Towards climate-smart sustainable management of agricultural soils

SCALE

Managing Sediment Connectivity in Agricultural Landscapes for reducing water Erosion impacts

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Guideline on how to improve the representation of erosion control measures and other connectivity elements in models

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1. Introduction

In surface hydrology, connectivity is a key concept to understand and predict the fluxes of water and associated materials. In a previous report (SCALE, 2023), we reviewed the connectivity elements and the erosion measures (CEEM) in the landscapes that could affect water and sediment connectivities. We also looked at the actual implementation of these CEEM in various soil erosion models. We concluded that 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.

In the present report, we outline the CEEM, and then identify them in a structured classification. This classification can help in selecting the proper CEEM for a specific scenario. Based on this classification, ways to include the CEEM that are not built-in in the models are suggested. That allows for extending the capacities of the users to better use erosion models. It helps answering issues such as: how can I represent the CEEM C1 in the model M1? CEEM C2 is similar to CEEM C3; what should I alter in the model M2 to account for it? Hence, the present report should result in better simulations, whether they are for a current situation or for future scenarios.

When overland flow causes soil erosion, not only mineral particles are removed from the field and transported downstream. Associated materials, such as organic carbon are relocated too. It means the modeling effort could require considering the water, the mineral particles, plus organic carbon. Currently, there is no standard way to do this. In the present report, based on existing models of soil erosion and existing models of organic carbon fate, a proposition for a coupled modelling is given (section 4).

2. Connectivity elements and erosion measures

2.1. Considered connectivity elements and erosion measures

This section gives an outline of the connectivity elements and erosion measures (CEEM) considered in the following parts of the report. A thorough description of these CEEM is given in SCALE (2023).

CEEM are divided in land use changes, agronomic measures, buffering measures and other connectivity elements (Table 1).

Table 1: Short description of the considered connectivity elements and erosion measures

Land use changes	
Afforestation	Typically, the conversion of an agricultural land or a pasture to a forest. It results in an increase of the soil cover.
Permanent grassland	The permanent soil cover of grassland generally means it is less prone to erosion. Grassland decreases the runoff formation, velocity and concentration, and the high vegetation cover reduces the runoff’s ability to cut into the soil and create rills.

Perennial crops	Perennial crops produce more ground cover, have longer growing seasons, and have more extensive root systems, in comparison to annual crops.
Crop rotations, crop diversification and set-aside	Some crops are more erosion-prone than others. Hence, crop rotation allows for altering the risk of erosion. Crop diversification at landscape level results in a more diverse patchwork of land use, altering the risk of erosion too. Set-aside means that parts of arable land are temporarily taken out of production which can lower soil erosion (if transformed into a less erodible land use).
Intercropping	Intercropping is defined as the relay or simultaneous cultivation of two or more crops on the same field. Crops can be freely mixed, seeded in alternative rows (simultaneously or after the main crop).
Agroforestry	Agroforestry indicates land use in which perennial woody plants and agricultural crops (or animals) are intentionally housed on the same parcel of land, mainly in some form of spatial arrangement.
Parcel size	Soil loss can be influenced by the parcel size, as longer slopes lead to increased erosion due to accumulation of runoff.
Terracing	Terracing involves some land leveling and results is a stepping landscape.
Agronomic measures	
Cover crops	Cover crops are designed to provide soil cover during the winter season and fallow periods. Hence the soil does not remain bare. Roots of the cover crop are efficient too in limiting erosion.
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. Residue management defines the way crop residues of the preceding crop are managed. If tillage is to be applied, different practices can be selected. Mulching and no-till practices cause transport limiting conditions for water and sediment due to higher soil cover, increased soil aggregate stability, lower runoff generation and lower flow velocity.
Contour farming and sowing practices	Contour farming is the practice of tillage, planting, and other farming operations performed along the contour of the field slope. The sowing practice (single, double, widespread, inline) can alter the effects of a given crop on soil erosion.
Micro-dams between ridges	In a ridge-and-furrow tillage system, small earthen dams can be build-up in the furrows. The implementation of micro-dams has been shown to reduce surface runoff and sediment yield.
Soil surface roughness	A rougher soil surface can present more depressions and barriers, and hence favor infiltration and trap sediments. Soil surface roughness can be related to clodiness, ridges-and-furrows, etc.

Reduction of subsoil compaction	The increase of bulk density of the subsoil can be remediated in the long term by mechanical or biological techniques. It can be avoided by lowering wheel loads and tyre pressures, as well as by avoiding excessive soil moisture conditions for field operations.
Increase of soil organic matter	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). It has diverse effects on soils, including decreasing their erodibility.
Buffering measures	
Grass buffer strips	Buffer strips made of grass can be implemented within fields, at field margins and adjacent to rivers. Different widths and species can be used.
Grass and shrub hedges	Grass hedges are vegetative barriers constructed from different species of grass. Both aerial parts and roots have important effects. A shrub hedge consists of a line of shrubs (possibly with some trees), growing on a base of herbaceous vegetation.
Grassed waterways	Grassed waterways are grassed areas specifically sets in thalwegs, along the waterways. They are planted with sod-forming grasses which reduce runoff, sediment transport and gully formation by slowing water flow.
Fascines	A fascine is a vegetative barrier made of bunches of stems (fagots) held in place by two lines of posts. They are positioned across the flow.
Dams in organic materials	Dams in organic materials are vegetative barriers made of plant residues. They can be constructed by using wood chips, coconut-fibers bales or straw bales.
Silt fences	Silt fences are made of wooden posts and geotextile fabric. They are positioned across the flow.
Sediment retention ponds	Sediment retention ponds are located upstream of rivers. Their design should control the runoff during most events without overtopping or using the emergency outflow, and delay runoff and increase sediment settling duration.
Other connectivity elements	
Tillage direction	Tillage induces a ridge-and-furrow roughness that can alter the flow pattern on a field scale. The flow direction can be very different from the topographic slope.
Wheel tracks	Wheel tracks are the 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.

Parcel borders	Parcel borders often represent a change in vegetation/surface cover and can thus act as a barrier to hinder water and sediment flow. Although, concentrated waterflow can also be observed along parcel borders, potentially creating higher erosion rates.
Subsurface drainage	Subsurface drainage is the implementation of porous pipes inside the soil to drain the water from the soil, and hence improve the drainage conditions in the field.
Roads	Roads can act as both connective and disconnective elements in the landscape. Disconnectivity may occur when the road acts as a parcel border and causes deposition upslope of the road. Increased connectivity may occur as roads generate concentrated runoff from their denser surface and lead the runoff into drainage networks, streams or downhill fields.
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. Ditches collect surface and subsurface water.
Topographic changes	Fields have natural variation in the surface topography. 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.

2.2. Classification based on geometry (line, surface)

When dealing with modelling at the field scale, catchment scale, and regional/national scale, the presence of connectivity elements and erosion measures (CEEM) are to be represented in a GIS.

The geometry of CEEM is always a surface when looking at close range. When dealing with national scales, they may not be represented explicitly because of the too coarse resolution. Instead, they can be represented, for example, as a percentage of the surface area.

Small-catchment scale is often used for soil erosion management scenarios because it allows to consider the diversity of the landscape while still representing it explicitly (Table 2). The spatial resolution used in the modelling effort has an effect of the actual representation of a CEEM. If the resolution is coarse, some CEEM that we typically consider as a line (such as a ditch) may cover a significant surface, while some others that are typically considered as a surface (such a sediment retention pond) may cover a single grid cell. Conversely, if the resolution is fine, a line may be indeed represented as a surface: a grass buffer strip might be represented with its actual surface. These resolution issues are always a challenge when modelling soil erosion: the parametrization of a given CEEM may need to be modified depending on the resolution in use.

Table 2: Typical geometry of the considered connectivity element or erosion measure, when carrying out modelling at the small catchment scale

	Line	**Surface**
Land use changes		
Afforestation		●
Permanent grassland		●
Perennial crops		●
Crop rotations, crop diversification and set-aside		●
Intercropping		●
Agroforestry		●
Parcel size		●
Terracing		●
Agronomic measures		
Cover crops		●
Mulching, crop residue management and tillage practices		●
Contour farming and sowing practices		●
Micro-dams between ridges		●
Soil surface roughness		●
Reduction of subsoil compaction		●
Increase of soil organic matter		●
Buffering measures		
Grass buffer strips	●	
Grass and shrub hedges	●	
Grassed waterways	●	
Fascines	●	
Dams in organic materials	●	

Silt fences	●	
Sediment retention ponds		●
Other connectivity elements		
Tillage direction		●
Wheel tracks	●	
Parcel borders	●	
Subsurface drainage		●
Roads	●	
Ditches	●	
Topographic changes		●

2.3. Classifications based on the effects

Four classifications are provided, in the form of tables:

- Does the connectivity element or erosion measure affect both water and sediment transfers, or mostly on sediment? (Table 3).
- Main effects of the connectivity element or erosion measure on water (Table 4).
- Main effects of the connectivity element or erosion measure on sediments (Table 5).
- Main effects of the connectivity element or erosion measure on connectivity of both water and sediments (Table 6).

The specific effects (and their magnitudes) can vary depending on the context (soil, climate, etc.) and the implementation. Relevant literature references are given in SCALE (2023).

These classifications will help users of models to estimate beforehand the potential effects that the CEEM present in their study area may have. They could use them when building management scenarios. In all cases, the type and location of a CEEM should be thought about before actually running a soil erosion model, whatever its choice. We wish that the tables below will help in achieving this purpose.

CEEM have effects on the water and sediment transfers. Some have a combined effect, i.e. affecting both water and sediment transfers, while others have an effect on sediment only (Table 3). Having an effect on “sediment only” means that the flow of water is mostly unaffected. No features were identified as having an effect on water only, likely because, when the water flow is affected, its sediment content is too.

When a given CEEM affects water, it may be related to several effects, namely infiltration, surface storage, flow velocity or flow direction (Table 4). Accounting for these effects in the modelling effort consists in altering the suitable variables. For infiltration, it can be the infiltration rate, the porosity, etc. For surface storage, it should alter the surface storage capacity. Flow velocity is often controlled by the hydraulic roughness in physical-based models, but it can translate into another variable in other model types. Finally, flow direction is related to the topography of the study area. For most of CEEM and modelling effort, altering the digital topography is unlikely to bring the expected results: the resolution of the topography is often too coarse to represent the CEEM (like the tillage direction). In such a case, the modeller can decide not to account for this effect of the CEEM, to the cost of higher inaccuracies in the results, or to select a model able to account for this effect, to the cost of learning the usage of a new model. In the long term, that last choice can be rewarding.

When a CEEM affects sediments, it can be because this CEEM increases or because it decreases the detachment of the soil at the surface (Table 5). So, this affects the source of sediments. The variable related to detachment can be named “erodibility” or “critical shear stress” depending on the model. A CEEM can alter the capability of the water flow to transport sediments. So, this concerns only sediments that are already in the flow and are coming from upstream. In erosion models, this can be parametrized as a transportation capacity, a maximum concentration or a settling velocity.

Finally, a CEEM can alter the connectivity intensity (connecting or a disconnecting effect) or its connectivity direction (Table 6). In most cases, a disconnecting effect was found, and, indeed, when trying to limit water or sediment transfers, we thought of disconnecting the upstream from the downstream. A few CEEM may be able to alter the connectivity direction, i.e. change the direction the water flow is moving. That can be used to make flow paths more complex, to direct the flow in a specific location purposely.

Table 3: Does the connectivity element or erosion measure affect both water and sediment transfers, or mostly on sediment?

	Mainly on sediment	**Water and sediment**
Land use changes		
Afforestation		●
Permanent grassland		●
Perennial crops		●
Crop rotations, crop diversification and set-aside		●
Intercropping		●
Agroforestry		●
Parcel size		●
Terracing		●
Agronomic measures		
Cover crops		●
Mulching, crop residue management and tillage practices		●
Contour farming and sowing practices		●

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Micro-dams between ridges		●
Soil surface roughness		●
Reduction of subsoil compaction		●
Increase of soil organic matter		●
Buffering measures		
Grass buffer strips	●	
Grass and shrub hedges	●	
Grassed waterways		●
Fascines	●	
Dams in organic materials	●	
Silt fences	●	
Sediment retention ponds		●
Other connectivity elements		
Tillage direction		●
Wheel tracks		●
Parcel borders		●
Subsurface drainage		●
Roads		●
Ditches		●
Topographic changes		●

Table 4: Main effects of connectivity elements and erosion measures on water.

	Infiltration	**Surface storage**	**Flow velocity**	**Flow direction**
Land use changes				
Afforestation	●	●	●	
Permanent grassland	●		●	

Perennial crops	●		●	
Crop rotations, crop diversification and set-aside	●	●	●	
Intercropping	●		●	
Agroforestry	●		●	
Parcel size			●	
Terracing	●	●	●	●
Agronomic measures				
Cover crops	●		●	
Mulching, crop residue management and tillage practices	●		●	
Contour farming and sowing practices	●	●	●	●
Micro-dams between ridges	●	●	●	
Soil surface roughness	●	●	●	
Reduction of subsoil compaction	●			
Increase of soil organic matter	●			
Buffering measures				
Grass buffer strips			●	
Grass and shrub hedges			●	
Grassed waterways	●		●	
Fascines			●	
Dams in organic materials			●	
Silt fences			●	
Sediment retention ponds		●	●	
Other connectivity elements				
Tillage direction	●	●	●	●

Wheel tracks	●	●	●	●
Parcel borders			●	●
Subsurface drainage	●			
Roads			●	●
Ditches			●	●
Topographic changes			●	●

Table 5: Main effects of connectivity elements and erosion measures on sediments.

	Detachment	**Transport**
Land use changes		
Afforestation	●	●
Permanent grassland	●	●
Perennial crops	●	●
Crop rotations, crop diversification and set-aside	●	●
Intercropping	●	●
Agroforestry	●	●
Parcel size		●
Terracing	●	●
Agronomic measures		
Cover crops	●	●
Mulching, crop residue management and tillage practices	●	●
Contour farming and sowing practices	●	
Micro-dams between ridges		●
Soil surface roughness	●	

Reduction of subsoil compaction		
Increase of soil organic matter	●	
Buffering measures		
Grass buffer strips		●
Grass and shrub hedges		●
Grass waterways	●	●
Fascines		●
Dams in organic materials		●
Silt fences		●
Sediment retention ponds		●
Other connectivity elements		
Tillage direction		
Wheel tracks	●	●
Parcel borders		●
Subsurface drainage		
Roads	●	●
Ditches		●
Topographic changes	●	●

Table 6: Main effects of connectivity elements and erosion measures on connectivity of both water and sediments.

	Connecting	**Disconnecting**	**Altering connectivity direction**
Land use changes			
Afforestation		●	
Permanent grassland		●	
Perennial crops		●	

Crop rotations, crop diversification and set-aside		●	
Intercropping		●	
Agroforestry		●	
Parcel size	●	●	
Terracing		●	
Agronomic measures			
Cover crops		●	
Mulching, crop residue management and tillage practices		●	
Contour farming and sowing practices		●	●
Micro-dams between ridges		●	
Soil surface roughness		●	
Reduction of subsoil compaction		●	
Increase of soil organic matter		●	
Buffering measures			
Grass buffer strips		●	
Grass and shrub hedges		●	
Grassed waterways		●	
Fascines		●	
Dams in organic materials		●	
Silt fences		●	
Sediment retention ponds		●	
Other connectivity elements			
Tillage direction			●
Wheel tracks	●		

Parcel borders	●	●	●
Subsurface drainage		●	
Roads	●	●	●
Ditches	●		●
Topographic changes	●	●	

3. Accounting for these erosion measures as (dis)connecting elements in models

The features, capabilities and limitations of each considered model is given in SCALE (2023), please see this deliverable for further information on each model. Section 3.1 looks at how connectivity elements and erosion measures (CEEM) are currently accounted for in models. The readily-available CEEM for each model is given in Table 7.

Section 3.2 goes behind the prescribed features of the models. It shows that CEEM that were not considered during model development can be included, in numerous cases, in the modelling effort (Table 9). This means that connectivity can be simulated even with models that did not have this concept included.

3.1. How connectivity elements are currently included in models

RUSLE capabilities can be separated depending on the version used (Table 7). The original version (Renard et al. 1997) is one at a field slope/plot scale, and assesses net erosion. However, spatially-distributed versions of RUSLE are popular. They are grid-based and estimates gross erosion. These two versions also differ in their capabilities to account for connectivity elements and erosion measures. While the field slope/plot scale RUSLE accounts for all listed land use changes except agroforestry, the spatially-distributed RUSLE does not readily include intercropping and terracing too. A similar observation is valid for agronomic measures, where the field slope/plot scale RUSLE accounts for five of the seven listed CEEM and the spatially-distributed RUSLE accounts only for 4 CEEM. Both versions of RUSLE have a limited capability to include buffering measures (only grass buffer strips can be represented, and only for the field slope/plot scale RUSLE) and other connectivity elements (subsurface drainage is the only element accounted by both versions; tillage direction only by the field slope/plot scale RUSLE, and the topographic changes only by spatially-distributed RUSLE). It means that, beyond the estimation of net or gross erosion, choosing the field slope/plot scale or spatially-distributed RUSLE limits the CEEM to be accounted for. If some CEEM are already present on the field (simulation of the current situation) or intended to be incorporated (simulation of a management scenario), it would be preferable to choose the RUSLE version that includes them.

WATEM-SEDEM uses the a spatially-distributed RUSLE to estimate the potential amount of soil loss. Then, following the flow network, sediments are rooted downstream, where they may deposit. It can account, to some extend, for land use changes, agronomic measures, buffering measures and other connectivity elements (Table 7).

Among the listed models, the physically-based ones (i.e. CASE2, Iber, Mhydas-Erosion, OpenLisem, SHETRAN and WEPP) present some common features (Table 7): among the already-included connectivity elements and erosion measures, the land use changes are well incorporated. Most land use changes can be readily simulated, if one has the typical parameters of these land uses (basically: hydraulic roughness, infiltrability and erodibility). These land use CEEM are polygons in these models, and each polygon can have its own spatial extent and parameter values based on the land use. The land uses that can be readily simulated are the ones that are homogeneous inside a given polygon. "Parcel size" is a special case of land use, which is directly related to the polygon geometry, a feature that the model user is expected to provide in all cases of physical-based models. Agroforestry may be more demanding in terms of parametrization because it is characterized by large heterogeneities inside a given field. So, here, the limitation is about representing the spatial heterogeneity related to agroforestry in a "homogeneous" polygon. Terracing is also a land use that involves large heterogeneities. The introduction of terraces involves modifications of the topography that go beyond the typical inputs expected when using physically-based models. Agronomic measures are not set by default in the models, except for the soil surface roughness. To be accounted for, specific modification of the land use parameters are to be implemented. This is an extra task for the model users. Even the agronomic measure "soil surface roughness" requires a specific thinking from the model users because it could alter several parameters in the land use. Indeed, "soil surface roughness" does not simply alter the hydraulic roughness. Finally, among the buffering measures and other connectivity elements, only a few are readily available.

Table 7: Connectivity elements and erosion measures readily included in models.

	RUSLE (field slope/plot scale; net erosion)	**RUSLE (spatially distributed; gross erosion)**	**WaTEM/SEDEM**	**CASE2**	**Erosion3D**	**Iber**	**MhydasErosion**	**OpenLisem**	**SHETRAN**	**WEPP**
Land use changes										
Afforestation	●	●	●	●	●	●	●	●	●	●
Permanent grassland	●	●	●	●	●	●	●	●	●	●
Perennial crops	●	●		●	●	●	●	●	●	●
Crop rotations, crop diversification and set-aside	●	●		●	●	●	●	●	●	●
Intercropping	●			●	●	●	●	●	●	●
Agroforestry										
Parcel size	●	●	●	●	●	●	●	●	●	●
Terracing	●									
Agronomic measures										
Cover crops	●	●								

Mulching, crop residue management and tillage practices	●	●								
Contour farming and sowing practices	●		●							
Micro-dams between ridges										
Soil surface roughness	●	●								
Reduction of subsoil compaction										
Increase of soil organic matter	●	●								
Buffering measures										
Grass buffer strips	●							●		●
Grass and shrub hedges										
Grassed waterways										
Fascines										●
Dams in organic materials										●
Silt fences										●
Sediment retention ponds			●							
Other connectivity elements										
Tillage direction	●		●							
Wheel tracks								●		
Parcel borders			●							
Subsurface drainage	●	●								
Roads			●					●		
Ditches							●			
Topographic changes		●	●							

3.2. How to better include connectivity elements in models

The previous subsection showed that, while land use changes can generally be accounted for, agronomic measures, buffering measures and other connectivity elements are scarcely available in soil erosion models. This may lead to incomplete representation of the actual connectivity elements and erosion measures or limit their inclusion in managements scenarios. In both cases, this may mean inaccurate simulation or incomplete scenarios, hampering our capability to simulate soil erosion adequately.

However, as shown in the following subsections, even connectivity elements and erosion measures that are not readily available can, in many cases, be included in the modelling effort. We first consider RUSLE, then WaTEM/SEDEM, and finally the physically-based models.

3.2.1. RUSLE

For the spatially distributed (R)USLE the implementation of sediment transport in the modelling framework is essential for considering erosion measures and connectivity elements more broadly. In the following we describe three methods for addressing this limitation. These are based on sediment delivery ratio (SDR), transport capacity (T), and a combination of index of connectivity (IC) and SDR. In the following we provide a general description of approaches and some modelling tools where these methods have been implemented. The method based on IC and SDR was found promising as it is easily implemented with (R)USLE as a post-processing method, the additional data requirements are small, it allows implementation of various land cover types that affect sediment connectivity, and it has potential for further development. Therefore, we also introduce a case study on the use of IC and SDR combination in agricultural settings. The research on implementing specific erosion measures and connectivity elements (e.g., buffer strips) within the combined IC and SDR approach is limited.

SEDIMENT DELIVERY RATIO (SDR)

SDR describes the fraction of gross erosion of a computational unit (e.g., a grid cell) that reaches the outlet of the modelled area. The sediment yield (Y) of a computational unit can be estimated by multiplying the SDR with gross erosion estimated by RUSLE. The Sediment Delivery Distributed (SEDD) Model (Ferro and Porto, 2000) uses this approach, but also other formulations of SDR exist. In the SEDD model the SDR is described as a function of travel time as

$$SDR_i = \exp(-\beta t_i) = \exp\left(-\beta \frac{l_{p,i}}{\sqrt{s_{p,i}}}\right) = \exp\left[-\beta\left(\sum_{j=1}^{N_p} \frac{l_{p,i}}{\sqrt{s_{p,i}}}\right)\right]$$

where β is a watershed-specific coefficient, t_i is the travel time (h) for cell I to the nearest outlet, $l_{p,I}$ is the length of the hydraulic path, $s_{p,I}$ slope of the hydraulic path, and N_p is the number of computational units along the hydraulic path j. The Y (t yr^{-1}) can be then computed as

$$Y_i = SDR_i A_i SU_i$$

where A_i is the gross erosion estimated by (R)USLE (t ha^{-1} yr^{-1}), SU_i is the area (ha) of computational unit.

In the SEDD model the effects of erosion measures and connectivity elements on gross erosion are considered in the gross erosion estimate of (R)USLE (A). For considering effects of these measures and elements on sediment transport at a specific computational unit, a potential entry point would be the travel time parameter (t). However, we don’t know whether this would be feasible, and we are not aware of approaches that would have tried to implement erosion measures and connectivity elements in SEDD in such a way.

TRANSPORT CAPACITY (T)

T approach has been implemented at least in Unit Stream Power - based Erosion Deposition (USPED) model (Mitasova et al., 1996) and in the Water and Tillage Erosion Model and Sediment Delivery Model (WaTEM/SEDEM) (Van Oost et al., 2000; Van Rompaey et al., 2001). T describes maximum amount of sediment that is transported through a computational unit (e.g., a grid cell) and how much of the sediment is deposited. This information can then be used for identifying erosion and deposition areas and downstream routing the sediments. In USPED the T (t ha^{-1} yr^{-1}) is described as

$$T = RKLS_T CP$$

where R is rainfall-runoff erosivity factor (MJ mm ha^{-1} hr^{-1} yr^{-1}), K is the soil erodibility factor (Mg hr^{-1} MJ^{-1} mm^{-1}), C is the cover-management factor (dimensionless), P is the support practice factor (dimensionless); and LS_T is the topographic sediment transport factor described as

$$LS_T = U^m (sin\, b)^n$$

where U is the upslope contributing area per unit width (m^2 m^{-1}), β is the slope, and m and n are empirical coefficients. The net erosion/deposition (ED, t ha^{-1} yr^{-1}) can then be computed as

$$ED = \nabla(TS_0) = \frac{\partial(T cos\alpha)}{\partial x} + \frac{\partial(T sin\alpha)}{\partial x}$$

where α is the aspect of the topography or the direction of flow, and S_0 is the unit vector in the steepest slope direction.

The entry point for considering the effects of erosion measures and connectivity elements in USPED would be the topographic sediment transport factor (LS_T). However, we don't know whether this would be feasible, and we are not aware of approaches that would have tried to implement erosion measures and connectivity elements in SEDD in such a way.

In WaTEM/SEDEM the erosion is calculated as in RUSLE and the displaced sediments are routed to downstream using T (t ha^{-1} yr^{-1}), which is formulated as

$$T = k_{tc} E_{PRG} = k_{tc} RK(LS - 4.1 \cdot S_{IR})$$

where k_{tc} is the transport capacity coefficient (-), E_{PRG} is the potential gully erosion, R is rainfall-runoff erosivity factor (MJ mm ha^{-1} hr^{-1} yr^{-1}), K is the soil erodibility factor (Mg hr^{-1} MJ^{-1} mm^{-1}), LS is the combined slope length and steepness factor (dimensionless), and S_{IR} is the inter-rill slope gradient (m m^{-1}) described as

$$S_{IR} = 6.8 \cdot {S_g}^{0.8}$$

where S_g is the slope gradient (m m^{-1}). The routing of sediments is then done using single flow-direction routing, which limits the routed sediments according to the T. The effects of erosion measures and connectivity elements can be considered in the transport capacity coefficient (k_{tc}) of T, but the coefficient values need to calibrated for a specific combination of computational grid size and routing method (Van Rompaey et al., 2001). The use of WaTEM/SEDEM for simulating riparian buffer strips is demonstrated, for example by Verstraeten et al. (2006).

INDEX OF CONNECTIVITY (IC) COMBINED WITH SDR

The approach combining IC (Borselli et al., 2008) and SDR (e.g., Zhao et al., 2020) has been implemented at least in Integrated Valuation of Ecosystem Services and Tradeoffs (InVest) model (Natural Capital Project, 2022), and in adapted form in SedInConnect (Crema and Cavalli, 2018), and used independently in several applications. The IC is an index describing sediment and water flux connectivity based on topographical and land cover information. High IC values describe areas with high degree of connectivity compared to lower IC values. The IC [-] is calculated as

$$IC = log_{10}(\frac{D_{up}}{D_{down}})$$

where D_{up} (-) is the upslope factor and D_{down} (-) is the downslope factor. D_{up} is

$$D_{up} = \bar{W}\bar{S}\sqrt{A}$$

where W (-) is the mean weighing factor (upslope area), S (m m^{-1}) is the mean slope of the upslope area and A (m^2) is the upslope area. D_{down} is

$$D_{down} = \sum_{i=1}^{n} \frac{d_i}{W_i S_i}$$

where d_i (m) is the length of ith pixel (along the downslope flowpath), W (-) is the weighing factor and S_i (m m^{-1}) is the slope of ith pixel. W describes the impacts of vegetation cover and land use on the connectivity. W is often parameterized by using the RUSLE C factor (e.g., Borselli et al., 2008).

The SDR is then calculated as

$$SDR_i = SDR_{max}\,(1 + \exp\left(\frac{IC_0 - IC_i}{K_{IC}}\right))^{-1}$$

where SDR_{max} (-) is the maximum SDR (from 0.0 to 1.0), IC_i (-) is the IC value of the ith grid cell, IC_0 (-) and K_{IC} (-) are empirical parameters. The sediment delivery Q_i (t yr^{-1}) can be then computed as

$$Q_i = SDR_i A_i SU_i$$

where A_i is the gross erosion estimated by (R)USLE (t ha^{-1} yr^{-1}), SU_i is the area (ha) of computational unit.

In the combined IC and SDR approach, the effects of erosion measures and connectivity elements on sediment transport can be considered in the weighing factor (W). W is often parameterized by using the RUSLE C factor. For example, Borselli et al. (2008) and Foerster et al. (2014) considered W=C for different land cover types in catchment scale simulations. However, we are not aware of combined IC and SDR modelling approaches that would have analyzed in detail the implementation of buffer strips or other erosion measures or connectivity elements in W within-field scale, and this direction requires further research.

EXPLORATION OF COMBINED IC AND SDR APPROACH IN FINNISH AGRICULTURAL LANDS

First large-scale structural connectivity estimate in Finland was conducted at agricultural lands of two topographically contrasting sub-catchments (Tähtikarhu et al., 2022). RUSLE data of Räsänen et al. (2023) was combined with IC and SDR to estimate the sediment yield from the field parcels. Furthermore, we estimated the proportion of field areas that are structurally connected to open ditches and streams, based on flow direction and flow accumulation computations. The computations were subjected to sensitivity analyses and were done with the ArcGIS software (e.g., Borselli et al., 2008). The computations were conducted with the spatial resolution of 2×2 m^2. The data and the analyses are shortly presented below and are described in more detail in Tähtikarhu et al. (2022).

The studied sub-catchments of Aurajoki (60.12°N, 23.74°E) and Mustionjoki (60.53°N, 22.44°E) located in southwestern Finland. The Mustionjoki sub-catchment had the area of 116 km^2 while the area of the Aurajoki sub-catchment was 147 km^2. Clay soil was the dominating soil type at both sub-catchments. The topography of the Mustionjoki sub-catchment was gently undulating (mean slope 4.9%), while the topography of the Aurajoki sub-catchment was gentle with steep slopes near the streams and rivers (mean slope 2.7%). Spring and winter cereal production were the dominant agricultural activities within the sub-catchments (about 60% of the agricultural land area, according to data from Finnish Food Authority), but also perennial grasses and hay-type crops were grown. The agricultural fields are well drained in Finland, and they are typically surrounded by open ditches and artificial subsurface drainage is a common practice.

Agricultural field parcels borders were taken from the field parcel data of the Finnish Food Authority. The data contains vectorized field parcels and covers nearly all agricultural lands of Finland. The average parcel area was 2.8 ha and 3.4 ha at the Aurajoki and Mustionjoki sub-catchment, respectively. The parcel boundaries typically closely match with the open ditches or stream locations. Thus, in our computations, the ditches and streams were represented by pixels which were located adjacent to the parcel boundaries.

Elevation differences were described by a 2×2 m^2 lidar-based digital elevation model (DEM), taken from National Land Survey of Finland (National Land Survey of Finland, 2020). The root mean square error (RMSE) of the DEM is <0.3 m on slopes ≤47%. Only 0.5% of the studied land area had the slope >47%. Mean RMSE has been shown to be 0.11 m and lower on the mildest slopes (Oksanen, 2013). DEMs typically include topographical depressions (sinks) of different sizes. Some sinks are too small to practically induce structural disconnectivity and some sinks are caused by the inaccuracies in the data. Since the sinks can influence the connectivity computations, we determined threshold values which determine which sinks can practically induce structural disconnectivity. Sinks with a lower depth than a threshold value, were considered as small depressions or noise and were thus filled in the DEM. We estimated 0.1 m to be a plausible threshold value (Tähtikarhu et al., 2022). However, to understand the sensitivity of our results to the choice of the threshold, we produced 4 different DEMs with the threshold values of 0.05, 0.10, 0.15 and 0.2 m, and they are hereafter called DEM5, DEM10, DEM15 and DEM20, respectively.

The IC and sediment delivery and connected field area computations were subjected to sensitivity analyses. Firstly, all computations were conducted with the four different DEMs (DEM5-DEM20) to produce a range of possible connectivity scenarios. Secondly, the sensitivity of the results to ditch width variability and possible inaccuracies in ditch locations was studied with two additional scenarios. In these two scenarios, the ditches were widened 2 and 3 pixels (4 and 6 meters) and the scenarios are hereafter called DITCH4 and DITCH6, respectively. Finally, we also studied the sensitivity of the results to parameter variations. The empirical parameters were parameterized based on previous studied and local observations. The parameterizations (P1-P7) are shown in Table 8 and see Tähtikarhu et al. (2022) for details.

Table 8: Parameterizations in the sensitivity analysis (sediment delivery computations).

Parameterization	**P1**	**P2**	**P3**	**P4**	**P5**	**P6**	**P7**
Description	Widely used literature value	Literature value	Literature value	Literature value	Reflects local data	Reflects local data	Reflects local data
IC_0	0.5	0.5	0.5	0.1	-4.7	-3.3	-5.7
K_{IC}	2.0	1.8	3.5	2.0	1.0	1.0	1.0
SDR_{max}	0.8	0.8	0.8	0.8	0.8	0.8	0.8

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While the results were computed at the 2×2 m^2 pixel scale, the index of connectivity and sediment delivery were also aggregated to field parcel scale. The aggregation was conducted by calculating mean values for each field parcel. The field parcel scale is of particular interest since that is a typical scale for agricultural and erosion management choices.

The results of the exploration are summarized here in the following and described in more detail in Tähtikarhu et al. (2022). The conducted computations showed how the pixel scale (2×2 m^2) IC values within the two sub-catchments largely overlapped. The values varied between -8.6...-1.2 in the Mustionjoki and -8.1...-0.4 in the Aurajoki sub-catchment. The distributions were centered around the median values (-6.0...-5.9 at Mustionoki and –5.9...-5.8 at Aurajoki subcatchment) and were slightly skewed. At both sub-catchments, the IC values correlated with log-transformed erosion values (Pearson r=0.58-0.59). Moreover, IC values typically formed tree-like drainage networks within field parcels and thus were not evenly distributed within the landscape. Impacts of the computational scenarios (DEM5-20 and DITCH4-6) on the results were low compared to the variability in the IC values within the DEM10 scenario. The distributions and relationships were qualitatively similar at the parcel scale as compared to the pixel scale.

Most of the agricultural areas within the sub-catchments were connected to the ditches and streams surrounding the field parcels. The share of connected field areas was sensitive to the sink treatment scenario and the share of connected area varied from 65% to 92% at Mustionjoki and from 78% to 97% at Aurajoki sub-catchment in the computational scenarios (DEM5-DEM20 and DITCH4-DITCH6). Disconnected field areas were mostly due to depressions on the soil surface, and they were sporadically located within the sub-catchments.

Parcel scale sediment delivery magnitudes with the different parameterizations P1-P7 varied by several orders of magnitude. This demonstrates that the computed sediment delivery magnitude predictions include high uncertainties. The parcel index of connectivity values correlated significantly with the erosion values (Pearson r≥0.49) at both sub-catchments (Figure 1).

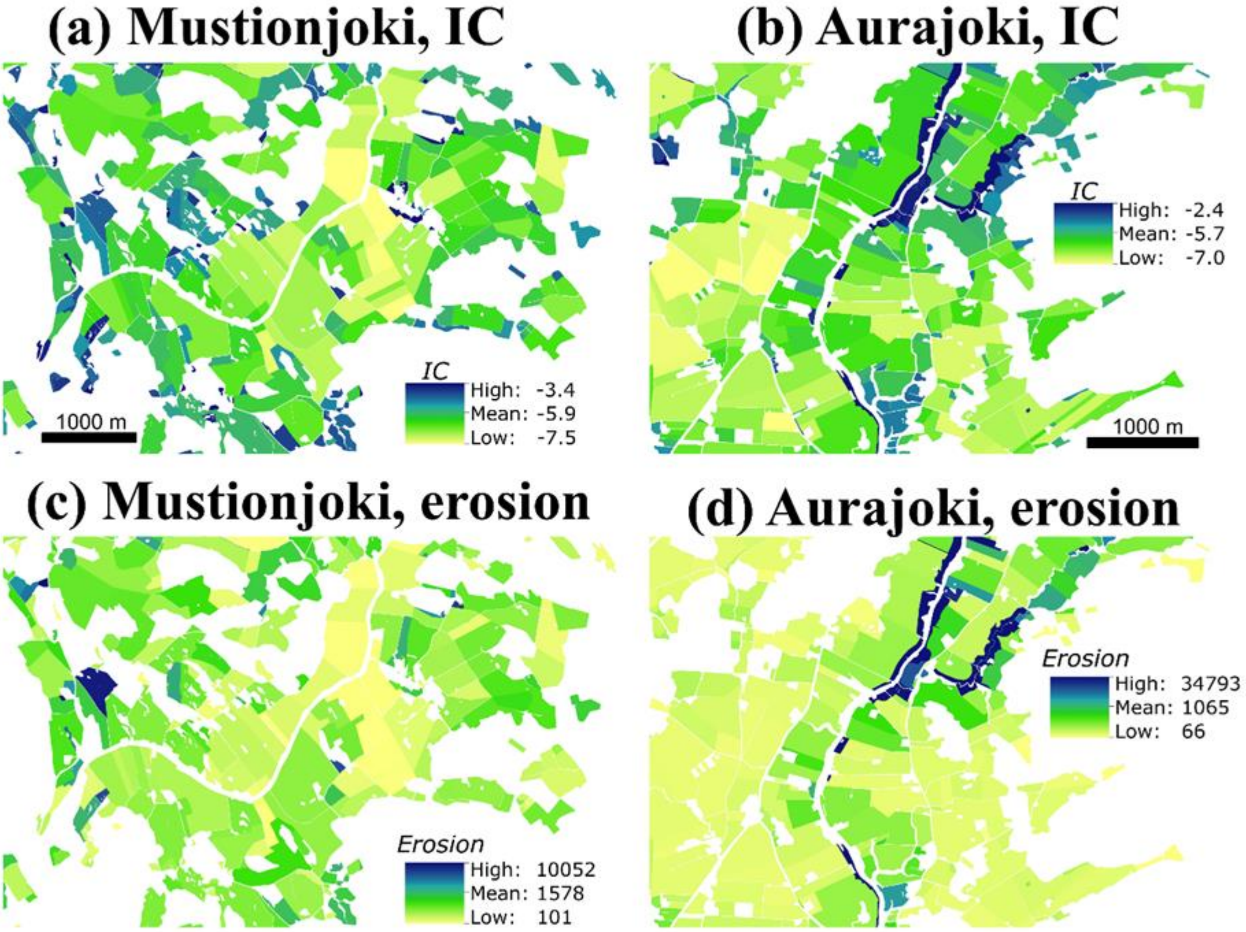

Figure 1: A representative snapshot on the spatial distribution of the mean plot-scale index of connectivity (IC) and erosion values (RUSLE) at the (a) Mustionjoki and (b) Aurajoki sub-catchments. Figure adapted from Tähtikarhu et al. (2022).

Finally, we analyzed rank correlations between the calculated parcel scale sediment delivery in different scenarios. The rank correlations between the sediment delivery results of parameterizations P1-P7 were high (Spearman r_s>0.95, p-value < 0.01). Thus, the model produced consistent relative estimates of sediment delivery of the field parcels with different parameterizations, although the absolute delivery magnitudes were uncertain.

3.2.2. WATEM-SEDEM

LAND USE CHANGES

To better include the impact of land use changes main attention should be given to a more accurate calculation of the cover-management factor (C-factor), that takes into account the (evolution of) the new land cover. Afforestation means a decreasing C-factor in time, depending on the type and evolution of the vegetation cover and of the combination with understory grasses. Permanent grassland and perennial crops can be from the first year on represented by a low crop-dependent C-factor. The impacts of crop rotations, crop diversification, set-aside, intercropping and agroforestry are all dependent on the used values for the respective C-factors, which means that C-factors should be estimated as accurately as possible. C-factor calculations can be improved by gathering better crop data that are needed for the C-factor calculation (from measurements, expert knowledge or remote sensing data).

Another improvement can be made by refining the estimation of the upstream area (A). Depending on the runoff that is created on a type of land cover, a higher or lower value for the Parcel Trapping Efficiency (PTEF) can be used. A refinement of the PTEF in function of the land cover type would be an improvement in taking into account the upstream land use, based on the reduction on runoff (%) compared to arable land.

Changing arable land to forest, grassland or perennial crops also means lower parcel connectivity on the parcel borders in comparison with parcel borders between two parcels of arable land. Decreasing parcel sizes creates more parcel borders in an arable area, which reduces the upstream area (A) and has an influence on the routing. Nevertheless, the impact is dependent on the type of parcel border. More knowledge about the types of parcel borders and their impact on parcel connectivity and routing would be very useful to refine the calculation of the upstream area (A).

Intercropping, agroforestry and terracing cannot easily be taken into account by the C-factors as crops are intermingled and topography is changed. The P-factor is more appropriate for these types of land use change, but the impact of these practices on the P-factor should be further investigated. Additionally, the estimation of the impact of these practices on the PTEF would improve the calculation of the downstream impact of these measures. In the case of terraces, the digital elevation model (DEM) can be adapted in case the available DEM represents the initial topography before the construction of the terraces.

AGRONOMIC MEASURES

Similar to the land use changes, the implementation of agronomic measures is simulated in WaTEM/SEDEM by the cover-management factor (C-factor) and the Parcel Trapping Efficiency (PTEF). A more refined knowledge about all the relevant crop and soil related factors that determine the C-factor, is automatically translated in a better calculation of the soil loss by water erosion. In the case the influence on the different parameters for the calculation of the C-factor is not known, the C-factor may also be reduced by a general reduction factor, e.g. for reduced tillage, mulching or leaving crop residues, and microdams between ridges. If the amount of runoff that is reduced is known, this can be translated in a higher PTEF.

Tillage practices and contour farming have an influence on the tillage direction, which can be considered in WaTEM/SEDEM. Nevertheless, input data about tillage direction should be available to incorporate this impact in the modelling, which is often not the case. Data collection about tillage directions would be an improvement of the modelling.

Soil organic carbon and soil structure can be translated to the K-factor when data availability allows this. Reduced infiltration and increased runoff due to subsoil compaction would also be an interesting factor to translate to a value for the Parcel Trapping Efficiency in order to incorporate the downstream impact of a landscape with lower infiltration rates.

BUFFERING MEASURES

Linear landscape elements like grass buffer strips and hedges, can be translated to another land use for these limited surfaces, which has the already described influence on the cover-management factor (C-factor), the Parcel Trapping Efficiency (PTEF), the parcel connectivity and the Transport Capacity coefficient (kTC). When working with rather coarse pixel size, efforts should be made to incorporate the impact of the width of grass buffer strips. Care should also be taken that the prioritization of land use assignation in favor of other land use, does not make the grass buffer pixels disappear in the final land use raster that is used as an input file. The border of the grass buffer strip adjacent to the arable land also should not be considered as a normal parcel border, because grass buffer strips are constructed without leaving a plow furrow to assure the inflow of sediments. This means that the routing has to conduct the sediments directly to the grass buffer strip.

The impact of grass buffer width can be taken into account by the following approach, that has an impact on both the C-factor and the kTC-factor. When the grass buffer strip width (W_{gbs}) is lower than the resolution, the C-factor (C_{gbs}) of the grass buffer pixels can be calculated by the following equation:

$$C_{gbs} = C_{grass}*(W_{gbs} /\text{resolution}) + (C_{arable}*(\text{resolution}-W_{gbs})/\text{resolution}))$$

with C_{gbs} the C-factor of a grass buffer strip pixel, C_{grass} the C-factor of grassland, W_{gbs} the width of the grass buffer strip, C_{arable} the C-factor of the adjacent arable land and resolution the grid cell size.

Based on Verstraeten et al. (2006) a relationship between kTC_{gbs} and kTC_{high} can be deduced which takes into account the Sediment Trapping Efficiency of the grass buffer strip:

$$kTC_{gbs} = kTC_{high}\,(1-SVE_{gbs}/100)$$

with kTC_{gbs} the transport capacity coefficient of the grass buffer strip, kTC_{high} the calibrated highest transport capacity coefficient and STE_{gbs} the Sediment Trapping Efficiency (%) of the grass buffer strip. By using a function that determines the relationship between the STE_{gbs} and the grass buffer strip width, the kTC_{gbs} can be estimated.

All buffering elements that retain sediments, like dams, silt fences and sediment retention ponds, are incorporated in WaTEM/SEDEM by using a Sediment Trapping Efficiency (STE). This STE not only reduces the sediment transported to the lower pixel, but also the upstream area (A). If A is not reduced, a clear water effect is created downstream the buffer which induces incorrect high amounts of erosion. Concerning the routing, all pixels of the buffer should be routed to the outlet of the buffer. When buffer pixels are assigned by an all-touched algorithm, the risk of a routing missing the buffer is reduced. Connectivity elements that guide a sediment flow into a buffer can be implemented by a forced routing algorithm that deviates the calculated routing to another direction.

OTHER CONNECTIVITY ELEMENTS

Tillage direction can be taken into account in WaTEM/SEDEM, but the necessary input data should be collected. Wheel tracks could be included by forcing the routing algorithm in the directions of the tracks, but other impacts of wheel tracks (e.g. reduced infiltration) cannot be included. The impact of parcel borders is already included in the parcel connectivity and the routing algorithm, but can be refined in function of the type of parcel border. In the case of subsurface drainage, the P-factor can incorporate a reduction of erosion.

Roads influence the routing algorithm in a way that the routing preferentially follows the road. Nevertheless, no sedimentation is calculated on roads. To calculate sedimentation on roads, a kTC-factor for infrastructure should be calculated, demanding appropriate calibration data to calculate accurate values.

When entering ditches and sewers the routing and sediment transport should be stopped as transport processes in ditches and sewers are very different for transport processes on the land. Assumptions of the percentage of sediment that reaches the water courses by ditches and sewers can be made, based on experimental data. Or these processes can be modelled by another model. Large underground pipes conducting watercourses or ditches should be excluded from sediment delivery in the case sediment cannot enter the water system at these sections. Small underground pipes can be included by a forced routing algorithm. Data availability of ditches, underground pipes and sewers in the landscape is a crucial factor for the incorporation of these elements; for the sewers data should be available about their inlets that capture runoff and sediments. Wastewater sewers with no inlets should be excluded.

Topographic changes can be translated to the digital elevation model (DEM), and will act in this way on the topographic factor (LS-factor).

3.2.3. Physically-based models

The physically-based models considered in the present guidelines (CASE2, Erosion3D, Iber, Mhydas Erosion, OpenLisem, SHETRAN and WEPP) come with similar features: they are spatially-distributed, and simulate overland flow first and then erosion. However, they differ in some ways. For example, Mhydas Erosion is not DEM based: it considered objects defined by the users to account for space distribution and connectivity. Since the models share multiple common characteristics, this section treats them concomitantly.

LAND USE CHANGES

Agroforestry and terracing were not readily-available land uses for the physically-based models (Table 9). However, they can be included in the simulations. Agroforestry is a heterogeneous land use. Thereare two ways to include agroforestry: 1) by giving as inputs equivalent parameters that account for the tree effects. These equivalent parameters (implicit representation) could come for specific experiments or from the literature, 2) with an explicit spatial representation of the tree locations and the crops. In such case, the actual spatial heterogeneity is preserved. The spatial resolution should be fine enough to allow for representing the spatial heterogeneity. In some cases, an explicit representation is the only sensible choice, such as when the tree lines are arranged along the slope. Since the tree lines and the surrounding crops do not interact in the overland flow and erosion processes, tree lines and crops may be better simulated by representing them as alternating land uses across the slope.

All the listed physically-based models can potentially account for terraces. Here too, an implicit representation can be used, assuming a homogenous behavior. Such behavior may not be expected for a few terraces on a short hill slope. However, it may be valid on a long hill slope, where the overland flow will encounter numerous terraces before reaching the foot slope. However, if the terraces have a significant effect of flow concentration, the simulations may not represent it. Since terraces are discontinuities in the topography (low slopes alternating with abrupt slopes), it may be challenging for some models. While a DEM with a fine resolution will allow for a visually-convincing representation of the terrace landscape, the model may not be able to simulate properly the abrupt changes in the topography. Modellers have to test their models before implementing this kind of landscape. This is especially true for Iber – the most physically-based model included in the present guideline – because its numerical scheme assumes a continuous topography to solve the shallow water equations. Abnormal behaviors in the water flow or sediment flux have to be looked for on a benchmark case before running actual simulation. The model may produce outputs that are meaningless, leading to interpreting simulation results that do not actually make sense. While we do not advocate against the use of Iber, its state-of-the-art numerical scheme should not be considered by modellers as a guaranty to have the best simulations.

AGRONOMIC MEASURES

None of the agronomic measures were readily-available in the seven process-based models. However, all models have the capabilities to simulate them (Table 9). Basically, the agronomic measures are modifying some parameters of the underlying land use. Hence, the polygon geometry do not have to be altered (if they are applied to the whole area), only the overland flow and erosion parameters have to be modified. These agronomic measures can alter infiltration, surface storage, hydraulic roughness, flow erosivity, soil erodibility, transport capacity, sedimentation, etc. Hence implementing them in the simulations is equivalent to altering the corresponding parameter.

However, some agronomic measures could be partially simulated only: e.g. if contour farming is shown to alter flow direction for a given landscape, this may be challenging to simulate (more on this below on tillage direction). As always, carrying out a simulation is setting up a simplified representation of reality. It is up to the modeller to assess that the representation is not over-simplified, and hence, that the results can be trusted.

BUFFERING MEASURES

All buffering measures can be incorporated in physically-based models (Table 9). WEPP was the model that could readily incorporate a significant part of them. In all cases, these measures require delineating landscape polygons with specific parameters. Grassed buffer strips and grass-and-shrub hedges have a typical width of several meters, that matches the typical spatial resolution of common DEMs. The same goes with grassed waterways, while their orientation is along the slope.

This is not true for fascines, dams in organic materials and silt fences: in most typical cases, their width is much narrower than the DEM resolution (even if their effect can be much wider than their own width). A direct implementation would be to consider them as having a one-cell width, even if this width may be too large. The parameters of the corresponding cells may need to be adapted depending on the cell size: for the same landscape and the same model, representing a fascine (for example) may lead to the use of different values depending on if the resolution is five meters or ten meters. Not doing so may lead to underestimation or overestimation of the effect of the fascine, biasing the results. Before running simulation on a whole catchment, preliminary tests should be carried out to ensure the implementation of buffering measures reproduce the expected behavior.

Sediment retention ponds may be the most challenging buffering measure to simulate: depending on their design, their consequences on water flow and sediment flux can be quite different. Discharge of water can be limited while the pond is not full, but then the flow of water is unaffected. Sediment may be trapped only partially, and after the sediments have filled the pond, its sediment retention capacity becomes negligible. None of the seven models have an implementation of sediment retention ponds. So modellers are left to simulate part of their behavior if they are present in the watershed or expected in the management scenarios. Depending on their size and location, some effects could be omitted. For example, in the case of a small retention pond, its capacity to store water could be considered negligible. This could lead to simulating the retention as a silt fence. Finally, if may be tempting to simulate the retention pond by creating a depression in the DEM. While that seems to make sense at first, not all models are able to cope with a depression in the DEM. In many cases, one of the DEM preprocessing steps involve the “filling of the depression” even if this depression is not an artifact. This is by design in some models, their algorithm assuming a hydrologically-connected DEM. For Iber, a preliminary test should be carried out. While it can deal with depressions, the size of the depression matters: if it is too small, the assumption of continuous topography may not be satisfied, leading to unexpected results.

OTHER CONNECTIVITY ELEMENTS

Among the other connectivity elements, none of the seven considered were readily able to account for tillage directions. Tillage direction has a significant effect on flow direction on low slopes. This means the flow does not simply follow the DEM slopes. This is a challenging issue for overland flow simulation. Dedicated algorithms have been proposed (Souchère et al., 1998; Takken et al., 2001a; Takken et al., 2001b; Couturier et al., 2013). They alter the flow direction depending on the slope gradient and the angle between the slope direction and the tillage direction. These algorithms are not implemented in the considered models. Hence, accounting for tillage direction would be quite challenging. We consider tillage direction cannot be simulated in the seven models. This is quite a drawback considering tillage directions can have a significant impact in low slope landscape. The use of a model able to account for tillage directions should be considered, even if this may lead to a significant effort to learn a new model. We believe the choice of a given model should be based on the main processes occurring in the landscape to be considered, and not on the past experience of the modeller only.

Wheel tracks can be readily simulated in OpenLisem only. Wheel tracks affects flow direction and limits infiltration. Because of their limited width, they cannot be explicitly represented. However, the effect of wheel tracks on infiltration could be implemented in all other models as a change in the infiltration capacity of the land use. Hence, they can be treated as the agronomic measures. Of course, this set aside the effects of wheel tracks on the flow direction. If this effect is to be simulated, OpenLisem could be chosen.

Parcel borders may have various effects on water and sediment, even in a given landscape. Hence, they may be challenging to parametrize. However, if a typology of parcel borders can be set up (with a few types only), they could be included in the simulations. Nevertheless, their inclusion will be specific for every model. They may be based on creating polygons of one-cell width at the parcel borders. This could be quite time-consuming. Here again, modellers should reflect on the importance of the parcel borders on water and sediment before deciding upon their implementation.

Subsurface drainage involves two processes: 1) the infiltration of water inside the fields, and 2) the supply of water at some points along ditches. The first process can be simulated by increasing the infiltration capacity of the drained fields. From a modeller point-of-view, this can be treated as the agronomic measures. To our knowledge, the supply of water to the ditches by the subsurface drainage cannot be simulated.

Roads are implemented in OpenLisem. For the six other models, roads have to be considered as specific areas. Roads have a zero infiltration. They can alter the flow direction when they do not conform to the surrounding topography. The input of a zero infiltration is straightforward. This will directly account for the fate of the rain, providing the width of the roads are well represented. However, the incoming and outgoing flows may not be so well accounted for because the alteration of flow direction can be more difficult to account for. The DEM may need to be altered, and preliminary tests will need to be carried out.

Open ditches are implemented in Mhydas Erosion. For the other models, they could be represented by alteration of the DEM, and of the local parameters (infiltration, roughness, etc.). Because ditches are usually narrower than the typical DEM resolution, the simulated behavior must be tested beforehand.

Topographic changes can be implemented as changes in the DEMs. As extensively discussed above, this does not guarantee that the simulated behavior can be trusted. The experience of the modeller will be critical in judging the adequacy of the simulation results.

CONCLUDING REMARKS

Physically-based models are often seen as closer to reality because of their representation of the landscape and of the processes. While they can, indeed, account for numerous connectivity elements and erosion measures, it is clear that most of the work is left to the modellers: in most of cases, it is up to the modellers to find a way to implement connectivity elements and erosion measures. While physically-based models have been developed since many decades, it is surprising that they do not provide more help to modellers. While they may be sound from the point of view of physics and spatial representation, they tend to lack behind in terms of user interface. Developing modeler-friendly user interface may be the bigger challenge physically-based models are currently facing.

In all cases, the experience of the modeller is a critical issue. The past experience with various models, the understanding of a model implementation, the critical view on the simulation outputs are the main guarantee of the result quality. In some sense, the modeller may seem more important than the model itself.

Table 9: Connectivity elements and erosion measures that are not initially set for inclusion, but that can be included in models.

	RUSLE (field slope/plot scale; net erosion)	**RUSLE (spatially distributed; gross erosion)**	**WATEM-SEDEM**	**CASE2**	**Erosion3D**	**Iber**	**Mhydas Erosion**	**OpenLisem**	**SHETRAN**	**WEPP**
Land use changes										
Afforestation										
Permanent grassland										
Perennial crops			●							
Crop rotations, crop diversification and set-aside			●							
Intercropping			●							
Agroforestry			●	●	●	●	●	●	●	●
Parcel size										
Terracing				●	●	●	●	●	●	●
Agronomic measures										
Cover crops			●	●	●	●	●	●	●	●
Mulching, crop residue management and tillage practices			●	●	●	●	●	●	●	●
Contour farming and sowing practices				●	●	●	●	●	●	●
Micro-dams between ridges	●		●	●	●	●	●	●	●	●

Soil surface roughness			●	●	●	●	●	●	●	●
Reduction of subsoil compaction			●	●	●	●	●	●	●	●
Increase of soil organic matter			●	●	●	●	●	●	●	●
Buffering measures										
Grass buffer strips			●	●	●	●	●		●	
Grass and shrub hedges	●		●	●	●	●	●	●	●	●
Grassed waterways			●	●	●	●	●	●	●	●
Fascines	●		●	●	●	●	●	●	●	
Dams in organic materials	●		●	●	●	●	●	●	●	
Silt fences	●		●	●	●	●	●	●	●	
Sediment retention ponds				●	●	●	●	●	●	●
Other connectivity elements										
Tillage direction										
Wheel tracks				●	●	●	●		●	●
Parcel borders				●	●	●	●	●	●	●
Subsurface drainage				●	●	●	●	●	●	●
Roads				●	●	●	●		●	●
Ditches		●	●	●	●	●		●	●	●
Topographic changes				●	●	●	●	●	●	●

4. Accounting for organic carbon transfer in soil erosion model

4.1. Modelling of the fate of organic carbon

Soil organic matter (SOM) models are mathematical representations and computational tools used to 1) improve scientific knowledge on SOM dynamics functioning (mechanistic approach), 2) predict carbon (C) storage in soils (pseudo-mechanistic and empirical approaches), and 3) support policy-makers and stakeholders (pseudo-mechanistic and empirical approaches) for planning management and practices regarding agricultural production, land use changes and climate change mitigation strategies (Derrien et al., 2023). SOM models can include the dynamics of nitrogen, porosity, aggregates, water as well as plant growth, harvest and socio-economic aspects. These holistic approaches allow for predicting and managing soil fertility, primary production, water pollution and soil health.

SOM models are highly diverse (Campbell and Paustian, 2015). They differ in their spatial and time resolutions, the mechanisms taken into account and the data requirements. Models formalize mechanisms at different degrees, from process-based models (mechanistic) explicitly representing processing to data-driven models (empirical, statistical) based on a reduced number of equations and parameters and based on statistical relationships between input and output variables. Many SOM models are intermediate, and can be considered as "pseudo-mechanistic" models (Derrien et al., 2023). These models are considered the most effective tools to develop management plans and policies. They are a robust trade-off between the complexity of processes formalized, the challenge in parametrization (experimental validation difficulties), the uncertainty associated with predictions and the parsimony in data requirements and mechanisms. In recent years, breakthroughs have been made in a better understanding and the integration of functional diversity and spatial organization of the soil matrix (Derrien et al., 2023). An additional approach is organism-oriented models, which explicitly simulate the dynamics of C and N through food webs and explore the specific role of soil biota in C and N dynamics (Stockmann et al., 2013). Process-oriented models (pseudo-mechanistic and mechanistic), which have been developed and tested on long time data sets, have dominated the modeling efforts.

SOM dynamics are based on conceptual pools of SOM differing in their size, mean residence time, decomposition rates and accrual. Usually, SOM decomposition is based on first-order kinetic rates. Running time steps range from daily to monthly and the space resolution usually includes the 30 cm topsoil layer from field to regional scale. Their predictive ability strongly depends on their calibration (Stockmann et al., 2013).

The most commonly used models for simulating SOM dynamics are (ranked by publication numbers, Campbell and Paustian, 2015; Stockmann et al., 2013):

1. Century (Carbon-Nitrogen-Terrestrial Ecosystem Response to Environmental Change): it simulates vegetation and the cycling of carbon and nitrogen in soils at the monthly time step in the long-term (Parton, 1996). It can be applied to forests, grasslands, savanna and cultivated ecosystems with various management practices. Another version of CENTURY is DAYCENT (Daily Century), which is process-based model that simulates daily carbon and nitrogen dynamics (CH4, N2, N2PO, NOx gas fluxes, NO3 leaching) in agricultural and natural ecosystems (Del Grosso et al., 2009). It accounts for land management, weather, and vegetation. It takes into account 2 types of litter and 3 SOM compartments. The soil clay content is important and controls the separation of C from active SOM to CO2 or slow SOM pool.

2. RothC (Rothamsted Carbon): monthly time step model that estimates soil organic C stocks based on climate (temperature, moisture), land use and management practices (Coleman and Jenkinson, 1996; Jenkinson and Rayner, 1977). It is often used to assess the effects of land use and management changes on C content. Originally developed for agricultural systems it has been extended to temperate grasslands and forests. It takes into account 5 SOM compartments. The soil clay content is important and controls the decomposition rate, ratio of humus, microbial biomass and CO2. The conceptual bases are similar to CENTURY.
3. DNDC (DeNitrification-DeComposition): simulates soil C and N cycling processes, including decomposition, mineralization, nitrification, and denitrification. It integrates 3 submodels (soil climate, decomposition, denitrification, plant growth). It is used for assessing the impact of land use and management practices on SOM and greenhouse gas emissions (CO2, CH4, N2O).
4. EPIC (Erosion-Productivity Impact Calculator): it is not a SOM model per se. It was developed by USDA, it simulates the impact of various land management practices, climate and weather on soil erosion, crop productivity, nutrient cycling (e.g., N, P) in the soil-plant system but also in waters (out-site pollution), and soil quality over time for assessing the long-term sustainability of agricultural systems (Williams et al., 1997). A submodel simulating C dynamics has been added (Izaurralde et al., 2012, 2006). EPIC has been widely used by researchers, agricultural professionals and decision-makers to inform decisions and make predictions at a regional and watershed scale.
5. DDSAT (Decision Support System for Agrotechnology Transfer): it is a crop modeling system used for simulating and managing agricultural systems, in particular nutrient management and crop selection (Jones et al., 2003, 1998). This model is used for training, education and advising the stakeholders as well as researchers and policymakers to develop trade-offs between food security, climate variability and environmental sustainability.
6. DAISY model (Danish Agroecological Integrated Simulation Model): comprehensive agroecological model that includes crop production, soil water and C and N dynamics (Bruun et al., 2003; Hansen et al., 1991). It has been developed as an agricultural land management tool. The clay content controls the SOM decomposition rates. It is used for a range of agroecosystems and agricultural practices.

Other models among the ~75 existing are: AMG (Andriulo et al., 1999; Clivot et al., 2019; Levavasseur et al., 2020), LPJ (Lund– Potsdam–Jena, Sitch et al., 2003), INCA (Integrated Nitrogen in CAtchments, Whitehead et al., 1998), CASA (Carnegie-Ames-Stanford Approach designed for global C cycle studies, Potter et al., 1993), C-N-SIM (Carbon-Nitrogen-Sulfur Interactions in managed grasslands, Petersen et al., 2005a, 2005b), PnET-CN (Photosynthesis and Evapotranspiration-Carbon and Nitrogen for forest ecosystems, Postek et al., 1995), ICBM (Introductory Carbon Balance Model; simple model for outreach and result sharing with society, Andrén and Kätterer, 1997).

Models need to be calibrated and validated with local data to ensure accurate predictions adapted to specific soil and land use contexts and to specific research and management goals, considering available data and detail requirements.

4.2. Coupled modelling of organic carbon and soil erosion

Lateral transfers of soil organic carbon (C) due to erosion can significantly influence regional and global C budgets (Doetterl et al., 2016). Globally, significant amounts of soil organic C (between 0.47 and 0.61 Pg/yr) move laterally with erosion (Van Oost et al., 2007). However, the question of whether water erosion is a source or a sink of atmospheric C remains uncertain (Van Oost and Six, 2023).

Estimates vary considerably, from a source of 0.8 to 1.2 Pg of C per year (Lal, 2003) to a sink of 0.12 to 1.5 Pg of C per year according to other studies (Stallard, 1998; Van Oost et al., 2007). To quantify the role of water erosion in ecosystem C balance, approaches coupling erosion and SOM dynamics models are essential tools. These approaches combine representations of processes involved in erosion and SOM dynamics aiming to assess and predict the erosion-driven dynamics of organic C.

Recent examples of spatially explicit models are: Changing Relief and Evolving Ecosystems Project (CREEP) model (Rosenbloom et al., 2001), SPEROS-C (Van Oost et al., 2005, Fiener et al., 2015), Erosion-Deposition-Carbon Model (EDCM) (Liu et al., 2003). The CREEP model focuses on the long-term development of the landscape (i.e. on a millennial scale) and on the diffusive geomorphological processes that occur in undisturbed grasslands. It also simulates textural differentiation and the preferential transport of the finest fractions by runoff. SPEROS-C focuses on shorter time scales (i.e. years to decades) and on agricultural landscapes. It includes water erosion and tillage spread over space, and dynamically couples C renewal with soil erosion (Dlugoß et al., 2012). The EDCM model is based on the SOM model CENTURY. EDCM allows vertical soil organic C distribution patterns to change over time, influenced by factors such as rooting properties, soil erosion and deposition (Liu et al., 2003). It simulates SOM dynamics for different soil layers, with layer-dependent properties and processes.

LSM Land Surface Models (LSM) have been developed to study the transport of organic C from soils through terrestrial ecosystems and aquatic environments (Zhang et al., 2022). In particular, the Dynamic Land Ecosystem Model (DLEM) (Tian et al., 2015) simulates the loss of particulate organic carbon (POC) induced by soil erosion to the river, and the transport and decomposition of POC in river systems. However, it does not represent POC deposition in floodplains or the impacts of soil erosion and floodplain deposition on vertical soil organic C profiles. The Carbon Erosion DYNAMics (CE-DYNAM) model (Naipal et al., 2020) simulates the erosion of soil organic C and its redeposition on the toe slope or in floodplains, the transport of POC along river channels and the impact on C dynamics at erosion and deposition sites. It operates on an annual running time step for simulations at centennial time scale. However, it does not represent the deposition and decomposition of carbon in river channels (Zhang et al., 2022).

Approaches aiming to couple erosion and organic C simulations in a coherent model face certain challenges to better predict organic C lateral and vertical redistribution and SOM dynamics affected by erosion. One is to bring together spatial and temporal scales, which differ between process-based erosion models (event-based, short time steps) and SOM models (annual time steps for long term predictions) (Doetterl et al., 2016). Process-based erosion models depend on data from specific events and focus on local conditions and short-term processes (< hours) whereas SOM storage evolution takes place over long timescales (>10 years), requiring observations along multiple decades (Wilken et al., 2017a).

Another challenge concerns the consideration of different textural classes and soil aggregates. Process-based models of water erosion, in particular those that incorporate the selective redistribution of textural classes (e.g. selective diffuse erosion and deposition) are crucial for modeling the lateral redistribution of soil organic C and the erosion-driven SOM dynamics. However, to be effective, these models must also be able to estimate soil aggregation and aggregate stability, integrate the different classes of aggregates into sediment transport and allocate organic C to these different classes (Doetterl et al., 2016). SOM dynamics is controlled by texture and aggregation. SOM can be separated in two main fractions differing in stabilization processes: particulate organic matter (POM), which is mineral-free and the mineral-associated organic matter (MAOM), which includes both organic compounds bound to mineral surfaces and occluded in aggregates represented by high density, clay (<2µm) and silt-sized particles (<50µm). POM is mainly composed of plant fragments of diverse decomposition stages while MAOM is composed of both small plant fragments occluded in aggregates and organic compounds of plant and microbial origins bound to mineral surfaces. Consequently POM and MAOM differ in composition and mean residence time in soils, which is illustrated by higher C to nitrogen (N) ratios and low residence time of POM than MAOM (Cotrufo and Lavallee, 2022).

The selective mobilization of particles and aggregates by water erosion also affect SOM dynamics as the mobilization of aggregates is more frequent during rill erosion while sheet erosion tends to favor the preferential mobilization of fine particles richer in organic C than sands (Wilken et al., 2017a). The allocation of C contents and residence times to different texture and aggregate classes is crucial for pairing erosion and SOM modeling. To date, the MCST-C model (Wilken et al., 2017b) is one of the few attempts to combine the Multi-Class Sediment Transport Model (MCST, Wilken et al., 2017a, Van Oost et al., 2004) and the SPEROS-C model (Van Oost et al., 2005, Nadeu et al., 2015, Fiener et al., 2015).

4.3. Example with Mhydas_Erosion_C

Here, we present a scheme to represent the lateral redistribution of organic carbon (C) into an event-based mechanistic erosion model. The model studied is MHYDAS_Erosion, which has been developed for sloping Mediterranean landscapes and considering the catchment as a series of interconnected field parts linked to the ditch network (Gumiere et al., 2011; Moussa et al., 2002).

4.3.1. A short description of 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 has been developed for agricultural headwater catchments covering a few km². The model spatially represents the catchment into homogeneous hydrological units, ‘surface units’ (SU) for fields and ‘reach segments’ (RS) for segments of the ditch network. The originality of the model stems from its capacity to integrate the impact of land management practices as key elements controlling the sediment connectivity in agricultural catchments. The model takes 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).

4.3.2. Conceptualization of integration of lateral transfers of organic C in MHYDAS_Erosion

The integration of the redistribution of soil organic C into the MHYDAS_Erosion model can only be conducted by considering variations in C concentrations as a function the size of the transported sediments (through the median size of sediments, d50 parameter) and differentiating the different soil organic matter (SOM) fractions (i.e., particulate organic matter - POM, mineral-associated organic matter - MAOM, dissolved organic matter- DOM) in short-term variations of sediment fluxes. In Mediterranean regions, extreme events (time compression) can be the main drivers of the total solid fluxes and particulate organic carbon (POC) fluxes. Due to the highly intermittent flow in Mediterranean headwater catchments, POC fluxes are generally greater than dissolved organic carbon (DOC) fluxes. The use of event-based modeling (e.g., MHYDAS_Erosion) for simulating lateral transfers of SOM is therefore a relevant approach in this context. This conceptual approach proposed here is adapted to the current configuration (i.e., variables, parameters, formalized processes) of MHYDAS_Erosion and the context of Mediterranean headwater catchments. MHYDAS_ Erosion is not able to consider large-scale spatial and temporal processes of SOM dynamics such as the SOM turnover and its evolution with detachment, transport and deposition or with depth. Therefore, no proposition about SOM dynamics (e.g., decomposition rates, turnover times) are made.

4.3.3. Carbon enrichment of sediments and preferential detachment of MAOM in SU

The rate of sheet erosion is often modeled as a function of the combined effects of soil erodibility, slope and rainfall erosivity. However, soil erosion affects soil particles differently depending on their size. Finer particles, which display higher C content, are detached before coarser particles.

To take into account the possibility of an increase in organic C content due to the preferential detachment of fine articles during diffuse erosion, an enrichment factor could be used. It is incorporated into models such as SPEROS-C (Oost et al., 2005) or SWAT-C (Zhang, 2018). The enrichment factor is determined empirically as a function of texture, event intensity and spatial scale (Fiener et al., 2015).

We suggest implementing an enrichment factor in MHYDAS_Erosion to explicitly consider the selective nature of fine organic-rich particles during diffuse erosion. To adapt this implementation to the current configuration of MHYDAS_Erosion (in which diffuse erosion is only simulated within SU), we decided to consider the preferential detachment only in SU. The enrichment factor should be empirically calibrated with field experiments. The comparison between the C contents of bulk soils and MAOM fraction could be used to calculate the enrichment factor. In addition, the sediment C contents could evolve with particle size and event size with sediment C contents could be inversely correlated with the d50 parameter and the intensity of the rain event (i.e., lower the d50 higher the sediment C content).

4.3.4. Mobilization of different fractions of soil organic matter

SOM is composed of a great variety of organic compounds of different nature originating from both plant and microbial origins. Differences in SOM stabilization processes between pedoclimatic conditions can be evaluated by dividing the bulk SOM pool into fractions differing in turnover times and composition. The importance of characterizing three main fractions has been demonstrated: POM, which is mineral-free, the DOM, which is extracted with water and of size < 0.7 µm, and the MAOM, which includes both organic compounds bound to mineral surfaces and occluded in aggregates represented by high density, clay (<2µm) and silt-sized particles (<50µm) (Poeplau et al., 2018). POM is mainly composed of plant fragments of diverse decomposition stages while MAOM is composed of both small plant fragments occluded in aggregates and organic compounds of plant and microbial origins bound to mineral surfaces (Cotrufo and Lavallee, 2022). DOM is composed of dissolved compounds from plant and microbial origins.

Simulating how soil erosion processes alter these different fractions will allow for better describing the different mechanisms of SOM dynamics (Doetterl et al., 2016). Important knowledge gaps remain on the different mobilization and transformation of fractions during erosion. The organic C enrichment of eroded sediments can be attributed to MAOM due to its association with clays and silts (Koiter et al., 2017). However, the enrichment due to MAOM is debated. MAOM could be preferentially mobilized by diffuse erosion compared to POM, and its enrichment rate decreased with increasing sediment concentration (Wang et al., 2013). Another study showed that POM was enriched by 47% and MAOM was depleted by 26% in sediments compared to the original soils (Holz and Augustin, 2021).

For MHYDAS_Erosion, we suggest implementing three SOM fractions: MAOM, POM and DOM. To implement these different MOS fractions in our model, we relied on an experimental study in the Roujan watershed by Lahens et al. ,2023 (under review). The study showed that MAOM contributes the most to the total SOM (around 85%), this contribution is similar in RS and SU, although the C content of POM within ditches (RS) are higher than in fields. The current configuration of MHYDAS_ Erosion only considers the suspended sediment load, neglecting floating particles on the flow surface. This implies a priori that the model would not be able to take into account the POM fraction. To do this, we introduced an empirical runoff value that would trigger the transfer of POM within the various spatial units. In addition, we also assumed that POM and DOM are rapidly exported from the small headwater catchments. For DOM and POM, we assume their transfer is rapid as runoff is usually driven by extreme and intense events. Therefore, we assume that POM and DOM concentrations in the runoff are constant along the watershed to the outlet. AS the POM transfer is quick, the POM contribution to enrichment of deposited sediments could be ignored in MHYDAS_ Erosion. The POM and DOM dynamics will be included in MHYDAS hydrological model rather than MHYDAS_Erosion (Gumiere et al., 2011; Moussa et al., 2002).

5. Conclusions

Models are key tools in soil erosion management. They are used to simulate the effects of connectivity elements and erosion measures. A previous report (SCALE, 2023) showed how connectivity was accounted for in models. The goal of the present report is to go beyond. It shows that, in numerous cases, connectivity features can be simulated even if they are not readily available in the models, whether they are empirical or physically-based. This should help in generating better simulations, and hence better management scenarios.

However, it may be challenging to implement some measures, and in some cases, their implementation may be impossible. In any case, this is a time-consuming task. If models were implementing more connectivity elements and erosion measures, the work of model's users would be made easier. That would lead them to spend more time on scenario design, and less time on model "tweaking" and testing.

Beyond soil erosion, it is still a challenge to simulate carbon-associated transfer in small watersheds. While there is a huge demand, models are far behind on this point, even if future development looks promising.

The present report is also a call to model developers to better account for modeller's needs. While the models' core features may be satisfying, the user interface should be improved to facilitate scenario parametrization.

6. References

Andrén, O., Kätterer, T., 1997. Icbm: The Introductory Carbon Balance Model for Exploration of Soil Carbon Balances. Ecological Applications 7, 1226–1236.

Andriulo, A., Mary, B., Guerif, J., 1999. Modelling soil carbon dynamics with various cropping sequences on the rolling pampas. Agronomie 19, 365–377.

Bruun, S., Christensen, B.T., Hansen, E.M., Magid, J., Jensen, L.S., 2003. Calibration and validation of the soil organic matter dynamics of the Daisy model with data from the Askov long-term experiments. Soil Biology and Biochemistry 35, 67–76.

Borselli, L., Cassi, P., Torri, D., 2008. Prolegomena to sediment and flow connectivity in the landscape: A GIS and field numerical assessment. CATENA 75, 268–277. https://doi.org/10.1016/j.catena.2008.07.006

Campbell, E.E., Paustian, K., 2015. Current developments in soil organic matter modeling and the expansion of model applications: a review. Environ. Res. Lett. 10, 123004.

Clivot, H., Mouny, J.-C., Duparque, A., Dinh, J.-L., Denoroy, P., Houot, S., Vertès, F., Trochard, R., Bouthier, A., Sagot, S., Mary, B., 2019. Modeling soil organic carbon evolution in long-term arable experiments with AMG model. Environmental Modelling & Software 118, 99–113.

Coleman, K., Jenkinson, D.S., 1996. RothC-26.3 - A Model for the turnover of carbon in soil, in: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models, NATO ASI Series. Springer, Berlin, Heidelberg, pp. 237–246.

Crema, S., Cavalli, M., 2018. SedInConnect: a stand-alone, free and open source tool for the assessment of sediment connectivity. Computers & Geosciences 111, 39–45. https://doi.org/10.1016/j.cageo.2017.10.009

Cotrufo, M.F., Lavallee, J., 2022. Soil organic matter formation, persistence, and functioning: A synthesis of current understanding to inform its conservation and regeneration, in: Advances in Agronomy. pp. 1–66. https://doi.org/10.1016/bs.agron.2021.11.002

Couturier, A.; Daroussin, J.; Darboux, F.; Souchère, V.; Bissonnais, Y. L.; Cerdan, O. & King, D. (2013). Improvement of surface flow network prediction for the modeling of erosion processes in agricultural landscapes. Geomorphology, 183: 120-129. https://doi.org/10.1016/j.geomorph.2012.07.025

Del Grosso, S.J., Ojima, D.S., Parton, W.J., Stehfest, E., Heistemann, M., DeAngelo, B., Rose, S., 2009. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global and Planetary Change, Changes in land use and water use and their consequences on climate, including biogeochemical cycles 67, 44–50.

Derrien, D., Barré, P., Basile-Doelsch, I., Cécillon, L., Chabbi, A., Crème, A., Fontaine, S., Henneron, L., Janot, N., Lashermes, G., Quénéa, K., Rees, F., Dignac, M.-F., 2023. Current controversies on mechanisms controlling soil carbon storage: implications for interactions with practitioners and policy-makers. A review. Agron. Sustain. Dev. 43, 21.

Dlugoß, V., Fiener, P., Van Oost, K., Schneider, K., 2012. Model based analysis of lateral and vertical soil carbon fluxes induced by soil redistribution processes in a small agricultural catchment. Earth Surface Processes and Landforms 37, 193–208. https://doi.org/10.1002/esp.2246

Doetterl, S., Berhe, A.A., Nadeu, E., Wang, Z., Sommer, M., Fiener, P., 2016. Erosion, deposition and soil carbon: A review of process-level controls, experimental tools and models to address C cycling in dynamic landscapes. Earth-Science Reviews 154, 102–122. https://doi.org/10.1016/j.earscirev.2015.12.005

Fiener, P., Dlugoß, V., Van Oost, K., 2015. Erosion-induced carbon redistribution, burial and mineralisation — Is the episodic nature of erosion processes important? CATENA 133, 282–292. https://doi.org/10.1016/j.catena.2015.05.027

Ferro, V., Porto, P., 2000. Sediment Delivery Distributed (SEDD) Model. Journal of Hydrologic Engineering 5, 411–422. https://doi.org/10.1061/(ASCE)1084-0699(2000)5:4(411)

Foerster, S., Wilczok, C., Brosinsky, A., Segl, K., 2014. Assessment of sediment connectivity from vegetation cover and topography using remotely sensed data in a dryland catchment in the Spanish Pyrenees. J Soils Sediments 14, 1982–2000. https://doi.org/10.1007/s11368-014-0992-3

Gumiere, S.J., Raclot, D., Cheviron, B., Davy, G., Louchart, X., Fabre, J.-C., Moussa, R., Bissonnais, Y.L., 2011. MHYDAS-Erosion: a distributed single-storm water erosion model for agricultural catchments. Hydrological Processes 25, 1717–1728. https://doi.org/10.1002/hyp.7931

Hansen, S., Jensen, H.E., Nielsen, N.E., Svendsen, H., 1991. Simulation of nitrogen dynamics and biomass production in winter wheat using the Danish simulation model DAISY. Fertilizer Research 27, 245–259. https://doi.org/10.1007/BF01051131

Holz, M., Augustin, J., 2021. Erosion effects on soil carbon and nitrogen dynamics on cultivated slopes: A meta-analysis. Geoderma 397, 115045. https://doi.org/10.1016/j.geoderma.2021.115045

Izaurralde, R.C., Mcgill, W.B., Williams, J.R., 2012. Development and application of the EPIC model for carbon cycle, greenhouse-gas mitigation, and biofuel studies (No. PNNL-SA-83721). M Liebig, AJ Franzluebbers and R Follett; Academic Press, Waltham, MA, United States(US).

Izaurralde, R.C., Williams, J.R., McGill, W.B., Rosenberg, N.J., Jakas, M.C.Q., 2006. Simulating soil C dynamics with EPIC: Model description and testing against long-term data. Ecological Modelling 192, 362–384.

Jenkinson, D.S., Rayner, J.H., 1977. The turnover of soil organic matter in some of the Rothamsted classical experiments. Soil Science 123, 298–305. https://doi.org/10.1097/00010694-197705000-00005

Jones, J.W., Hoogenboom, G., Porter, C.H., Boote, K.J., Batchelor, W.D., Hunt, L.A., Wilkens, P.W., Singh, U., Gijsman, A.J., Ritchie, J.T., 2003. The DSSAT cropping system model. European Journal of Agronomy, Modelling Cropping Systems: Science, Software and Applications 18, 235–265.

Jones, J.W., Tsuji, G.Y., Hoogenboom, G., Hunt, L.A., Thornton, P.K., Wilkens, P.W., Imamura, D.T., Bowen, W.T., Singh, U., 1998. Decision support system for agrotechnology transfer: DSSAT v3, in: Tsuji, Gordon Y., Hoogenboom, Gerrit, Thornton, Philip K. (Eds.), Understanding Options for Agricultural Production, Systems Approaches for Sustainable Agricultural Development. Springer Netherlands, Dordrecht, pp. 157–177.

Koiter, A.J., Owens, P.N., Petticrew, E.L., Lobb, D.A., 2017. The role of soil surface properties on the particle size and carbon selectivity of interrill erosion in agricultural landscapes. CATENA 153, 194–206. https://doi.org/10.1016/j.catena.2017.01.024

Lal, R., 2003. Soil erosion and the global carbon budget. Environment International 29, 437–450. https://doi.org/10.1016/S0160-4120(02)00192-7

Levavasseur, F., Mary, B., Christensen, B.T., Duparque, A., Ferchaud, F., Kätterer, T., Lagrange, H., Montenach, D., Resseguier, C., Houot, S., 2020. The simple AMG model accurately simulates organic carbon storage in soils after repeated application of exogenous organic matter. Nutr Cycl Agroecosyst 117, 215–229.

Liu, S., Bliss, N., Sundquist, E., Huntington, T.G., 2003. Modeling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. Global Biogeochemical Cycles 17. https://doi.org/10.1029/2002GB002010

Mitasova, H., Hofierka, J., Zlocha, M., Iverson, L.R., 1996. Modelling topographic potential for erosion and deposition using GIS. International Journal of Geographical Information Systems 10, 629–641. https://doi.org/10.1080/02693799608902101

Moussa, R., Voltz, M., Andrieux, P., 2002. Effects of the spatial organization of agricultural management on the hydrological behaviour of a farmed catchment during flood events. Hydrological Processes 16, 393–412. https://doi.org/10.1002/hyp.333

Oost, K.V., Quine, T.A., Govers, G., Heckrath, G., 2005. Modeling Soil Erosion Induced Carbon Fluxes between Soil and Atmosphere on Agricultural Land Using SPEROS-C, in: Soil Erosion and Carbon Dynamics. CRC Press.

Nadeu, E., Gobin, A., Fiener, P., van Wesemael, B., van Oost, K., 2015. Modelling the impact of agricultural management on soil carbon stocks at the regional scale: the role of lateral fluxes. Global Change Biology 21, 3181–3192. https://doi.org/10.1111/gcb.12889

Naipal, V., Lauerwald, R., Ciais, P., Guenet, B., Wang, Y., 2020. CE-DYNAM (v1): a spatially explicit process-based carbon erosion scheme for use in Earth system models. Geoscientific Model Development 13, 1201–1222. https://doi.org/10.5194/gmd-13-1201-2020

National Land Survey of Finland, 2020. Elevation model 2 m [Data set] [WWW Document]. URL https://www.maanmittauslaitos.fi/en/maps-and-spatial-data/expert-users/product-descriptions/elevation-model-2-m (accessed 5.29.20).

Natural Capital Project, 2022. InVEST User’s Guide. Stanford University, University of Minnesota, Chinese Academy of Sciences, The Nature Conservancy, World Wildlife Fund, and Stockholm Resilience Centre.

Oksanen, J., 2013. Can binning be the key to understanding the uncertainty of DEMs. Proceedings of the GISRUK 2013 3–5.

Parton, W.J., 1996. The CENTURY model, in: Powlson, D.S., Smith, P., Smith, J.U. (Eds.), Evaluation of Soil Organic Matter Models, NATO ASI Series. Springer, Berlin, Heidelberg, pp. 283–291.

Petersen, B.M., Berntsen, J., Hansen, S., Jensen, L.S., 2005a. CN-SIM—a model for the turnover of soil organic matter. I. Long-term carbon and radiocarbon development. Soil Biology and Biochemistry 37, 359–374.

Petersen, B.M., Jensen, L.S., Hansen, S., Pedersen, A., Henriksen, T.M., Sørensen, P., Trinsoutrot-Gattin, I., Berntsen, J., 2005b. CN-SIM: a model for the turnover of soil organic matter. II. Short-term carbon and nitrogen development. Soil Biology and Biochemistry 37, 375–393.

Postek, K.M., Driscoll, C.T., Aber, J.D., Santore, R.C., 1995. Application of pnet-cn/chess to a spruce stand in Solling, Germany. Ecological Modelling, Modelling Water, Carbon and Nutrient Cycles in Forests 83, 163–172.

Potter, C.S., Randerson, J.T., Field, C.B., Matson, P.A., Vitousek, P.M., Mooney, H.A., Klooster, S.A., 1993. Terrestrial ecosystem production: A process model based on global satellite and surface data. Global Biogeochemical Cycles 7, 811–841.

Räsänen, T.A., Tähtikarhu, M., Uusi-Kämppä, J., Piirainen, S., Turtola, E., 2023. Evaluation of RUSLE and spatial assessment of agricultural soil erosion in Finland. Geoderma Regional. https://doi.org/10.1016/j.geodrs.2023.e00610

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K., Yoder, D.C. (1997). Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). Agricultural Handbook 703. US Department of Agriculture, Washington, DC, pp. 404.

Rosenbloom, N.A., Doney, S.C., Schimel, D.S., 2001. Geomorphic evolution of soil texture and organic matter in eroding landscapes. Global Biogeochemical Cycles 15, 365–381. https://doi.org/10.1029/1999GB001251

SCALE (2023). Guideline on the current implementation of erosion measures and other connectivity elements depending on scale and modelling approach. 52 p.

Sitch, S., Smith, B., Prentice, I.C., Arneth, A., Bondeau, A., Cramer, W., Kaplan, J.O., Levis, S., Lucht, W., Sykes, M.T., Thonicke, K., Venevsky, S., 2003. Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. Global Change Biology 9, 161–185.

Souchère, V.; King, D.; Daroussin, J.; Papy, F. & Capillon, A. (1998). Effects of tillage on runoff directions: Consequences on runoff contributing areas within agricultural catchments.. Journal of Hydrology, 206: 256-267. https://doi.org/10.1016/S0022-1694(98)00103-6

Stallard, R.F., 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. Global Biogeochemical Cycles 12, 231–257. https://doi.org/10.1029/98GB00741

Stockmann, U., Adams, M.A., Crawford, J.W., Field, D.J., Henakaarchchi, N., Jenkins, M., Minasny, B., McBratney, A.B., Courcelles, V. de R. de, Singh, K., Wheeler, I., Abbott, L., Angers, D.A., Baldock, J., Bird, M., Brookes, P.C., Chenu, C., Jastrow, J.D., Lal, R., Lehmann, J., O'Donnell, A.G., Parton, W.J., Whitehead, D., Zimmermann, M., 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystems & Environment 164, 80–99.

Tähtikarhu, M., Räsänen, T., Oksanen, J., Uusi-Kämppä, J., 2022. Exploring structural sediment connectivity via surface runoff in agricultural lands of Finland. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science 72, 957–970. https://doi.org/10.1080/09064710.2022.2136583

Takken, I., Jetten, V., Govers, G., Nachtergaele, J., & Steegen, A. (2001a). The Effect of Tillage-Induced Roughness on Runoff and Erosion Patterns. Geomorphology 37(1–2): 1–14.

Takken, I.; Govers, G.; Steegen, A.; Nachtergaele, J. & Guérif, J. (2001b). The prediction of runoff flow directions on tilled fields. Journal of Hydrology, 248: 1-13. https://doi.org/10.1016/S0022-1694(01)00360-2

Van Oost, K., Beuselinck, L., Hairsine, P.B., Govers, G., 2004. Spatial evaluation of a multi-class sediment transport and deposition model. Earth Surface Processes and Landforms 29, 1027–1044. https://doi.org/10.1002/esp.1089

Van Oost, K., Govers, G., Desmet, P., 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landscape Ecology 15, 577–589. https://doi.org/10.1023/A:1008198215674

Van Oost, K., Quine, T.A., Govers, G., Heckrath, G., 2005. Modeling Soil Erosion Induced Carbon Fluxes between Soil and Atmosphere on Agricultural Land Using SPEROS-C, in: Soil Erosion and Carbon Dynamics. CRC Press.

Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., da Silva, J.R.M., Merckx, R., 2007. The Impact of Agricultural Soil Erosion on the Global Carbon Cycle. Science 318, 626–629. https://doi.org/10.1126/science.1145724

Van Oost, K., Six, J., 2023. Reconciling the paradox of soil organic carbon erosion by water. Biogeosciences 20, 635–646. https://doi.org/10.5194/bg-20-635-2023

Van Rompaey, A.J.J., Verstraeten, G., Van Oost, K., Govers, G., Poesen, J., 2001. Modelling mean annual sediment yield using a distributed approach. Earth Surface Processes and Landforms 26, 1221–1236. https://doi.org/10.1002/esp.275

Verstraeten, G., Poesen, J., Gillijns, K., Govers, G., 2006. The use of riparian vegetated filter strips to reduce river sediment loads: an overestimated control measure? Hydrological Processes 20, 4259–4267. https://doi.org/10.1002/hyp.6155

Wang, Z., Govers, G., Oost, K.V., Clymans, W., den Putte, A.V., Merckx, R., 2013. Soil organic carbon mobilization by interrill erosion: Insights from size fractions. Journal of Geophysical Research: Earth Surface 118, 348–360. https://doi.org/10.1029/2012JF002430

Whitehead, P.G., Wilson, E.J., Butterfield, D., Seed, K., 1998. A semi-distributed integrated flow and nitrogen model for multiple source assessment in catchments (INCA): Part II — application to large river basins in south Wales and eastern England. Science of The Total Environment 210–211, 559–583.

Wilken, F., Fiener, P., Van Oost, K., 2017a. Modelling a century of soil redistribution processes and carbon delivery from small watersheds using a multi-class sediment transport model. Earth Surface Dynamics 5, 113–124. https://doi.org/10.5194/esurf-5-113-2017

Wilken, F., Sommer, M., Van Oost, K., Bens, O., Fiener, P., 2017b. Process-oriented modelling to identify main drivers of erosion-induced carbon fluxes. SOIL 3, 83–94. https://doi.org/10.5194/soil-3-83-201

Williams, J.R., Greenwood, D.J., Nye, P.H., Walker, A., 1997. The erosion-productivity impact calculator (EPIC) model: a case history. Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences 329, 421–428.

Zhang, H., Lauerwald, R., Regnier, P., Ciais, P., Van Oost, K., Naipal, V., Guenet, B., Yuan, W., 2022. Estimating the lateral transfer of organic carbon through the European river network using a land surface model. Earth System Dynamics 13, 1119–1144. https://doi.org/10.5194/esd-13-1119-2022

Zhang, X., 2018. Simulating eroded soil organic carbon with the SWAT-C model. Environmental Modelling & Software 102, 39–48. https://doi.org/10.1016/j.envsoft.2018.01.005

Zhao, G., Gao, P., Tian, P., Sun, W., Hu, J., Mu, X., 2020. Assessing sediment connectivity and soil erosion by water in a representative catchment on the Loess Plateau, China. CATENA 185, 104284. https://doi.org/10.1016/j.catena.2019.10428