Towards climate-smart sustainable management of agricultural soils

SCALE

Managing Sediment Connectivity in Agricultural Landscapes for reducing water Erosion impacts

Deliverable WP4-D2

Guideline on the current implementation of erosion measures and other connectivity elements depending on scale and modelling approach

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1 Introduction

Soil erosion and sediment transport through the landscape are strongly influenced by the presence of connectivity elements. These elements influence whether and to which degree the transport of water and sediment is facilitated between landscape features during hydrologic events. Erosion control measures, like conservation tillage, grass buffer strips or retention ponds, are explicitly implemented to prevent soil loss and reduce the connectivity of the landscape but are not the only connectivity elements that need to be considered in soil erosion and sediment transport modelling. Other connectivity elements, like roads, ditches, and parcel borders, can either increase or reduce the connectivity of a landscape.

Erosion control measures not only avoid soil loss but also loss of soil organic matter. For example, the measure of the sediments accumulated for 36 years (1954-1990) in an artificial reservoir located in the clayey hill environment of central Italy (Volterra – Pisa) (Bazzoffi and Pellegrini, 1992), which drains an 85 ha watershed under a five-year rotation of cereals and forage legume, allowed to quantify an average soil loss of 9.75 Mg ha^{-1} $year^{-1}$, and an associated loss of 123.8 kg ha^{-1} $year^{-1}$ of organic matter (Bazzoffi et al., 1997).

In deliverable WP4-D2 we discuss which erosion control measures and other connectivity elements exist and we give an overview of their implementation in a set of erosion models. The incorporation of erosion control measures and other connectivity elements is crucial to calculate the water flow and sediment transport through a landscape and simulate the impact of current land use and future mitigation scenarios for reducing on-site and off-site consequences of soil erosion. In the first part of this deliverable the different types of elements and their impact is discussed. This includes four types of elements: land use changes, agronomic measures, buffering measures and other connectivity elements. Also, research evidence on the effectiveness of these elements is also introduced. The second part gives an overview of the current implementation of erosion control measures and other connectivity elements in three types of models: RUSLE, WaTEM/SEDEM and process-based models. Based on this assessment we will detect and suggest model improvements in deliverable WP4-D3.

2 Impact of erosion control measures and other connectivity elements

2.1 Land use changes

2.1.1 Afforestation

Afforestation reduces soil loss by erosion as the soil is more covered by vegetation than in the case of arable land. Nevertheless, the effectiveness in reducing soil erosion is dependent on the development of the forest cover. Sparse tree cover has only a limited effect. During the first stages of land use conversion to forest it is very important to provide a sufficient vegetation cover density. Liu et al. (2020) examined the benefits of soil erosion control in forests using published data. They proved that the benefits of vegetation restoration increase with increasing the vegetation cover and tend to be stable when the coverage exceeds 60%. When considering afforestation, the preservation of forest understory grasses should be enforced to effectively reduce soil erosion. Forest harvesting by clearcutting causes significant short-term increases in sediment mobilization and sediment yield (Porto et al., 2009).

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2.1.2 Permanent grassland

Conversion of cropland to permanent grassland may function as a mitigation measure to reduce soil loss from an erosion-prone area. Grassland is generally subject to less erosion due to the permanent vegetation cover, which reduces the detachment of the soil particles from the soil surface. The high vegetation cover also favors infiltration and reduces the runoff's ability to cut into the soil and create rills. In this way, grassland decreases the overland flow formation, velocity, and concentration (Seibert & Auerswald, 2020). However, the mitigating effect depends on the actual green vegetation cover of the grassland as well as on the grazing pressure. In Switzerland, the soil loss from grasslands with 100% vegetation cover was modelled to result in a soil loss of 0.14 t ha^{-1} yr^{-1}. The actual green vegetation cover of grassland in Switzerland is estimated at 54 %, which increased the modelled soil loss considerably to 4.55 t ha^{-1} yr^{-1} (Schmidt et al., 2019). Still, there is a lack of knowledge of the complex erosion processes in permanent grassland (Milazzo et al., 2023) and the impact of soil erosion from these areas is underestimated, also regarding lateral carbon transfer (Borrelli et al., 2018).

2.1.3 Perennial crops

Perennial crops produce more ground cover and perform longer growing seasons, and have more extensive root systems, which make them an appropriate choice to reduce soil erosion (Zhang et al., 2011). Promising win-win strategies include the production of perennial bioenergy crops on recently abandoned cropland or on cropland prone to land degradation, as perennial crops typically reduce soil erosion rates (Naess et al., 2023).
Research at two experimental fields (Kukkonen et al., 2004; Puustinen et al., 2005) and a modelling study with observational data from seven experimental fields (Räsänen et al., 2023) in Finland found that perennial grass type crops reduce erosion effectively compared to annual crops. According to these studies, the erosion was 59-73% lower with perennial grass than with spring cereals (wheat, oat, barley) with conventional autumn till. A large share of erosion in Finland occurs outside the growing season during the rainy late autumn-early winter and during spring snowmelt (Puustinen et al., 2007), and consequently perennial crops have an important role in erosion reduction.

2.1.4 Crop rotations, crop diversification and set-aside

Crop choices on arable land determine the severity of soil erosion. On agricultural parcels with a high erosion risk, the cumulative soil loss by erosion over multiple years can be reduced by choosing crop rotations that include crops giving a better protection of soils against erosion, e.g., winter cereals. On the contrary, crops like potatoes, onions or corn are known as erosion sensitive crops. Shifting the most erosion sensitive crops to agricultural parcels with a lower erosion risk and vice versa has a substantial impact on total erosion at catchment scale.
Research at two experimental fields (Kukkonen et al., 2004; Puustinen et al., 2005) and a modelling study with observational data from seven experimental fields (Räsänen et al., 2023) in Finland found that winter crops, that are sown in the late summer and are in the seedling or tillering growth stages over the winter are also a means to reduce erosion. According to these studies, winter cereals (wheat, rye) reduced erosion by 25-29% compared to spring cereals (wheat, oat, barley) with autumn till.
Crop rotations are important for reducing the cumulative loss of fertile soil at parcel level. Another type of erosion control can be realised by crop diversification at landscape level. More different crops in a catchment lowers soil erosion and sediment transport as a more divers patchwork of soil covered by vegetation is realised.

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Set-aside means that parts of arable land are temporarily taken out of production and transformed to a 'non-erodible' type of land use (e.g., forests, grasslands), which lowers both soil erosion and sediment transport.

2.1.5 Intercropping

Intercropping is defined as the relay or simultaneous cultivation of two or more crops on the same field (Burgess et al., 2022). It may concern the growing of two or more cash crops together, or of a cash crop with a cover crop or other non-cash crop.
Several types of intercropping exist. Mixed cropping means that one or more crops are freely mixed. The term strip cropping is used when multiple rows of one crop are alternated with multiple rows of another crop. Undersowing indicates the sowing of a cover crop with or after a main crop. A fast-growing crop can be sown with a slow-growing crop, so that the fast-growing crop is harvested before the slow-growing crop starts to mature. Relay cropping signifies that the second crop is sown during the growth of the first crop.
In either case intercropping implies an additional soil cover and thus a lower soil erosion risk. As an example, maize can be undersown with grass to reduce erosion.

2.1.6 Agroforestry

Agroforestry is the term used to indicate those land use systems and technologies in which perennial woody plants (trees, shrubs, palms, or bamboo) and agricultural crops or animals are intentionally housed on the same parcel of land in some form of spatial and temporal arrangement (FAO and ICRAF, 2019). According to Burgess et al. (2022), in agroforestry systems, trees are incorporated on at least 10% of agricultural land. The adoption of agroforestry can offer several environmental benefits compared to conventional agricultural systems (Dupraz and Newman, 1997). It is well known that trees and bushes effectively contribute to reduced soil loss by intercepting rain drops and dissipating their kinetic energy (Grove and Rackham, 2001), improving soil structural characteristics beneath the canopies (Joffre and Rambal, 1998), regulating surface runoff and drainage (Vandermeer, 1989), and acting as disconnectivity elements between the cultivated fields and the permanent hydrographic network.
A field experiment was carried out in Italy, in the valley of the Era River (Pisa), on a clay soil marginal area, to evaluate the effects of a Sulla (Hedysarum coronarium L.) + Saltbush (Atriplex halymus L.) agrosilvopastoral system (ASPS) in improving the soil physical characteristics and to provide environmental protection by controlling runoff and reducing soil erosion on slopes (Chisci et al., 2001). The results showed that the tested ASPS was able to increase, over a period of about 4 years, soil aggregate stability and soil macroporosity in a soil previously ploughed every year for continuous winter wheat (Triticum durum L.) cultivation. The average annual runoff amount was reduced to 5% under ASPS compared to winter wheat. It was inferred that the canopy of the Sulla/Atriplex shrub plant association dissipated the storm kinetic energy, reducing consistently the detachment of soil particles by raindrop impact, reduced runoff velocity and prolonged lag-time of peak discharge. In addition, ASPS reduced the average annual soil loss by about 7% respect to continuous wheat cultivation. Thus, ASPS has the potential to reduce the contribution of pollutants from slopes to river network, with the overall effect of decreasing the environmental impact downstream.
However, investigating the environmental performance of agroforestry through field experiments is expensive and time-consuming because trees and bushes take years to mature and, consequently, the initiation of such experiments is difficult (Poulton, 1995). Computer models can be useful instruments

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for overcoming these problems. A computer simulation, carried out at 19 landscape test sites in the Mediterranean and Atlantic regions of Europe, showed that silvoarable agroforestry could significantly reduce erosion by up to 65% when combined with contouring practices at sites showing erosion values in the 0.5-3 Mg ha^{-1} y^{-1} range (Palma et al., 2007).

2.1.7 Field parcel size

Parcel size can influence the resulting soil loss, as longer slopes lead to increased erosion due to accumulation of overland flow (Renard et al., 1997). Large field parcels increase the flow route length and structural changes in the agricultural landscape towards larger parcels have been shown to result in increased erosion (Devátý et al., 2019). The parcel effect has been shown to increase with increasing flow rates and to be primarily responsible for sediment production under high-flow conditions (Wang et al., 2022). Dividing large parcels into smaller segments increases spatial heterogeneity. This may reduce erosion by interrupting flow paths and increasing sediment trapping between parcels e.g., by adding other erosion measures such as grass buffer strips. The key factor in erosion control is that parcel structure together with other landscape elements such as field boundaries, hedges, etc. highly affect connectivity in the landscape.

2.1.8 Terracing

Terraces reduce erosion by breaking the slope into shorter slope lengths. Also, terraces may also effectively contribute to deposition of sediments. The deposition behind the terraces is dependent on the spacing and slope of the terraces. For example, Renard et al. (1997) estimate that terraces with 33 m spacing and slopes of 0.1-0.3 % can reduce sediment yields by 40%, and that the reduction effect with terrace slopes > 0.8 % is negligible. They should be well maintained, otherwise, adverse effects are likely.

2.2 Agronomic measures

2.2.1 Cover crops

Cover crops provide soil cover during the winter season and fallow periods are a very effective erosion control technique. Winter crops reduce runoff and erosion during the intercropping period that precedes spring crops, but also during the subsequent spring crop (Laloy et al., 2010). Laloy et al. (2010) measured that cover crops reduced runoff and erosion by more than 94% compared with untilled, post-maize harvest plots. Moreover, plots that had been covered during the previous intercropping period lost 40 to 90% less soil compared with maize plots that had been left bare during the intercropping period. When cover crops freeze at the beginning of the winter period, the above-ground biomass becomes less effective in protecting the soil from water erosion, but roots can still play an important role in improving soil strength. Cover crops with thick roots (e.g., white mustard and fodder radish) are less effective than cover crops with fine-branched roots (e.g., ryegrass and rye) in preventing soil losses by concentrated flow erosion. Moreover, after frost, the erosion-reducing potential of phacelia and oats roots decreased (Baets et al., 2011). Panagos et al. (2015) estimated the erosion reduction by cover crops to be around 20% in their estimation of a soil erosion cover-management factor at the European scale.

2.2.2 Mulching, crop residue management and tillage practices

Mulching is applying a natural or artificial layer of plant residues or other materials on the soil surface to protect the soil. Crop residues of the preceding crop can also be left on the field surface. No tillage, reduced tillage and strip-tillage are tillage practices that contribute substantially to erosion reduction, especially in combination with mulching or leaving crop residues on the field.

Mulch and no-till practices cause transport limiting conditions due to higher soil cover, increased soil aggregate stability, lower runoff generation and lower flow velocity (Klik & Rosner, 2020). A review by Strauss et al. (2003) showed that the mean runoff reduction of mulch and minimum tillage was 28 %, while the mean reduction in soil loss was 66 %. In Switzerland, a significant decrease in soil loss over a 20-year field monitoring period could be directly linked to the increased use of conservation tillage (Prasuhn, 2020).

Research at two experimental fields (Honkanen et al., 2021; Puustinen et al., 2005) and a modelling study with observational data from seven experimental fields (Räsänen et al., 2023) in Finland found that no-till (winter-time stubble) is one of the most effective ways to reduce erosion, but the observations on the effect of reduced tillage (disc cultivator till, shallow stubble till) on erosion are limited. According to these studies, no till reduced erosion by 42-70% and reduced tillage reduced erosion by 32% compared to conventional autumn till in cereal (wheat, oat, barley) cultivation.

The effects of reduced tillage practices on runoff and erosion processes were investigated on a long-term basis at three sites in Lower Austria (Klik & Rosner, 2020). Compared to conventional tillage reduced tillage practices decreased surface runoff from silt loam by 25-55 % and 49-60 % for mulch tillage and no-tillage, respectively. However, for a not well-drained silty clay loam soil an increase in runoff was observed, which could be related to higher bulk densities of reduced tillage practices. Soil loss was reduced between 38 % to 93 % depending on soil type. Reductions in nutrient losses (of nitrogen and phosphorus) as well as soil organic carbon (34 -89 %) were also observed under the conservation tillage practices.

Leys et al. (2007) conducted 184 rainfall simulation experiments on small scale plots (0.73 m²) on fields planted with maize and sugar beet. The results clearly confirmed the beneficial effects of reduced tillage in terms of runoff and soil loss, but they also indicated that the effect of the application of conservation tillage was highly variable. The mean reduction of erosion was 38%. However, Leys et al. (2010) clarified in a literature review that that differences in runoff and erosion between conservation and conventional tillage are scale-dependent: the difference in runoff and erosion response increases with the length of the plot/field considered. The relative scale effect is more important for erosion than for runoff. While a mean erosion reduction of 48% was found for very small plots (<2 m), this reduction increases to ca. 67% for plots between 2 m and 30 m and it is about 86% for plots exceeding 30 m in length.

The Gomeros project (Vanden Nest et al., 2019) confirmed that 85% reduction is an adequate assessment of the impact of conservation tillage on field parcel level for the most common arable crops, but that the reduction potential is also crop dependent. Determining factors are the crop characteristics and the methods of seedbed preparation and seeding.

Strip-tillage is a suitable cultivation technique to reduce erosion for row crops. Only the small sowing strip is tilled, and the remaining surface stays untilled. The Gomeros project (Vanden Nest et al., 2019) measured a median reduction of soil erosion of 95%.

Panagos et al. (2015) used a reduction factor of 65% for conservation/ridge tillage and 75% for no-till practices; leaving adequate residue on the ground after the harvest is incorporated as an additional 12% reduction.

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2.2.3 Contour farming and sowing practices

Contour farming is the practice of tillage, planting, and other farming operations performed on or near the contour of the field slope. Brandhuber and Kistler (2014) mention a reduction of ca. 10% of surface damaged by soil erosion.

Double sowing and widespread sowing offer also possibilities for a higher soil cover and better protection against soil erosion. Experiments of Vanden Nest et al. (2019) showed that widespread sowing of maize can lead to 30-50% of erosion reduction for an interrow distance of 15 cm in comparison with 75 cm while keeping the sowing density constant.

2.2.4 Micro-dams between ridges

The effects of micro-dams in potato farming have been investigated in various countries and climatic zones across Europe in the last decade. Olivier et al. (2014) studied the effects over one year in Belgium, Vejchar et al. (2019) and Konzett et al. (submitted) over three years in Czech Republic and Austria, respectively. Through the implementation of micro-dams surface runoff was decreased by 75% (Vejchar et al., 2019) up to 95% (Konzett et al., submitted) compared to the conventional potato cultivation. Similar to those results, the sediment yield was also decreased from 66% (Olivier et al., 2014) to 99% (Konzett et al., submitted). It has to be considered that the effectiveness of micro-dams is crucially influenced by the micro-dam characteristics (height, width between two micro-dams) as well as by the slope of the field and the precipitation intensity. Konzett et al. (submitted) also investigated the effect of an additional reinforcement of micro-dams through a cover-crop and found that reinforced micro-dams withhold higher precipitation intensities than conventional micro-dams.

2.2.5 Soil surface roughness

A rough soil surface has many depressions and barriers. They trap water and sediment, causing rough surfaces to erode at lower rates than do smooth surfaces in similar conditions. Roughness and cloddiness also affect the degree and rate of soil sealing by rainfall impact (Renard et al., 2017). In this way, fine seedbed preparations lead to higher erosion risks. Vanden Nest et al. (2019) measured an erosion reduction for onions and peas when opting for a rougher seedbed preparation. Similarly, the soil roughness on and in-between ridges impacts the degree of runoff and erosion between the ridges (Vanden Nest et al., 2019).

2.2.6 Reduction of subsoil compaction

Soil compaction has a major influence on the infiltration rates in soils. Consequently, measures that prevent or remediate soil compaction contribute to the reduction of water runoff and lower the soil erosion risk. Measures for preventing soil compaction are e.g., lowering wheel loads and tyre pressures as well as avoiding bad soil moisture conditions for field operations. Remediation of soil compaction is a difficult task and can on the long term be achieved by mechanical or biological decompaction techniques (Elsen et al., 2014).

2.2.7 Increase of soil organic matter

Soil erosion can be reduced by increasing organic matter content of the soil (Renard et al., 1997). Soil organic matter can be increased by reducing tillage, permanent soil cover, crop diversification, and by application of organic amendments, such as manure, compost, and by-products from agroindustry

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(Francaviglia et al., 2023). The potential for increasing soil organic matter may, however, vary by soil type and characteristics (Börjesson et al., 2018).
The relationship of soil organic matter and erosion can be evaluated by a nomograph that is used to estimate the soil erodibility factor of USLE and RUSLE (Wischmeier and Smith, 1978; Renard et al., 1997). The nomograph is an empirical equation describing the propensity of the soil to erode by water. The equations contain a variable for soil organic matter content, in addition to particle size fractions, and soil structure and permeability.
A study in the UK made laboratory experiments with 30 soil samples with varying soil organic matter contents and estimated erosion under a rainfall simulator (Guerra, 1994). The study found that of the various measured soil properties, organic matter and the proportion of water stable aggregates played the main role in erosion. At lower rainfall intensities soils with less than 3.5% of organic matter were found to have more unstable aggregates, but with higher rainfall intensities all soils suffered soil loss.
A study in Finland estimated the effect of organic soil amendments on erosion in a four-year combined experimental field and laboratory study (Rasa et al., 2021). Three different organic sludges from the pulp and paper industry were applied on clay soil, and it was found that they significantly reduced the suspended solids in percolation water (>25%) even four years after the sludge applications. The reduction effect was attributed to lower particle detachment by rain and to more stable soil aggregates due to interactions of soil minerals with the added particulate organic matter and microbe-derived compounds.

2.3 Buffering measures

2.3.1 Grass buffer strips

Vegetated buffer strips are implemented with various widths and vegetation covers, but often grass buffer strips are used. Grass buffer strips reduce erosion from their own area, and they trap sediments flowing from upslope areas. They can be implemented within fields, at field margins and adjacent to rivers. Grass buffer strips can be planted in agricultural fields along contours at predefined height differences (Mekonnen et al., 2015). They are placed in catchments to reduce the exportation of sediments, soil and pollutants from agricultural activities (Gumiere et al., 2011). Grass buffer strips is a measure that is fast to set up.
Herbaceous vegetation reduces soil erosion by protecting the soil against the raindrops impacts; furthering infiltration; stabilising soil; increasing surface roughness; reducing runoff velocity; boosting evapotranspiration; and inducing sediment retention. The plant efficiency towards runoff and soil erosion reduction depends on the species used (Kervroedan et al., 2021). The plant canopy decreases the raindrop kinetic energy (Kervroedan et al., 2021). Grass buffer strips reduce the velocity and sediment transport capacity of flowing water by retarding and spreading the concentrated surface runoff, which enhances sediment deposition within and upslope of the grass buffer strip (Mekonnen et al., 2015). When the soil reaches its saturated hydraulic conductivity, the main process inducing sediment retention and the reduction of the runoff flow velocity is the hydraulic resistance created by the vegetation (Kervroedan et al., 2021). The aboveground biomass of the herbaceous vegetation reduces the flow velocity, creating a backwater area in front of the vegetation where sediments settle as the sediment transport capacity of the flow is reduced (Kervroedan et al., 2021). Hence, grass buffer strips act as filters, trapping sediment particles suspended in overland flow, thus reducing sedimentological connectivity (Gumiere et al., 2011). Gumiere et al. (2011) presented studies that observed a runoff reduction of 50 to 80 %. The efficiency of sediment removal of grass buffer strips

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increases when flow depth is of the same order as the vegetation height (Gumiere et al., 2013). Grass buffer strips increase the infiltrability of soil. Furthermore, infiltration fluxes double when doubling grass buffer strip width (Gumiere et al., 2011). The existence of backwater regions upslope from vegetated filters acts in favour of sedimentation processes and may increase sediment removal efficiency (Gumiere et al., 2011). The physical characteristics of the different grass species are important in retarding runoff through upslope ponding (Mekonnen et al., 2015). In an agricultural catchment, the global runoff reduction and sediment removal efficiency changes depending on the distribution of grass buffer strips (Gumiere et al., 2011).

Literature reviews suggest that the trapping effect of buffer strips is highest at the first five meters, does not substantially increase after 10m width (Figure 1), and the sediment trapping efficiency was approximately 80% for all buffer widths of greater than 5 m (Liu et al., 2008; Yuan et al., 2009). Grass buffer strips are recommended to be 5–15 m wide and every 50–150 m downslope. The width of the strip should be increased and the gap between strips decreased, as erosion risk increases (Fullen, 2003). The buffer strips are the most effectively located in areas where the buffer strip area itself has high erosion (e.g., sloping field sections near water bodies) and where a large field area drains over the buffer strip. For steep fields, highly concentrated flows and sediment fluxes, and very small soil aggregate sizes the sediment trapping efficiencies of vegetated buffer strips can be lower (Liu et al., 2008; Yuan et al., 2009; Verstraeten et al., 2006).

In Finland, two experimental studies reported that on gently sloping fields 14 m and 10 m wide managed grass buffer strips (with autumn moving) reduced sediment loads from the field 22-72% with larger reduction effects with wider grass buffer strip (Puustinen et al., 2005; Uusi-Kämppä and Jauhiainen, 2010). The other study experimented also with natural vegetation buffer strip (hay and shrubs) and found that they reduced sediment loads 4-5% more than managed grass buffer strip (Uusi-Kämppä and Jauhiainen, 2010). These reduction rates are lower compared to rates reported in review of buffer strips (Liu et al., 2008; Yuan et al., 2009), which is possibly explained by winter conditions, and particularly during the spring snowmelt period when the trapping efficiency of buffer strip vegetation may be lower.

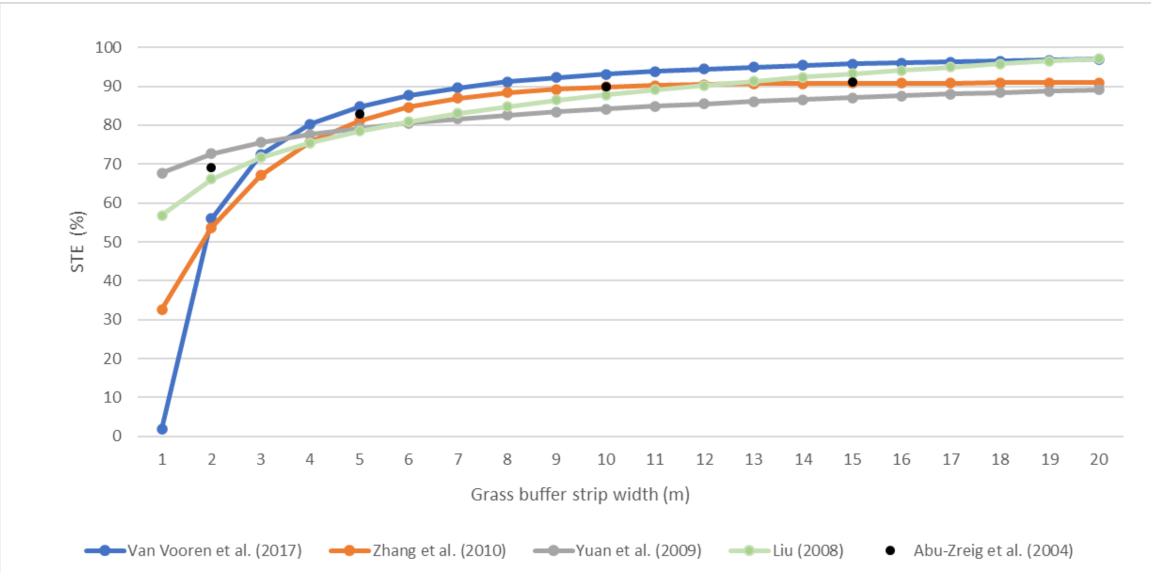

Figure 1: Sediment trapping efficiency (STE) in function of grass buffer strip width (m)

More sediment can be trapped using a combination rather than single measures, as an example, the combination of grass buffer strips and check dams or terraces (Mekonnen et al., 2015). Grass hedges

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decreased concentrated runoff and deposited sediment above the barriers more than grass buffer strips did (Blanco-Canqui et al., 2006). The complexity of interactions between sediment flow and land management practices makes it difficult to take grass buffer strips into account in catchment-scale erosion models (Gumiere et al., 2011).

2.3.2 Hedges and hedgerows

2.3.2.1 Grass hedges

Grass hedges are vegetative barriers constructed from different species of grass. They have been used as a soil and water conservation practice for over 50 years (Dabney et al., 2004). This technique is used in tropical environments and in high culture fields in the USA (Ouvry et al., 2012). They are constructed with many different plant species, including vetiver grass and switchgrass (that can support temperatures < -15°C) (Dabney et al., 2004). For the best performance in soil protection from erosion, grass hedges must be placed at 0.5 m vertical intervals and comprised of a variety of species (Dabney et al., 2004). There is a practical upper limit to the gully slope that can be successfully treated with grass hedges even when vertical intervals are < 0.5 m: this maximum slope is 33% (18°). This slope may also be a practical limit to the ability of a series of vegetative hedges to dissipate energy and resist washout by concentrated flows (Dabney et al., 2004). The root system of hedges adds cohesion to the soil and can re-grow when partially buried by sediments (Dabney et al., 2004). Switchgrass hedges remain erect in unit discharges as large as 0.2 m^2 s^{-1}, producing backwater depths as large as 0.4 to 0.5 m and reducing water velocity to non−erosive values (Dabney et al., 2004). Erosive flow velocities would be expected to develop downslope of each hedge, in the case of lower flows, when backwaters would not fully cover the regions between hedges (Dabney et al., 2004).

Hedges have three flow regimes (Dabney et al., 2004): During low flow, there is erosion below the hedge and deposition above the hedge. For intermediate flow, the erosion/deposition processes are substantially damped. At high flow, the flow is concentrated and has a reduced velocity near the bed. A grass hedge causes the formation of a backwater (Richet et al., 2017), with a horizontal level between the hydraulic jump and the hedge. The base of the hedge has an earth ridge a few centimeters high (due to the development of roots). The flow velocity is lower than 0.10 m/s upstream of the hedge.

Grass hedges reduce ephemeral gully erosion (Dabney et al., 2004). They cause reductions in water runoff of 30-60%, and reductions in erosion rates at the plot level of 70-90% (Ouvry et al., 2012). The percentage of reduction in overland flow increases with precipitation intensity (Wu et al., 2010). Grass hedges reduce soil erosion from 65% to 95 % according to the type of grass that constitutes the hedge, the slope, and the rain intensity (Wu et al., 2010). The increased soil cohesion due to hedge's roots protects from gully erosion (Dabney et al., 2004).

The reduction of runoff depends on hedge's different development rate and plant density. Therefore, maximum efficiency will be expected in years for some grass hedge types (Wu et al., 2010). Mass failure occurs if a maximum downslope (27 °) is present downstream of the hedges (Dabney et al., 2004). The success of the practice is uncertain because some erosion would be expected to occur between the hedges before the slope stabilized, and because the steps created between hedges might be unstable (Dabney et al., 2004).

If successful, this solution is less capital-intensive than rock dams and fascines (Dabney et al., 2004). Grass hedges could re-grow when partially buried by sediment (rock dams and fascines do not present this ability) (Richet et al., 2017).

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2.3.2.2 Shrub hedges

A shrub hedge consists of a line of shrubs or shrubs with some trees, growing on a base of herbaceous vegetation. They are used to prevent erosion and flooding (Ouvry et al., 2012). While grass buffers are widely used, shrub hedges are not. This is mainly due to a lack of knowledge about their hydraulic properties and the situations to which they are most suited (Richet et al., 2017). Shrub hedges can be placed across a thalweg to mitigate rill and ephemeral gully erosion (Richet et al., 2017). Shrub hedges reduce flow velocity. The hedge causes the formation of a backwater, with a horizontal level between the hydraulic jump and the hedge (upstream of the hedge). The base of the hedge has an earth ridge a few centimeters high due to the development of roots. The slowdown is proportional to the density of stems opposing the flow per unit of width crossed. The presence of foliage reinforces this effect (Ouvry et al., 2012). Runoff coefficients are reduced by 65 to 85% (Ouvry et al., 2012). Shrub hedges infiltration is evaluated in the order of 400 +/- 100 mm h^{-1}. To infiltrate an inflow of 1 l s^{-1}, you need an area of 9 m² (Richet et al., 2017). The soil hydraulic conductivity at saturation near a shrub hedge in Peru varies from 180 to 500 mm h^{-1} (Ouvry et al., 2012).
The role of tree and shrub hedges on the sedimentation of particles from water erosion is important. Sediment trapping has been evaluated as 50–85% of soil loss when contour hedges are planted on steep slopes; at the same time, runoff is dramatically reduced (Richet et al., 2017). When planted along contour lines at 10 m intervals, shrub hedges can reduce erosion at the field's outlet by more than 95% (König (1992) and Roose (1994), according to Richet et al., (2017)). Trapping efficiency is independent of inflow concentration, but linked to the size and density of undispersed sediment (Richet et al., 2017). At flow rates over 10 l s^{-1} m^{-1}, Manning's hydraulic roughness values are between 0.4 and 1.0 s $m^{-1/3}$ (Ouvry et al., 2012). The flow velocities are lower than 0.10 m s^{-1} upstream of the hedges (Richet et al., 2017). The Manning's hydraulic roughness values for hedges at flow rates < 12 l s^{-1} m^{-1} are from 0.36 to 0.43 s $m^{-1/3}$ (Richet et al., 2017).
Shrub hedges have a good efficiency in the long term (Richet et al., 2017). They have a small footprint and support biodiversity (Richet et al., 2017). They have a very long settling time (almost 15 years) (Richet et al., 2017). The conditions of efficiency of hedges requires a high maintenance: e.g., initial planting and follow-up maintenance should aim to reach the threshold of 50 stems per linear meter as quickly as possible, so maintenance should be done carefully in the first ten years (Richet et al., 2017). Shrub hedges require less land use than grass buffers, and their location can be restricted to the edge of the field. Thus, less cumbersome, these vegetative barriers are more easily accepted by farmers than grass buffers (Richet et al., 2017). They can be associated with fascines, as the association between them can ensure the efficiency from the moment they are installed and for many years to come (Ouvry et al., 2012).

2.3.3 Grassed waterways

Grassed waterways are a common measure to drain surface overland flow from fields without gullying along the thalweg (Fiener & Auerswald, 2005). Grassed waterways are set in areas where water is concentrated in flows from neighboring fields. They are planted with sod-forming grasses which reduce runoff, sediment transport and gully formation by slowing water flow (Dermisis et al., 2010). They are located in thalwegs, along the waterways. Stem area density, which is determined by number of stems and stem diameter per unit area, increases the sediment trapping efficiency of vegetation. Plant height influences the Manning's n coefficient (Lees et al., 2021). A stem density of more than 10,000 stems per m² reduces detachment by flow (Lees et al., 2021).
Grassed waterways reduce runoff by 47% (Fiener & Auerswald, 2005). The potential range of factors affecting the efficiency of a grassed waterway include the magnitude of the storm events, the grassed

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waterway dimensions (length and width), prevailing soil conditions, gradient of the contributing hillslopes and condition of the grass cover (cut or amenaged) (Dermisis et al., 2010). Vegetation traits affect the detachment of soil by rain splash as they facilitate dissipation of kinetic energy from rainfall (Lees et al., 2021). Mean root diameter, total length of roots, and total root surface area are important as they influence both soil cohesion and aggregate stability (Lees et al., 2021). Grassed waterways decrease flow velocities and promote sedimentation due to increased hydraulic retention (Lees et al., 2021). The critical length for the effectiveness of a grassed waterway to reduce sediment yield for different flow peaks is 500 m (Dermisis et al., 2010).

Grassed waterways are recommended to avoid soil erosion at slopes higher than 1% (Mishra et al., 2006a). Salts, acidity, or root restrictions are some substratum features that limit plant growth of the grassed waterway, so recommendations from a soil scientist have to be sought (NRCS, 2010). The construction of a grassed waterway can disturb large areas and potentially affect cultural resources (NRCS, 2010). Compared with other erosion control measures, a grassed waterway is commonly much longer than a vegetative filter strip (Fiener & Auerswald, 2005).

2.3.4 Vegetative barriers

2.3.4.1 Fascines

A fascine is a vegetative barrier made of bunches of stems (fagots) held in place by two lines of posts (Richet et al., 2017). It is a linear arrangement of branches. A distinction is generally made between fascines called dead weirs, all of whose components are made of dead wood (piles and faggots), from the so-called living fascines which include at least one living element: either the stakes or the stems staked around the faggot (Ouvry et al., 2012). Positioned across the flow, they constitute a permeable obstacle, which slows down the water (Richet et al., 2017). They have the particular benefit of slowing flows and trapping sediment upstream (Richet et al., 2017). Fascines can be placed across the thalweg to mitigate rill and ephemeral gully erosion (Richet et al., 2017).

Fascines cause the formation of a backwater with a horizontal level between the hydraulic jump and the fascine. There, the flow velocity is lower than 0.08 m s^{-1}. The hydrologic effect of fascines of reducing flow velocities is due to the increase of the hydraulic roughness, up to a factor of three (Frankl et al., 2018). If the flow rate is greater than 10 l s^{-1} m^{-1} then the Manning's coefficient will be between 0.4 and 1 s $m^{-1/3}$. For lower flow rates, the range of Manning coefficients is wider between 0.15 and 2.9 s $m^{-1/3}$. The mechanical resistance for a four-year-old living fascine is between 80 and 250 N m^{-2} (Frossard, 2009).

Fascines are instantly effective for regulating both water and sediment flux (Richet et al., 2017). They are very efficient in terms of flow slowdown and sediment trapping (Richet et al., 2017). Efficiency decreases with increasing deposits (filling might occur in one event) (Richet et al., 2017). Fascines have a short lifetime (mean. 7 years): after 5 to 10 years, the fascine bundle has completely disappeared (unless the branches have been replaced over time). In the absence of intervention, only the (dead or live) stakes remain, depending on the type of fascine. The filtering efficiency of the fascine then becomes very low (Ouvry et al., 2012; Richet et al., 2017). Fascines have a high cost of set-up (> 50 €/m) and maintenance (Richet et al., 2017).

Fascines can be associated with shrub hedges, as the association between them can ensure the efficiency of the installation from the moment it is installed and for many years to come (Ouvry et al., 2012). The "hedge + fascine" association can be completed on the downstream side by a grassy zone of 3 to 5 m wide. This association allows transforming a concentrated flow of decimetric depth into a thin laminar flow that flows between the blades of grass as on a grassed strip of a field edge (Ouvry et al., 2012).

2.3.4.2 Dams made of plant residues

Vegetative barriers made of plant residues have the advantage of being immediately effective in protection against erosion, but have a short life expectancy (Frankl et al., 2020). Vegetative dams can be constructed by using wood chips, coconut-fibers bales or straw bales (Vandekerckhove, 2010; Frankl et al., 2021). These dams differ in capacity to retain water and in sediment trapping efficiency. Experiments of Frankl et al. (2021) showed that barriers made of coconut-fiber bales performed markedly better than those of straw bales or wood chips. Sediment deposition ratios were 19%, 38% and 64% for barriers made of straw bales, wood chips and coconut-fiber bales respectively. These values increased during subsequent experiments demonstrating the effect of sediment accumulating inside the structures.

2.3.5 Silt fences

Generally, silt fences are made of wooden posts and geotextile fabric (Secci et al., 2014). Silt fences have been used to control surface erosion for several decades (Robichaud & Brown, 2002). A silt fence is a synthetic geotextile fabric that is woven to provide structural integrity (tensile strength between 0.3 to 0.4 kN) with small openings (0.3 to 0.8 mm) that pass water but not sediment (Robichaud & Brown, 2002). Contributing area into a silt fence needs to be designed so it does not overwhelm or overtop the silt fence. Hence its size varies depending on expected flow and sediment yield and a boundary at the top of the plot is often needed (Robichaud & Brown, 2002). Typically, silt fences have a length between 3 and 15 m across the hillslope, and plot lengths upslope are 5 to 61 m, while contributing areas vary from 15 to 930 m^2 (Robichaud & Brown, 2002). If the contributing area is large, a second silt fence located below the first silt fence may be used to trap any sediment that overflows the first silt fence (Robichaud & Brown, 2002). Silt fences need sufficient tensile strength to withstand the forces exerted by the storm runoff and collected silt (Wyant, 1980).

Average sediment removal efficiencies are of 92% for silty soil and 97% for sandy soil (Wyant & Virginia Highway & Transportation Research Council, 1980). Silt fences have low water permeability rates, which make them suitable to form temporary detention storage areas allowing sediment to settle and water to pass through slowly (Robichaud & Brown, 2002). For the hydraulic performance, the maximum flow rate through silt fences is a function of the head and is generally small (0.00028 to 0.013 $m^3\ s^{-1}$) (Robichaud & Brown, 2002). For the trapping efficiencies, studies have shown they range from 68% to 98% (Robichaud & Brown, 2002).

Silt fences work best when they are located on uniform slopes with minimal obstructions (Robichaud & Brown, 2002). Silt fences are inexpensive and relatively easy to remove (Robichaud & Brown, 2002). Typical problems observed with silt fence installations include tearing, broken or bent support poles, insufficient fencing, uncompacted soil near the base, material piled on or against fencing, and vandalism, so it is important to maintain a regular inspection and maintenance (Cooke et al., 2015). Compared with other erosion control measures, silt fences may have certain advantages over check dams for small catchments, like higher trap efficiency (Luffman et al., 2018).

2.3.6 Sediment retention ponds

Sediment retention ponds located upstream of rivers — considering their sediment retention capacity — can create a state of imbalance in the geochemical and hydro-sedimentary status of the underlying

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rivers (Al Sayah et al., 2019). Retention pond design should control the runoff during most events without overtopping or using the emergency outflow, and delay runoff and enhance sediment settling time as long as possible without damaging field crops or the waterway grass, which happens approximately after 3-4 days of submergence (Fiener et al., 2005).

According to Fiener et al. (2005), retention ponds trap 50-80% of the incoming sediments. Retention ponds have shown to retain as much as 90% of sediments transported in basins (Al Sayah et al., 2019). The disruptive effect of retention ponds is due to the increase of residence time of waters, resulting in a decline in the temporal variation of the main discharge (Al Sayah et al., 2019). In agricultural areas, retention ponds, that hold water only during storms, are used to protect infrastructure and private properties from flooding and damages by muddy floods (Fiener et al., 2005). Storm water detention and retention ponds or basins are common features in storm water management, to retain storm runoff for a certain time and to reduce peak discharge to a level that is bearable for the drainage system (Fiener et al., 2005). Besides the reduction of peak runoff rates, there are several additional purposes, like sediment trapping, prevention of downstream linear erosion, or water quality management, which have been addressed in a variety of retention pond sizes, constructions, and storage strategies (Fiener et al., 2005). These retention ponds typically maintain a permanent pool of water between storms to improve water quality by the settling of suspended solids and sediment bound substances (Fiener et al., 2005). Additionally, farm ponds are important to preserve biodiversity in the agricultural landscape (Miracle et al., 2010).

Conservation of ponds is a recognized need due to increasing impacts of environmental alterations as a result (Miracle et al., 2010). Compared to other surface waters, ponds still receive little effective protection from legislation or policy (Céréghino et al., 2014). Ponds create high costs for construction, area, and maintenance. Especially regular dredging is cost intensive (Fiener et al., 2005).

2.4 Other connectivity elements

2.4.1 Tillage direction

Topography is not the only determining factor of flow direction. Tillage induced roughness can also significantly impact runoff and erosion, and therefore flow patterns on a field scale can be very different from the flow pattern that would be predicted from the topography alone. On tilled fields, water flow is often directed along the tillage lines instead of in topographic direction. Due to the fact that the runoff pattern defines the locations where water will concentrate, as well as the effective slope gradient (i.e., the slope in flow direction), erosion patterns and rates are affected by tillage direction (Takken et al., 2001)).

2.4.2 Wheel tracks

Wheel tracks are wheel footprints left by the tractor on fields, during operations such as seeding. Wheel tracks are a linear feature found inside agricultural fields that tends to increase overland flow and erosion. The tractors can traffic over the same area each time also known as ‘controlled traffic’ (OptiSurface, 2021). Wheel tracks increase the compaction of the soil, which leads to less infiltration, and more runoff and erosion (Soane et al., 1982). Imposed wheel loads caused an increase in soil bulk density (Ahmadi & Ghaur, 2015). Mechanical compaction can cause harmful effects on water infiltration rate of any type of soil. With wheel-induced compaction, soil water infiltration rate decreased by 84 to 400% together with decreases in plant-available water. In these circumstances, the

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risk of flooding is augmented (Ahmadi & Ghaur, 2015). Increasing tractor wheeling intensity decreases soil permeability (Ahmadi & Ghaur, 2015). Runoff from rainfall and irrigation concentrates in the wheel tracks, and flow along the furrows towards a low point (OptiSurface, 2021). Wheel tracks, when present, can be responsible for a very large part of the runoff. Best track management practice, like breaking up the tracks, is recommended (Laloy & Bielders, 2008). By preventing wheel track compaction and improving, among others, seedbed properties (roughness), soil aggregate stability and residue or vegetation cover, the runoff and its consequences are reduced (Laloy & Bielders, 2008).

2.4.3 Parcel borders

Parcel borders can act as a barrier in the landscape due to differences in vegetation cover that may hinder water and sediment transport (Takken et al., 1999). However, concentrated waterflow is also observed along parcel borders, which may produce higher erosion rates (Takken et al., 2001). It has thus been suggested that the effect of parcel borders on connectivity in the landscape cannot be fully predicted from satellite images, but that field mapping and observations are needed (Boardman et al., 2019).

2.4.4 Subsurface drainage

Subsurface drainage is usually implemented to improve the drainage conditions in the field and to increase the crop yields. It can also be considered as erosion mitigation measure, but it also affects sediment connectivity. Research in different regions and climates suggest that subsurface drainage reduces erosion by 16-84% (Bengtson et al., 1988, 1984; Bengtson and Sabbagh, 1990; Bottcher et al., 1981; Formanek et al., 1987; Grazhdani et al., 1996; Istok and Kling, 1983; Schwab et al., 1980, 1977). Research in Finland in turn showed that substituting poorly functioning old drainage pipes and trenches with new ones reduced sediment loads by up to 15% on gently sloping (2.6%) clay soil (Turtola and Paajanen, 1995). These studies attribute the erosion reduction effect of subsurface drainage to reduced surface runoff, increased soil infiltration, changes in soil moisture, and increased crop yield. The research on the effect of subsurface drainage on erosion is however limited, and the effect of different subsurface drainage types and conditions are not well known.

The subsurface drainage alters the sediment transport pathways, which may affect the role and cost-effectiveness of erosion management measures targeted to trap erosion loads in surface runoff. Research suggests that the soil eroded at the soil surface is transported via macropores, such as cracks, fissures, and earthworm burrows in the soil matrix to the subsurface drainage pipes (Foster et al., 2003; Øygarden et al., 1997; Turunen et al., 2017; Uusitalo et al., 2001) causing a bulk of erosion material to travel from the field via subsurface drainage network instead of via surface runoff. Under Finnish conditions research and data from subsurface drained experimental fields shows, that 33–98% of the total sediment loads (surface runoff + subsurface drainage load) are observed via subsurface drainage flow, and that higher drainage densities increase the loads via subsurface drainage flow and decrease the loads via surface runoff (Finnish Environment Institute, 2019; Turunen et al., 2017; Warsta et al., 2014, 2013; Turtola et al., 2007; Äijö et al., 2018; Koskiaho et al., 2002).

2.4.5 Roads

Roads can act as both connective and disconnective elements in the landscape. Runoff from a field may be routed along the parcel border created by roads (Takken et al., 2001), and cause deposition upslope of roads (Takken et al., 1999), thereby functioning as a disconnective element. On the other hand, roads can amplify connectivity by generating concentrated runoff from the denser surface which

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impedes water infiltration. The runoff may be led into a drainage network, a stream or spill over onto downhill fields, thereby making them vulnerable to soil erosion (Bracken, 2013; Harden, 2013).

2.4.6 Ditches

Ditches are man-made channels created primarily for agricultural purposes, and which usually, have a linear planform, follow linear field boundaries, often turning at right angles, and showing little relationship with natural landscape contours (Biggs et al., 2017). Farm ditches are human-made linear elements that constitute the upstream parts of the permanent hydrographic networks in agricultural landscapes (Dollinger et al., 2015). Ditches are vital for the sustainable functioning of agricultural land (Aviles et al., 2020). The principle environmental variables influencing ditch vegetation types are salinity, water depth, substrate and hydroseral stage (Biggs et al., 2017).

Ditches collect surface and subsurface water in order to drain excess water and/or to prevent soil erosion (Dollinger et al., 2015). Ditches, which are located on field margins, only play a role in surface runoff reduction (Dollinger et al., 2015). Ditch networks play a key role in the prevention of soil erosion by surface runoff in agricultural landscapes (Dollinger et al., 2015). The combination of terraces and ditches decreases the length over which overland flow occurs along the catchment slope (Dollinger et al., 2015). To reduce soil erosion, ditches are built in the bottom of the terrace risers, to accommodate all runoff created by the terrace itself as well as any tributary runoff that enters the terrace drain (Tarolli et al., 2015). Vegetation on the banks not only increases the shear strength of the soil but is also likely to reduce the forces caused by water flow in the ditch, decreasing its erosive force (Aviles et al., 2020).

Ditches perpendicular to the slope direction is typical land conservation techniques (Dollinger et al., 2015). Some ditches may efficiently trap sediments; their mean sediment retention capacity ranged from 8.6 to 107.2 kg m^{-1} $year^{-1}$ (Dollinger et al., 2015). Within the ditches, sediment retention is mainly due to sedimentation processes or to the infiltration of particle-loaded water fluxes and, more marginally, to the sieving of particles by vegetation and litter (Dollinger et al., 2015). Ditches are characterized by an efficient surface runoff collection and a maximal infiltration allowed by a slow downstream conveyance (Dollinger et al., 2015). Agricultural ditches degrade over time by the action of multiple processes, including rain, overland flow, bank erosion, and mass movement (Aviles et al., 2020). They also require a lot of maintenance work, especially where erosion is more likely to occur (Aviles et al., 2020).

Ditches can therefore act as both connectivity and disconnectivity elements. Ditches can effectively transport eroded soil material from fields and agricultural areas to watercourses, but at the same time ditches can disconnect land units, such as field parcels, in terms of surface runoff. For example, in Finland efficient drainage of fields is important for crop productivity and therefore the field parcels are typically surrounded by open ditches. The erosion rate in Finland is relatively low in European context (Panagos et al., 2015), but the ditch and drainage network enhance sediment connectivity between the fields and the water courses.

When containing buffering elements like check dams, ditches can retain water and sediments. Small dams are constructed at several places in a ditch to lower the flow velocity and enhance sedimentation. Ditches can also be implemented to guide water and sediments to retention ponds.

Swales are shallow ditches that are combined with a vegetated verge. These swales can slow down runoff and catch sediments in the landscape.

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2.4.7 Topographic changes

Soil erosion is highly dependent on local topography, and longer and steeper slopes contribute to higher erosion rates (Renard et al., 1997). Fields have natural variation in the surface topography and surface run-off in long-term may have resulted to lowered elevations in some parts of the fields where surface runoff concentrates and consequently contributes to higher erosion. Leveling of field surface is a common practice and it is performed often to improve the management of the fields, but it can also contribute to reduction of erosion if it reduces slope length and steepness, particularly in high-erosion locations.

3 Implementation of erosion control measures and other connectivity elements in models

In this chapter we go through selected soil erosion models and their current capabilities and limitations regarding implementation of erosion control measures and connectivity elements in the model. The models include the empirical model RUSLE, which is one of the most applied soil erosion models, WaTEM/SEDEM, which builds on the RUSLE, but also incorporates sediment transport, and finally, a (non-exhaustive) selection of process-based models are considered. The two main questions asked are:

- Which erosion measures and other connectivity elements are already implemented?
- Which erosion measures and other connectivity elements are not implemented?

3.1 RUSLE

3.1.1 Revised Universal Soil Loss Equation (RUSLE)

RUSLE (Renard et al., 1997) is an empirical model for predicting sheet and rill erosion by water, and it is the most widely used erosion model with an increasing trend in its use (Alewell et al., 2019; Batista et al., 2019; Borrelli et al., 2021). RUSLE is a revised version of USLE (Wischmeier and Smith, 1978), and it was originally developed for assessing soil loss at field slope/plot scale but has been widely used later as a spatially distributed model. The RUSLE equation is (Eq. 1):

$$A = R \times K \times LS \times C \times P \qquad (1)$$

where A is the annual average erosion (t ha^{-1} yr^{-1}). R is the rainfall-runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}) describing the effect of rainfall and run-off on erosion, and it is defined by the energy intensity of rainfall events. K is the soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}) describing the propensity of soil to detach by the energy of the rainfall and runoff, and it is affected by soil properties, including particle size fractions, organic matter content, soil structure, soil permeability and soil freezing. LS is the topographic factor (dimensionless) describing the effect of the slope length (L) and steepness (S) on erosion. C is the cover-management factor (dimensionless) considering the effects of different cropping and tilling practices on erosion, and it is described by the energy intensity of rainfall, prior-land-use, canopy cover, surface cover, and the surface roughness. P is the support practice factor (dimensionless) accounting for the effect of various support practices on erosion, including contouring, strip cropping, terracing and subsurface drainage. For a more detailed description of RUSLE factors, see Renard et al. (1997).

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The original field slope/plot scale RUSLE predicts soil loss, or the amount of sediment transported to the end of the slope (Renard et al., 1997), whereas the spatially distributed RUSLE predicts soil loss at a spatially discrete unit, such as a grid cell, but does not account for sediment transport between the spatial units. Therefore, the predictions of spatially distributed RUSLE over a landscape are commonly considered as gross erosion predictions. The spatial units are, however, connected in the computation of LS factor.

3.1.2 Erosion control measures and connectivity in RUSLE

The handbook (Renard et al., 1997) of RUSLE demonstrates how different crop and tillage practices, contour tillage, cross-slope strip cropping, buffer strips, filter strips, terraces, and subsurface drainage can be implemented in the model and how they affect the soil loss or the sediment delivery to the end of the slope (Table 1).

The spatially distributed RUSLE, in turn, is more limited in the possibilities for considering structural sediment connectivity, due to lack of description of the sediment transport. Therefore, many of the features in the original field slope/plot scale RUSLE that involve sediment transport cannot be considered, or can be considered only partially, in the spatially distributed RUSLE (Table 1). For example, in the case of buffer strips, their sediment retention cannot be considered, but the effect on erosion reduction at the buffer strip area can be considered.

In the distributed RUSLE, the LS factor is the only element connecting different computational spatial units, and therefore it is the main entry point for considering connectivity within the current RUSLE framework. According to Desmet and Govers (1996), hydrologically isolated areas (in terms of surface runoff) can be considered in the LS factor by calculating the factor separately for the hydrologically isolated land units. For example, field parcels are often surrounded by open ditches, leading to hydrological isolation. Also, surface runoff in certain land cover (e.g., forest) can be considered minimal or non-existent, which may require the consideration of hydrological isolation, for example, when fields are situated downslope a forest (Desmet and Govers, 1996). However, isolation of different land cover is often partial and its consideration in the LS factor is not developed.

Certain sediment connectivity elements may be considered in the spatially distributed RUSLE by modifying the Digital Elevation Model (DEM) underlying the LS calculation. For example, the DEM can be modified to contain ditches and embankments, which in turn will affect the computation of the LS factor and consequently the estimation of gross erosion. The feasibility of the DEM modification, however, depends on the resolution of the DEM and characteristics of the connectivity element.

Otherwise, spatially distributed RUSLE is a highly limited model for considering structural sediment connectivity, and its use is mainly limited to estimation of gross erosion and the effect of different land covers and soil management types on gross erosion. Further considering sediment transport or structural connectivity elements in spatially distributed RUSLE additional approaches are needed to supplement the RUSLE framework. The difference between field slope/plot scale and spatially distributed RUSLE, in terms of sediment connectivity are further described in Table 1.

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Table 1. Comparison of field slope/plot scale and spatially distributed RUSLE in terms of type of predicted erosion and possibility to consider erosion measures and connectivity features.

	Field slope/plot	Spatially distributed
Type of erosion		
Net erosion	Yes	No
Gross erosion	No	Yes
Measures and connectivity features		
Crop/land cover	Yes	Only the effect on gross erosion can be considered
Tillage	Yes	Only the effect on gross erosion can be considered
Contour tillage	Yes	Only the effect on gross erosion can potentially be considered if tillage direction is exactly according to contours. Off-grade contouring cannot be considered
Cross-slope strip cropping	Yes	Only the effect on gross erosion can be considered, the variable land cover difficult to consider in the LS calculation
Buffer strips	Yes	Only the effect on gross erosion can be considered, the variable land cover difficult to consider in the LS calculation
Filter strips	Yes	Only the effect on gross erosion can be considered, the variable land cover difficult to consider in the LS calculation
Terraces	Yes	Can be considered if terraces are hydrologically isolated, and potentially if the flow pathways between terraces can be described in DEM/LS factor and there is no partial reduction of connectivity in the flow pathways
Subsurface drainage	The reduction effect on net erosion can be considered, but the yield estimation at the end of slope is difficult as RUSLE does not include description of transport via sub-surface drainage	The reduction effect on gross erosion can be considered
Hydrological isolation (surface runoff) of land units	Yes	Yes
Topographical features through DEM modification	Can potentially be considered	Yes

3.2 WaTEM/SEDEM

WaTEM/SEDEM is a spatially distributed soil erosion and sediment delivery model. This model was developed by the KULeuven through the aggregation of WaTEM (Van Oost et al., 2000), calculating soil erosion, and SEDEM (Van Rompaey et al., 2001), simulating sediment transport to rivers (Verstraeten et al., 2002). This model is able to calculate the impact of soil conservation and sediment control measures as well as land use changes.

An extensive description of the WaTEM/SEDEM methodology is provided by Van Rompaey et al. (2001), Van Oost et al. (2000) and Verstraeten et al. (2002). In summary, the model has 3 major components:

1. A two-dimensional flow algorithm by Desmet and Govers (1996) simulates the runoff patterns, taking into account the Digital Elevation Map (DEM), fields borders and road infrastructure (Desmet and Govers, 1996). The possibility of considering tillage direction is incorporated by Takken et al. (2001). This flow algorithm is used for the calculation of the upstream area at each location in the landscape and for the routing of the sediment through the landscape.
2. The gross erosion is calculated by an adapted version of the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997). For a two-dimensional approach the slope length factor (L) is replaced by contributing area (A), which is calculated by the flux decomposition algorithm of Desmet and Govers (1996). The contributing area (A) is influenced by the land use and parcel borders of the upstream area by means of the Parcel Trapping Efficiency (PTEF) and parcel connectivity.
3. Sediment is transported along the calculated flow paths to the nearest river. The maximum sediment mass that can be transported by the overland flow is defined as the Transport Capacity (TC) term, which can be assumed to be proportional to the potential rill erosion rate. Transport capacity coefficients (kTC) for different types of landuse need to be assessed by means of calibration. For each grid cell the amount of sediment input is added to the amount of soil erosion in that cell. This total amount is transported downslope when TC is not exceeded. In the other case, the transport is limited to TC and the net erosion is lower than the gross erosion. If the sediment input is higher than TC, sedimentation takes place and no net erosion occurs.

The initial version of this model has been adapted and recalibrated several times. Recalibration of the transport capacity coefficients (kTC) is needed whenever algorithms or parameters are adapted or input data with different accuracy or resolution are used (Van Rompaey et al., 2001; Verstraeten, 2006; Deproost et al., 2018).

3.2.1 Inclusion of land use and topographic changes

WaTEM/SEDEM makes a distinction between arable land, pasture, road infrastructure and build-up areas and rivers. Land use changes (afforestation, permanent grassland, perennial crops) are reflected in the land use map and influence the Parcel Trapping Efficiency (PTEF), parcel connectivity, transport coefficient (kTC) and cover-management factor (C). Crop choices, crop rotations and crop diversification on arable land are translated to the use of PTEF and C-factor in function of the crop. Intercropping, agroforestry and terraces can be translated to an adapted PTEF and P-factor. Crop choices, crop rotations, intercropping, agroforestry, and terracing have no influence on parcel connectivity and kTC as WaTEM/SEDEM only makes a difference between arable land parcels, pasture and forest. Parcel sizes have an important influence by the parcel connectivity and the routing algorithm. Levelling or other topographic changes are incorporated by changing the Digital Elevation Map (DEM).

Table 2 gives an overview of the parameters that can be adapted in WaTEM/SEDEM to model the influence of land use changes.

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Table 2: Types of land use change that can be modelled in WaTEM/SEDEM (WS)

Type of land use change	**Modelled in WS**	**Parameters**
Afforestation	Yes	C-factor, PTEF, parcel connectivity, kTC
Permanent grassland	Yes	C-factor, PTEF, parcel connectivity, kTC
Perennial crops	Yes	C-factor, PTEF, parcel connectivity, kTC
Crop rotations, crop diversification and set-aside	Yes	C-factor, PTEF
Intercropping, agroforestry	Yes	P-factor, PTEF
Parcel size	Yes	parcel connectivity, routing algorithm
Terracing	Yes	P-factor, PTEF

3.2.2 Inclusion of agronomic measures

All crop and crop management related measures can be translated to the C-factor and PTEF. Tillage and sowing practices have a complementary influence on the routing as tillage direction is includes in the algorithm. The increase of soil organic matter can be taken into account when calculating the K-factor. The influence of the reduction of subsoil compaction cannot be modelled by WaTEM/SEDEM.

Table 3 gives an overview of the parameters that can be adapted in WaTEM/SEDEM to model the influence of agronomic measures.

Table 3: Types of agronomic measures that can be modelled in WaTEM/SEDEM (WS)

Type of agronomic measure	**Modelled in WS**	**Parameters**
Cover crops	Yes	C-factor, PTEF
Mulching, crop residue management and tillage practices	Yes	C-factor, PTEF, routing algorithm
Contour farming and sowing practices	Yes	C-factor, PTEF, routing algorithm
Microdams between ridges	Yes	C-factor, PTEF
Soil surface roughness	Yes	C-factor
Reduction of subsoil compaction	No	-
Increase of soil organic matter	Yes	K-factor

3.2.3 Inclusion of buffering measures

Grass buffer strips, hedges, hedgerows and grassed waterways are implemented as a grassland or forest landuse, with corresponding C-factor, PTEF, parcel connectivity and kTC. The impact of retention ponds is calculated for a given sediment trapping efficiency (STE) of the pond or pool. This is the percentage of the incoming sediment that is deposited. Dams in organic material, silt fences, sediment retention ponds and buffering ditches can be considered as ponds with a STE.

Table 4 gives an overview of the parameters that can be adapted in WaTEM/SEDEM to model the influence of buffering measures.

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Table 4: Types of buffering measures that can be modelled in WaTEM/SEDEM (WS)

Type of buffering measure	Modelled in WS	Parameters
Grass buffer strips	Yes	C-factor, PTEF, parcel connectivity, kTC
Hedges and hedgerows	Yes	C-factor, PTEF, parcel connectivity, kTC
grassed waterways	Yes	C-factor, PTEF, parcel connectivity, kTC
Dams in organic materials	Yes	STE
Silt fences	Yes	STE
Sediment retention ponds	Yes	STE
Buffering ditches	Yes	STE

3.2.4 Inclusion of other connectivity elements

Tillage direction can be taken into account by the routing algorithm in WaTEM/SEDEM (Takken et al., 2001), but the influence of wheel tracks cannot be modelled. As each agricultural field has its own identifier, the impact of parcel borders on the routing is also incorporated. The model knows when a parcel border is reached, and the overland flow will follow the parcel border to the lowest point before crossing the parcel border. In the same way, the overland flow on a road will follow the road until a lowest point on the land use infrastructure is reached. WaTEM/SEDEM calculates the sediment export to river segments which can be linked. Ditches can be incorporated when included in the river map. WaTEM/SEDEM does not predict sediment transport within a river, bank erosion or floodplain sediment deposition. The sediment export to a sewer system is not simulated. Topographic changes are reflected in changes in the Digital Elevation Map (DEM).

Table 5 gives an overview of the parameters that can be adapted in WaTEM/SEDEM to model the influence of other connectivity elements that are not implemented as erosion control measures.

Table 5: Types of connectivity elements that can be modelled in WaTEM/SEDEM (WS)

Type of connectivity element	Modelled in WS	Parameters
Tillage direction	Yes	routing algorithm
Wheel tracks	No	-
Parcel borders	Yes	parcel connectivity
Subsurface drainage	Yes	P-factor
Roads	Yes	routing algorithm
Ditches	Yes	river routing
Sewers	No	-
Topographic changes	Yes	DEM

3.3 Process-based models

3.3.1 CASE2

CASE2 (Calculator for Soil Erosion) is a spatially distributed, partially physically-based surface runoff and soil erosion model for individual rainfall events. Model structure and assumptions are based on RMMF and MMMF models (Morgan, 2001; Morgan & Duzant, 2008) – in contrast to these, it is spatially distributed, event-based and focused on infiltration-excess as the dominant process of surface runoff generation. Applications are so far limited to research (e.g., inverse calibration of parameters with measured runoff and soil loss data from experimental data). CASE2 can, in principle, be applied to very large catchments (>50 km² with 1m grid size), but the model is not suitable for runoff and sediment routing within the river network – so larger catchments might be split. Typical spatial resolutions are 1 m or 10 m grids. Multiple events can be concatenated to simulate longer time periods.

Model input parameters are a mixture of physically based and empirical origin.

CASE2 needs a minimum of 8 raster inputs: flow directions (D8 in ArcGIS coding, sewage inlets with ID of sewage inlet), slope (°), land use patterns (ID of land use pattern), rainfall patterns (ID of rainfall pattern), catchment outlet (single cell with value 1), river (ID of river), sewage inlets (ID of connected outlet), sewage outlets (ID of outlet). These rasters have to be in netCDF format, have to share the exact same extent, cell size and resolution. Additionally, text input in .dat format is needed to parametrize all land use and rainfall patterns, and the river that are contained in the respective raster:

- Rainfall pattern ID, timeslice, event rainfall sum, max 30 min intensity of event. Rainfall sum in fixed 5-minute intervals for each rainfall pattern ID.
- Land use pattern ID, timeslice, interception of rainfall, C-factor, canopy cover, ground cover, plant height, soil water content at field capacity, bulk density, erodibility, cohesion, Manning's n, saturated hydraulic conductivity, effective net capillary drive, soil water content at saturation, soil water content before event, concentration factor. An arbitrary number of land use classes can be specified, if parameters are known. Typically, forest, paved, water and grassland areas would each be realized with only 1 land use class and constant parameters, while cropland is parametrized in more detail (multiple management/development stages, different crops etc.).
- River parameters (experimental): ID, timeslice, ground cover, erodibility, slope.

The required inputs related to erosion are: soil cohesion (controls soil detachment by runoff), soil erodibility (controls soil detachment by raindrop impact), proportion of intercepted rainfall (only affects splash detachment), canopy cover (only affects splash detachment), plant height (only affects splash detachment), ground cover (only affects runoff detachment), crop factor (only affects runoff detachment), soil water contents at saturation and before the event, saturated hydraulic conductivity, and effective net capillary drive (control runoff, which is dominating soil detachment in most cases, and limits the available transport capacity)

Optional inputs are: sewage in- and outlets (runoff and sediment are routed through connected in- and outlets, with an optional fraction of Q and E retained in the process), modified flow directions (e.g. tillage controlled runoff pattern (TCRP) according to Takken et al. (2001) can be used instead of standard flow directions – they have to be hydrologically sound), mapped (runoff concentrating) linear features (these have to be burned into a DEM, the resulting modified flow directions can be used in the model), Manning's n for flow velocity and travel time calculations (experimental), concentration factor (intended to represent runoff concentration at a sub-cell level and scales runoff detachment and transport capacity accordingly).

The outputs of CASE2 are in form of raster maps (netCDF format), per timeslice of inputs: runoff, erosion, deposition, flow velocity (very rough estimate), flow path distance (for concentration time analysis or isochrones), and various intermediate results (runoff and splash detachments, total detachment, transport capacity). These raster outputs can be summarized as needed – e.g. SUM per catchment or per parcel, or sample value of routed quantity at catchment outlet – depending on the

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actual goal of the application. Note that for each unique combination of rainfall and runoff pattern, output of the infiltration sub-module at 5 min resolution is saved for checking plausibility.

CASE2 has the advantages of:

- quantifying on-site soil erosion rates and sediment yield (cell, parcel or catchment).
- assessing downstream propagation of erosion mitigation effects that reduce either amount of runoff, runoff detachment or transport capacity
- including “shortcuts” in runoff routing – i.e. “sewage networks” (culverts, pipes)
- having most parameters shared with other erosion models and easily estimated.

The main limitations of CASE2 are:

- the current lack of graphic user interface. All data pre- and post-processing steps have to be done manually or scripted. This is mainly due to different sources for input data (e.g., scripts exist for using Austrian Cadastral map to create land use patterns). Output post-processing will also be highly dependent on the purpose of the modelling effort.
- although the runoff calculation by the infiltration sub-model is performed at high temporal resolution (5 min), the resulting output is lumped into total event runoff, which is then routed. Similarly, runoff and sediment at the catchment outlet are lump sums for the event or chain of events and do not represent actual hydrographs or sedigraphs.
- calibration and validation with measurements at river gauges are difficult to perform and require runoff separation of the hydrograph to make it somewhat comparable
- the model contains no dedicated plant growth and soil water balancing. Plant parameters and soil water contents have to be calculated or measured by other means and inserted accordingly.
- uncertainties about the initial soil water content before an event presumably are the greatest risk to realistic results
- no subsurface processes are considered, the infiltration sub-module uses only 1 layer
- the assumption of infiltration excess runoff makes the model inclined to short high-intensity (=erosive) rainfall events.

CASE2 is able to account for erosion control measures. By adapting the erosion- or runoff-related inputs accordingly, many mitigation measures might be simulated, given that the parameters have been calibrated beforehand, or they are known from literature. The easiest way of including measures is variation of C (=C*P), P, CC, GC parameters, for which literature is usually available. Technical mitigation measures (runoff/sediment traps, e.g. retention ponds) can be simulated by introducing sewage inlets with specified trapping efficiency – which would have to be calibrated. However, no specific -erosion control measures are directly implemented; the user has to do it by pre-processing the input data.

3.3.2 EROSION-3D

EROSION-3D is a physically-based distributed hydrological model which simulates runoff and soil erosion for watersheds (Singh et al., 2022). The model mainly consists of two sub-models: infiltration and erosion (Singh et al., 2022). Schmidt et al. (1999) presents the theoretical concept underlying EROSION-2D/3D erosional processes:

- generation of runoff

- detachment of particles by raindrop impact and runoff
- transport of detached particles by runoff
- routing of runoff and sediment through the catchment
- sediment deposition.

The erosion sub-model represents erosion processes in the form of detachment, transport and deposition of soil particles. The erosion sub-model calculates erosion using the momentum flux approach, which means that erosion will occur if the sum of mobilizing forces acting on particles is larger than that of the resisting forces (Singh et al., 2022). The temporal resolution of the model depends on the rainfall data available, and can range from 1 to 15 min. More than 5x105 grid elements can be processed (Schmidt et al., 1999). Due to its raster basis, EROSION-3D can be linked to various geographical information systems (Schmidt et al., 1999).

As the EROSION-3D model is a raster-based model, it requires a grid-cell presentation of the watershed (Singh et al., 2022). The model offers several modes for the computation of slope and flow distribution (Werner, 2003). All input files must have the same number of rows and columns as well as the same grid size and corner coordinates. The files can have different formats (Arc/Info, Grass, Idrisi and Surfer) (Werner, 2003). In the first step, a DEM file has to be entered as a relief data (Werner, 2003). For each soil parameter a raster file is required (Werner, 2003).

We can insert data (raster) about Erodibility, Roughness, Cover, Particle, Bulk density, Organic content, Moisture and Skin factor of the soil, in the Soil/Landuse window (Werner, 2003). The simulation requires the two pre-processed data sets with relief and soil parameters and a precipitation file (Werner, 2003). Schmidt et al. (1999) lists EROSION-3D input parameters:

- Digital elevation data
- Texture
- Bulk density
- Organic matter content
- Initial soil moisture
- Surface roughness (Manning's n)
- Resistance to erosion
- Rainfall intensity and duration

As outputs, the model provides the sediment budget result (to find out how much sediment has left the watershed for example), and the runoff (Werner, 2003). If we do the long-term calculation which means that the model does the calculation event per event, then the model can give us grid maps of cumulative sediment volume and the erosion/deposition values for each grid cell (Werner, 2003). Schmidt et al. (1999) present the output parameters of the model as:

- Related to the cross-section of a selected grid element:
  - Runoff
  - Sediment discharge
  - Grain size distribution of the transported sediment
- Related to the catchment of a selected grid element:
  - Erosion/deposition
  - Net erosion

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The modeling approach has unlimited transferability to other situations (Singh et al., 2022). EROSION-3D needs fewer input soil parameters as compared with other hydrological models such as WEPP and EUROSEM (Singh et al., 2022). Single event modeling is permitted (Singh et al., 2022). Simulations generated by the model contain a high spatial and temporal resolution (Singh et al., 2022). A detailed parameter database is available for the input used by the model based on extended field results for various types of soils and land uses (Singh et al., 2022). EROSION-3D can recognize several file formats automatically (Arc/Info ASCII files, Grass Raster files and Surfer 6 Grid files) (Werner, 2003). The snow can be modeled by EROSION-3D (Werner, 2003). Usually all spatially distributed data are imported from a Geographical Information System database (e.g., ArcInfo). But direct data access is still possible within the model without using any external software (Schmidt et al., 1999).

To calibrate the model, the influence of the soil parameters and the grid size have to be taken into consideration, or different results will be given by the model (Starkloff & Stolte, 2014). Some instabilities are observed in results obtained by the model when a small catchment is modeled using a large grid size (Starkloff & Stolte, 2014).

No erosion control measures are prescribed. However, rasters can be modified to include erosion control measures.

3.3.3 IBER

Iber is a numerical model for simulating turbulent free surface unsteady flow and environmental processes in rivers. The ranges of application of Iber cover river and open channel flow, dam-break simulation, evaluation of flood inundation, sediment transport and erosion processes and water quality in rivers and non-stratified estuaries (Cea Gómez et al., 2018).

Iber uses the two-dimensional St. Venant equations as a hydrodynamic module (Cea Gómez et al., 2018). For spatial and time resolutions, Cea Gómez et al. (2010) chose for every situation its adequate space resolution and time step (which is dependent on the space resolution). Problems that can be solved by Iber (Ancey, 2020):

- Hydrodynamics of rivers, canals, and hydraulic installations (taking into account structures such as gates, culverts, weirs).
- Dam failure, with breach formation and flood propagation.
- Sediment transport study.
- Runoff and flooding problems, drainage, infiltration.

In order to carry out a simulation with Iber, the following steps are required:

- Create or import a Geometry, which is a representation of the zone of study and its parameters (using lines or arcs…).
- Assign a series of input parameters (bed roughness, turbulence model, etc.).
- Assign boundary and initial conditions.
- Set the problem data (time of calculation, numerical scheme parameters, additional modules requirements).

The inputs related to erosion are:

- Precipitation. Data can be in the form of raster, hyetograph, or rasters interpolation.

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- Soil characteristics of the entire zone. They can be assigned automatically by using raster files in a GIS software (which means that this file will contain the soil characteristics of the entire zone), or manually by identifying multiple zones with different soil characteristics.
- Roughness of land use. Manning coefficients can be added manually or by raster files made with a GIS software.
- Boundary conditions, for both suspended load and bed load transport.

Erosion can be driven by 2 mixture models:

- A single non-cohesive layer (the layer can be eroded as much as it can, which give the model the amount of the erodible material that will be in this layer).
- A non-cohesive layer over a cohesive layer (the process of erosion is then different, because its particles are cohesive).

Sediment distribution is identified as different classes of particles (different diameters, erodibility coefficient, porosity, sediment density, etc.). For every zone a sediment class or a mixture of 2 sediment classes can be prescribed. Moreover, Iber gives the possibility to choose between 2 resuspension models (Hairsine's or Van Rijn's).

As outputs, Iber provides maps for each variable selected: depth, velocity, specific discharge, water elevation, Froude number, etc.

Iber considers if an element is wet or dry, which means that it can handle the transitions between dry and wet zones. The model is relatively simple to understand (while being physically elaborated) and has a dedicated graphic interface. Iber considers many hydrological processes (for example precipitation data that can be defined in many ways, also infiltration losses that can be considered by many ways). Iber is able to suggest model corrections, like bed slope correction.

There is no erosion control measure such as grass or plant height that is implemented in Iber (users have to define them in the input maps beforehand). One of the main limitations of Iber at the present time is the CPU time needed to perform simulations over complex and large spatial domains (of several km²) (García-Feal et al., 2018).

3.3.4 MHYDAS-Erosion

MHYDAS-Erosion is a dynamic and spatially-distributed single-storm erosion model (Gumiere et al., 2011). It has been developed under the OpenFLUID software development environment (https://www.openfluid-project.org/) as a module of the hydrological MHYDAS model (Moussa et al., 2002). It was initially developed for agricultural headwater catchments covering a few km² (Gumiere et al., 2011) but there is nothing inherently limiting the size of the simulated watershed. The catchment is subdivided into homogeneous hydrological units labelled 'SUs' for 'surface units' and 'RSs' for 'reach segments'. Its originality stems from its capacity to integrate the impact of land management practices (LMP) as key elements controlling the sedimentological connectivity in agricultural catchments. The model allows taking into account the main processes contributing to soil erosion such as: interception of rainwater by vegetation, rain splash erosion, overland flow, flow detachment, sediment transport or deposition by rill and interrill processes (Gumiere et al., 2011).

Table 6 shows the input parameters required to run MHYDAS-Erosion, in addition to rainfall pattern.

Most of these parameters can be derived from the following source maps:

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- Topography (raster format)
- Land use/cover (vector format)
- Soil type (vector format)
- Infrastructure (vector format)
- Drainage network (vector format, optional)

The maps have to be in the GIS format.

Table 6: Input parameters necessary to run MHYDAS-Erosion (from Gumiere et al., 2015)

Parameter	Description	Unit
SU		
ks	Saturated hydraulic conductivity	$m\ s^{-1}$
hc	Air entry potential	m
θ_r	Soil residual moisture	$m^3\ m^{-3}$
θ_s	Soil saturation moisture	$m^3\ m^{-3}$
n_{SU}	Manning's roughness coef.	$s\ m^{-1/3}$
A_s	Aggregate stability index	—
N_{rill}	Number of rills	—
W	Rill width	m
$d50_{sed}$	Median sediment diameter	m
τ_c	Critical soil shear stress	*Pa*
Transfcode	Interface type indicator	—
K_r	Rill erodibility	$s\ m^{-1}$
Cetimax	Max transport coef from interrill erosion	—
Strip.width	Vegetated filter width	m
Strip.density	Density of vegetation on the filter	—
RS		
ks	Saturated hydraulic conductivity	$m\ s^{-1}$
n_{RS}	Manning's roughness coef.	$s\ m^{-1/3}$
K_r	Rill erodibility	$s\ m^{-1}$
τ_c	Critical soil shear stress	*Pa*

GROOV'Scape tool (Lagacherie et al., 2021) has been designed to assist the user in data preparation. The optimum time step for rainfall and discharge data for small catchments is about 1–5 min for rainfall and 5–10 min for discharge. These timesteps vary with each catchment and depend also on the type of rainfall and on the SU and RS sizes. Sensitivity analysis of MHYDAS-Erosion has been published (Cheviron et al., 2010, 2011).

The required inputs are:

- Aggregate stability for splash erosion (without unit)
- d50sed: Median sediment diameter (m)
- critical shear stress (Pa)
- rill erodibility (s m-1)
- maximal transport coefficient from interrill erosion (without unit)

Optional inputs are:

- number of rills (without unit)
- width of rills (m)

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- vegetated filter width (m)
- density of vegetation in vegetated filter (without unit)

MHYDAS-Erosion can provide the following outputs:

- time series of discharge and erosion.
- Soil loss map.

MHYDAS-Erosion can:

- Identify source areas of sediment yield.
- Quantify on-site soil erosion rates.
- Include a river routing module.
- Distinguish interrill from rill/channel erosion.

The model does not simulate 'slower' processes such as evapotranspiration, groundwater flow or changes in vegetation because of crop growth. While other connectivity measures could be accounted for, only vegetative filters are implemented as LMP in the current version of MHYDAS-Erosion (Gumiere et al., 2011, 2015).

3.3.5 OpenLisem

OpenLisem (Open Limburg Soil Erosion Model) is a spatially-distributed physically-based model simulating overland flow and erosion in response to individual rainfall events (de Vente et al., 2013). Although initially developed for small catchments in The Netherlands (de Vente et al., 2013), it is increasingly applied world-wide (de Vente et al., 2013). The model can handle any size catchment (tested up to several hundred km²). The model has been designed to simulate runoff and erosion as a consequence of single rainstorms in agricultural catchments of a size ranging from 1 ha up to approximately 100 km² (De Roo & Jetten, 1999). It allows taking into account the main processes contributing to soil erosion such as: interception of rainwater by vegetation, rain splash erosion, water storage in micro-depressions of ground surface, infiltration of ground surface, overland flow, flow detachment, sediment transport or deposition, and runoff on impervious surfaces (Hessel et al., 2003).

The OpenLisem model uses physically-based mathematical relationships where possible. Infiltration and soil water transport are simulated using a direct solution of the Richard's equation, and a kinematic wave approximation is used for runoff routing (De Roo & Jetten, 1999). In some cases, the model uses other equations for the surface flow simulation, like kinematic wave equation, diffusive flow approximation and dynamic flow approximation (Documentation & User Manual, 2018)

OpenLisem needs a minimum of 24 maps depending on the input options selected in the interface, and these maps can be derived from 6 base sources (Documentation & User Manual, 2018):

- Rainfall
- Topography
- Land use/cover
- Soil type
- Infrastructure

The maps have to be in the PCRaster GIS format (must be represented as a raster, if not, PCRaster can convert the maps to raster). The rainfall file must be in ASCII format, which is the most common format

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for text files on computers and the Internet, and most GIS software packages support. In addition, it can be converted to PCRaster maps (Documentation & User Manual, 2018). It is crucial that all input maps share the same extent, resolution, cell size and referencing. PCRaster calculations automatically generate maps with these same properties (Documentation & User Manual, 2018). The grid cell size has to be smaller than 1 ha (Documentation & User manual). Cell sizes should be used that allow spatial variation to be taken into account (De Roo & Jetten, 1999). The optimum timestep for rainfall and discharge data for small catchments is about 1–5 min for rainfall and 5–10 min for discharge. These timesteps vary with each catchment and depend also on the type of rainfall (De Roo & Jetten, 1999). The infrastructure accounted within OpenLisem consists of three different types: Buildings, Roads and Bridges or culverts (Documentation & User Manual, 2018).

The inputs related to erosion are: soil cohesion, aggregate stability (for splash erosion), D50 (Median grain diameter), vegetation height, LAI (Leaf Area Index). Additionally, D90 (90 percent grain diameter), available material for erosion and flooding barrier height can be set.

As outputs, OpenLisem can provide discharge graph (sediment discharge, discharge, concentration and rainfall), soil loss map, flood level map and flow velocity map (Documentation & User Manual, 2018). Apart from the erosion/deposition maps, OpenLisem also generates time series of discharge and erosion for the outlet of the catchment (Hessel et al., 2003).

According to de Vente et al. (2013), OpenLisem can identify source areas of sediment yield, quantify on-site soil erosion rates, include a river routing, and distinguish hillslope erosion from channel erosion. For landuse, a classification can be made using satellite images or national datasets (Documentation & User Manual, 2018). The soil class, that is one of the inputs, can estimate erosion hazard, which means soil erodibility (these maps are publicly available in Europe) (Documentation & User Manual, 2018).

OpenLisem might not predict erosion patterns correctly, therefore further research into the distribution of erosion as simulated by OpenLisem is needed (Hessel et al., 2003). The OpenLisem model does not distinguish in its algorithms between interrill and rill erosion as water simply flows from cell to cell following a drainage network related to ground topography (Baumann et al., 2020). It does not simulate ‘slower’ processes such as evapotranspiration, groundwater flow or changes in vegetation because of crop growth (Documentation & User Manual, 2018).

As erosion control measures, OpenLisem can include maps of grass strips (with grass strip Mannings n), and buffers (Documentation & User Manual, 2018). Although they are not erosion control measures, wheeltracks and roads can be accounted for as connectivity elements (Documentation & User Manual, 2018).

3.3.6 SHETRAN

SHETRAN is a physically-based spatially-distributed hydrological model. It is a model for water flow, solute and sediment transport in river catchments (User Guide and Data Input Manual, 2021). The surface area of the catchment in SHETRAN is usually in the range of one to a few thousand square kilometers. The depth of the subsurface is usually less than 100 m (User Guide and Data Input Manual, 2021). There are two versions of SHETRAN available for download:

- a version with a Graphical User Interface (GUI) in a Windows environment
- the Standard version which uses text-based files (User Guide and Data Input Manual, 2021).

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Using the GUI version in a Windows environment, a basic catchment can be set-up rapidly using data straight from a GIS (User Guide and Data Input Manual, 2021).

The timestep is modified in SHETRAN based on user-defined criteria for rates of rainfall. During dry periods the timestep can be relatively large (a basic timestep of one or two hours is often used). During a storm event rates of infiltration into the ground and surface runoff can both vary rapidly, so a smaller timestep must be used (User Guide and Data Input Manual, 2021). The SHETRAN model is proposed for the hydrological simulation due to its capacity of explicitly considering spatial variability to a level of grid scales (Zhang et al., 2019). The physical processes are modeled by finite difference representations of the partial differential equations for mass, momentum, and energy balances, as well as by empirical equations (Zhang et al., 2019). The basin is discretized by an orthogonal grid network in the horizontal view and by a column of layers at each grid square in the vertical view; the river network is simplified as the links run along the edges of the grid squares (Zhang et al., 2019).

In SHETRAN, the catchment geometry and basic simulation control parameters are set up in the frame module data set (User Guide and Data Input Manual, 2021). SHETRAN has four components: FR frame, WAT water flow, Sediment yield, Solute transport. Every frame is responsible for a component of the model (Naseela et al., 2015). Input data required are: precipitation, potential evaporation, DEM, soil maps and vegetation details. All the input data used in the model are in Arc ASCII grid format (Naseela et al., 2015). The main spatial data inputs are:

- catchment geometry
- ground surface elevation
- river network
- river network elevation
- rainfall/meteorological station distribution
- land cover distribution
- soil distribution and depths
- geology

These inputs can be entered by hand or by a GIS. For erosion, there are a number of parameters to be defined by the user, such as saturated hydraulic conductivity, relative hydraulic conductivity, porosity, specific storage, etc. Initial and boundary conditions need to be defined too (Naseela et al., 2015). The vegetation library of SHETRAN has 7 types (grass, deciduous forest, urban, bare ground, arable, evergreen forest, shrub) (Naseela et al., 2015). The vegetation properties considered by the model are: vegetation cover indices, canopy storage, drainage parameters, root density function, evapotranspiration parameters. The values of these properties are given in the user guide for each type of vegetation (Data Requirements, Data Processing & and Parameter Values, 2021). The soil properties that are taken into consideration are: porosity, soil moisture characteristic function, saturated conductivity, conductivity function relationships, specific storativity. The user guide gives indications about the processing methods of these properties.

Outputs from SHETRAN are ASCII files containing lines of information about net rainfall, soil water potential, total depth of sediment, sediment discharge rate, total depth of sediment, vertical flows, etc.

SHETRAN has the ability to simulate the hillslope erosion processes, and the continuity in simulation necessary for the prediction of land use and climate change (Op de Hipt et al., 2017). In the study of

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Janes et al. (2018) the bank erosion component within the SHETRAN model has been further developed to incorporate both temporal and spatial variability of bank erosion by inclusion of additional controlling factors; removal of bank vegetation and bank collapse after a flood event and subsequent recovery, and channel sinuosity.

For geotechnical stability of the banks, the one-dimensional infinite-slope analysis ignores all upslope, downslope and lateral boundaries, and is therefore subject to uncertainty where groundwater flow or topography produce forces that are significant in directions other than slope-normal (Bathurst et al., 2010). The model subgrid resolution (usually determined by the resolution of the available digital elevation model) limits both the ability of the model to account for topographic controls on landsliding, such as hillslope hollows and gullies, and the representation of local slope angle (Bathurst et al., 2010). The SHETRAN landslide model is likely to be most relevant to basins where slopes are laterally homogeneous and gradient dominates topography as the landslide control (Bathurst et al., 2010).

For the following vegetation types, suggested root density function (RDF) for standard vegetation types is given. The function depends on the depth below ground, in SHETRAN it is input according to the proportion of roots in each cell (arable, bare ground, grass, deciduous forest, evergreen forest, shrub, urban). Shrub hedges, grass hedges, and grass strips can be modeled by the data suggested, but the user must specify the location and enter the data for each measure (User Guide and Data Input Manual, 2021).

3.3.7 WEPP

Water Erosion Prediction Project (WEPP) was initiated in August 1985 to produce a new generation of water erosion prediction technology, for use by federal action agencies involved in soil and water conservation (Flanagan et al., 2007). Developed as a replacement for empirically based erosion prediction technologies (USLE), the WEPP model simulates many of the physical processes important in soil erosion, including infiltration, runoff, raindrop and flow detachment, sediment transport, deposition, plant growth, and residue decomposition (Flanagan et al., 2007). WEPP was designed for continuous simulation of the hydrologic process using a daily time step, and it can handle only one rainfall or irrigation event per day (Kincaid, 2002). A hillslope can be divided into several sections (overland flow elements, or OFEs) that can have different soil properties and management practices (Kincaid, 2002). The WEPP watershed model combines several hillslopes to form a watershed and routes runoff through a series of concentrated flow channels (Kincaid, 2002). The WEPP hillslope model calculates infiltration using the Green–Ampt model. The main parameters in this model are the effective hydraulic conductivity, soil water deficit, and wetting front suction (Kincaid, 2002). Surface runoff is modeled using a simplified kinematic wave procedure (Kincaid, 2002). The hillslope model predicts the amount and duration of runoff and peak runoff rate, which are then used in the hillslope interrill and rill erosion component (Kincaid, 2002). The main parameters in the erosion model are the interrill erodibility, rill erodibility, and critical shear. The erosion model provides suggested values for each soil type (Kincaid, 2002).

According to WEPP user summary (1995), the hillslope component of the WEPP erosion model requires a minimum of four input data files to run:

- a climate file: (CLIGEN V4.0 format) includes daily values for precipitation, temperatures, solar radiation, and wind information.

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- a slope file: includes slope orientation, slope length, and slope steepness at points down the profile
- a soil file (see below)
- a plant/management file: contains all the information needed by the WEPP model related to plant parameters (rangeland plant communities and cropland annual and perennial crops), tillage sequences and tillage implement parameters, plant and residue management, initial conditions, contouring, subsurface drainage, and crop rotations.

Each of the input files has its own file builder, which may be accessed from the WEPP interface (WEPP user summary, 1995). The key parameter for WEPP in terms of infiltration is the Green and Ampt effective conductivity parameter (Ke). This parameter is related to the saturated conductivity of the soil.

WEPP takes a Soil input file (WEPP user summary, 1995): Information on soil properties to a maximum depth of 1.8 meters is input to the WEPP model through the soil input file. The user may input information on up to 8 different soil layers. WEPP internally creates a new set of soil layers based on the original set parameter values. As with the slope file, soil parameters must be input for each and every Overland Flow Element (OFE) on the hillslope profile and for each channel in a watershed, even if the soil on all OFEs is the same. Baseline interrill erodibility (Ki), rill erodibility (Kr), and critical hydraulic shear (τc): three soil erodibility parameters to input in the model. Roughness Coefficients: Calculated according to the type of land: cropland rills, Cropland Interrill Areas, Rangeland Rills, Rangeland Interrill Areas, for every type we can estimate the roughness's friction coefficient (USDA, 1995).

The WEPP computer program produces many kinds of output, in various quantities, depending upon the wishes of the user. The most basic output contains the runoff and erosion summary information, which may be produced on a storm-by-storm, monthly, annual, or average annual basis. The time-integrated estimates of runoff, erosion, sediment delivery, and sediment enrichment are contained in this output, as well as the spatial distribution of erosion on the hillslope (WEPP user summary, 1995).

For applications across a wide range of spatial and temporal scales, Renschler (2003) developed a geospatial interface for WEPP called "GeoWEPP". GeoWEPP is a tool that can be used to generate the watershed's topography characteristics and input them in the WEPP model (Dun et al., 2009). Irrigation can be simulated using the WEPP model.

WEPP has no predefined graphical user interface that can use GIS. Even though the model considers a large number of soil layers, some input data cannot be specified for each layer (e.g., Ks) (Dun et al., 2009).

The model provides an interface where we can add filter strip defined by its width and its management. The model has a front-end interface that's called WEPPSIE, where the user can enter information on one or more of the following possible structures (USDA, 1995):

- Drop spillway
- Perforated riser
- Two sets of identical culverts
- Emergency spillway or open channel
- Rock-fill check dam
- Filter fence or a straw bale check dam.

3.3.8 Summary: Representation of connectivity elements in process-based models

Connectivity elements are represented in the process-based models as spatially distributed points, lines, and surfaces. All the seven process-based models are theoretically able to represent the connectivity elements when they are provided with data having a sufficient spatial distribution (i.e., a small pixel size). However, only a few of these models come with a set of predefined connectivity elements, and when they exist, their variety is quite limited (see WP4.D1). Hence, we are in a situation where the models are able to account for connectivity elements, but this work is to be accomplished by the users. This implies that users have a strong expertise in the effect of connectivity elements and which parameters are altered. Table 7 outlines the parameters altered by the connectivity elements. This also causes an extra-work that will need to be repeated by each user: first, define the parameters altered by each connectivity element of interest, and second, implement these changes in properties at the location of each connectivity element. Considering the expertise and the time needed, this is a limiting factor in developing watershed management scenarios. This tends to limit the variety of connectivity elements accounted for, and the number of scenarios to be simulated. By lowering the quality of watershed management reports, it restricts our capability to preserve soils from erosion. As a conclusion, while the process-based models are able to simulate overland flow and soil erosion, it is difficult to account for connectivity features. This hampers watershed management.

Table 7. Parametrization of various connectivity elements in process-based erosion models.

Connectivity element	**Soil infiltrability**	**Hydraulic roughness**	**Soil cohesion**	**Topography**
Fascine		X		
Grass strip	X	X	X	
Grass hedges		X	X	
Shrub hedges	X	X		
Grassed waterway	X	X	X	
Ponds	X	X		X
Silt fences		X		
Wheel tracks	X	X	X	

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4 Conclusions

This deliverable gives an overview of different types of erosion control measures and connectivity elements, categorized into land use changes, agronomic measures, buffering measures and other connectivity elements. The current implementation of erosion control measures and other connectivity elements in different types of models (RUSLE, WaTEM/SEDEM, process-based models) is summarised. Improvements to this implementation will be elaborated in deliverable WP4-D3.

The overview of erosion control measures and connectivity elements which were presented in this deliverable clearly shows that a diverse range of measures and elements exist. However, their effect on connectivity in the landscape is not always straightforward, as some elements can be either connective or disconnective depending on how they are spatially situated in the landscape. This is why it is important to include them in erosion modelling: this allows accounting for sediment transport and connectivity through the landscape and be able to put erosion control measures in the right places. The reviewed erosion measures and connectivity elements affect different stages of the erosion-transport-deposition process in different ways. Some affect the erosion generation by influencing the erosive impact of rain drops and surface runoff, while others affect the transport and deposition of sediments by influencing the hydrological and sediment connectivity. The physical mechanisms behind these effects are also many and complex, and the capacity of the models to incorporate erosion measures and connectivity elements ultimately depend on how well the models are able to describe and take into account these physical mechanisms in their model structure.

RUSLE is an empirical model for predicting sheet and rill erosion by water. It was originally developed for assessing soil loss at field slope/plot scale but has been widely used later as a spatially distributed model. It is the most widely used erosion model with an increasing trend in its use. The predicted erosion and the possibility to include erosion measures and connectivity elements in the RUSLE depends on whether you use the field/plot scale model or the spatially distributed model. The spatially distributed RUSLE is more limited in the possibilities for considering structural sediment connectivity, due to lack of description of the sediment transport. In this way, the effect of an erosion control measure on sediment retention cannot be considered, but the effect on erosion reduction of the measure can be estimated. Thus, the use of spatially distributed RUSLE is mainly limited to estimation of gross erosion and the effect of different land covers and soil management types on gross erosion. Further consideration of sediment transport or structural connectivity elements in spatially distributed RUSLE approaches are needed to supplement the RUSLE framework.

WaTEM/SEDEM is a spatially distributed soil erosion and sediment delivery model. This model is able to calculate the impact of erosion control measures and can incorporate connectivity elements in its modelling procedure. This is mainly due to the combination of the RUSLE model with a sediment transport model, based on the calculation of a spatially distributed sediment transport capacity. The sediment transport equation uses transport coefficients that are determined by calibration. Erosion control and connectivity elements influence the routing, the subfactors of the RUSLE, the parcel trapping efficiency, the parcel connectivity and the transport capacity coefficients. Buffers are included by means of a sediment trapping efficiency. Possible refinements and improvements of the WaTEM/SEDEM model concern specific and detailed adaptations of the relevant parameters for the various erosion control and connectivity elements and the automatization of the necessary adaptations in elaborated preprocessing scripts.

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Seven process-based models (CASE2, EROSION-3D, IBER, MHYDAS-Erosion, OpenLisem, SHETRAN, WEPP) and their capability of including erosion control measures and connectivity elements were presented in this deliverable. They are all theoretically able to represent the erosion control measures and connectivity elements as spatially distributed points, lines, and surfaces, when they are provided with data having a sufficient spatial distribution. However, only a few of these models come with a set of predefined connectivity elements, and when they exist, their variety is quite limited.

From this presentation of some commonly used erosion models, we see that, although not all models are capable of including them, there is a range of possibilities to incorporate erosion control measures and connectivity elements in the modelling procedure. However, often this requires great expertise from the model user on the effect of connectivity elements and the alteration of parameters for certain erosion control scenarios. Both the incapability of models to include a certain erosion control measure or connectivity element, and the lack of required data or knowledge of the model user to include it, ultimately hamper the capability to model the mitigation of soil erosion and sediment transport in the landscape.
Recommendations on how to improve the representation of erosion control measures and other connectivity elements in models will be further elaborated in deliverable WP4-D3 of the SCALE project.

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