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Soil pitting mitigates runoff, erosion and pesticide surface losses in maize crops in the Belgian loess belt

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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.

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., 2007) 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 likely 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

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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., 2012; 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 finally 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.

2. Materials and methods

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 1; Fig. 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, 2022a; 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 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 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. 1.

Erosion plots (54–75 m^2; Table 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

Table 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 = m^2 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, 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	Slope [%]	Soil text. (Sa-Si-Cl)	OC cont. [%]	Trial year (s)	Plot size [m*m]	# exp. blocks	Sow. & harv. date	Tot. rain. [mm]	# meas.	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[1] (Hen)	10	Sandy loam (31–55–14)	1.6	2012	25 * 3 = 75	2	28/6 – 24/10	240	10	24/7: 300 g/ha flufenacet (wat. & sed.)	91	8 (4)	Soignies (8.8)
Virginal[1] (Vir)	11	Silt loam (unknown)	Unknown	2013	25 * 3 = 75	3	6/5 – 4/10	306	5	No	N/A	N/A	Tubize (4.9)
Ottignies[1] (Ott)	16	Silt loam (12–70–18)	1.0	2013	25 * 3 = 75	3	26/4 – 16/10	272	5	No	N/A	N/A	Louvain-la-Neuve (LLN) (3.2)
				2014	24 * 3 = 72	3	30/4 – 16/10	403	10	10/6: 350 g/ha flufenacet & 583 g/ha therbutylazine (wat.)	92	15 (17)	
				2015		3	16/4 – 8/10	254	3	9/6: 350 g/ha flufenacet & 583 g/ha therbutylazine (wat.)	63	18 (13)	
				2016		3	4/5 – 10/10	238	3	7/6: 350 g/ha flufenacet & 80 g/ha mesotrione (wat. & sed.)	86	22 (16)	
				2017		1 (mean of 3)	25/4 – 2/10	219	2	No	N/A	N/A	
Ittre[1] (Itt)	11	Sandy loam (32–52–16)	1.1	2015	24 * 3 = 72	3	28/4 – 3/10	233	6	9/6: 350 g/ha flufenacet & 583 g/ha therbutylazine (wat.)	76	9 (9)	Tubize (8.0)
				2016	25 * 3 = 75	3	5/5 – 3/10	289	7	6/6: 350 g/ha flufenacet & 75 g/ha mesotrione (wat. & sed.)	87	29 (1)	
				2017		1 (mean of 3)	28/4 – 21/9	215	4	No	N/A	N/A	
Huldenberg (steep slope)[2] (Hul_SS)	16	Silt loam (11–74–15)	Unknown	2013	24 * 3 = 72	1	unknown	318	9	No	N/A	N/A	Wavre (9.8)
Huldenberg (medium slope)[2] (Hul_MS)	9			2013	24 * 3 = 72	1	unknown	318	9	No	N/A	N/A	
				2018		1	unknown	214	13	No	N/A	N/A	
				2019	18 * 3 = 54	1	unknown	385	10	No	N/A	N/A	Nossegem (10.0)

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

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 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, 2022b) 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. 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. 2 and Table 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

Fig. 1. (a) Adapted roller from the seeding unit used to form the pits between maize rows (left) and subsequent pattern at plot scale (right). (b) Source: CIPF.

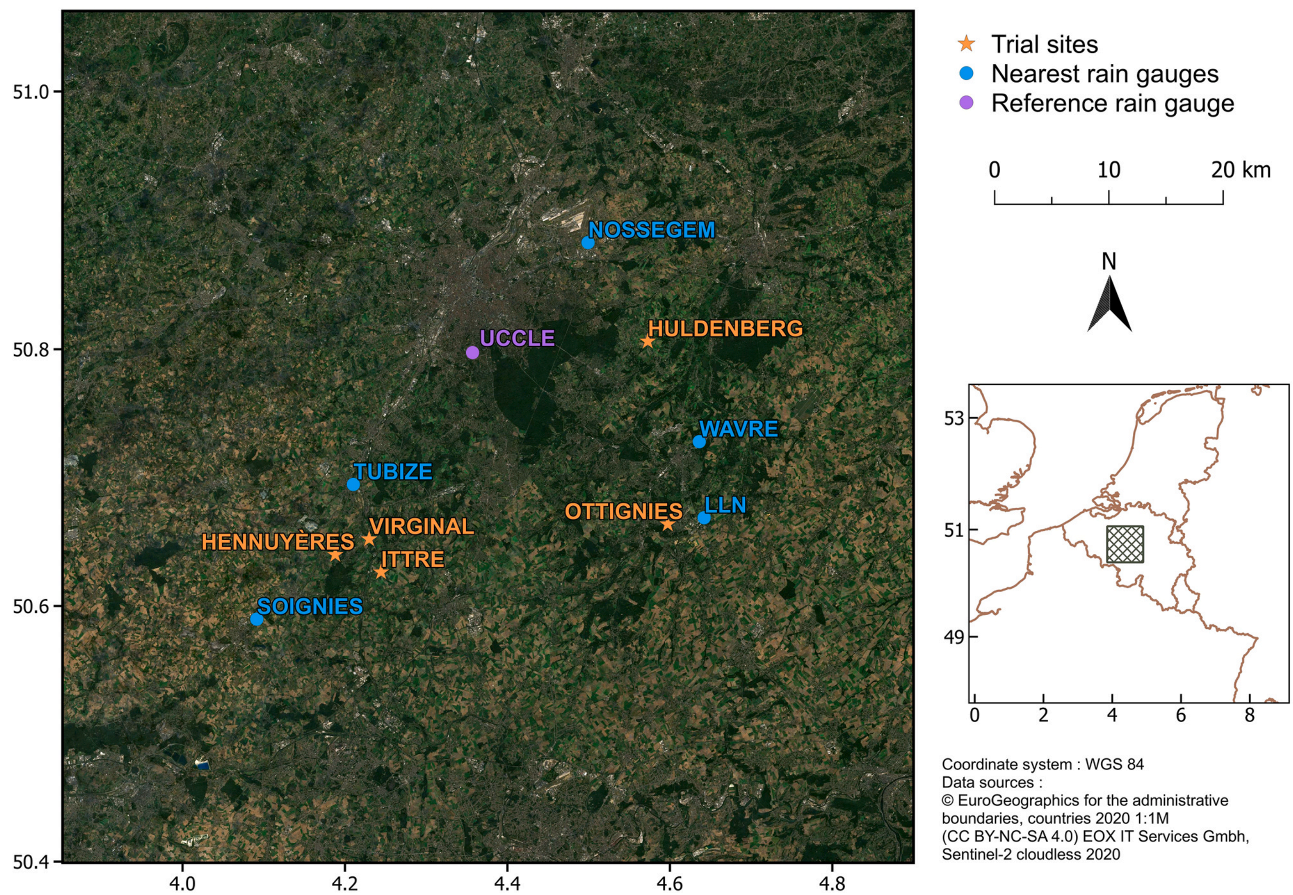

Fig. 2. Map of trial sites and rain gauges considered in the dataset. “Nearest rain gauges” are used for rainfall estimates at trial sites (see Section 2.1 and Table 1), while the “reference rain gauge” is used for assessment of the long-term representativeness of the rainfall intensities recorded during the trials (see Section 2.1).

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 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.

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 1).

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. 1 and 2):

$$Q = \frac{(P - \lambda S)^2}{P + (1 - \lambda)S} \quad if \quad P > I_a = \lambda S;\ else \quad Q = 0 \tag{1}$$

$$S = \frac{25\,400}{\mathrm{CN}} - 254 \tag{2}$$

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 0.01–0.2 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.

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). 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).

$$Runoff_j \approx \frac{\left(\sum_{j=1}^{n} Runoff_j\right) * Imax_j}{\sum_{j=1}^{n} Imax_j} \tag{3}$$

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.

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. 4):

$$\ln RR_{runoff_{i,j,k}} = \ln\left(\frac{Runoff_{soil\quad pitting_{i,j,k}}}{Runoff_{control_{i,j,k}}}\right) \tag{4}$$

where $Runoff_{treatment_{i,j,k}}$ 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) 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 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. Results

3.1. Seasonal runoff, soil and pesticides losses

Mean measured seasonal runoff, erosion rates, and pesticide surface losses are shown for each site*year in Figs. 3, 4 and 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.

Table 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 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).

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. 6). A λ value of 0.05 suits better to our field data; the residual sum of squares (RSS) with $\lambda = 0.05$ are more than 3 times lower compared to RSS values with $\lambda = 0.2$. 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 at least one order of magnitude smaller than the measured runoff.

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)

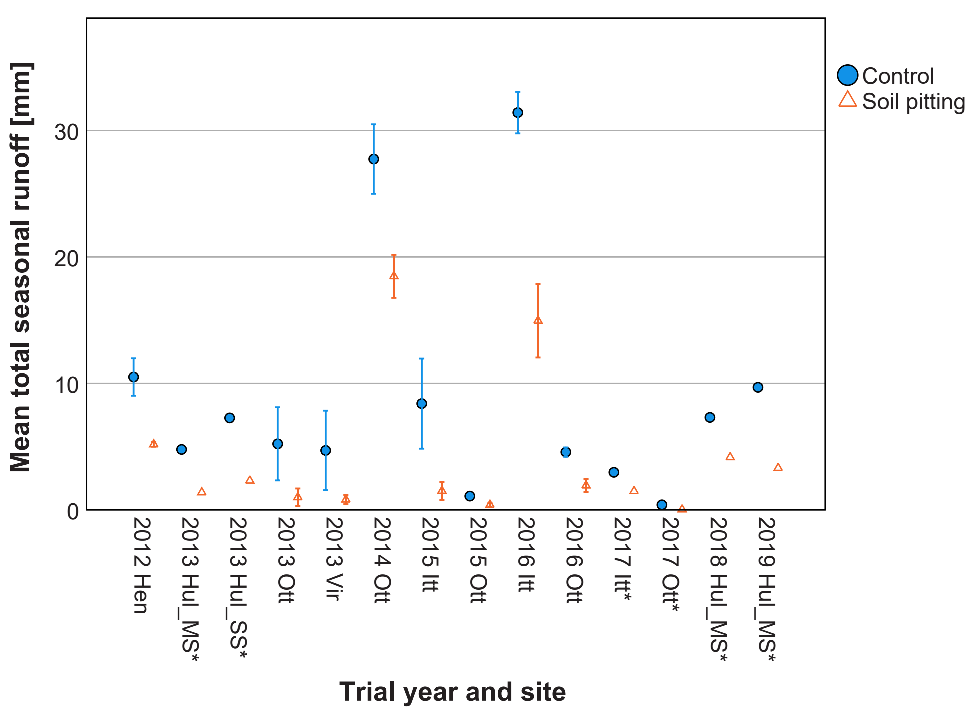

Fig. 3. Mean (across plot replicates) total seasonal runoff for all soil pitting trial years and sites. Error bars correspond to standard deviation. Site*years marked with an asterisk (*) correspond to trials without replicates (only 1 plot per treatment) thus without error bars.

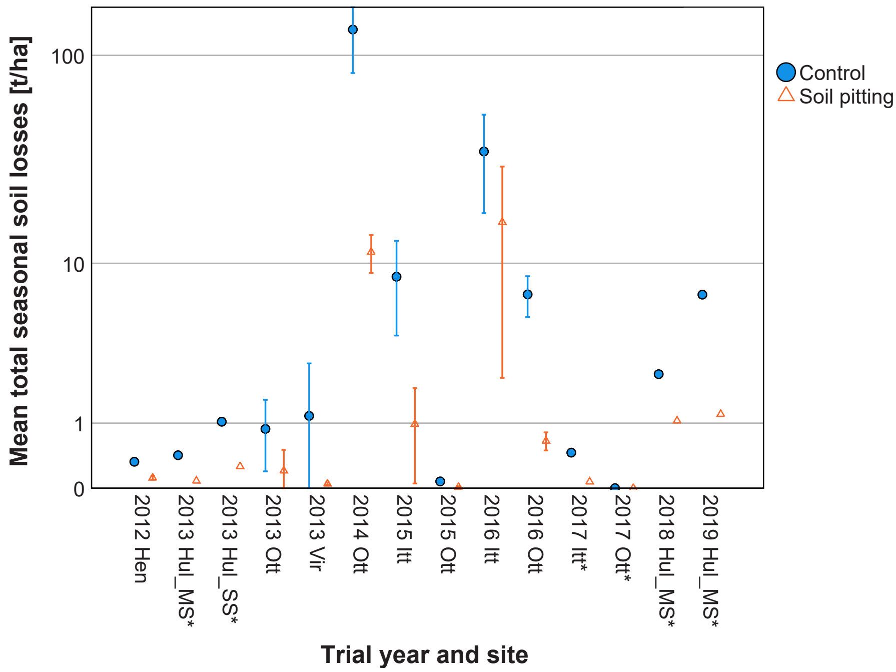

Fig. 4. Mean (across plot replicates) total seasonal soil losses for all soil pitting trial years and sites. Logarithmic scale for y-axis. Error bars correspond to standard deviation. Site*years marked with an asterisk (*) correspond to trials without replicates (only 1 plot per treatment), thus without error bars.

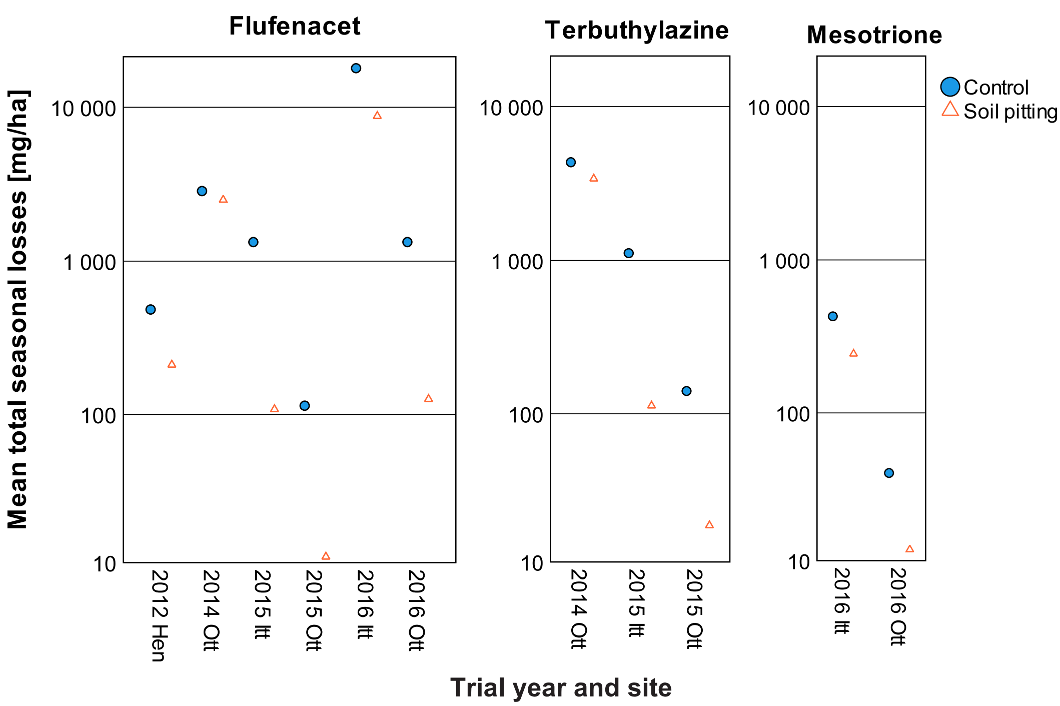

Fig. 5. Mean (across plot replicates) total seasonal pesticide surface losses for all soil pitting trial years and sites for which pesticides were monitored. Logarithmic scale for y-axis. The figure does not feature any error bars because only means of replicates for each treatment were available.

(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.

4. Discussion

Overall, the mean seasonal runoff and erosion rates in control plots

Table 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 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"

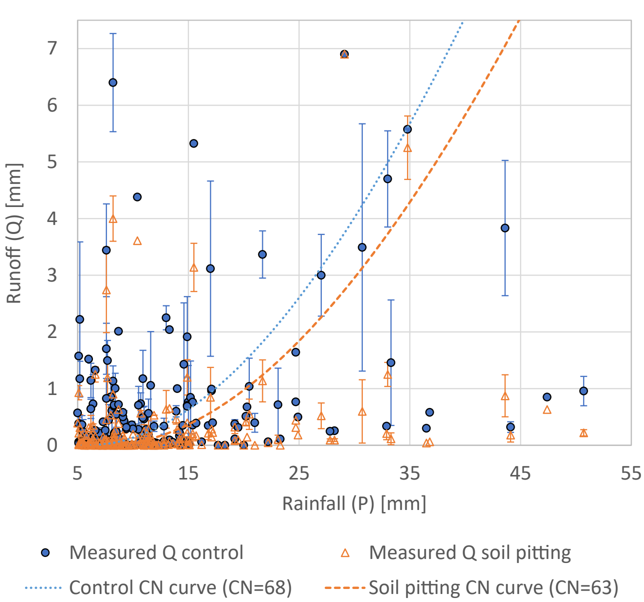

Fig. 6. Measured rainfall-runoff data and curves for median-fitted curve numbers ($\lambda = 0.05$), for control and soil pitting treatments in maize crop. Measured runoff values are means across plot replicates. Error bars correspond to standard deviations. Points without error bars are single plot (no replicates) experiments. Only rainfall events > 5 mm were considered (see Section 2.1).

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 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., 2012). 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 likely 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 3.1) 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 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 a greater dilution effect once the flow reaches 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 $\lambda = 0.2$). 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 calibrated in this article are typically 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 take into account 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).

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 rainfall characteristics and 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).

5. Conclusions

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.

Declaration of Competing Interest

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

Data Availability

Data will be made available on request.

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References

Belis, J., 2018. Etude du combiné de semis ERuiStop dans la lutte contre l'érosion en culture de maïs (MSc thesis). Université catholique de Louvain, Louvain-La-Neuve.

Boardman, J., Poesen, J., 2006. Soil Erosion in Europe. Wiley–Blackwell, Chichester, England.

Bresson, É., Laprise, R., Paquin, D., Thériault, J.M., de Elía, R., 2017. Evaluating the ability of CRCM5 to simulate mixed precipitation. Atmos. Ocean 55, 79–93. https://doi.org/10.1080/07055900.2017.1310084.

Casamitjana, M., D'Haene, K., Loaiza Usuga, J.C., Gabriels, D., De Neve, S., 2009. Influence of tillage systems on soil structural properties, in: No-Till Farming: Effects on Soil, Pros and Cons and Potential. Nova Science Publishers.

CIPF (Centre indépendant de promotion fourragère), Unpublished results. 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 (2012 – 2017). Louvain-La-Neuve, Belgium.

CIPF asbl , 2023. CIPF - Centre indépendant de promotion fourragère. URL ⟨https://cipf.be/fr⟩ (accessed 3.22.23).

Cooper, C.M., 1993. Biological effects of agriculturally derived surface water pollutants on aquatic systems—a review. J. Environ. Qual. 22, 402–408. https://doi.org/10.2134/jeq1993.00472425002200030003x.

CRA-W, 2022. Réseau pameseb. ⟨https://agromet.be/fr/reseau-pameseb/⟩ (accessed 21 October 2022).

Dabney, S.M., Moore, M.T., Locke, M.A., 2006. Integrated management of in-field, edge-of-field, and after-field buffers1. J. Am. Water Resour. Assoc. 42, 15–24. https://doi.org/10.1111/j.1752-1688.2006.tb03819.x.

Darly, S., Torre, A., Olivier, C., 2020. Smart land use for smart rural development: The case of rural and peri-urban regions in Europe, in: Torre, A., Corsi, S., Steiner, M., Wallet, F., Westlund, H., Smart Development for Rural Areas. Routledge, London.

European Commission, 1997. Council Directive 97/57/EC of 22 September 1997 establishing Annex VI to Directive 91/414/EEC concerning the placing of plant protection products on the market, OJ L.

European Food Safety Authority (EFSA), 2011. Conclusion on the peer review of the pesticide risk assessment of the active substance terbuthylazine. EFSA J. 9, 1969. https://doi.org/10.2903/j.efsa.2011.1969.

European Food Safety Authority (EFSA), 2016. Peer review of the pesticide risk assessment of the active substance mesotrione. EFSA J. 14, 4419. https://doi.org/10.2903/j.efsa.2016.4419.

Evrard, O., Bielders, C.L., Vandaele, K., van Wesemael, B., 2007. Spatial and temporal variation of muddy floods in central Belgium, off-site impacts and potential control measures. Catena 70, 443–454. https://doi.org/10.1016/j.catena.2006.11.011.

Farr, T.G., Rosen, P.A., Caro, E., Crippen, R., Duren, R., Hensley, S., Kobrick, M., Paller, M., Rodriguez, E., Roth, L., Seal, D., Shaffer, S., Shimada, J., Umland, J., Werner, M., Oskin, M., Burbank, D., Alsdorf, D., 2007. The shuttle radar topography mission. Rev. Geophys. 45 https://doi.org/10.1029/2005RG000183.

Feng, C., Wang, H., Lu, N., Tu, X.M., 2013. Log transformation: application and interpretation in biomedical research. Stat. Med. 32, 230–239. https://doi.org/10.1002/sim.5486.

European Commission, 2003. Review report for the active substance flufenacet (No. 7469/VI/98-Final). European Commission Health & Consumer Protection Directorate E1 Plant health.

Gentile, A.R., Barceló-Cordón, S., Van Liedekerke, M., 2009. Soil country analyses: Belgium (No. EUR 23959 EN/2–2009), JRC Scientific and Technical Reports. European Commission. Joint Research Centre. Institute for Environment and Sustainability, LU.

Gillijns, K., Govers, G., Poesen, J., Mathijs, E., Bielders, C., 2005. Erosion des sols en Belgique, état de la question (No. Cahier n°10). KINT - IRGT, Bruxelles.

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. Agric. Water Manag. 98, 1782–1788. https://doi.org/10.1016/j.agwat.2011.07.007.

Harrison, X.A., Donaldson, L., Correa-Cano, M.E., Evans, J., Fisher, D.N., Goodwin, C.E. D., Robinson, B.S., Hodgson, D.J., Inger, R., 2018. A brief introduction to mixed effects modelling and multi-model inference in ecology. PeerJ 6, e4794. https://doi.org/10.7717/peerj.4794.

Hawkins, R.H., Ward, T.J., Woodward, D.E., Mullem, J.A.V., 2009. Curve Number Hydrology: State of the Practice. American Society of Civil Engineers, Reston.

Heikkinen, J., Ketoja, E., Nuutinen, V., Regina, K., 2013. Declining trend of carbon in Finnish cropland soils in 1974-2009. Glob. Change Biol. 19, 1456–1469. https://doi.org/10.1111/gcb.12137.

Hitzfeld, K.L., Müller, A., Knillmann, S., Weisner, O., Pickl, C., Liess, M., Vormeier, P., Liebmann, L., Reemtsma, T., 2022. small streams - big problems. German event-driven monitoring reveals alarming pesticide pollution and regulatory deficiencies. Presented at the International Interdisciplinary Conference on Land Use and Water Quality. Agriculture and the Environment, Maastricht, the Netherlands.

Huber, A., Bach, M., Frede, H.G., 2000. Pollution of surface waters with pesticides in Germany: modeling non-point source inputs. Agric. Ecosyst. Environ. 80, 191–204. https://doi.org/10.1016/S0167-8809(00)00145-6.

Jones, O.R., Baumhardt, R.L., 2007. Furrow dikes. In: Trimble, S.W. (Ed.), Encyclopedia of Water Science. Marcel Dekker, New York, pp. 317–320.

Jones, O.R., Clark, R.N., 1987. Effects of furrow dikes on water conservation and dryland crop yields. Soil Sci. Soc. Am. J. 51, 1307–1314. https://doi.org/10.2136/sssaj1987.03615995005100050039x.

Jones, O.R., Stewart, B.A., 1990. Basin tillage. Soil Tillage Res. 18, 249–265. https://doi.org/10.1016/0167-1987(90)90064-K.

Jones, R.J.A., Spoor, G., Thomasson, A.J., 2003. Vulnerability of subsoils in Europe to compaction: a preliminary analysis. Soil Tillage Res 73, 131–143. https://doi.org/10.1016/S0167-1987(03)00106-5.

Karlsson, A.S., Lesch, M., Weihermüller, L., Thiele, B., Disko, U., Hofmann, D., Vereecken, H., Spielvogel, S., 2020. Pesticide contamination of the upper Elbe River and an adjacent floodplain area. J. Soils Sediment. 20, 2067–2081. https://doi.org/10.1007/s11368-020-02571-w.

Koricheva, J., Gurevitch, J., Mengersen, K., 2013. Handbook of meta-analysis in ecology and evolution, Princeton University Press. ed. Princeton, New Jersey. Woodstock, Oxfordshire.

Lahmar, R., 2010. Adoption of conservation agriculture in Europe. Land Use Policy 27, 4–10. https://doi.org/10.1016/j.landusepol.2008.02.001.

Laloy, E., Bielders, C.L., 2008. Plot scale continuous modelling of runoff in a maize cropping system with dynamic soil surface properties. J. Hydrol. 349, 455–469. https://doi.org/10.1016/j.jhydrol.2007.11.033.

Laloy, E., Bielders, C.L., 2010. Effect of intercropping period management on runoff and erosion in a maize cropping system. J. Environ. Qual. 39, 1001–1008. https://doi.org/10.2134/jeq2009.0239.

Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., Verhoeven, J.T.A., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environ. Evid. 5, 9 https://doi.org/10.1186/s13750-016-0060-0.

Ling, L., Yusop, Z., 2014. A micro focus with macro impact: Exploration of initial abstraction coefficient ratio (λ) in Soil Conservation Curve Number (CN) methodology, in: IOP Conference Series: Earth and Environmental Science. Presented at the 8th International Symposium of the Digital Earth (ISDE8), p. 012121. https://doi.org/10.1088/1755–1315/18/1/012121.

Liu, Y., Xin, Y., Xie, Y., Wang, W., 2019. Effects of slope and rainfall intensity on runoff and soil erosion from furrow diking under simulated rainfall. Catena 177, 92–100. https://doi.org/10.1016/j.catena.2019.02.004.

Maetens, W., Poesen, J., Vanmaercke, M., 2012. How effective are soil conservation techniques in reducing plot runoff and soil loss in Europe and the Mediterranean? Earth-Sci. Rev. 115, 21–36. https://doi.org/10.1016/j.earscirev.2012.08.003.

Makowski, D., Piraux, F., Brun, F., 2019. From Experimental Network to Meta-analysis. Methods and Applications with R for Agronomic and Environmental Sciences, Quae. ed. Springer, Versailles, France.

Martin, P., 2009. De la trajectoire d'états des écosystèmes cultivés aux espaces territorialisés dynamiques: contribution à la prise en compte de la dimension temporelle dans une agronomie des territoires (PhD thesis). Institut national polytechnique de Toulouse, Paris.

Martin, P., Ouvry, J.F., Bockstaller, C., 2010. Adaptation of the curve number approach to runoff estimation for loamy soils over a growing season for winter wheat: Comparison with the STREAM approach: CN APPROACH TO RUNOFF ESTIMATION IN AN AGRICULTURAL CONTEXT. Land Degrad. Dev. 21, 376–387. https://doi.org/10.1002/ldr.957.

Martinez, M.N., Bartholomew, M.J., 2017. What does it "mean"? A review of interpreting and calculating different types of means and standard deviations. Pharmaceutics 9, 14. https://doi.org/10.3390/pharmaceutics9020014.

Mhazo, N., Chivenge, P., Chaplot, V., 2016. Tillage impact on soil erosion by water: discrepancies due to climate and soil characteristics. Agric. Ecosyst. Environ. 230, 231–241. https://doi.org/10.1016/j.agee.2016.04.033.

Momm, H.G., Bingner, R.L., Yuan, Y., Locke, M.A., Wells, R.R., 2014. Spatial characterization of riparian buffer effects on sediment loads from watershed systems. J. Environ. Qual. 43, 1736–1753. https://doi.org/10.2134/jeq2013.10.0413.

Monnier, G., Boiffin, J., 1985. Effect of the agricultural use of soils on water erosion: the case of cropping systems in western Europe. Presented at the Impact of change of land use and management. Atelier CEE, Cesena, Italy.

Morgan, R.P.C., 2005. Soil Erosion and Conservation, 3rd ed. ed. Wiley-Blackwell, Malden, MA.

Nearing, M.A., Govers, G., Norton, L.D., 1999. Variability in soil erosion data from replicated plots. Soil Sci. Soc. Am. J. 63, 1829–1835. https://doi.org/10.2136/sssaj1999.6361829x.

Oberpriller, J., de Souza Leite, M., Pichler, M., 2022. Fixed or random? On the reliability of mixed-effects models for a small number of levels in grouping variables. Ecol. Evol. 12, e9062 https://doi.org/10.1002/ece3.9062.

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. Commun. Agric. Appl. Biol. 79, 513–524.

Osman, K.T., 2014. Soil Degradation, Conservation and Remediation. Springer, Dordrecht. https://doi.org/10.1007/978-94-007-7590-9.

Panagos, P., Meusburger, K., Ballabio, C., Borrelli, P., Alewell, C., 2014. Soil erodibility in Europe: a high-resolution dataset based on LUCAS. Sci. Total Environ. 479–480, 189–200. https://doi.org/10.1016/j.scitotenv.2014.02.010.

Panagos, P., Borrelli, P., Spinoni, J., Ballabio, C., Meusburger, K., Beguería, S., Klik, A., Michaelides, S., Petan, S., Hrabalíková, M., Olsen, P., Aalto, J., Lakatos, M., Rymszewicz, A., Dumitrescu, A., Perčec Tadić, M., Diodato, N., Kostalova, J., Rousseva, S., Banasik, K., Alewell, C., 2016. Monthly rainfall erosivity: conversion factors for different time resolutions and regional assessments. Water 8, 119. https://doi.org/10.3390/w8040119.

Panagos, P., Ballabio, C., Meusburger, K., Spinoni, J., Alewell, C., Borrelli, P., 2017. Towards estimates of future rainfall erosivity in Europe based on REDES and WorldClim datasets. J. Hydrol. 548, 251–262. https://doi.org/10.1016/j.jhydrol.2017.03.006.

Panagos, P., Ballabio, C., Poesen, J., Lugato, E., Scarpa, S., Montanarella, L., Borrelli, P., 2020. A soil erosion indicator for supporting agricultural, environmental and climate policies in the European Union. Remote Sens. 12, 1365. https://doi.org/10.3390/rs12091365.

Patault, E., Ledun, J., Landemaine, V., Soulignac, A., Richet, J.-B., Fournier, M., Ouvry, J.-F., Cerdan, O., Laignel, B., 2021. Analysis of off-site economic costs induced by runoff and soil erosion: example of two areas in the northwestern European loess belt for the last two decades (Normandy, France). Land Use Policy 108, 105541. https://doi.org/10.1016/j.landusepol.2021.105541.

Pimentel, D., Kounang, N., 1998. Ecology of soil erosion in ecosystems. Ecosystem 1, 416–426. https://doi.org/10.1007/s100219900035.

Pisa, P.R., Preti, F., Rossi, M., Ventura, F., Mazzanti, B., 1999. Water, soil and chemical losses: field experiments and model analysis. Water Sci. Technol. 39, 93–102. https://doi.org/10.2166/wst.1999.0144.

Reyniers, M., Maertens, K., Vrindts, E., De Baerdemaeker, J., 2006. Yield variability related to landscape properties of a loamy soil in central Belgium. Soil Tillage Res. 88, 262–273. https://doi.org/10.1016/j.still.2005.06.005.

Rhoton, F.E., 2000. Influence of time on soil response to no-till practices. Soil Sci. Soc. Am. J. 64, 700–709. https://doi.org/10.2136/sssaj2000.642700x.

Richet, J.-B., Ouvry, J.-F., Saunier, M., 2017. The role of vegetative barriers such as fascines and dense shrub hedges in catchment management to reduce runoff and erosion effects: experimental evidence of efficiency, and conditions of use. Ecol. Eng. Wetl. Buffer Zones Watershed Manag. 103, 455–469. https://doi.org/10.1016/j.ecoleng.2016.08.008.

Sittig, S., Sur, R., Baets, D., Hammel, K., 2020. Consideration of risk management practices in regulatory risk assessments: evaluation of field trials with micro-dams to reduce pesticide transport via surface runoff and soil erosion. Environ. Sci. Eur. 32, 86 https://doi.org/10.1186/s12302-020-00362-1.

Sittig, S., Sur, R., Baets, D., 2022. Runoff mitigation via micro-dams and conservation tillage—Numerical modeling of runoff and erosion from maize field trials. Integr. Environ. Assess. Manag. 18, 1348–1363. https://doi.org/10.1002/ieam.4546.

Soane, B.D., Ball, B.C., Arvidsson, J., Basch, G., Moreno, F., Roger-Estrade, J., 2012. No-till in northern, western and south-western Europe: A review of problems and opportunities for crop production and the environment. Soil Tillage Res 118, 66–87. https://doi.org/10.1016/j.still.2011.10.015.

SPW (Service public de Wallonie), 2022a. Parcellaire agricole anonyme (situation 2021) - Service de visualisation WMS. Géoportail de la Wallonie. 〈http://geoportail.wallonie.be/catalogue/00916a35–786b-4569–9da6–71ca64ca54b1.html〉 (accessed 4.4.23).

SPW (Service public de Wallonie), 2022b. Direction générale opérationnelle de la Mobilité et des Voies hydrauliques (DGO2) - Annuaires et statistiques. voies-hydrauliques.wallonie.be. 〈http://voies-hydrauliques.wallonie.be/opencms/opencms/fr/hydro/Archive/annuaires/index.html〉 (accessed 3.31.22).

Stehle, S., Elsaesser, D., Gregoire, C., Imfeld, G., Niehaus, E., Passeport, E., Payraudeau, S., Schäfer, R.B., Tournebize, J., Schulz, R., 2011. Pesticide risk mitigation by vegetated treatment systems: a meta-analysis. J. Environ. Qual. 40, 1068–1080. https://doi.org/10.2134/jeq2010.0510.

Stehle, S., Dabrowski, J.M., Bangert, U., Schulz, R., 2016. Erosion rills offset the efficacy of vegetated buffer strips to mitigate pesticide exposure in surface waters. Sci. Total Environ. 545–546, 171–183. https://doi.org/10.1016/j.scitotenv.2015.12.077.

Sui, Y., Ou, Y., Yan, B., Xu, X., Rousseau, A.N., Zhang, Y., 2016. Assessment of micro-basin tillage as a soil and water conservation practice in the black soil region of Northeast China. PLoS One 11. https://doi.org/10.1371/journal.pone.0152313.

Tan, W.J., Khor, J.F., Ling, L., Huang, Y.F., 2018. Misuse of Lambda (λ) in NRCS-CN Model. In: Huang, Y.F., Tan, K.W., Ling, L., Leong, K.H. (Eds.), E3S Web of Conferences. Presented at the International Conference on Civil and Environmental Engineering (ICCEE). Kuala Lumpur, Malaysia, p. 07006. https://doi.org/10.1051/e3sconf/20186507006.

Trnka, M., Olesen, J.E., Kersebaum, K.C., Skjelvåg, A.O., Eitzinger, J., Seguin, B., Peltonen-Sainio, P., Rötter, R., Iglesias, A., Orlandini, S., Dubrovský, M., Hlavinka, P., Balek, J., Eckersten, H., Cloppet, E., Calanca, P., Gobin, A., Vučetić, V., Nejedlik, P., Kumar, S., Lalic, B., Mestre, A., Rossi, F., Kozyra, J., Alexandrov, V., Semerádová, D., Žalud, Z., 2011. Agroclimatic conditions in Europe under climate change. Glob. Change Biol. 17, 2298–2318. https://doi.org/10.1111/j.1365-2486.2011.02396.x.

Truman, C.C., Nuti, R.C., 2010. Furrow diking in conservation tillage. Agric. Water Manag. 97, 835–840. https://doi.org/10.1016/j.agwat.2010.01.004.

USDA-SCS (United States Department of Agriculture- Soil Conservation Service), 1954. National Engineering Handbook, Section 4: Hydrology.

van Dijk, A.I.J.M., Bruijnzeel, L.A., Rosewell, C.J., 2002. Rainfall intensity–kinetic energy relationships: a critical literature appraisal. J. Hydrol. 261, 1–23. https://doi.org/10.1016/S0022-1694(02)00020-3.

Vejchar, D., Stehlik, M., Mayer, V., 2017. Influence of tied ridging technology on the rate of surface runoff and erosion in potato cultivation. Agron. Res. 15, 2207–2216. https://doi.org/10.15159/AR.17.039.

Verro, R., Finizio, A., Otto, S., Vighi, M., 2009. Predicting pesticide environmental risk in intensive agricultural areas. I: screening level risk assessment of individual chemicals in surface waters. Environ. Sci. Technol. 43, 522–529. https://doi.org/10.1021/es801855f.

Verstraeten, G., Poesen, J., 1999. The nature of small-scale flooding, muddy floods and retention pond sedimentation in central Belgium. Geomorphology 29, 275–292. https://doi.org/10.1016/S0169-555X(99)00020-3.

Virto, I., Imaz, M., Fernández-Ugalde, O., Gartzia-Bengoetxea, N., Enrique, A., Bescansa, P., 2014. Soil degradation and soil quality in Western Europe: current situation and future perspectives. Sustainability 7, 313–365. https://doi.org/10.3390/su7010313.

Wauters, E., Bielders, C., Poesen, J., Govers, G., Mathijs, E., 2010. Adoption of soil conservation practices in Belgium: an examination of the theory of planned behaviour in the agri-environmental domain. Land Use Policy 27, 86–94. https://doi.org/10.1016/j.landusepol.2009.02.009.

Wischmeier, W.H., Smith, D.D., 1978. Predicting Rainfall Erosion Losses: A Guide to Conservation Planning. USDA, Science and Education Administration, Hyattsville, Maryland, USA.

Wiyo, K.A., Kasomekera, Z.M., Feyen, J., 2000. Effect of tied-ridging on soil water status of a maize crop under Malawi conditions. Agric. Water Manag. 45, 101–125. https://doi.org/10.1016/S0378-3774(99)00103-1.

Zhang, D., Lin, Q., Chen, X., Chai, T., 2019. Improved curve number estimation in SWAT by reflecting the effect of rainfall intensity on runoff generation. Water 11, 163. https://doi.org/10.3390/w11010163.

Zhang, X., Liu, X., Zhang, M., Dahlgren, R.A., Eitzel, M., 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. J. Environ. Qual. 39, 76–84. https://doi.org/10.2134/jeq2008.0496.