place before herbicide spraying, preventing the high flow mitigation potential of this trial to be reflected in sulcotrione loss mitigation.

With regard to maize yields, we found a statistically significant average decrease of 11% under strip-tillage compared to the control treatment, although from an economic standpoint this slight yield loss may be compensated for by reduced tillage costs. In their meta-analysis covering Europe, Achankeng & Cornelis (2023) also reported an average crop yield loss following strip-tillage on silt loam soils, although the decrease (3%) was less than in the present study. Besides the fact that their review includes crops other than maize, the discrepancy may also arise from the strip-tillage operation in our experiments being performed very close in time to maize sowing – either on the same day or the day before. As has been shown for conservation (no or reduced) tillage practices in general, in humid climates such as Belgium, soil

temperature is a main factor limiting crop development, especially in the early season. Soil with minimal or no tillage tends to remain wet and cold for a longer time, delaying crop germination (Van den Putte et al., 2010; Achankeng & Cornelis, 2023). In a temperate European climate, performing strip-tillage 1-2 weeks before sowing may enhance soil warming in the row, thereby minimizing crop yield losses following strip-tillage (Luna & Staben, 2003; Adee et al., 2016). This approach may have been adopted in most primary studies collected by Achankeng & Cornelis (2023).

4.4.2 Modeling and scenario analysis

Based on the field experiments, the two most sensitive parameters from the CREHDYS model were calibrated for each treatment (control and strip-tillage) and each trial: *Ksat* and Manning's *n*. According to this methodology and for the sake of parsimony, we therefore assumed that other parameters do not depend on treatment, including RR, θ_{sat}, As, or Coh (see Table 4.2). For random roughness RR, this appears fairly justified based on in situ observations of soil surface and the literature (Paz-Ferreiro et al., 2008; Panachuki et al., 2015). For the other three parameters, the assumption is more tenuous, as (the absence of) tillage is known to affect total porosity, and aggregate stability and cohesion (Tangyuan et al., 2009; Casamitjana et al., 2009). Nevertheless, these parameters are (very) little sensitive compared to *Ksat* or Manning's *n* (Laloy & Bielders, 2009a; De Roo & Offermans, 1995). The CREHDYS model is also highly sensitive to the soil matric potential ψ at the wetting front and to the median particle diameter $d50$ (Laloy & Bielders, 2009a). However, since they are both supposed to be mainly related to soil granulometry, it is little relevant to adjust their value according to cropping practices, at least when studying short-term impacts. These key parameters thus had to be estimated from measured textural fractions but may fail to capture the specific soil structural state of each trial, which may partly explain the limited quality of model fits (see below).

Furthermore, Laloy & Bielders (2009b) already demonstrated that, as is frequently the case for other (complex) models and due to structural inadequacies (Efstratiadis & Koutsoyiannis, 2010), there exists

a space of Pareto-optimal solutions, i.e. multiple sets of parameters which meet a trade-off between the two objectives of the CREHDYS model: accurately simulating runoff on the one hand, and soil loss on the other. In other words, the *Ksat* that optimizes runoff prediction is probably not exactly the same as the one that optimizes erosion prediction, and the opposite is equally true for Manning's *n*. Given the methodology adopted here, it is not guaranteed that the calibrated parameter set fall within this Pareto-optimal solution. Since *Ksat* is the most sensitive parameter, our calibration favors runoff prediction over soil loss. In addition, neither the errors on observed nor simulated flows are explicitly incorporated in the calibration process. As discussed by Beven & Freer (2001), the concept of equifinality implies that there could be several sets of parameters – or even several different models – that "acceptably" reproduce the observed system behavior given these errors. In the present case, this issue is, however, of limited impact given the small number of calibrated parameters and their high sensitivity (see Figure B.3 in the Appendices).

Notwithstanding these methodological concerns, the calibrated values of saturated hydraulic conductivity *Ksat* (0.9 - 4.2 mm/h) are consistent with other studies on silt loam soils with low organic carbon content, highly sensitive to crusting (Cerdan et al., 2002; Laloy & Bielders, 2009b; Elhakeem et al., 2018). The calibrated Manning's *n* values are also compatible with the literature overall (Kwaad et al., 1998; Laloy & Bielders, 2009b), although covering a fairly wide range of values (Figure 4.4b) and leaning towards the higher end values. Interestingly, when focusing on the control treatment, trials with lower calibrated Manning's *n* ($< 0.05\ \mathrm{s.m^{-1/3}}$ for Itt22, Reb21, and Bonl21) are those on which ploughing was performed on the control treatment and/or where weeds were well controlled (see Table 4.1), thus are conducive to a hydraulically smoother soil surface, as opposed to the other trials (Manning's $n > 0.05\ \mathrm{s.m^{-1/3}}$ for Ott21, Ott22, and RM23).

The application of the CREHDYS model to simulate the flows observed on our trials yielded good or satisfactory fits for runoff and erosion prediction on control treatment, and for runoff on strip-tillage, both for the calibration and cross-validation steps (see 4.3.4 and Figure 4.5). However, we acknowledge that the simulation of erosion under

strip-tillage is hardly satisfactory following calibration, and unsatisfactory according to cross-validation (see 4.3.4 and Figure 4.5b, d, f). Specifically for the erosion outcome, even though three other major erosion events appear to be well predicted by the model (Figure 4.5b), the highest erosion event under strip-tillage (from Ott21 trial) is not predicted at all. Indeed, no runoff is simulated and hence no erosion either, in spite of the fact that rill erosion was observed in the strip-tilled maize rows (see Figure B.1l in the Appendices), resulting in substantial soil losses. In terms of the Pareto-optimal solutions space, this particular case is an example of a parameters set that strongly favors the runoff prediction objective at the expense of the erosion one. Furthermore, on two trials (Bonl21 and Ott22), the upper limit for Manning's *n* set for calibration was reached for the strip-tillage treatment (max Manning's $n = 0.1\ \mathrm{s.m^{-1/3}}$), suggesting that the change in (*Ksat* and) hydraulic roughness is not sufficient by itself to explain the erosion mitigation under strip-tillage. For the reasons outlined above, calibration of additional parameters was not considered appropriate. Overall, whatever the treatment and the step (calibration or cross-validation) considered, model performances lie below those previously obtained with the same model (Laloy & Bielders, 2009b; $\mathrm{NSE_{runoff,validation}}$ from 0.64 to 0.77) or with other process-based models (e.g. Saia et al. (2013) achieved $\mathrm{NSE_{runoff}}$ = 0.79 and 0.84, with WEPP model). However, such studies generally collect flow data for several years, on a limited number of sites (typically one or two) and sometimes with a sub-event scale resolution, which allows for a more robust calibration and enables to calibrate a larger number of parameters. When looking at a comparable application of parsimonious calibration of a process-based model (WEPP) using event plot data from diverse sites and conditions, as carried out by Zhang et al. (1996), similar fit performances are found ($R^2_{\mathrm{runoff}} = 0.77$ and $R^2_{\mathrm{erosion}} = 0.36$), even with a much more extensive dataset (556 plot-years). Our trials also feature a wide range of conditions (soil type, (cover) crop history, tillage, weeds) as well as limited measured data (including surface flows from only one cropping season for most trials), thereby requiring a parsimonious calibration procedure.

We also highlighted a more general issue with the ability of CREHDYS model to simulate the observed seasonal pattern in soil erosion. Indeed, early-season events produced (much) higher erosion

rates than late-season events in our trials (not shown), which is the well-known consequence of increased splash erosion at the start of the cropping season, when vegetation cover is low thus raindrop energy available for soil detachment is high (Cerdan & Le Bissonnais, 2002; Morgan, 2005). Looking at the model outputs following calibration, even when the event runoff was accurately simulated, soil losses were systematically underestimated in the early season events, and overestimated at the end (see Section 4.3.4). As a matter of fact, similar rain events modeled at the beginning (low vegetation cover) or the end of the season (high vegetation cover) resulted in similar simulated soil losses. We noticed that, throughout the season, splash erosion contributed very little (a few percent) to total simulated erosion compared to flow detachment. This has also been reported by other authors using the LISEM model (Takken et al., 1999; Leduc, 2017) from which the erosion processes implemented in CREHDYS stem. As discussed by Takken et al. (1999), because models such as LISEM and CREDHYS do not explicitly model concentrated (rill) flow (de Roo et al., 1996), a fairly thick layer of water will be predicted as interrill flow for high intensity rainstorms, which in turn leads to very low predicted splash detachment rates. These authors argued that simulated splash detachment rates are about 4 to 30 times lower than measured ones. In the case of the original LISEM, this flaw may have been frequently missed and/or corrected for by calibration when applied in other studies, as it is an event-based model and not a time-continuous model. The issue may, however, have been rectified in recent versions of openLISEM, by introducing a differentiated calculation of splash detachment in ponded and dry area (Jetten, 2024).

The results of the scenario analysis show limited effects of soil texture on surface flows, with changes in average soil loss (across all other factors) between the different simulated soil types (Table 4.3) staying below 10% (see Figure B.4m-p in the Appendices). This would mean that, in our simulated scenario's, the higher $Ksat$ on the loamy soil compared to the silt loam soil are offset by lower ψ and θ_{sat} (see Table 4.3), resulting in similar infiltration rates between different textural classes, hence similar runoff.

Focusing on the treatment effect, the average seasonal erosion rate simulated in the conventional tillage scenario (6.2 t/ha) is almost double that with strip-tillage (3.4 t/ha). The flood risk arising from maize fields would also be cut in half following strip-tillage management: in the control treatment, 6 years out of 30 (i.e. 1 year out of 5, on average), the maximum daily runoff (averaged across the other factors) exceeds 7 mm, whereas under ST this same runoff amount is exceeded for only 3/30 years (i.e. 1 year out of 10). The erosion mitigation effect is comparable in magnitude with those provided by other conservation farming practices such as cover crops or deep tillage (Maetens et al., 2012a), and appear even more effective than "intensive reduced tillage systems" (no ploughing but repeated use of powered shallow tillage operations; Clement et al., 2024).

Though already satisfactory, the overall mitigation effects of strip-tillage against runoff and erosion are probably underestimated in this study since improvements in soil structure could be expected by the technique when applying strip-tillage for several years (Han et al., 2024; Jaskulska et al., 2020b). This should also apply to strip-tillage implemented as part of a crop rotation, provided that other (winter) crops are cultivated under conservation tillage as well (Packer et al., 1992; Clement et al., 2024). In most of our trials, the conservation technique was indeed only applied as a one-shot intervention (i.e. for a single year), in a previously conventional system. The slow processes enhancing soil structure following conservation management – e.g. concentration of organic matter in the topsoil, increased biological activity, etc. – are not captured in the field experiments used for model calibration, therefore not depicted in the modelling and scenario analysis either.

4.5 Conclusion

We evaluated through field experiments the effectiveness of undersowing (fescue or clover) and strip-tillage, at mitigating runoff, soil erosion, and herbicide surface losses in a maize crop. The undersowing technique showed serious failures regarding weed control, no mitigation effect on surface flows, and reduced maize yields. Even in the case of adequate weed management, undersowing as implemented in this study appears to be of little interest for controlling erosion and

muddy floods, since it does not substantially improve the total vegetation cover during the critical stages of the maize growing season. Moreover, even when another spring crop follows the maize in the rotation, the undersown crops do not seem to offer any advantage over standard winter cover crops (sown after maize harvest), since the former suffered from poor development during the winter.

As opposed to undersowing, strip-tillage reduced all measured surface flows on average, statistically significantly for runoff and soil losses. A 30-year scenario analysis for the Belgian context using the CREDHYS model revealed that strip-tillage would reduce both the average erosion rate at the plot scale as well as the risk of storm-induced muddy floods by around a factor of 2. Despite its suitability for different row crops and evidence of effectiveness both in terms of crop yields and soil conservation, strip-tillage specifically is still less adopted than other conservation tillage practices in Europe (Heller et al., 2024). A future avenue could be to explore the underlying causes for this gap, and to promote endorsement of the practice by farmers accordingly.

CHAPTER 5: GENERAL DISCUSSION AND PERSPECTIVES

This general discussion outlines the implications of the outcomes of this thesis relative to the stated objectives and explores potential future directions for the conservation practices studied, including both implementation strategies and areas for further research.

5.1 How effective are the well-documented conservation practices conservation tillage, cover crops, and micro-basin tillage?

In the second chapter of this thesis, we quantitatively reviewed the effectiveness of three well-known conservation farming practices in mitigating runoff and erosion, under NW European conditions: conservation tillage, winter cover crops, and tied-ridging (micro-basin tillage) in potato crops. Our meta-analysis provides estimates of their overall effectiveness against water and sediment overland flows and, more importantly, highlights the influence of various factors, both related to the environment and modalities of practices implementation. In this respect, the findings of this work provide a useful resource for informing decisions made by farmers, agricultural advisors, and policymakers, thereby facilitating the promotion and implementation of conservation practices under favorable conditions. However, our results offer only limited insight into which soil (physical) properties are impacted by the conservation practices studied, as well as the extent of these effects. Similarly, this limitation extends to the specific hydro-pedological processes involved in runoff and erosion mitigation (see Section 1.2 and Figure 1.3). In this regard, while it is theoretically possible to confirm hypotheses by identifying significant factors in the meta-analysis, it remains challenging in practice due to some key factors poorly reported in primary studies (e.g. residue cover), or because the limited datasets hinder the estimation of (complex) interactions between factors. For example, if a positive interaction between “intensive non-inversion tillage” and “time since practice application” had been established on the flow mitigation effect of conservation tillage, it might have been primarily attributed to the rising concentration of organic carbon at the soil surface, assuming that intensive conservation tillage affects other soil properties similarly to

conventional tillage. Thus, this meta-analysis represents a highly applied research study. It is therefore crucial to continue conducting experiments designed to test specific hypotheses about the impact of conservation practices on soil physical properties, which should then be measured (e.g. Schlüter et al., 2018). Relating changes in soil properties that result from a conservation practice to flow mitigation effects – particularly through the use of physically-based models – represents a powerful opportunity but also an additional challenge (see Section 5.3).

In the meantime, our meta-analysis can also serve as a valuable resource for empirical modeling approaches aimed at appraising larger-scale (e.g., catchment) scenarios of conservation practice implementation, particularly those using USLE-based models, as commonly performed (see Section 5.3). The meta-regression equations (see Tables A.2.2 and A.3.3 in Appendix A) allow for the calculation of a seasonal erosion response ratio ($= \frac{\text{Erosion}_{\text{conservation}}}{\text{Erosion}_{\text{conventional}}}$) for a conservation practice under specific conditions. This response ratio differs from the USLE P-factor in that it does not account for soil losses over the whole year. By applying the erosion response ratio to the time-varying version of the USLE C-factor – which is based on a weighted periodic "soil loss ratio" according to the proportion of erosivity in each period (Renard et al., 1997) – a "CP-factor" (combined crop-management and conservation practice factor) can be calculated for an entire year or rotation:

$$[\mathrm{CP}-\text{factor}]_{\text{rotation,conservation}} = \frac{\sum_i (\mathrm{SLR}_i * \mathrm{EI}_{30\,i} * \mathrm{RR}_i)}{\sum_i (\mathrm{EI}_{30})} \qquad \text{(Eq. 5.1)}$$

where SLR_i is the soil loss ratio for period i (e.g., month), defined as the ratio of soil losses under a specific cover (crop-dependent) and management practice (to be considered in this approach as conventional cropping and tillage) compared to bare soil. $\mathrm{EI}_{30\,i}$ is the rainfall erosivity in the period i. For example, Bielders et al. (2011) determined 10-day erosivity and SLR data for major crops and rotations in Belgium. RR_i represents the seasonal erosion response ratio of the conservation practice if period i falls within the season to which the RR applies, otherwise RR_i should be assumed to equal 1.

Regarding the assessment of runoff mitigation, most empirical hydrological models use the curve number ("CN") method (Deliman et

al., 1999). Deriving CN for the conservation practices ($\mathrm{CN}_{\mathrm{conservation}}$) examined in our quantitative review is not straightforward. The most practical approach would be to compute runoff for a given rain event under a conventional cropping scenario, such as using seasonal CNs as a function of cover and crusting stage (Evrard et al., 2008a). Then, if the practice is deemed effective, the simulated conventional runoff can be adjusted by multiplying it by the runoff response ratio calculated from our meta-regression equations (see Tables A.2.1 and A.3.1 in Appendix A), which serves as a simple reduction factor to estimate runoff under the conservation practice. For assessments of both runoff and erosion mitigation, these methods based on the use of response ratios in empirical models assume that the conservation practice maintains constant effectiveness throughout the applicable season, and has no impact beyond that period. They also assume that the mitigation effect of a practice is uniform across all spatial scales (a point that has been questioned; Langhans et al., 2019; Leys et al., 2010), from the plot level evaluated in the meta-analysis, to the field or catchment scale where it would be simulated.

Our meta-analysis in Chapter 2 also highlighted several explanatory factors influencing the effectiveness of the conservation practices. For winter cover crops and conservation tillage, the time elapsed since the beginning of practice implementation significantly increases their flow mitigation effectiveness, due to the cumulative benefits to soil structure improvement (see Sections 2.3.1, 2.3.3 and 2.4.2). As also demonstrated by other authors (e.g., Rhoton, 2000), the improvements in soil properties under conservation tillage are greatest in the first few (0-10) years compared to the long term, where the system tends towards a new state of equilibrium (Figure 2.4 a and b). Beyond this temporal trend, however, there appear to be cases where no substantial improvement in soil structure results from conservation tillage, as evidenced by Schlüter et al. (2018) even over a long-term basis (25 years). In addition to the rotation-related absence of deep rooting, the latter authors suggested that some sites might be less suitable for soil bioturbation by earthworms due to relatively dry periods, high soil pH (>7.2) and limited soil (loess) depth. We additionally suggest considering the possibility of an inherent bioturbation potential at each site, depending on initial earthworm population (in part related to

previous cropping practices), potential inoculum from surrounding natural areas, and predation pressure. Although finer-textured soils are known to favor earthworm populations (Briones & Schmidt, 2017), we found that these soils respond slightly less positively to conservation tillage than coarser-textured soils in terms of flow mitigation (Figure 2.4 e and f). The latter soil type being less sensitive to compaction, the (short-term) benefits of conventional tillage on soil structure are lower as well, resulting in a higher relative efficiency of conservation tillage.

Other factors influencing the mitigation effectiveness of conservation tillage were related to the choice of non-inversion tillage scheme. Although previous studies had already revealed differences between no tillage, reduced tillage and deep tillage systems (Maetens et al., 2012a; Xiong et al., 2018), our meta-analysis is the first one to highlight the following more specific factors: tillage depth, number of operations, and use of powered implements. The fairly intuitive explanation for these factors lies in the direct link between minimum tillage intensity and preserved soil structure, hence maximum flow mitigation, with two exceptions: no tillage results in more compacted soil and thus lower runoff mitigation, and deep tillage fractures the subsoil, thereby increasing water infiltration hence mitigating runoff.

Crop type (spring vs. winter crop) was the last factor identified in our meta-analysis as significantly affecting the effectiveness of conservation tillage. One initial assumption for the greater effectiveness of conservation practices in spring crops was a greater magnitude of runoff and erosion under spring crops, which allows a higher mitigation effect from conservation tillage to be identified (see Section 2.4.2). This explanation alone is not sufficient, however, because even when the absolute amount of soil loss is included as a moderator in the meta-regression model, the "crop type" variable remains included in the "best" model (lowest AIC) (not shown). An alternative explanation may lie in soil susceptibility to compaction, in the same way as for the "soil texture" effect: under a winter crop, most in-field traffic (harvesting of a previous spring crop, tillage, and sowing) takes place in autumn, when the soil is wet and therefore vulnerable to compaction. Conventional tillage is more beneficial to soil structure under such conditions than on a good load-bearing soil (i.e., in the spring when the soil is dryer), resulting in a

lower relative effectiveness of conservation tillage. Yet a last explanation can be put forward for this "crop type" effect: winter cover crops are commonly grown before spring crops (but not before winter crops) in NW Europe, and they are known to have a beneficial impact on soil structure and protection by residues (see Section 2.4.2), as also discussed in Section 5.2 with regard to the strip-tillage practice.

Although conservation farming practices have been studied individually in the meta-analysis (and more widely in this thesis), several different practices can be combined in the same field. In particular, the logic of conservation agriculture lies in the joint effects of vegetation-based practices (e.g., cover crops, intercrops) and minimal soil disturbance (tillage) (see Section 1.2.2). In order to maximize the benefits provided by the former practices to soil structure, cultivated species should be diversified (Holderbaum et al., 1990; See Section 1.2.1), and the produced non-harvested biomass should be maximized (Ranaivoson et al., 2017), through species mixtures adapted to sowing date and pedo-climatic conditions, as well as through early sowing and late destruction of non-harvested crops (Feng et al., 2024; Laloy & Bielders, 2009b).

More generally, the pitfalls of systematic reviews and meta-analyses must also be stressed. Although their principles and methods have been designed to enable greater replicability (a key tenet of the scientific process; Powers & Hampton, 2019) as well as objective and accurate quantitative analysis of results from primary studies, several obstacles to these goals persist. Commonly recognized issues include, for example, measurement errors and various types of literature bias, such as publication, citation, or selective (non-)reporting bias (Chandler et al., 2021). Beyond that, a major obstacle to our meta-analysis, as with many others, was the lack of data reporting by authors in primary studies, compounded by the widespread lack of response following attempts to contact authors. The most critical missing data was experimental variance (i.e., variance between plot replicates), either due to a lack of reporting or because the treatments were not replicated in the experimental design (n = 1 plot per treatment). Notably, 73% of the 165 trials in the conservation tillage meta-database for the erosion outcome lacked experimental variance (Table 2.1). Fortunately, in our case, other variance components could be estimated (i.e., between-trials

and between-studies variability) but the inter-plot variability had to be neglected, resulting in an underestimation of the uncertainty around the effect size. Encouragingly, awareness of the value of meta-analyses is growing, and standard reporting recommendations for experimental studies have recently been proposed (Hillebrand & Gurevitch, 2013; Fernández-Castilla et al., 2024), mainly focusing on general statistical considerations. Nonetheless, in our systematic review, several potential explanatory variables were also subject to missing data. General variables like slope, soil texture, and rainfall were fairly well reported, and when missing, they often can be retrieved afterwards from open databases, provided the locations of the trials are known. However, most missing information related to the specific conditions of practices implementation. For the winter cover crops meta-database, this primarily involved the maximum vegetation cover (35% missing) as well as sowing and destruction date (30% and 43% missing, respectively) moderators. For the conservation tillage meta-database, missing data primarily pertained to the presence of winter cover crops (at least 15% missing, see below) and the amount of surface residues (80% missing). It is certainly difficult to anticipate the potentially relevant moderators to be measured and reported in primary studies, especially when evaluating an innovative farming practice. One partial yet straightforward solution is to systematically append pictures of the (field) experiments to published articles. This allows future systematic reviews to perform a posteriori semi-quantitative assessments of some moderators of interest. Meanwhile, authors of meta-analyses should be aware that being more restrictive with the quality and reporting criteria of primary studies will severely limit the number of primary studies included in their meta-databases, especially those from grey literature which are valuable for preventing possible publication bias. Ultimately, the trade-off between the quantity and quality of primary studies is a challenging balance to strike.

Another, little acknowledged, methodological issue we sometimes faced during the systematic review process relates to the translation of textual language (used for reporting primary studies in journal articles or reports) into facts, when encoding potential explanatory variables in the meta-dataset. For example, Jaskulska et al. (2020a), a primary study incorporated in our conservation tillage meta-

dataset, report the crop rotation in their experiment as follows: “In successive years of the experiment, the crops grown were winter rape, winter wheat, and maize.” For some readers, the lack of mentioned winter cover crops in their description (and throughout the article) may be interpreted as an indication that no cover crops were grown in their experimental fields, and it could have been encoded as such in the meta-dataset. More cautious readers, might have considered it an unknown variable i.e., missing data. Such subjective interpretations, although almost inevitable, are an actual impediment to the perfect reproducibility of systematic reviews and meta-analyses.

In Chapter 3, we evaluated the effectiveness of soil pitting in a maize crop at mitigating runoff, erosion and pesticide transfer. This form of micro-basin tillage was carried out by an adapted roller mounted within a seeding unit, assessed in the field by the CIPF (2012-2017). Based on the data collected on their plot experiments, the technique proved highly effective, reducing runoff by 69% and erosion by 83%. Although not presented in Chapter 3, several other tillage tools had been (briefly) tested by the CIPF (2012-2017) for micro-basin tillage: an adapted disc harrow, a “rouleau à ailette”, the "Marnik Van Mello" machine. All these tools appeared less effective at reducing surface flows compared to the adapted roller. The development of the latter device is the result of years of collaboration between a farm equipment manufacturer (“Atelier LSM”) and maize-specialized agronomists from the CIPF, combining their valuable technical expertise and numerous field trials. When considering the implementation of micro-basin tillage, such as within a regulatory framework, special attention should be given to the type of equipment used or to the pattern and dimensions of the depressions, to ensure effective application. The technical drawings of the designed adapted roller could also be made publicly available, so that it can be easily reproduced.

Compared with other conservation farming practices (e.g., strip-tillage, see Figure 4.3), soil pitting appears to provide a more consistent flow mitigation effect across the various trials (Figures 3.3 to 3.5). Traditional conservation farming practices generally rely on protecting and improving soil structure through vegetation and biological activity (see Section 1.2.2), both of which vary widely across different sites and

years. In contrast, the micro-basin tillage technique involves only a "mechanical" process of storing water on the surface in depressions created by tillage, resulting in a mitigation effect that is steadier across different sites and years. Consequently, soil pitting can easily be integrated into a conventional farming system, which is a major advantage of the technique, as it overcomes most barriers to adoption commonly invoked for conservation agriculture practices: complicated weed and pest management, reduced crop yields, or farming traditions (Wauters et al., 2010; Heller et al., 2024). On the other hand, the underlying mechanism of micro-basin tillage is also its weakness, as it does not enhance soil structure in any way, and probably even requires intensive tillage. Indeed, soil pitting does not seem compatible with conservation tillage techniques, as the shaping of depressions could be hampered by consolidated soil and the presence of crop residues. Since occasional tillage can partially offset the accumulated benefits to soil structure from conservation tillage (Blanco-Canqui & Wortmann, 2020), it is likely of little interest to apply micro-basin tillage into a rotation that was previously under full conservation agriculture. It is in fact debatable whether micro-basin tillage should be considered as a conservation farming practice. While it meets the ultimate objective of conserving water and soil in the field, it does not rely on any of the principles of conservation agriculture: maximizing and diversifying vegetation, and reducing tillage (see section 1.2.2; Landers et al., 2021).

5.2 How effective are the undersowing and strip-tillage innovative conservation practices?

In Chapter 4, we conducted plot experiments to evaluate the effectiveness of undersowing and strip-tillage in maize crop at controlling overland flows of water, sediments and herbicides. Our findings revealed that, as implemented in our study, undersowing maize with red fescue or white clover did not mitigate surface flows but did reduced maize yields, and resulted in significant weed pressure on the trial sites. Our plot experiments suggest that the lack of flow mitigation is attributable to the limited additional vegetation cover provided by undersowing during the maize growing season, itself due to the slow growth rate of red fescue and white clover. These species were deliberately selected for this trait in order to avoid excessive competition

with maize, though a (limited) maize yield loss did occur. During the maize growing season, it thus appears highly challenging to achieve both low competition with maize and (high) surface flows mitigation for maize intercropping techniques, whether undersowing or relay cropping (see Section 4.1). Even at the end of the subsequent winter season, the undersown crops reached biomass and vegetation cover levels no higher than those achieved by standard winter cover crops sown after maize harvest. Therefore, as compared to such traditional winter cover crops, undersown crops do not seem to provide any additional soil conservation benefits during the following winter season either. However, caution is warranted with this conclusion, as only two winter seasons were monitored and only at a single trial site (Ottignies).

Nevertheless, there may still be opportunities to develop the intercropping technique if one focuses on its role as a cover crop during the winter period following the (intercropped) maize season, i.e., when maize is not followed by a winter crop in the rotation. Prospective benefits could include developing a higher biomass compared to (late sown) traditional winter cover crop sown after maize harvest for soil conservation purposes (Alonso-Ayuso et al., 2020), although our study questions this potential (see above), or achieving other environmental targets such as mitigating nitrate leaching (Manevski et al., 2015). However, the weed control issue associated with the undersowing practice as implemented in our study remains to be addressed for two reasons. Firstly, the maize yield reduction observed in our trials (averaging -12%) reflects only the competition between maize and undersown crops, but not the additional loss caused by weeds themselves, since both control and undersown plots experienced similar weed pressures due to the application of the same active ingredient (sulcotrione) on both treatments. Secondly, high weed pressure is not just a concern for the current cropping season. The soil seed bank, which is replenished by previous weeds that have gone to seed (Gallandt, 2006), further complicates weed management in subsequent year(s) as well, as we observed in the Ott22 trial (see Figure B.1g in the Appendices).

Given that a high pressure in weed grasses (especially *Echinocloa crus-galli*) was frequently observed in our trials, and when considering undersowing only with clover (and not fescue), one might

first presume that an active ingredient specifically targeting grass weeds could also be applied, typically nicosulfuron. However, CIPF (n.d.) report that the biomass of white clover would still be reduced by 66% with this herbicide, compared to an undersown white clover growing without any herbicide spraying. Similar intercropping studies include Kleinman et al. (2005), who used imazethapyr and bromoxynil as post-emergence treatments when interseeding corn with ryegrass, clover, or alfalfa. However, their approach also aimed to control the undersown crops to limit competition with maize. Given the drawbacks of post-emergence applications, pre-emergence herbicides could provide an effective solution for weed control in conjunction with undersowing. Previous studies have demonstrated success with thiocarbamate application followed by red clover undersowing (Wall et al., 1991) or pendimethalin followed by Fabacea intercropping (Bibi et al., 2019). Another promising implementation is reported by Kincl et al. (2022), who describe intercrop sowing (ryegrass, rye, clover, or *Phacelia*) combined with mixed weed control at the 3-4 leaf stage of maize. This approach involves hoeing and intercrop sowing between maize rows, alongside strip herbicide application on the maize rows. Eventually, to eliminate weeding constraints, one could simply look back to the relay cropping practice (see Section 4.1) where the crop associated with the maize (typically ryegrass; Conrad & Fohrer, 2016) can be sown after the post-emergence herbicide application. In fact, ryegrass is more vigorous than most other species frequently intercropped with maize (e.g., clovers), ensuring optimal growth during the autumn season and resistance to winter conditions (Alonso-Ayuso et al., 2020). From a longer-term perspective, advancements in precision agriculture could provide future solutions for weed management in intercropping systems through deep-learning-based plant species recognition. Selective control of weeds while preserving undersown crops would become feasible, similar to current capabilities for distinguishing weeds from a main crop, as demonstrated by Nasiri et al. (2022) on sugar beet.

Unlike undersowing, the strip-tillage practice effectively controlled runoff (-31% on average) and erosion (-60%) as measured in our plot experiments. These efficiencies are less pronounced than those measured by other studies conducted in a similar context (e.g. Ryken et al., 2018). A possible explanation may lie in the greater soil disturbance

induced by the strip-tiller used in our trials. Approximately 55% of the total area – a 40 cm wide row out of 75 cm – was tilled, whereas the literature refers to half as much: typically, no more than one-third of the surface should be tilled under strip-tillage practice (Morrison, 2002). It would be worthwhile to further investigate how the balance tips between production and conservation objectives as a function of the proportion of the tilled area under strip-tillage. However, the very limited number of studies conducted in NW Europe currently precludes such an analysis.

Furthermore, the mitigation effects provided by strip-tillage varied significantly between our trials (see Figure 4.3). Excluding trials where (very) low surface flows were observed, strip-tillage proved ineffective – or even counter-effective – at the Ottignies site, but effective at Bonlez and Ittre. One first hypothesis for this inter-trial variability is that conservation tillage techniques, including strip-tillage, generally perform better on coarser-textured soils (see Figure 2.4 and Section 2.4.2). However, the textural effect alone does not seem sufficient to explain such significant variations. Another hypothesis proposed is the role of winter cover crops (preceding the maize growing season in which strip-tillage is applied) in the flow mitigation effect provided by strip-tillage, through both: 1) improvement of soil structure (e.g. macroporosity) by vegetation and biological activity and 2) protection, by residues, of the soil surface from crusting and detachment (see Section 1.2.2 and Figure 1.3). In the Ittre (2022) trial, the strongest reduction in runoff and erosion is most likely the result of these combined benefits. Interestingly, in Bonlez (2021), residue cover was very similar between control and strip-tilled plots; therefore, the flow mitigation effect is primarily attributable to improved “internal” soil structure only. Unfortunately, the aforementioned hypotheses could not be strictly verified in our study from a statistical point of view, as some combinations of variables were not observed in the trials, such as a site with fine-textured soil and a well-developed previous winter cover crop. The respective contribution of enhanced "internal" soil structure on the one hand and of residue protection on the other hand, to the flow mitigation effect provided by conservation tillage techniques remains quantitatively unclear (Leys et al., 2010; Ranaivoson et al., 2017). This question can be linked with the diversity of definitions for conservation

tillage, some of which focus on the rate of soil cover, while others describe "acceptable" tillage methods to limit soil structure disturbance (see Section 1.2.2).

Overall, however, the literature attributes a predominant role to the presence of plant residues in achieving soil conservation, even independently of reduced/no tillage, i.e. through mulching (Kwaad et al., 1998; Lal, 2009; Maetens et al., 2012a). Thus, mulching could prove to be an effective conservation strategy for mitigating overland flows in situations where intensive tillage is almost inevitable, e.g. in organic farming systems or in potato cultivation (see below). The ongoing decline in livestock population (Eurostat, 2023) and the recent extensive planting of hedgerows in NW Europe could lead to increased availability of organic material suitable for mulching in the near future, such as straw (previously used as litter) and chipped wood, respectively. Given the widespread practice of growing winter cover crops in NW Europe (Heller et al., 2024), a theoretically sound yet imaginative practice would be to harvest and store the above-ground biomass of the winter cover crop (as for hay) in early spring. Afterwards, the soil could be conventionally tilled, the main spring crop sown, and the previously collected biomass spread on the field as mulch.

The two innovative conservation practices investigated in this thesis – undersowing and strip-tillage – were applied in maize crop, primarily because it is the most widespread spring crop (therefore erosion-sensitive) in Western Europe (see Figure 1.1). Sugar beet follows in second place. In this case, the lower row spacing – generally 45 cm instead of 75 cm for maize – limits the opportunity for micro-basin tillage, as the depressions would have to be small or might even be challenging to shape. However, similar to maize, strip-tillage offers promising prospects in sugar beet, and could be a suitable alternative to no-tillage which causes considerable yield losses (Geelen et al., 1996; Morris et al., 2007; Laufer & Koch, 2017). Potato is the third most widely grown spring crop in Western Europe. The cultivation on ridges and the need for a loose soil structure for proper tuber development require intensive tillage, relying on numerous operations and powered tillage implements. Therefore, conservation tillage can, at best, only be applied in the form of intensive non-inversion tillage, for which the expected

flow mitigation effect is very limited (see Figure 2.4g and h). In fact, within the framework of the Intell'eau project (see Section 1.1.3), intensive non-inversion and other conservation techniques were briefly investigated in a potato crop through plot experiments similar to those in Chapter 4 (not presented in this thesis nor published), including intercropping of faba beans, and autumn ridging (see also Henriksen et al., 2007; Haccart et al., 2021). None of these practices significantly mitigated runoff or soil erosion. The only remaining existing conservation technique for potato crop is tied-ridging (micro-basin tillage, see Chapter 2), which is highly effective for mitigating overland flows, but suffers from the same drawbacks as soil pitting (see Section 5.1). Therefore, it seems practically impossible to grow potatoes under conservation agriculture at present. A last possibility worth further investigation in potato crop would be to consider mulching (see above), potentially in combination with tied-ridging. Similarly, for organically farmed areas facing overland flows issues, mulching could be the ultimate soil conservation solution. Intensive tillage (including ploughing) is often required for weed control, and repeated operations for mechanical weed control during the season would damage depressions in the case of micro-basin tillage.

Aside from the outcomes of these plot experiments, we also wish to raise a methodological argument related to how treatments, whether conservation or control, were implemented in the field trials of Chapter 4:

a) Implementation of the conservation treatments: when it comes to testing different farming methods through field experiments, one may basically choose to compare (Büchi et al., 2019): 1) Different individual cropping practices, usually through a short-term experiment (one growing season), with a single independent variable, i.e., a unique factor to be tested such as a fertilization dose, or 2) Different cropping systems, each of which is a coherent, integrated set of management techniques, e.g., organic vs. conventional farming systems. This usually involves conducting the trial over a full rotation to capture the entire system. The case of conservation tillage practices is tricky with regard to such categorization, as also discussed by Giller et al. (2009). We aimed at

testing specific conservation practices – undersowing and strip-tillage – in a given crop, which was maize. Because of crop rotations, sites changed (almost) every year, which aligns our trials more closely with the first type of experiment, i.e., assessing individual practices. Meanwhile, conservation practices only really make sense as part of a comprehensive cropping system geared towards reducing tillage and maximizing vegetation (see Section 1.2.2); strip-tillage relies on the presence of residues, and undersowing could provide soil conservation benefits after the maize growing season as a winter cover crop (which was only minimally targeted by our monitoring, see above). Part of the benefits to soil structure are expected in the longer term (see Chapter 2), which our experiments were unable to highlight, except perhaps at Bonlez, where initial conditions may have influenced the outcomes. This site was historically cultivated under conservation tillage, so it may have held greater conservation potential favoring strip-tillage treatment from previously accumulated benefits to soil structure. In 2023, we expanded our approach to the undersowing practice to embrace a multi-annual application; specific winter cover crops of fescue and clover were sown in the future undersown fescue and clover plots, respectively, with the aim of mimicking two consecutive years of undersowing, to test for potential benefits of the practice when integrated into a maize monocropping system.

b) Implementation of the control treatment: For the sake of experimental convenience but also to represent the diversity of "conventional" practices, the cropping operations applied as part of the control treatment often followed the ones used by the farmer who owned the field. As an example, the management of winter cover crops or of primary tillage in the control plots differed greatly across trials (see Table 4.1). Manure was spread in all treatments on trials in 2023, while it was not in 2021 and 2022.

In a nutshell, implementing (conservation and control) treatments involved transposing the agronomic design of the treatment into a set of practices, considering the specific conditions of each trial. As a result, (comparisons of) conservation and control treatments

do not carry exactly the same meaning across trials, which also explains the great variability in the response of treatments to surface flows (see Figure 4.3 and 4.4).

5.3 How to account for conservation practices in models?

As discussed in Section 4.2.3.1, results from field experiments remain strongly dependent on the specific experimental conditions under which they were conducted. Given this limitation, modeling is frequently used to extrapolate the outcomes of field studies to other (broader) conditions, in other words, to generalize the findings. In the context of conservation farming practices, this typically involves studying scenarios of technique(s) implementation on a larger spatial and/or temporal scale, with the aim of predicting the consequences on the water balance. Such an effort requires the ability to model both the hydrological behavior of conventionally cultivated cropland and its reponse to a conservation practice, accross a variety of pedological, topographical and climatic conditions. Many hydrological models operate on an empirical basis, using the SCS-CN method to calculate runoff (and infiltration) and/or USLE (or derivatives: RUSLE, MUSLE) for soil loss (Deliman et al., 1999). Adjusting the curve number parameter or the USLE C- or P-factor for a given management technique is a common methodology in the literature (Wang et al., 2008; Musyoka et al., 2023; Sittig et al., 2022; Kincl et al., 2021). Such an approach is practical for application on a large scale when available information limits adequate parameterization. However, it assumes that the mitigation effect of the practice can be captured by a single parameter, which, in the case of the P-factor, corresponds to a simple proportional reduction in soil losses regardless of the conditions (independently of other USLE factors). Furthermore, when different conservation practices are applied on the same plot, a specific P factor for the practice combination should be empirically evaluated from plot experiments. Given the labor-intensive nature of these experimental setups, such values are often not available. It must then be assumed that the erosion mitigation effect of the combination of practices (overall P factor) is the product of their individual efficiencies (individual P factors), which in most cases probably overestimates the impact of the conservation management.

Consequently, the ability of such empirical modelling methods to reliably simulate (the effect of conservation practices on) runoff and soil losses remains limited, especially at the scale of plots and individual events (see Section 3.3.2 and Figure 3.6; Lal et al., 2017; Maski et al., 2008). Moreover, numerous soil properties are affected by conservation practices (see Section 1.2 and Figure 1.3), which in turn impact complex hydropedological processes that non-linearly drive overland flows (Vreugdenhil et al., 2022). This suggests the need to refine the representation of variables and processes in models, specifically by employing mechanistic models, and to set appropriate parameters according to soil management methods, with the goal of improving prediction accuracy. By modelling how specific processes are affected by conservation practices, physically-based models also open up the potential for a more accurate assessment of the hydrological response to implementing combinations of multiple conservation techniques. Note, however, that the distinction between empirical and mechanistic models is not always clear-cut but rather exists along a continuum. Some so-called “empirical” models incorporate basic representations of processes; for example, the curve number equation is derived from a simple water budget. Conversely, many “mechanistic” models include empirical components, such as the percolation calculation in the CREHDYS model, or recent hybrid approaches that combine physical modelling with machine-learning algorithms (Maity et al., 2024).

Soil properties, along with related processes, that are potentially impacted by conservation practices are generally already incorporated into physically-based models: macroporosity (Ksat), structural stability, shear strength, etc. Therefore, the primary limitation in such mechanistic modeling efforts is not so much the ability to represent the affected variables and processes, but rather the challenge of achieving proper parameterization. For example, the European « OPTAIN » project aimed to model water retention measures like reduced (“low”) tillage using the SWAP model (Mikołaj et al., 2023). To this end, Marval et al. (2022) reviewed tillage-induced relative changes in soil properties like saturated water content, α and n parameters from the water retention curve, or saturated hydraulic conductivity. However, they identified several contradictory findings, which they attributed to specific soil and geographical conditions. Even measuring soil physical properties under

defined conditions for model parameterization is not trivial, as we experienced in Chapter 4. When the CREHDYS model was parameterized with Ksat values measured on-site using an infiltrometer, it utterly failed to reproduce observed runoff. This discrepancy likely reflects that our Ksat measurements did not adequately capture the soil surface state (crust), which is crucial in overland flow processes. A similar issue could have arisen with the 'median particle diameter' parameter (d50) if we had sought to measure it on-site instead of estimating it from textural fractions (see Table 4.2). *d50* influences flow detachment/deposition, and therefore represents not only the distribution of individual soil particles but also, to some extent, the aggregate stability. Due to equifinality and overparameterization concerns (Whittaker et al., 2010; Beven & Freer, 2001), typically only a few parameters can be calibrated based on measured flow data (i.e., through inverse modeling, see Chapter 4). It is thus tricky but essential to understand the physical reality represented by any model parameter and, most importantly, to be able to reliably measure it, or at least estimate it. Such developments require the accumulation of modeling efforts under various conditions, validated by field experiments, and using models including common process representation (e.g. Green-Ampt for infiltration and runoff) for sharing common parameters between studies. Our work reported in Chapter 4 aligns with this approach, as it has benefited from numerous previous studies for parameterization (see Table 4.2). Furthermore, the pedotransfer function *Ksat* = f(sand) (Figure 4.4a) that we identified could be applied to estimate *Ksat* in future studies under similar cropping and soil conditions.

It is precisely the lack of background in the literature on process-based modeling of pesticides that has hindered us from modeling the surface flows of active ingredients associated with runoff and soil losses, in Chapter 4. Most hydrological models that incorporate a pesticide flow module (e.g., SWAT, PRZM, CREAMS/GLEAMS) operate, at least partially, on an empirical basis, and consequently suffer from the limitations discussed earlier (Deliman et al., 1999; Centanni et al., 2023). More recently, modeling of pesticide concentrations in overland flows have been integrated into mechanistic models, with encouraging results. However, these methods inevitably introduce additional parameters

such as the mixing layer depth or the soil-water partition coefficient. For example, Lefrancq (2014) implemented a first-order kinetic reaction for pesticide sorption/desorption in the LISEM model to simulate the surface transfer of (both dissolved and particulate) pesticide, and applied it to maize and sugar beet crops. Several additional parameters had to be calibrated for the specific active substances studied, i.e. metolachlor and acetochlor: film transport coefficient, desorption rate, mixing layer depth, and soil-water partition coefficient. Another example is reported by Saia et al. (2013), who integrated a pesticide module into the WEPP model, and applied it to atrazine in corn-soybean rotation. One of the module's parameters is the half-life of the active ingredient. However, due to the wide range of measured values across different soil types, climates, or soil pH levels, calibration of this parameter was still required in their study. As previously discussed with regard to soil-related parameters (e.g. Ksat or d50), assigning a parameter value from measurements of the corresponding physical reality – in this case a physico-chemical property of an active substance – remains challenging due to the high variability of measurements and/or their lack of alignment with the actual meaning conveyed by the parameter in the model. Beyond the limited availability of suitable models (LISEM is event-based and not continuous), we deemed that the restricted pesticide monitoring in our trials (see Chapter 4) as well as the literature background in terms of parameter estimation (restricted to a few studies and a few active substances) would not have allowed us to accurately model sulcotrione fluxes. Perhaps in the future, following the approach of pedotransfer functions for soil-related parameters, “pesti-transfer” functions could be developed to estimate some parameters frequently used in pesticide modelling formalisms, based on measured physico-chemical properties of active substances.

Whatever the approach (empirical or mechanistic), research into pesticide modeling in overland flows is valuable because predicting the environmental fate of active ingredients is crucial for assessing area-specific vulnerability and evaluating the impacts of conservation practices. From a purely on-site soil conservation perspective, focusing on a single agricultural field is inherently meaningful, as the amount of soil eroded from the plot directly indicates the threat to the soil resource. Tools such as the empirical USLE model facilitate the extrapolation of

results from measured to unmeasured plots. Upscaling from the field to a watershed level is not strictly necessary, although the transfer of flows and sediments between plots certainly affects sediment yields. In terms of flood risk, however, the amount of runoff from a specific plot alone provides limited information. It must be contextualized within the entire catchment, where upstream contributing area and downstream flow concentration towards urbanized zones are critical for flood risk assessment. A similar approach applies to pesticide flows, with the vulnerable targets downstream being surface water bodies.

In fact, when considering the impact of conservation practices on water quality, studies from the literature – including the present thesis – focus almost exclusively on surface water (Skaalsveen et al., 2019; Busari et al., 2015), probably because this is where the greatest benefits are expected. Looking back at Chapter 1, the soil's function of regulating and purifying water more generally refers to its buffering capacity, both quantitatively (by reducing runoff and holding water during heavy rainfall and drought periods, respectively) and qualitatively (by filtering pollutants). Given that conservation practices enhance water infiltration, particularly through macropores, they could increase groundwater recharge (Donigian & Carsel, 1986). Consequently, higher loads of agricultural contaminants (nutrients, pesticides) could also be leached into the aquifer, although part of it would be filtered (i.e., adsorbed, degraded) in the process. Significant environmental processes and key hydropedological variables to monitor differ between surface water and groundwater. Consequently, studies are generally confined to focusing on either one or the other compartment, whether through modeling or field experiments. For instance, in the FOCUS framework used for European regulation on plant protection products, ecological risk from active substances is assessed by two different models for surface water and groundwater (Holvoet et al., 2007; Bach et al., 2016). A combined appraisal of the impact of conservation practices on both compartments is scarce in the literature and warrants further investigation (Donigian & Carsel, 1986; Wang et al., 2014). According to the pollution swapping principle, no single mitigation option would reduce all pollutants (Stevens & Quinton, 2009).

5.4 Prospects for conservation policies and conservation practices adoption

The intensity of overland flow processes, and the associated risks, are expected to become increasingly severe in the near future in NW Europe due to the rising rainfall erosivity driven by climate change (Panagos et al., 2017). Addressing these challenges through the implementation of targeted conservation strategies is therefore of critical importance. As explained in Section 1.1 and reiterated in Section 5.3, overland flow processes lead to two sets of issues with different timeframes:

1. **Short-term (acute) risks**: Refers to the risks of surface flows from agricultural fields arising from an extreme event hazard, and discharging towards off-site vulnerable targets. Upscaling to the catchment level is necessary because these risks strongly depend on the contributing area upstream and flow concentration downstream. Such short-term issues can be further subdivided into two categories depending on the vulnerable target and the flows of interest:

    1.1 **(Muddy) flood risk**: Refers to the potential discharge of sediment-laden runoff to urbanized zones.

    1.2 **Ecological risk**: Refers to the potential sediment and pollutant (nutrients, pesticides) loads being discharged into natural areas hosting biodiversity, particularly surface water bodies.

2. **Long-term risk**: Refers to the risk of soil resource depletion, threatening productivity. The focus is on the balance between the rates of soil erosion and soil formation. Assessments at the scale of individual plots remains relevant.

Consequently, the assessment of each risk requires a distinct methodology, and a given field may exhibit varying levels of risk depending on the aspect considered. In Europe, since the 2000's, the FOCUS protocol has been implemented to appraise the ecological risk for any new plant protection product submitted for authorization. This process involves modelling the fate of the active ingredient in surface

water and groundwater compartments and evaluating possible toxicity for aquatic organisms and human health under scenarios representative of the diverse soil, climate, and landscape conditions across Europe. Based on the outcomes of the FOCUS assessment, mitigation measures – such as adjustements to the pesticide application rate or timing, or the required buffer strip width at the edge of the field – are prescribed for the use of the substance. However, beyond the empirical nature of the models used in this framework (e.g., PRZM), the evaluated scenarios do not necessarily account for the worst-case situations in each member state (Bach et al., 2017), indicating that the FOCUS protocol may be insufficient for thoroughly protecting water resources. To our knowledge, Belgium does not employ a systematic modeling tool to further assess the ecological risk during the national registration process for active substances. National authorities may impose use restrictions based on expert opinions, but this decision-making process lacks transparency.

Regarding flood risk assessment, the Walloon administration apparently adopted a bottom-up approach: stakeholders – including municipalities, individuals, and farmers – affected by muddy flood problems can seek advice from entities such as the GISER unit or Natagriwal to explore solutions, including soil conservation management techniques (see Section 1.1.3). Currently, the only restrictive agricultural policy in Wallonia designed to address overland flow issues is the “BCAE 5”, the conditionality rule of the Common Agricultural Policy that targets soil degradation. This policy imposes rules on crop type, grass strip width, and tillage methods based exclusively on the slope steepness of the plots. Consequently, the design of this policy primarily aligns with long-term risk management, although it also aims to mitigate the off-site impacts of overland flows. If scientifically feasible, assessing the levels of the three aforementioned risks for each plot, and incorporating all of them into the BCAE 5 framework, woud allow for a more comprehensive management of overland flow issues.

Recently, the Walloon administration attempted to target the long-term erosion risk more accurately when it drafted a new version of the Strategic Plan of the Common Agricultural Policy, which included

significant modifications to the BCAE 5 (SPW, 2022c). This project involved an assessment of erosion risk for each plot, accounting for multiple environmental factors: soil erodibility, local rainfall erosivity, and slope steepness and length, following the USLE modelling approach. This was made possible by the availability of high-resolution georeferenced data in Wallonia. The soil map, along with the Aardewerk, RéQuaSud, and CARBOSOL databases, enabled the mapping of textural fractions, rock fragment and soil organic carbon contents (SPW, 2020 & 2022d). The Pameseb (CRA-W, 2022) and SPW (2022a) weather station networks provide historical climate data, while a 1-m DEM is sourced from LIDAR measurements (SPW, 2015). Depending on the plot's risk category, farmers would have been required to choose between different cropping constraints: crop type, plot subdivision, buffer strip, and/or conservation practices. This initiative was ultimately abandoned following pressures from farmers (unions) (RTBF, 2023). Despite very limited scientific evidence regarding the undersowing practice in the NW Europe context at that time, it was still included as a potential mitigation measure within this framework. We now reveal that the technique is neither effective against overland flows, nor agronomically suitable in its current state (see Chapter 4). This situation highlights the need for caution when incorporating agricultural practices into a regulatory policy. Before such practices can be recommended, robust scientific evidence is required, based on field trials conducted over multiple sites and years.

As detailed in Section 5.3, scientific knowledge at this stage does not appear ready to support the incorporation of a (purely) physically-based modeling framework into policies regulating overland flows, as large-scale estimation of the (numerous) parameters, and consequently the risks, remains highly uncertain. However, empirical modeling could reasonably serve this purpose. This applies both to the assessment of risks themselves (see above and Section 5.3), and to the evaluation of the mitigation effects from conservation practices on these risks (see Section 5.1 and 5.3).

Apart from restrictive policies, it is generally assumed that a farmer is likely to adopt a new agricultural practice if there is a substantial benefit to be gained (Morgan, 2005). Given the depth of the

loess layer in most areas of NW Europe, soil productivity is not threatened in the medium term by soil erosion, and most of the consequences of overland flows are externalities (see Section 5.1). Among the barriers to adopting conservation practices, the literature frequently highlights non-economic factors such as lack of knowledge, availability of adequate equipment, and farming tradition (Heller et al., 2024; Wauters et al., 2010; Lahmar, 2010). However, it must be acknowledged that switching from a conventional to a conservation tillage system is expected to reduce crop yields in temperate regions, on average by approximately 7% for no-tillage and around 3% for reduced tillage (Van den Putte et al., 2010; Achankeng & Cornelis, 2023). This yield loss is typically more severe in erosion-sensitive crops such as maize and sugar beet.

However, crop yield refers only to the absolute volume of agricultural production. From a microeconomic perspective, a new farming practice will be adopted by farmers if it leads to an increase in gross margin, which is the difference between revenue (from production sales) and production costs (fuel, machinery, labor, etc). For example, a farm-scale bio-economic analysis by Townsend et al. (2016) in England concluded that a direct drilling system would be financially more profitable than a conventional tillage system, despite crop yield reductions of up to 14%. The outcomes of such analyses strongly depend on contextual factors, including farm size, land and equipment ownership, input and labor costs, as well as the type and market price of agricultural products. Following our implementation of strip-tillage in maize crop, an average yield loss of 11% was measured (Table 4.4). A basic cost-benefit analysis shows that the income loss resulting from switching from a conventional maize cultivation system (expected average yield of 17 t maize dry matter/ha) to a strip-tillage system (15.3 t/ha) would amount to approximately -250€/ha, assuming 150€/t. Excluding labor costs, this loss far exceeds the expected reduction in mechanization costs (about -50€/ha; CRA-W, 2021). However, most farmers in NW Europe have little flexibility regarding their income; in Wallonia, the average gross income of a farmer between 2000 and 2020 was 54% of the average income of the population working in other sectors (SPW, 2023).

If the adoption of conservation farming practices is to be encouraged outside a constraining policy framework, it will be necessary to compensate for such income losses through financial incentives. Indeed, the environmental benefits provided by conservation farming practices are typically not factored into a farmer's economic decision-making. The rationale behind economic incentives is that society pays farmers through state support for the positive externalities (i.e., ecosystem services) generated by adopting "good" farming practices. This could be achieved as part of the CAP subsidies from the 1st pillar, through "eco-schemes", which are good farming practices voluntarily committed to by farmers in exchange for payment (Runge et al., 2022). Although this thesis has focused solely on the hydrological function of the soil – specifically, the regulation of overland flows – the broader concept of "soil health" must be considered when planning the promotion of agricultural practices that support ecosystem services. This requires an integrated assessment of how all soil functions respond to the farming practice or system: food and economic production, habitat provision, carbon storage, pollutants filtration, etc. (Cárceles Rodríguez et al., 2022; Evangelista et al., 2023; Kibblewhite et al., 2018; Panagos et al., 2022; see also the discussion related to the pollution swapping principle in Section 5.3).

Finally, this whole thesis has focused on the effectiveness of conservation farming practices, considering that the other main agronomic components, i.e. fertilization and weed control, are managed 'conventionally'. In the context of Europe's goal to achieve synthetic pesticide-free agriculture by 2050 (Mora et al., 2023), conservation agriculture may need to transition towards organic conservation agriculture, which currently presents agronomic challenges, especially regarding weed control (Casagrande et al., 2016). Currently, most organic conservation farming systems rely on intensive (non-inversion) tillage, due to the repeated operations required to control weeds (Ferdinand & Baret, 2024). However, as we demonstrated in Chapter 2, beyond the abandonment of ploughing, a conservation tillage scheme is only truly effective if it significantly reduces overall tillage intensity, including the number of operations and the use of powered implements. Therefore, alternative strategies will be needed to destroy cover crops and control weeds while minimizing soil disturbance, including rolling (e.g., "roller-

crimper"), mulching, shredding/mowing, pulling (e.g., "Weedpuller"), electric weeding, and grazing (Ashford & Reeves, 2003; Peigné et al., 2016). Many of these alternative solutions also provide a new lever for preventive weed control, by providing a surface mulch that smothers juvenile weeds (Akhtar et al., 2022). For row crops, sowing in a living mulch by using a strip-tiller also offers promising prospects in this area (Hartwig & Ammon, 2002; Siller et al., 2016). The mulch, if not supplied by a previous winter (cover) crop, could be sourced from external organic materials such as straw from other fields or chipped wood from surrounding hedgerows. Nevertheless, it must be recognized that the weeding effectiveness of these methods is unlikely to match that of conventional methods like ploughing or herbicides.

Throughout the thesis, we mainly focused on crops commonly grown in NW Europe, particularly those that most contribute to overland flow issues, such as maize and potatoes. In any agricultural policy planning, beyond simply adaptating cultivation techniques alone, a more comprehensive reflection is required on farming systems as a whole, including the types of crops to be promoted. Certain crops (or crop associations), though less productive, would be relevant to self-sufficient farming systems with low environmental impacts, including soil erosion. These include for example permanent and temporary grasslands (possibly associated with legumes), hemp (*Cannabis sativa L.*) or innovative high-potential crops such as the cup plant (*Silphium perfoliatum*) and camelina (*Camelina sativa*) (Zanetti et al., 2021; Peni et al., 2020).

In fact, the farming and food system that we aim to develop through agricultural policies calls for a sociological question: to what extent do we allow our anthropic activities to deplete soil and pollute water resources, for the benefit of our current standards of living? On the one hand, should our fertile loess soils be exploited to produce high yields and high economic value crops, driving down food prices but resulting in environmental externalities, that society will ultimately pay for, either now or in the future? On the other hand, to what extent are we willing to substitute our current cultivation practices and crops for more extensive yet less productive ones, thereby accepting higher costs for food (or taxes)?

5.5 General conclusions

This thesis aimed to quantify the effectiveness of several conservation farming practices in mitigating runoff, soil erosion and pesticide surface losses in NW Europe. We have shown that over the long term (about more than 10 years), conservation (i.e., non-inversion) tillage remains the most effective in-field technique for controlling runoff and soil losses. Winter cover crops are strongly suspected to play a key role in supporting and protecting good soil structure in these agricultural systems. Among the various conservation tillage methods, direct drilling is the most effective at reducing soil erosion, whereas intensive (non-inversion) tillage is the least effective. However, for many non-cereal crops, no tillage significantly reduces crop yields. For row crops such as maize, strip-tillage offers an attractive trade-off between production and conservation objectives.

Micro-basin tillage techniques (e.g., soil pitting in maize, tied-ridging in potato) are very effective during the season in which they are implemented. However, they require intensive tillage which harms soil structure, therefore exposing the soil to overland flow issues during subsequent crop rotation. Therefore, micro-basin tillage practices are more relevant to be recommended as an alternative to conservation tillage, particularly in potato cultivation where intensive tillage is already required, or for further development in vegetable cultivation.

Crop association with maize appears to provide minimal benefits as a conservation practice. There seems to be no intercropping method (in terms of species and sowing timing) that achieves a balance between effective flow mitigation and minimal competition with maize.

Given the primary role of plant residues in soil conservation, as emphasized by the literature, mulching the soil with organic material (e.g., straw, chipped wood, cover crop residues) could be a promising avenue for conservation management, despite not being studied in this thesis. This approach might complement other practices such as micro-basin tillage, or conservation tillage following an underdeveloped winter cover crop. Moreover, mulching could prove to be the ultimate conservation solution in organic farming, where weed control

constraints compromise the implementation of both conservation and micro-basin tillage. Nevertheless, the specific potential of mulching in NW Europe should be further investigated, considering the availability of suitable organic material, economic and agronomic feasibility, and effectiveness against surface flows. Herbicide-free innovative methods are also being developed to destroy weeds (or cover crops) while minimizing soil disturbance.

APPENDICES

Appendix A: Supplementary material to Chapter 2

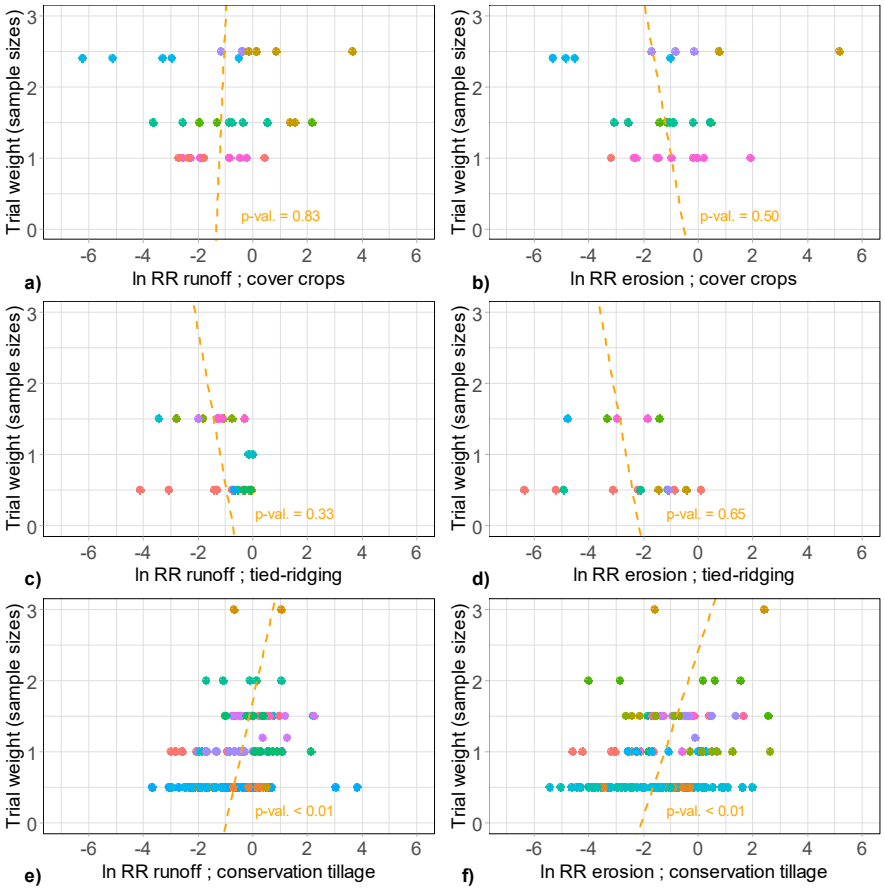


Figure A.1 : Funnel plots of the natural logarithm of the response ratio (see Eq. 2.1) for runoff or erosion, as function of trials weight (calculated based on sample sizes, see Eq. 2.3), for each conservation practice. Points in the same color are coming from the same study. The regression line between trial effect size and weight is drawn in dashed orange, with associated *p*-value for the slope significance.

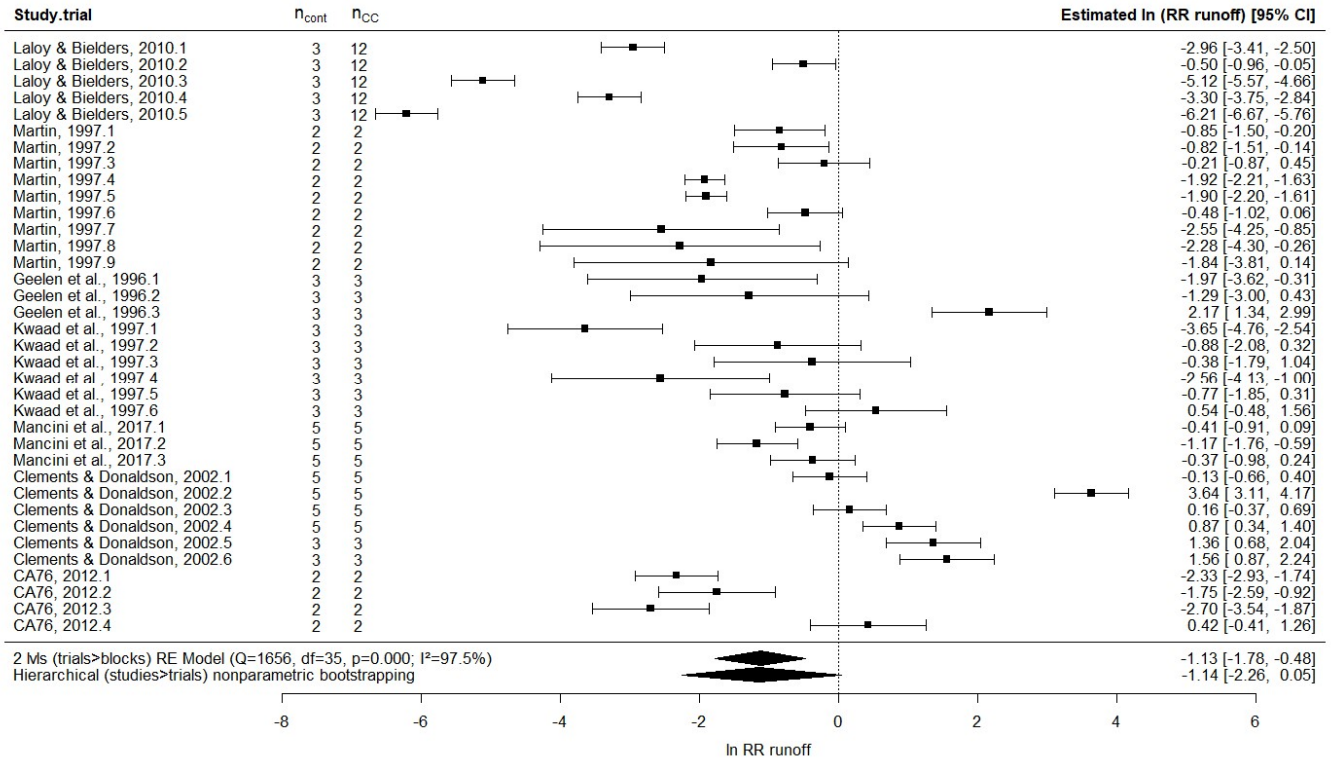


Figure A.2 : a) Forest plots of the meta-analysis for cover crops for the runoff outcome. Estimated effect size for each trial and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. Sample sizes are reported for each trial: n_{cont} and n_{CC} are the number of plot replicates for control and cover crop treatments respectively.

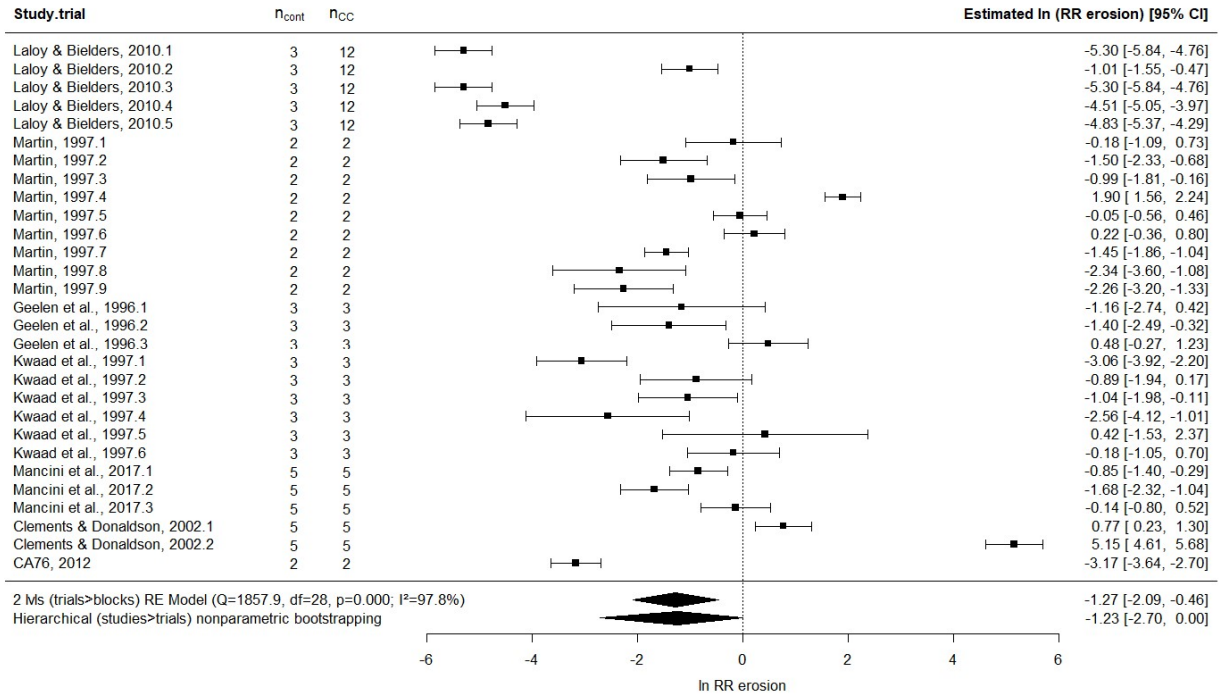


Figure A.2 (continued) : b) Forest plots of the meta-analysis for cover crop for the erosion outcome. Estimated effect size for each trial and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. Sample sizes are reported for each trial: n_{cont} and n_{CC} are the number of plot replicates for control and cover crop treatments respectively.

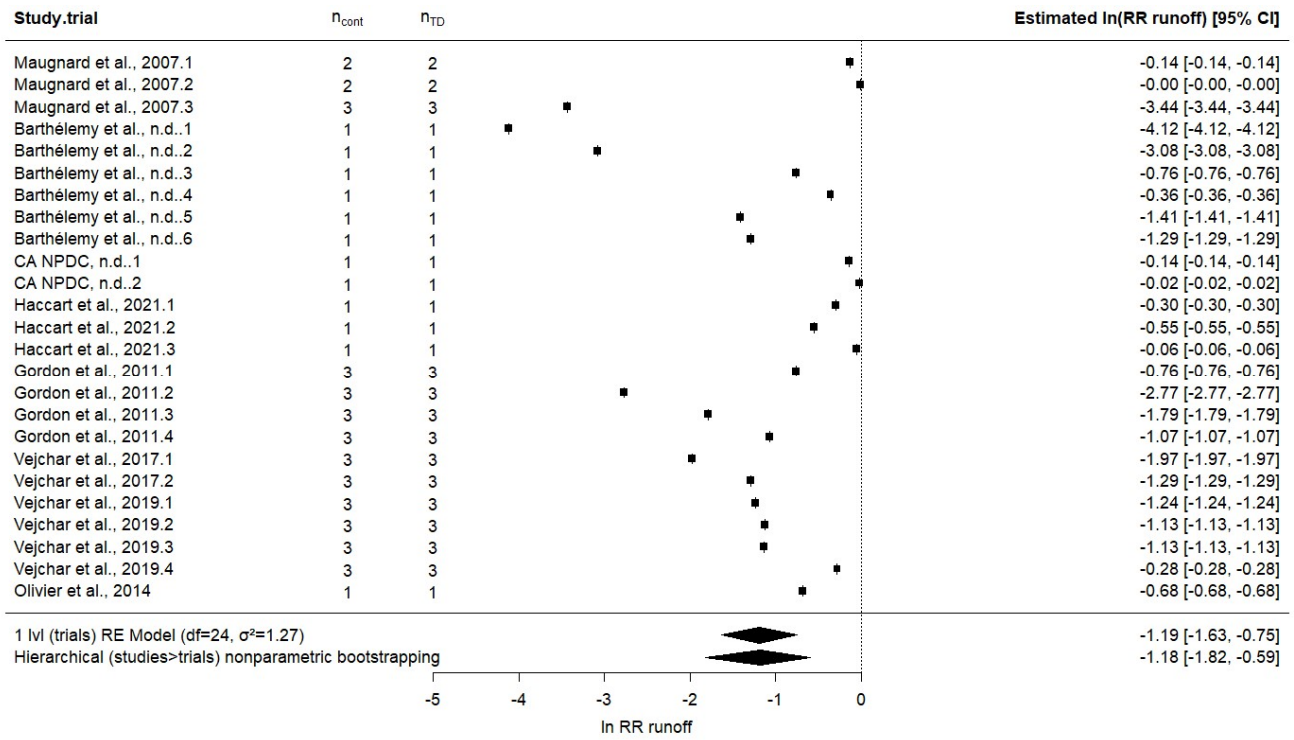


Figure A.2 (continued) : c) Forest plots of the meta-analysis for tied-ridging for the runoff outcome. Estimated effect size for each trial and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. No error bars are shown for each trial because inter-block variability has been neglected in the model (see Section 2.2.2.5). Sample sizes are reported for each trial: n_{cont} and n_{TD} are the number of plot replicates for control and tied-ridging treatments respectively.

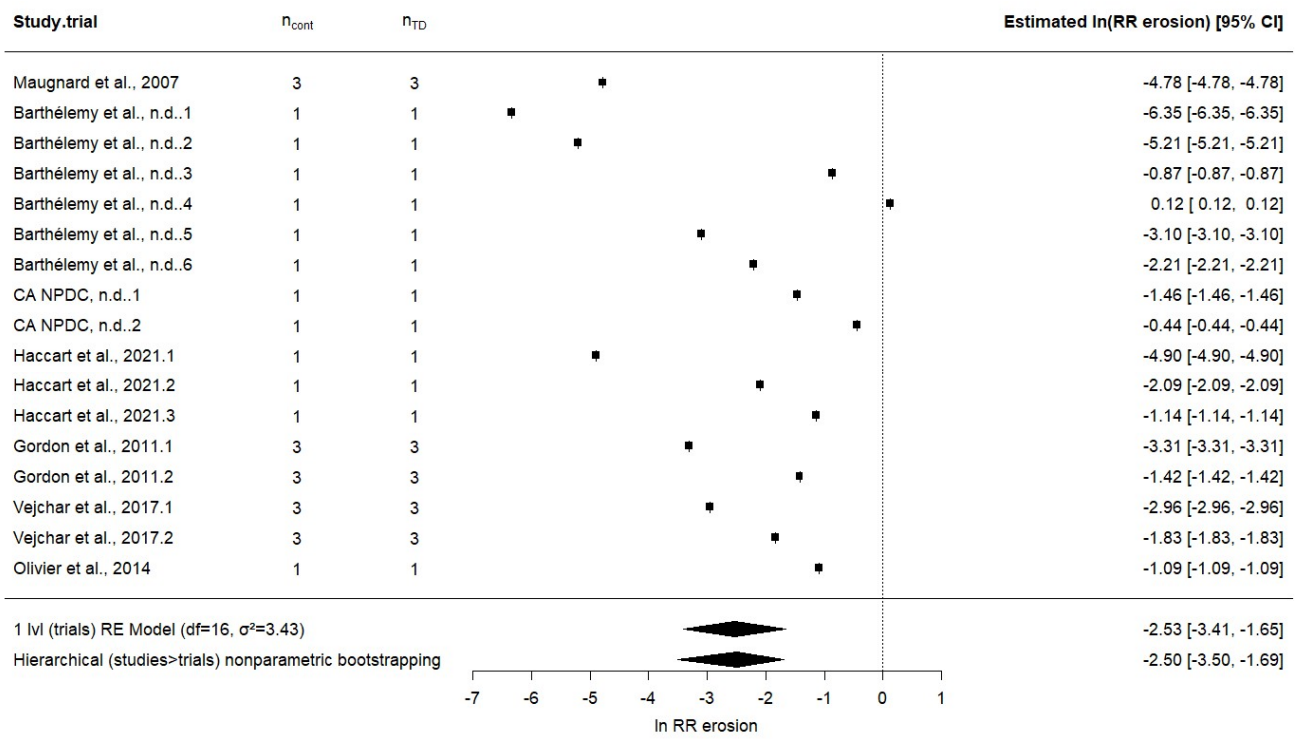


Figure A.2 (continued) : d) Forest plots of the meta-analysis for tied-ridging for the erosion outcome. Estimated effect size for each trial and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. No error bars are shown for each trial because inter-block variability has been neglected in the model (see Section 2.2.2.5). Sample sizes are reported for each trial: n_{cont} and n_{TD} are the number of plot replicates for control and tied-ridging treatments respectively.

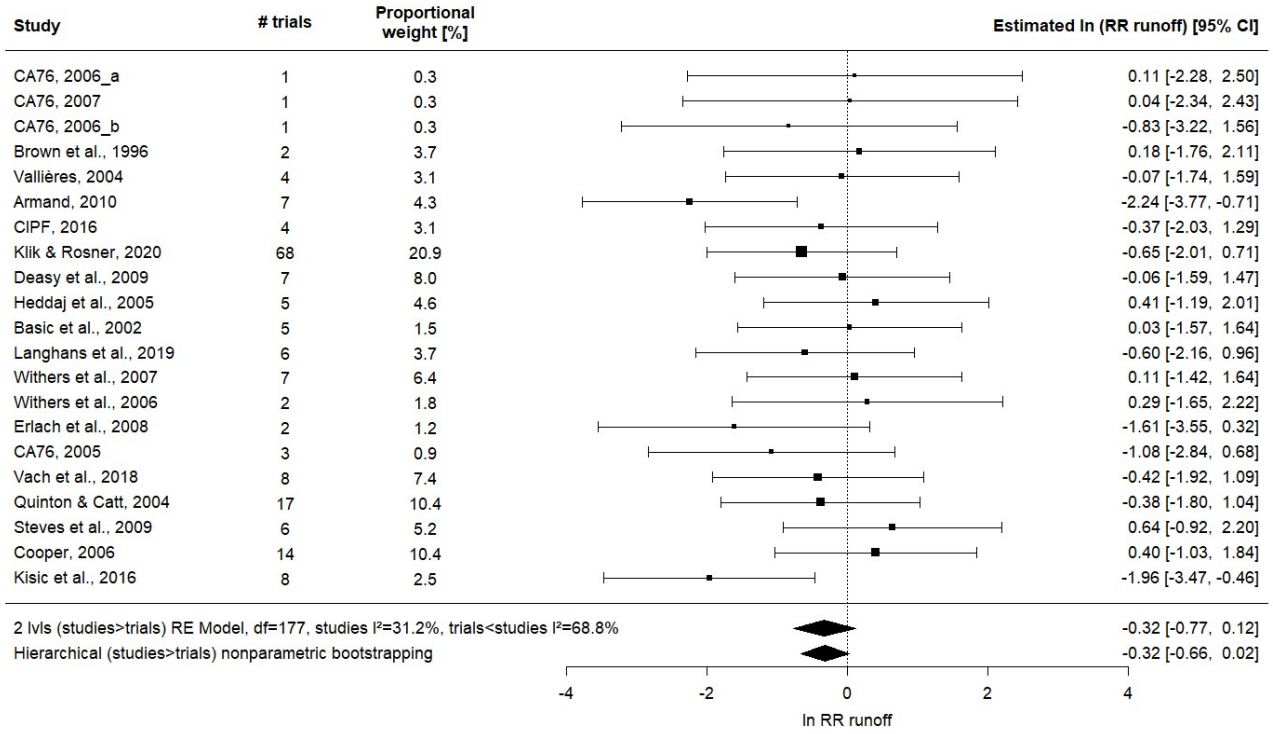


Figure A.2 (continued) : e) Forest plots of the meta-analysis for conservation tillage for the runoff outcome. Estimated effect size for each study, and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. Trials effect sizes were aggregated within studies for readability; the number of trials and the proportional weight in the calculation of the overall effect size (= number of trials weighted by the trial weight thus samples sizes, see Eq. 2.3, in relative format [%]) are shown for each study.

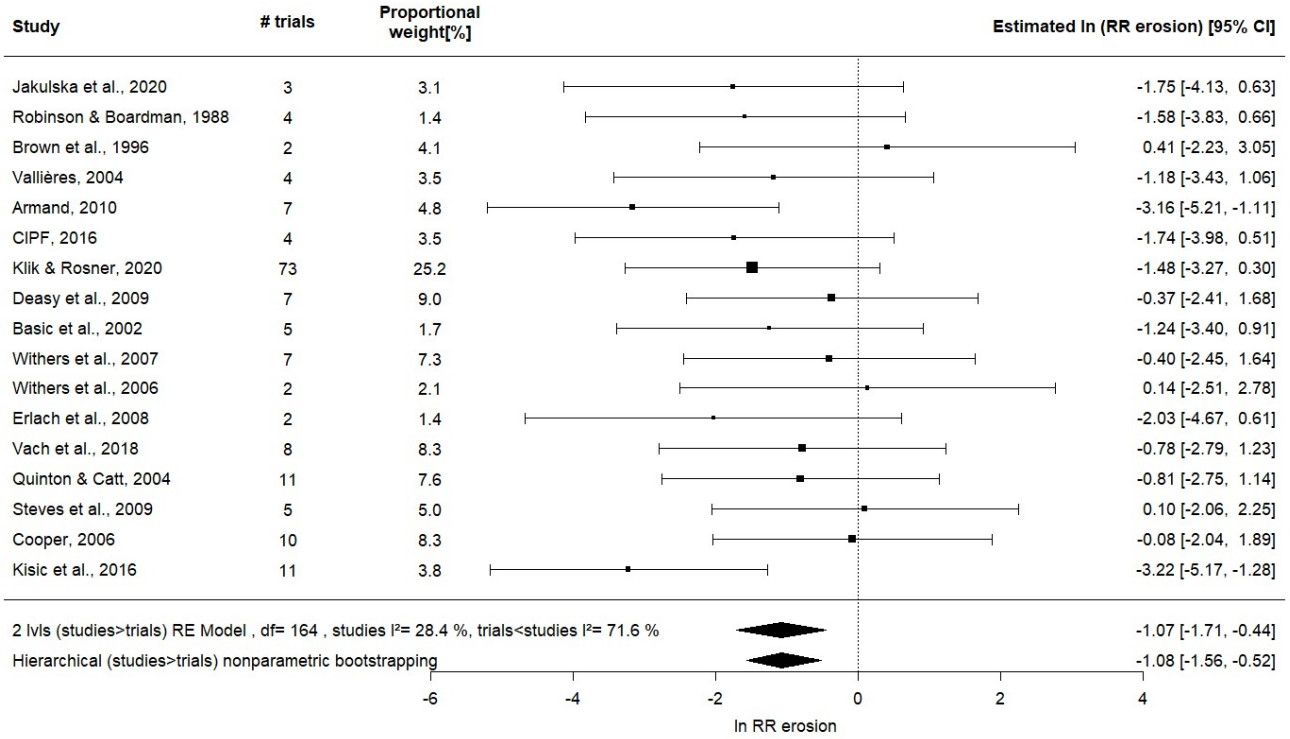


Figure A.2 (continued) : f) Forest plots of the meta-analysis for conservation tillage for the erosion outcome. Estimated effect size for each study, and estimated overall mean effect size computed by random effects model and hierarchical nonparametric bootstrapping are shown, associated with their 95% confidence interval. Trials effect sizes were aggregated within studies for readability; the number of trials and the proportional weight in the calculation of the overall effect size (= number of trials weighted by the trial weight thus samples sizes, see Eq. 2.3, in relative format [%]) are shown for each study.

Table A.1 : Exhaustive list of primary studies included in the meta-databases and their bibliographic reference(s)

N°	Conservation practice	Country	Reference(s)
1	Cover crops	Belgium	Laloy, E., & Bielders, C. L. (2010). Effect of Intercropping Period Management on Runoff and Erosion in a Maize Cropping System. Journal of Environmental Quality, 39(3), Article 3. https://doi.org/10.2134/jeq2009.0239
2	Cover crops	France	Martin, P. (1999). Reducing flood risk from sediment-laden agricultural runoff using intercrop management techniques in northern France. Soil & Tillage Research, 52, 233–245. Martin, P., Bissonnais, Y. L., Benkhadra, H., Ligneau, L., & Ouvry, J.-F. (1997). Mesures du ruissellement et de l'érosion diffuse engendrés par les pratiques culturales en Pays de Caux (Normandie)/Measurement of runoff and interrill erosion generated by cultivation practices in the Pays de Caux (Normandie). Géomorphologie : relief, processus, environnement, 3(2), 143–154.
3	Cover crops	The Netherlands	Geelen, P. M. T. M., Kwaad, F. J. P. M., van Mulligen, E. J., Wansink, A. G., van der Zijp, M., & van den Berg, W. (1996). The impact of soil tillage on crop yield, runoff and soil loss under various farming systems of maize and sugerbeet on loess soils (No. 211; p. 123). Proefstation voor de Akkerbouw en de Groenteteelt in de Vollegrond.
4	Cover crops	The Netherlands	Kwaad, F. J. P. M., van der Zijp, M., & van Dijk, P. M. (1998). Soil conservation and maize cropping systems on sloping loess soils in the Netherlands. Soil & Tillage Research, 46(1–2), Article 1–2. van Dijk, P. M., van der Zijp, M., & Kwaad, F. J. P. M. (1996). Soil Erodibility Parameters Under Various Cropping Systems of Maize. Hydrological Processes, 10(8), 1061–1067. https://doi.org/10.1002/(SICI)1099-1085(199608)10:8<1061::AID-HYP411>3.0.CO;2-V
5	Cover crops	UK	Mancini, A., Deeks, L. K., Simmons, R. W., Truckell, I., & Pawlett, M. (2017). Field trial to investigate the role of cover crops in reducing runoff and soil erosion in maize. Aspects of Applied Biology, 257–264.

N°	Conservation practice	Country	Reference(s)
6	Cover crops	UK	Clements, R. O., & Donaldson, G. (2002). Soil Erosion Control in Maize (R&D Technical Report No. P2-123/1). Environment Agency.
7	Cover crops	France	Ouvry, J.-F., Coufourier, N., Richet, J.-B., Lhériteau, M., Pivain, Y., & Martin, P. (2012). Maîtrise du ruissellement et de l'érosion des sols en Haute-Normandie. Expérimentations sur les pratiques culturales 2001-2010. Synthèse des résultats de ruissellement et d'érosion (Synthèse Des Résultats de Ruissellement et d'érosion, p. 36). Groupe Maîtrise du ruissellement par les pratiques culturales.
8	Tied-ridging	Belgium	Maugnard, A., & Bodson, B. (n.d.). AFPP – PROTECTION DES EAUX DE SURFACE CONTRE LES TRANSFERTS DIFFUS DE PRODUITS PHYTOSANITAIRES PARIS – 15 ET 16 NOVEMBRE 2007. 21.
9	Tied-ridging	Belgium	Barthélémy, J.-P., Fonder, N., Olivier, C., & Ver Eecke, P. (n.d.). Contrôle du ruissellement et de ses impacts en culture de pomme de terre en Wallonie.
10	Tied-ridging	France	Haccart, C., Carneau, A., Monchy, J., & Houillez, B. (2021). Pommes de terre. Expérimentations 2020 et références techniques. Chambre d'agriculture Nord-Pas-de-Calais. Chambre d'agriculture Nord-Pas de Calais. (n.d.). ESSAIS 2021. Impact des "inter-buttes" sur les phénomènes d'érosion et de ruissellement en culture de pommes de terre.
11	Tied-ridging	Canada	Gordon, R. J., VanderZaag, A. C., Dekker, P. A., De Haan, R., & Madani, A. (2011). Impact of modified tillage on runoff and nutrient loads from potato fields in Prince Edward Island. Agricultural Water Management, 98(12), 1782–1788. https://doi.org/10.1016/j.agwat.2011.07.007
12	Tied-ridging	Czech Republic	Vejchar, D., Stehlik, M., & Mayer, V. (2017). Influence of tied ridging technology on the rate of surface runoff and erosion in potato cultivation [PDF]. Agronomy Research, 15(5), 2207–2216. https://doi.org/10.15159/AR.17.039
13	Tied-ridging	Czech Republic	Vejchar, D., Vacek, J., Hájek, D., Bradna, J., Kasal, P., & Svobodová, A. (2019). Reduction of surface runoff on sloped agricultural land in potato cultivation in de-stoned soil. Plant, Soil and Environment, 65(No. 3), 118–124. https://doi.org/10.17221/736/2018-PSE
N°	**Conservation practice**	**Country**	**Reference(s)**

14	Tied-ridging	Belgium	Olivier, C., Goffart, J.-P., Baets, D., Xanthoulis, D., Fonder, N., Lognay, G., Barthélémy, J.-P., & Lebrun, P. (2014). Use of micro-dams in potato furrows to reduce erosion and runoff and minimise surface water contamination through pesticides. Communications in Agricultural and Applied Biological Sciences, 79(3), 513–524. Olivier, C., Goffart, J.-P., Baets, D., CropScience, B., Fonder, N., & Barthélemy, J.-P. (2012). 9.5 Le cloisonnement des interbuttes en culture de pomme de terre: Essai réalisé en collaboration avec Bayer CropScience (Huldenberg 2011).
15	Conservation tillage	Poland	Jaskulska, I., Romaneckas, K., Jaskulski, D., & Wojewódzki, P. (2020). A Strip-Till One-Pass System as a Component of Conservation Agriculture. Agronomy, 10(12), 2015. https://doi.org/10.3390/agronomy10122015
16	Conservation tillage	France	Chambre d'agriculture 76 (Seine-Maritime). (n.d.). Compte-rendu de l'essai blé mené au Lycée agricole d'Yvetot automne 2005-printemps 2006.
17	Conservation tillage	France	Chambre d'agriculture 76 (Seine-Maritime). (n.d.). Compte-rendu de l'essai labour / sans labour (betterave) Limésy—Printemps 2007.
18	Conservation tillage	France	Chambre d'agriculture 76 (Seine-Maritime). (n.d.). Compte-rendu de l'essai maïs mené au Lycée agricole d'Yvetot Printemps 2006.
19	Conservation tillage	UK	Robinson, D. A., & Boardman, J. (1988). Cultivation practice, sowing season and soil erosion on the South Downs, England: A preliminary study. The Journal of Agricultural Science, 110(1), 169–177. https://doi.org/10.1017/S0021859600079818
20	Conservation tillage	UK	Brown, L., Donaldson, G., Jordan, V. W. L., & Thornes, J. B. (1996). Effects and interactions of rotation, cultivation and agrochemical input levels on soil erosion and nutrient emissions. Aspects of Applied Biology, 47.
21	Conservation tillage	Canada	Vallières, D. (2004). Efficacité du mode de travail de sol et des bandes enherbées sur la diminution de la pollution diffuse [MSc thesis]. Université Laval.

N°	Conservation practice	Country	Reference(s)
22	Conservation tillage	France	Armand, R. (2010). Étude des états de surface du sol et de leur dynamique pour différentes pratiques de travail du sol. Mise au point d'un indicateur de ruissellement. 210. Armand, R. (2004). Observation, appréciation et quantification du ruissellement appliquées aux parcelles cultivées en techniques culturales sans labour (TCSL) [Master thesis]. Université Louis Pasteur.
23	Conservation tillage	Belgium	CIPF, UCLouvain, & Ateliers LSM. (2016). Evaluation de l'efficacité de techniques innovantes de préparation du sol dans la lutte contre le ruissellement et l'érosion en culture de maïs sur terres en pente (Rapports ERUISTOP, p. 36).
24	Conservation tillage	Austria	Klik, A., & Rosner, J. (2020). Long-term experience with conservation tillage practices in Austria: Impacts on soil erosion processes. Soil and Tillage Research, 203, 104669. https://doi.org/10.1016/j.still.2020.104669
25	Conservation tillage	UK	Deasy, C., Quinton, J. N., Silgram, M., Bailey, A. P., Jackson, B., & Stevens, C. J. (2009). Mitigation Options for Sediment and Phosphorus Loss from Winter-sown Arable Crops. Journal of Environmental Quality, 38(5), 2121–2130. https://doi.org/10.2134/jeq2009.0028
26	Conservation tillage	France	Heddadj, D., Gascuel-Odoux, C., & Cotinet, P. (2005). Mode de travail du sol, ruissellement et propriétés hydrodynamiques sur un dispositif expérimental de l'Ouest de la France. Etude et Gestion des Sols, 14.
27	Conservation tillage	Croatia	Basic, F., Kisic, I., Nestroy, O., Mesic, M., & Butorac, A. (2002). Particle Size Distribution (Texture) of Eroded Soil Material. Journal of Agronomy and Crop Science, 188(5), 311–322. https://doi.org/10.1046/j.1439-037X.2002.00567.x
28	Conservation tillage	Belgium	Langhans, C., Diels, J., Clymans, W., Van den Putte, A., & Govers, G. (2019). Scale effects of runoff generation under reduced and conventional tillage. CATENA, 176, 1–13. https://doi.org/10.1016/j.catena.2018.12.031
29	Conservation tillage	UK	Withers, P. J. A., Hodgkinson, R. A., Bates, A., & Withers, C. L. (2007). Soil cultivation effects on sediment and phosphorus mobilization in surface runoff from three contrasting soil types in England. Soil and Tillage Research, 93(2), 438–451. https://doi.org/10.1016/j.still.2006.06.004

N°	Conservation practice	Country	Reference(s)
30	Conservation tillage	UK	Withers, P. J. A., Hodgkinson, R. A., Bates, A., & Withers, C. M. (2006). Some effects of tramlines on surface runoff, sediment and phosphorus mobilization on an erosion-prone soil. Soil Use and Management, 22(3), 245–255. https://doi.org/10.1111/j.1475-2743.2006.00034.x
31	Conservation tillage	Germany	Erlach, F., & Entrup, N. L. (n.d.). Strategien zur Minderung diffuser Einträge von herbiziden Wirkstof- fen in Oberflächengewässer—Vergleich verschiedener Bestellver- fahren auf Parzellen mit Mais in Nordrhein-Westfalen. 10.
32	Conservation tillage	France	Chambre d'agriculture 76 (Seine-Maritime). (2005). Synthèse Guetteville 2005.
33	Conservation tillage	Czech Republic	Vach, M., Hlisnikovský, L., & Javůrek, M. (2018). The Effect of Different Tillage Methods on Erosion. Agriculture (Pol'nohospodárstvo), 64(1), 28–34. https://doi.org/10.2478/agri-2018-0003
34	Conservation tillage	UK	Quinton, J. N., & Catt, J. A. (2006). The effects of minimal tillage and contour cultivation on surface runoff, soil loss and crop yield in the long-term Woburn Erosion Reference Experiment on sandy soil at Woburn, England. Soil Use and Management, 20(3), 343–349. https://doi.org/10.1111/j.1475-2743.2004.tb00379.x
35	Conservation tillage	UK	Stevens, C. J., Quinton, J. N., Bailey, A. P., Deasy, C., Silgram, M., & Jackson, D. R. (2009). The effects of minimal tillage, contour cultivation and in-field vegetative barriers on soil erosion and phosphorus loss. Soil and Tillage Research, 106(1), 145–151. https://doi.org/10.1016/j.still.2009.04.009
36	Conservation tillage	UK	Cooper, S. (2006). The role of conservation soil management on soil and water protection at different spatial scales [PhD thesis]. Cranfield University.
37	Conservation tillage	Czech Republic	Kisic, I., Bogunovic, I., Birkás, M., Jurisic, A., & Spalevic, V. (2017). The role of tillage and crops on a soil loss of an arable Stagnic Luvisol. Archives of Agronomy and Soil Science, 63(3), 403–413. https://doi.org/10.1080/03650340.2016.1213815

Table A.2.1 : Equation and parameters of the cover crops meta-model for ln(RR runoff)

$$\ln(\mathrm{RR}_{\mathrm{runoff}}) = \ln\left(\frac{\mathrm{runoff}_{\mathrm{cover\ crops}}}{\mathrm{runoff}_{\mathrm{control}}}\right) = 0.8365 - 1.1217 * conshis - 3.5772 * 10^{-4} * maxcov^2 + 2.3809 * stub_{\mathrm{DS}} + 0.4603 * stub_{\mathrm{SS}}$$

Parameter	Estimate	Standard error	*p*-value
$intercept$	0.8365	0.6427	0.1931
$conshis$	-1.1217	0.2774	< 0.0001
$maxcov^2$	$-3.5772 * 10^{-4}$	$8.6506 * 10^{-5}$	< 0.0001
$stub_{\mathrm{DS}}$	2.3809	0.4695	< 0.0001
$stub_{\mathrm{SS}}$	0.4603	0.5180	0.3742

Table A.2.2 : Equation and parameters of the cover crops meta-model for ln(RR erosion)

$$\ln(\mathrm{RR}_{\mathrm{erosion}}) = \ln\left(\frac{\mathrm{erosion}_{\mathrm{cover\ crops}}}{\mathrm{erosion}_{\mathrm{control}}}\right) = 0.1099 - 1.0995 * conshis - 2.4831 * 10^{-4} * maxcov^2 + 2.1261 * stub_{\mathrm{DS}} + 1.0006 * stub_{\mathrm{SS}}$$

Parameter	Estimate	Standard error	*p*-value
$intercept$	0.1099	1.0458	0.9163
$conshis$	-1.0995	0.4622	0.0174
$maxcov^2$	$-2.4831 * 10^{-4}$	0.0001	0.0700
$stub_{\mathrm{DS}}$	2.1261	0.8531	0.0127
$stub_{\mathrm{SS}}$	1.0006	0.8753	0.2530

- The $\boldsymbol{intercept}$ is included in the model in all cases.
- $\boldsymbol{conshis}$ (*conservation history*) is the number of consecutive years of cover crops sowing. Its value should remain within the range of the original meta-dataset, i.e. from 0 to 2 years.
- $\boldsymbol{maxcov^2}$ is the square of the maximum vegetation cover [%] of the cover crop achieved during the intercropping season.
- $\boldsymbol{stub_{\mathrm{DS}}}$ and $\boldsymbol{stub_{\mathrm{SS}}}$ are dummy variables coding for the type of stubble tillage on control plots. If a deep stubble tillage (plough or deep cultivator) is performed, $stub_{\mathrm{DS}} = 1$ and $stub_{\mathrm{SS}} = 0$. If a shallow stubble (e.g. disk or tine cultivator) is executed on control plots, $stub_{\mathrm{DS}} = 0$ and $stub_{\mathrm{SS}} = 1$. If there is no stubble tillage in control plots (post-harvest condition, i.e. bare and untilled soil), both $stub_{\mathrm{DS}}$ and $stub_{\mathrm{SS}}$ are equal to 0. A value of 1 for both variables is not a valid combination.

Table A.3.1 : Equation and parameters of the conservation tillage meta-model for ln(RR runoff)

$$\ln(\mathrm{RR}_{\mathrm{runoff}}) = \ln\left(\frac{\mathrm{runoff}_{\text{conservation tillage}}}{\mathrm{runoff}_{\mathrm{control}}}\right) =$$
$$0.3613 - 0.0845 * conshis - 0.7222 * croptype + 0.0732 * tillop - 0.2761 * soiltext + 0.2963 * dep_{\mathrm{mid}} - 0.4642 * type_{\mathrm{pass}}$$

Parameter	Estimate	Standard error	*p*-value
$intercept$	0.3613	0.4151	0.3840
$conshis$	-0.0845	0.0217	< 0.0001
$croptype$	-0.7222	0.2305	0.0017
$tillop$	0.0732	0.1325	0.5807
$soiltext$	-0.2761	0.2864	0.3349
dep_{mid}	0.2963	0.2561	0.2473
$type_{\mathrm{pass}}$	-0.4642	0.3226	0.1502

Table A.3.2 : Equation and parameters of the conservation tillage meta-model for ln(RR erosion)

$$\ln(\mathrm{RR}_{\mathrm{erosion}}) = \ln\left(\frac{\mathrm{erosion}_{\text{conservation tillage}}}{\mathrm{erosion}_{\mathrm{control}}}\right) =$$
$$-0.7543 - 0.0751 * conshis - 0.8510 * croptype + 0.1750 * tillop - 0.7063 * soiltext + 0.4629 * dep_{\mathrm{mid}} + 0.2773 * preccrop$$

Parameter	Estimate	Standard error	*p*-value
$intercept$	-0.7543	0.7428	0.3099
$conshis$	-0.0751	0.0375	0.0453
$croptype$	-0.8510	0.3917	0.0298
$tillop$	0.1750	0.2168	0.4195
$soiltext$	-0.7063	0.5137	0.1691
dep_{mid}	0.4629	0.4042	0.2520
$preccrop$	0.2773	0.3530	0.4322

- The ***intercept*** is included in the model in all cases.
- ***conshis*** (*conservation history*) is the number of consecutive years of conservation tillage, i.e. since ploughing stopped. Its value should remain within the range of the original meta-dataset, i.e. from 1 to 25 years.

- $\boldsymbol{croptype}$ is a dummy variable coding for crop type. $croptype = 0$ for winter crops, and = 1 for spring crops.
- $\boldsymbol{tillop}$ is the total number of tillage passes in the conservation tillage system, including sowing. Its value should remain in the range of the original meta-dataset, i.e. from 1 to 5 tillage passes.
- $\boldsymbol{soiltext}$ is a dummy variable coding for soil texture. $soiltext = 0$ for silty(-clay) loams, and 1 for sandy loams.
- $\boldsymbol{preccrop}$ is a dummy variable coding for the crop type of the preceding crop. $preccrop = 0$ if the preceding crop type was a winter crop, and 1 if it was a spring crop. This variable is only present in the erosion meta-model.
- $\boldsymbol{dep_{\mathrm{mid}}}$ is a dummy variable coding for the maximum tillage depth in the conservation tillage system. If the maximum tillage depth is intermediate - between 5 and 15cm - $dep_{\mathrm{mid}} = 1$. For deep tillage (>15cm) OR for direct drilling (<5cm), $dep_{\mathrm{mid}} = 0$.
- $\boldsymbol{type_{\mathrm{pass}}}$ is a dummy variable coding for the maximum tillage intensity in the conservation tillage system. If the most intensive tillage operation is performed using a passive implement (e.g. tine, disk cultivator), $type_{\mathrm{pass}} = 1$. If it uses a powered tool (e.g. powered rotary harrow, rototiller) OR if no tillage is performed (direct drilling), $type_{\mathrm{pass}} = 0$. This variable is only present in the runoff meta-model.

NB: For predictions on direct drilling systems, the following variables are logically constrained to $tillop = 1$, $dep_{\mathrm{mid}} = 0$, $type_{\mathrm{pass}} = 0$.

Appendix B: Supplementary material to Chapter 4

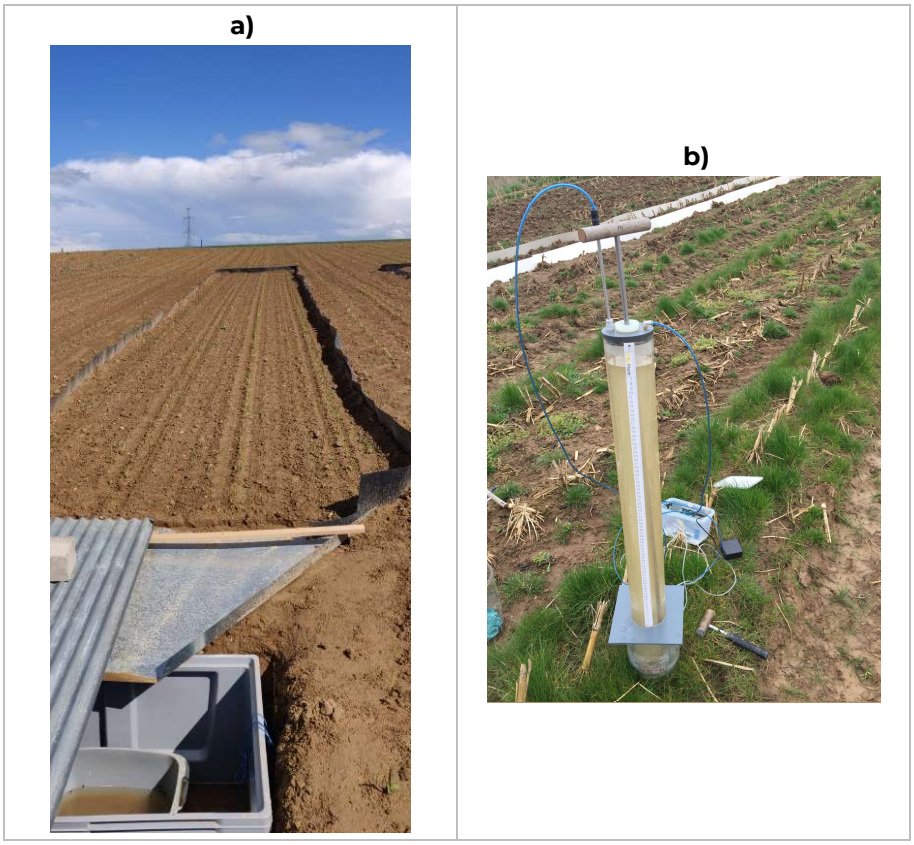


Figure B.1 : Pictures from field trials. a) Experimental plot, at Ittre in 2022. b) Automated single ring infiltrometer used for Ksat measurements.

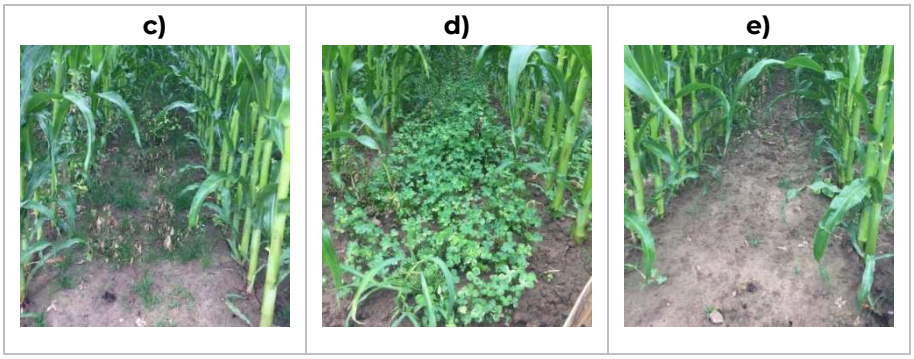


Figure B.1 (continued) : Pictures from undersowing field trials at Itt22. c) Undersown fescue. d) Undersown clover. e) Control maize.

f)



g)



h)



i)



Figure B.1 (continued) : Pictures from undersowing field trials at Ottignies. f) Adapted seeding unit for undersowing. g) High weed pressure in 2022 in an undersown plot. h) Undersown fescue in December 2022 (note prominent wheel tracks resulting from the maize harvest and heterogeneous vegetation cover). i) Undersown clover in December 2022.

j)





k)



l)



Figure B.1 (continued) : Pictures from strip-tillage trials. j) Strip-tiller used in the trials. k) Soil surface state at Itt2 trial for strip-tillage (left) and control (right) treatment. l) Rill erosion in strip-tilled maize rows at Ottignies in 2021.

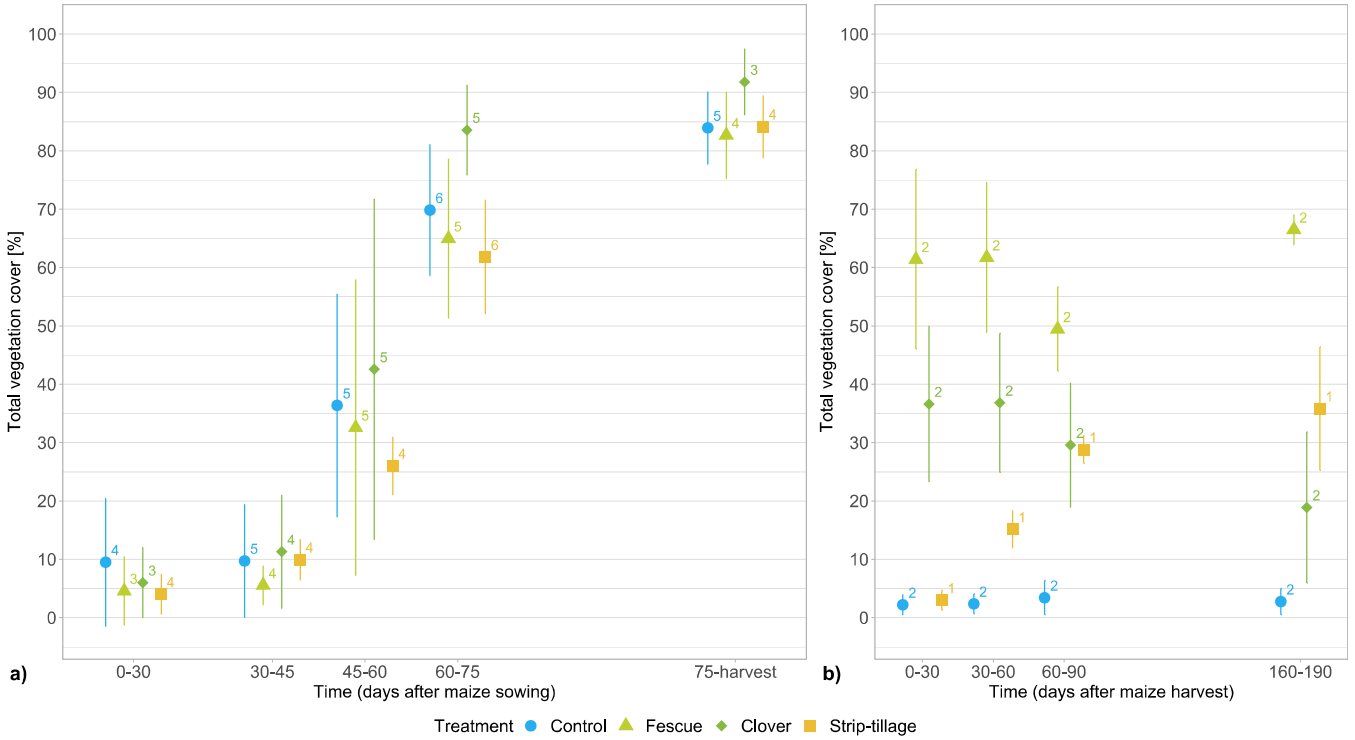


Figure B.2 : Total vegetation cover per time period and per treatment, averaged across all plots from all trials. Error bars are standard deviations. The number next to each point is the number of trials included, i.e. with cover data available for this period, out of the 7 trials. a) during the maize cropping season, as function of the number of days after maize sowing. b) during the winter season, as function of the number of days after maize harvest (see, winter monitoring of Ottignies in 2021-2022 and 2022-2023).

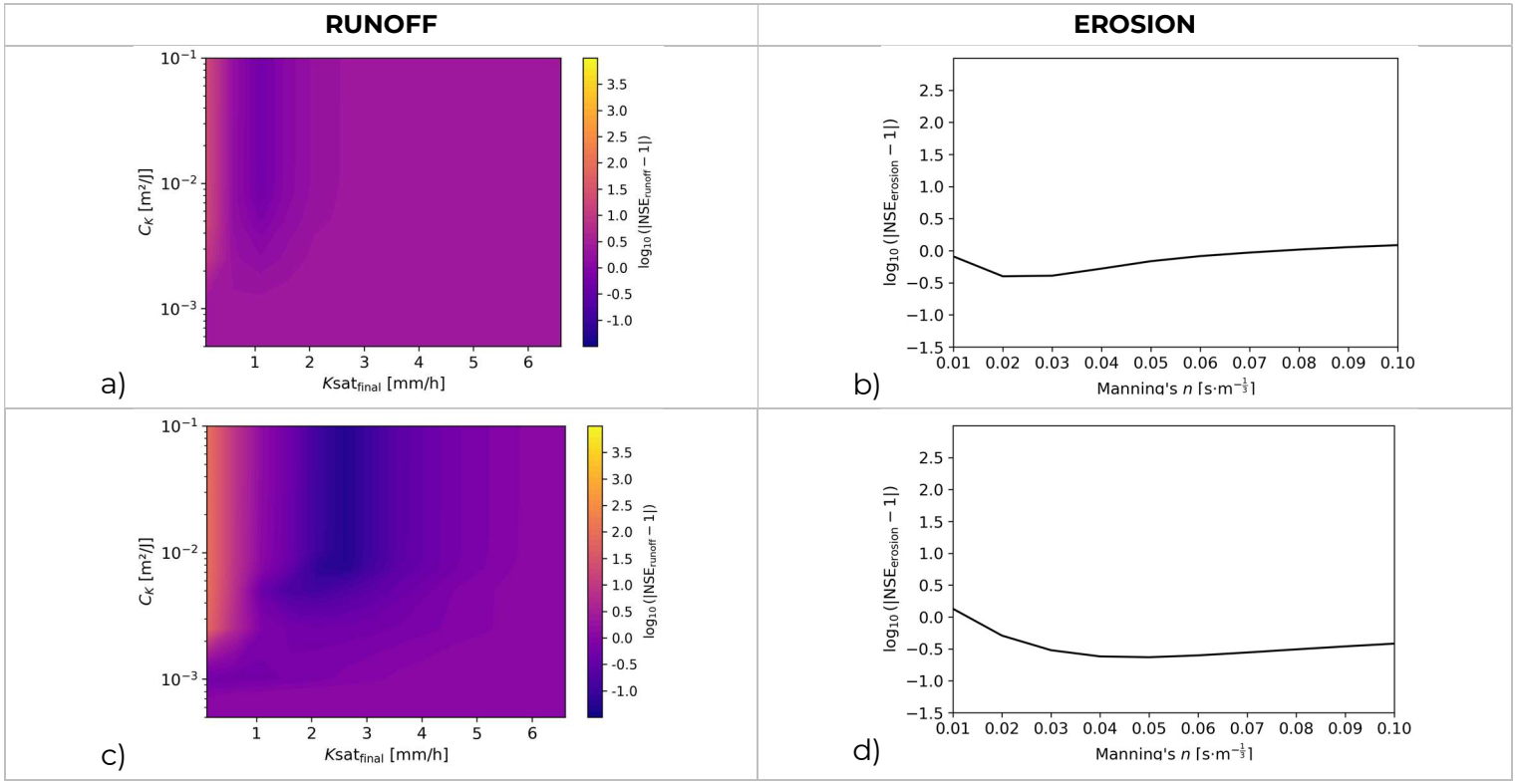


Figure B.3 : Inverse modelling optimization through global screening of the parameters $Ksat_{\mathbf{final}}$, C_K and Manning' s n, for each trial, for the calibration on runoff and erosion outcomes, for the control treatment. Note that model's accuracy is reported as $\log_{10}(|\mathrm{NSE} - 1|)$ for better vizualization, lower values meaning a better fit (higher NSE). a) Reb21 for the runoff calbration. b) Reb21 for the erosion calibration. c) Bonl21 for runoff. d) Bonl21 for erosion. See Table 4.1 for trials abbreviations.

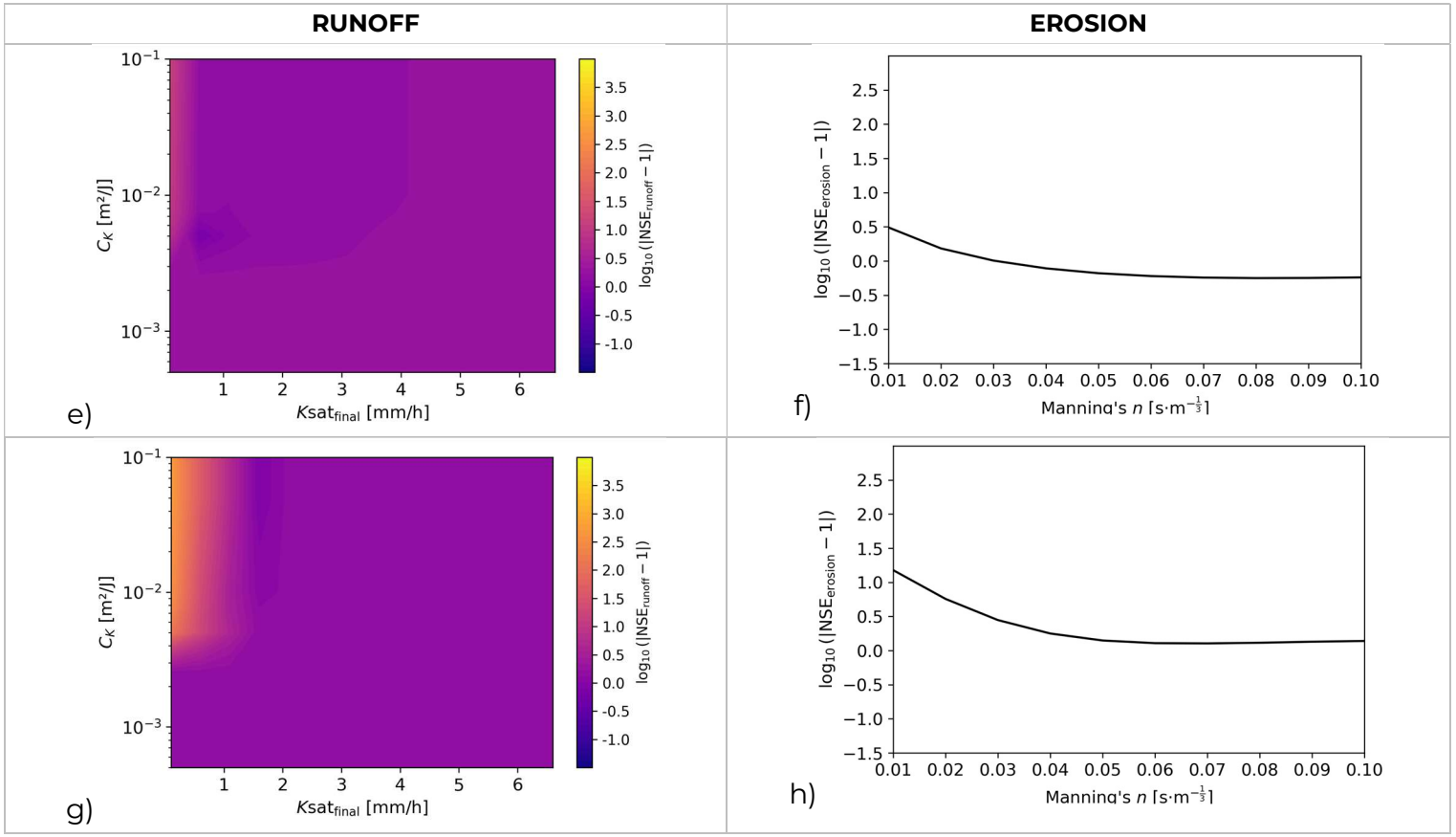


Figure B.3 (continued) : e) Ott21 for runoff. f) Ott21 for erosion. g) Ott22 for runoff. h) Ott22 for erosion.

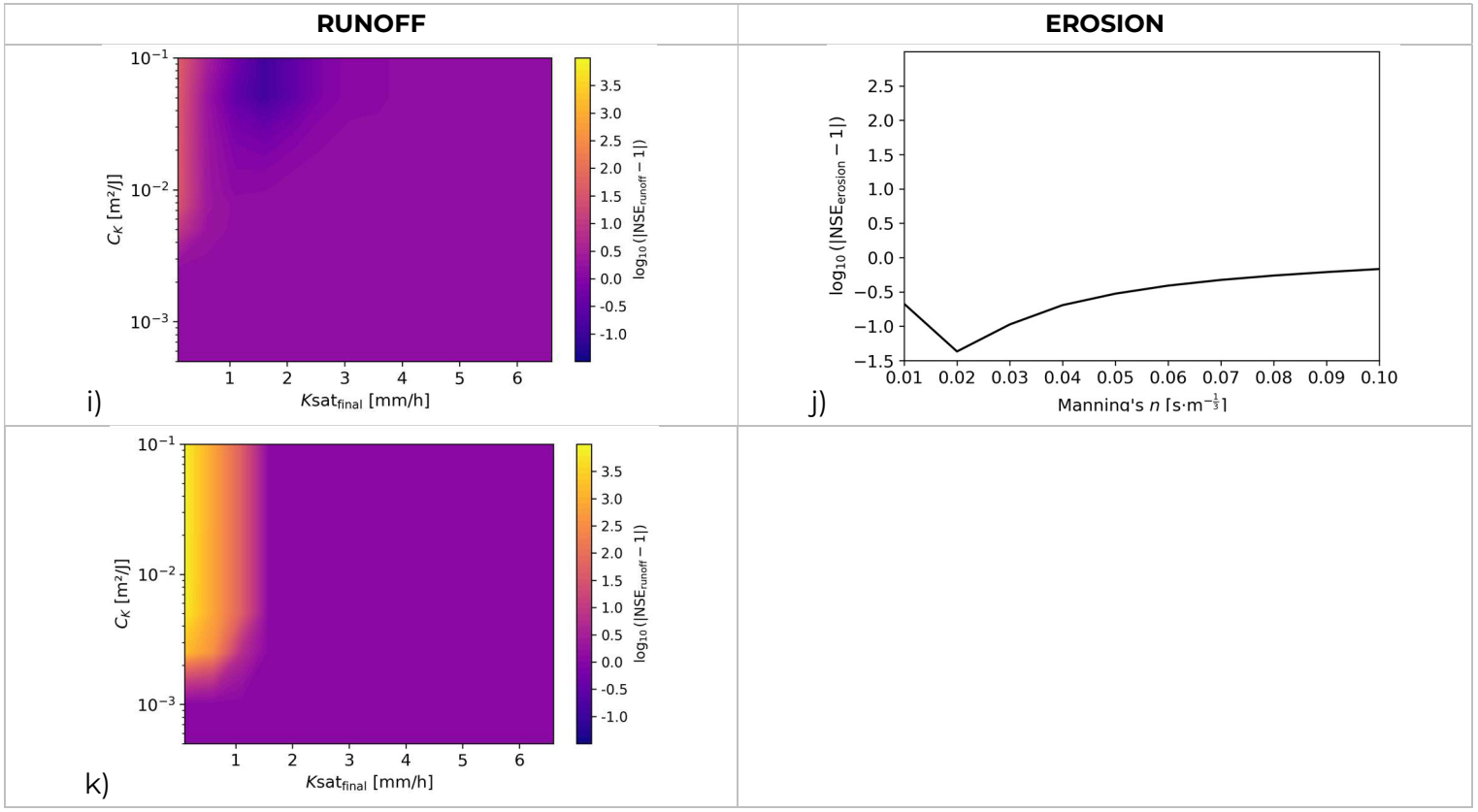


Figure B.3 (continued) : i) Itt22 for runoff. j) Itt22 for erosion. k) Both23 for runoff. No response function is provided for erosion calibration at Both23 because no Manning's *n* value could be calibrated for that trial (see Section 4.2.3.2).

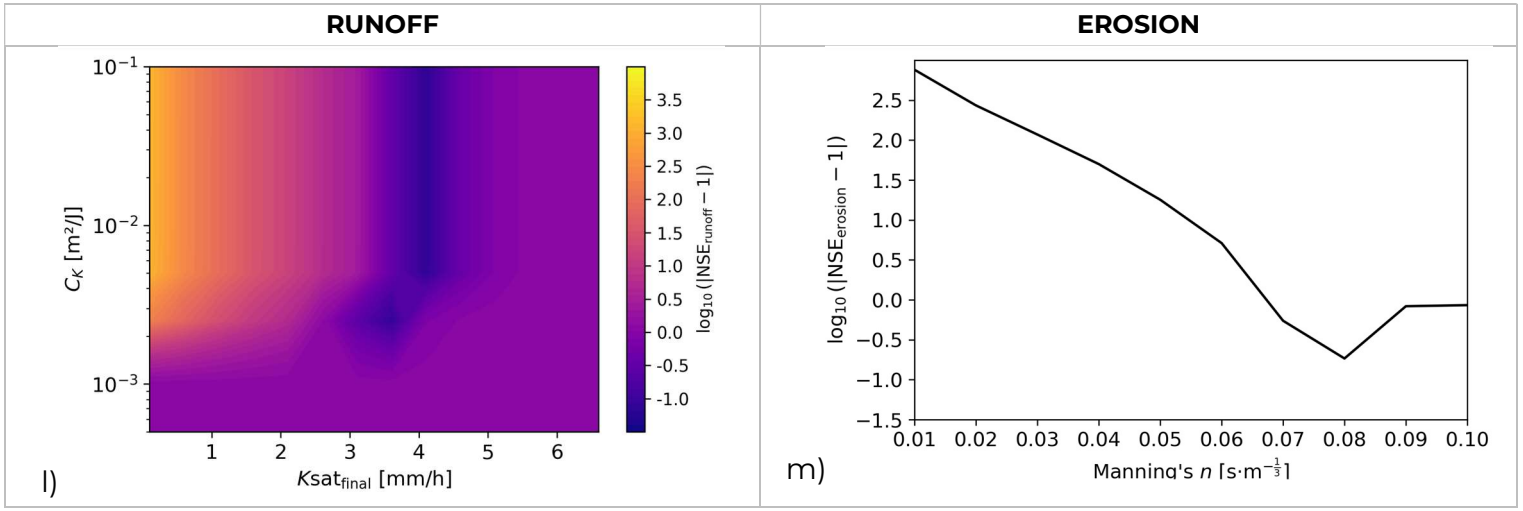


Figure B.3 (continued) : l) RM23 for runoff. m) RM23 for erosion.

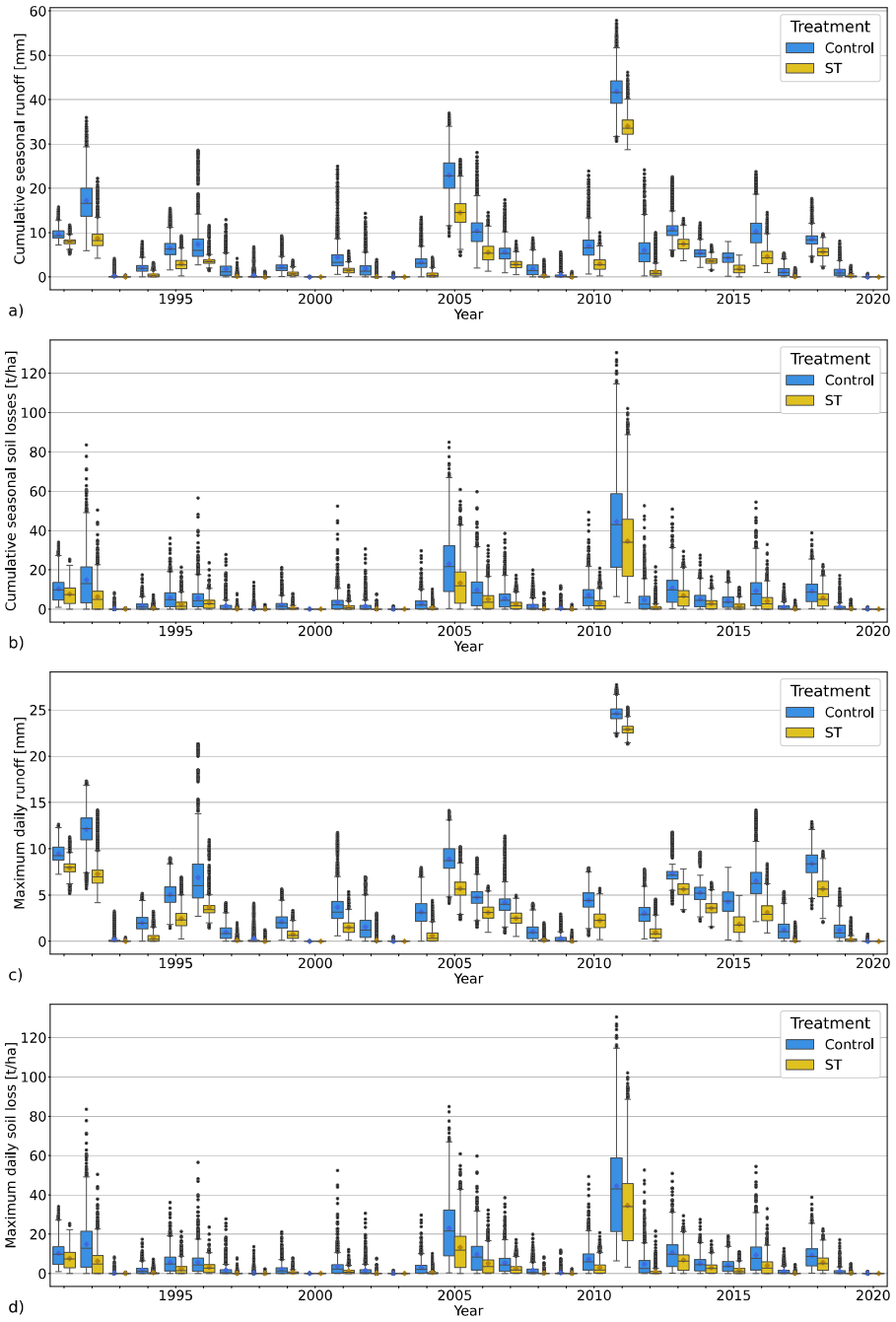


Figure B.4 : Flow outputs from the scenario analysis as function of simulated cropping season and treatment, all other factors combined. a) Total seasonal runoff. b) Total seasonal erosion. c) Maximum daily runoff. d) Maximum daily soil loss.

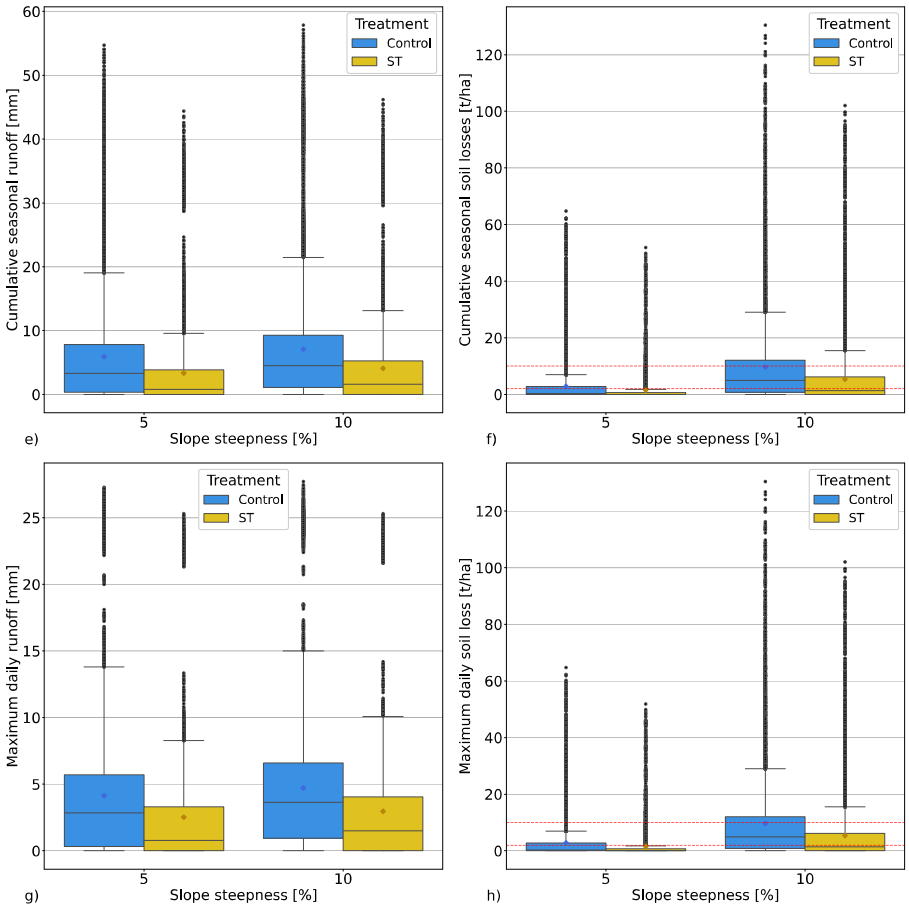


Figure B.4 (continued) : Flow outputs from the scenario analysis as function of simulated slope steepness and treatment, all other factors combined. e) Total seasonal runoff. f) Total seasonal erosion. g) Maximum daily runoff. h) Maximum daily soil loss. For f) and h), the two horizontal red dashed lines refers to the critical thresholds for "sustainable" erosion rate of 2 t/ha/year (Panagos et al., 2020) and 10 t/ha/year (Morgan, 2005).

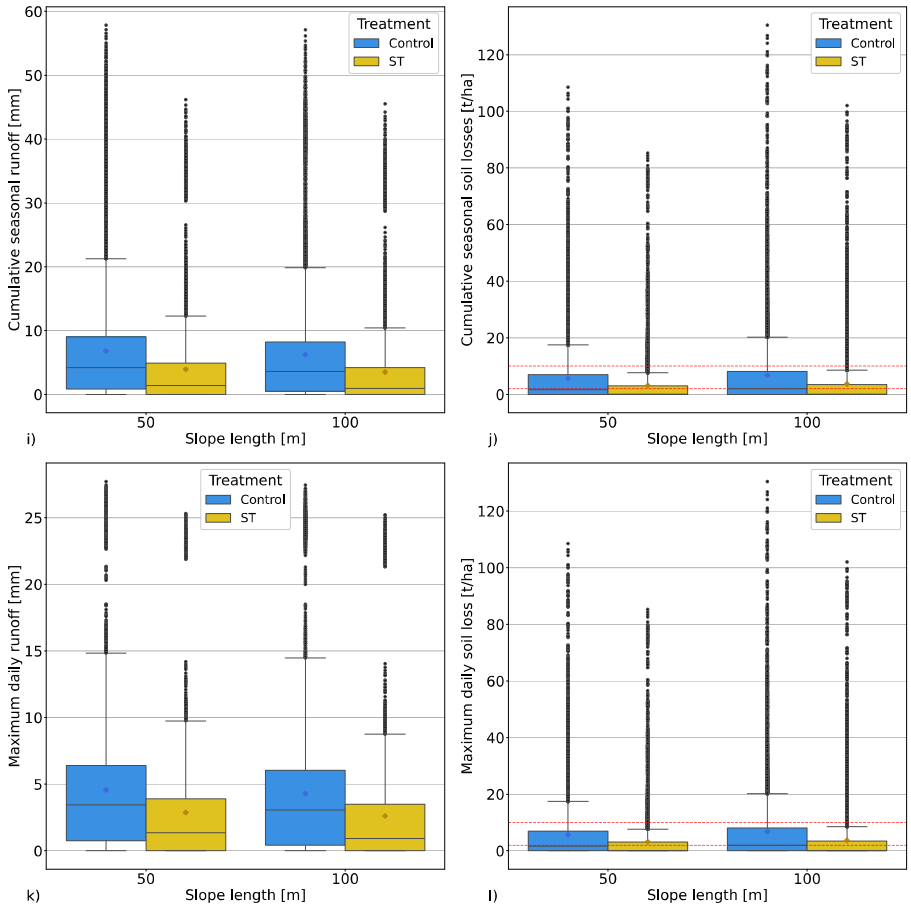


Figure B.4 (continued) : Flow outputs from the scenario analysis as function of simulated slope length and treatment, all other factors combined. i) Total seasonal runoff. j) Total seasonal erosion. k) Maximum daily runoff. l) Maximum daily soil loss. For j) and l), the two horizontal red dashed lines refers to the critical thresholds for "sustainable" erosion rate of 2 t/ha/year (Panagos et al., 2020) and 10 t/ha/year (Morgan, 2005).

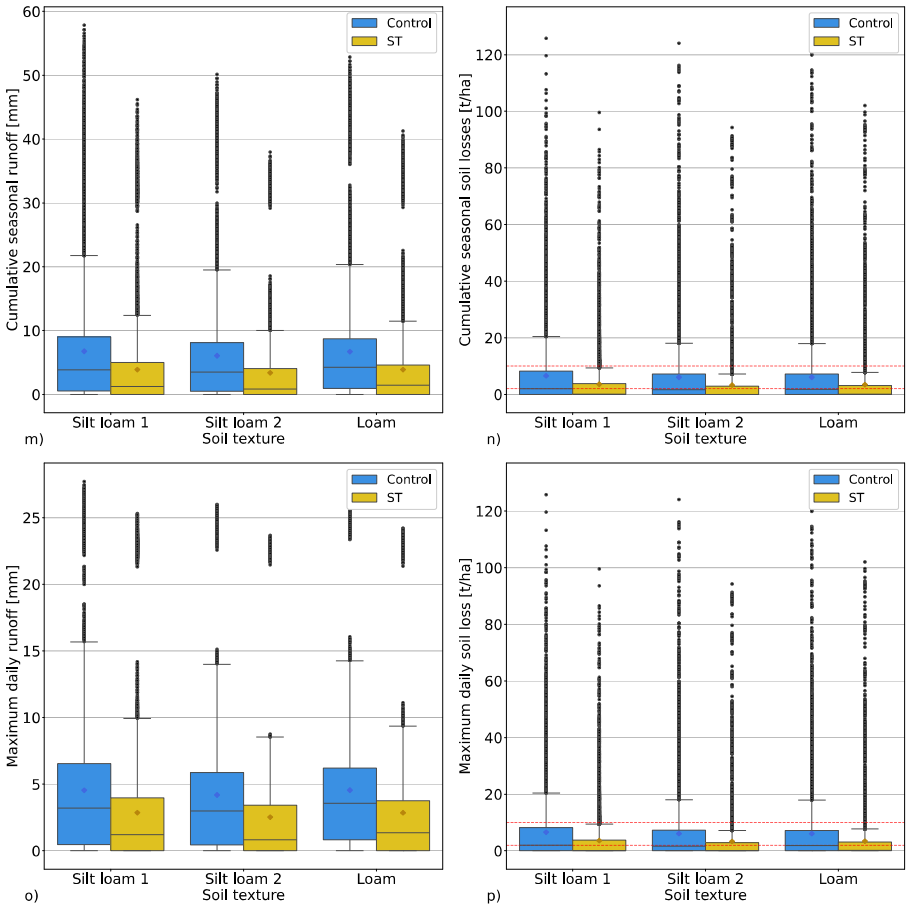


Figure B.4 (continued) : Flow outputs from the scenario analysis as function of simulated soil texture and treatment, all other factors combined. m) Total seasonal runoff. n) Total seasonal erosion. o) Maximum daily runoff. p) Maximum daily soil loss. For n) and p), the two horizontal red dashed lines refers to the critical thresholds for "sustainable" erosion rate of 2 t/ha/year (Panagos et al., 2020) and 10 t/ha/year (Morgan, 2005).

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