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Title: Connectivity elements and mitigation measures in policy-relevant soil erosion models: A survey across Europe

Year: 2023

Version: Published version

Please cite the original version:

M. Schmaltz, E., L. Johannsen, L., Thorsøe, M. H., Tähtikarhu, M., A. Räsänen, T., Darboux, F., & Strauss, P. (2024). Connectivity elements and mitigation measures in policy-relevant soil erosion models: A survey across Europe. CATENA, 234, 107600. https://doi.org/10.1016/j.catena.2023.107600

Catena 234 (2024) 107600

Contents lists available at ScienceDirect

Catena

journal homepage: www.elsevier.com/locate/catena

Connectivity elements and mitigation measures in policy-relevant soil erosion models: A survey across Europe

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ARTICLE INFO

Keywords:
Water erosion
Connectivity
Soil erosion models
Policy
Europe

ABSTRACT

The current use of soil erosion models in Europe was investigated through an exploratory survey of 46 model applications covering 18 European countries. This revealed novel information on erosion model applications, their parameterisation, incorporation of landscape elements and mitigation measures with implications for connectivity and their use in decision-making in Europe. The model application predictions were applied at national, regional, catchment or field scale. The majority of model applications used the USLE or versions thereof, but a range of semi-empirical, decision-tree and process-based models were also used. The majority of model applications were used for policy relevant purposes such as erosion risk assessment or mitigation measure implementation at a range of spatial scales. The analysis identified an evident prevalence towards the use of national or regional data sets and a highly varying parameterisation of model applications. Landscape elements and mitigation measures with effects on connectivity were implemented in most model applications, but not with a focus on modelling connectivity within the landscape. Altogether, the results demonstrate a need for improving connectivity modelling in diverse agricultural landscapes across multiple scales. Models should be chosen dependent on their ability to reflect erosion risk at different spatial scales. Albeit, harmonisation of data sets, parameterisation procedures and validation approaches is needed for certain modelling scenarios to ensure comparability of soil erosion risk assessment and suitable mitigation practices. Furthermore, we recommend that policy-relevant erosion risk maps should be verified by empirical data and thresholds derived from erosion risk maps should be adapted to regional conditions when used for policy guidelines. Hence, comparability, comprehensibility and regional adaptation are essential qualities of policy-relevant erosion maps.

1. Introduction

Agricultural soil erosion by water results in harmful changes in soil structure and loss of fertile topsoil (Pimentel et al., 1995) and contributes to muddy flooding of property and infrastructure, and eutrophication, pollution, and sedimentation of water bodies (Boardman et al., 2019). These effects can be categorised as on-site effects directly at the erosion site, and off-site effects. The relative significance of on– and off-site effects of soil erosion by water may vary by region, but usually, off-site effects by far exceed on-site, as exemplified in Western and Northern Europe (Boardman et al., 2019; Ulén et al., 2012). The severity of on-site effects is affected by local processes that affect the detachment of soil particles from the soil surface, whereas the severity of off-site effects is influenced by landscape features providing water flow and sediment transport pathways between arable systems and surface waters or infrastructure. Anthropogenic structures, such as roads, tracks, ditches, parcel borders, and erosion mitigation measures can have an important role in determining sediment transport distances.

The concept of connectivity describes the transport of water and sediment between linked landscape elements at different scales and to which degree the flow of water and sediment transport is facilitated (Bracken et al., 2015; Wainwright et al., 2011). Connectivity emphasises the role of landscape elements that enable sediment detachment and transport and act as connecting or disconnecting factors in sediment

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https://doi.org/10.1016/j.catena.2023.107600
Received 28 February 2023; Received in revised form 28 September 2023; Accepted 13 October 2023

transport during hydrologic events. Landscape connectivity becomes important once the soil is eroded by water and transported forward by water flow. Therefore, when it comes to the reduction of connectivity and the resulting off-site impacts, also erosion mitigation measures that are designed to reduce on-site effects have to be considered. The efficacy of the implementation of mitigation measures such as buffer strips, grassed waterways, reduced tillage or land use change can be investigated through targeted modelling scenarios (Devátý et al., 2019; Didoné et al., 2021; Fiener and Auerswald, 2005; Verstraeten et al., 2006).

Within the European Union (EU), Member States have to comply with regulations of both the EU's Common Agricultural Policy (CAP) and the Water Framework Directive (WFD) as well as other national policies. In the CAP, "good agricultural and environmental conditions" (GAECs) constitute a prerequisite for direct payments, in particular, GAEC 5 (minimising soil erosion) presupposes a sound knowledge basis on the erosion risk. Both policies require mitigation measures that target and have an effect on soil erosion. Thus, soil erosion risk maps produced by soil erosion models function as important tools to support policy-makers in designing policies for areas at risk of erosion. Furthermore, in the recently published Proposal for a Directive on Soil Monitoring (EU Commission, 2023), soil erosion is considered as one of the key soil threats in the EU. Several models have been used to produce soil erosion risk assessment maps to aid decision-making processes by identifying high-priority areas and/or which targeted measures to apply for maximum efficacy. Some examples include the USLE and CASE for Austria (Schmaltz et al., 2023; Brunner et al., 2023), Switzerland (Prasuhn et al., 2013), Czech Republic (Janeček et al., 2012), Hungary (Pásztor et al., 2016) and Spain (Martín-Fernández and Martínez-Núñez, 2011), WaTEM/SEDEM for Flanders, Belgium (Oorts et al., 2019), MESALES for France (le Bissonnais et al., 2002), PESERA for Norway (Kværnø et al., 2020), and RUSLE and VEMALA for Finland (Huttunen et al., 2016).

Soil erosion models can be divided into empirically-based, process-based, and a combination of both. The most widely used empirical model (Borrelli et al., 2021) is the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978, 1965) and its many derivatives, such as the MUSLE (Williams and Berndt, 1977), RUSLE (Renard et al., 1997), RUSLE-3D (Millward and Mersey, 1999) and RUSLE2 (USDA-ARS, 2008). Process-based models are models that intend to simulate processes, such as infiltration, runoff, soil detachment, sediment transport and deposition more explicitly than the simplified approaches used by empirical models. Prominent representatives of this group of models are WEPP (Laflen et al., 1991), LISEM (de Roo et al., 1996), EROSION-3D (Schmidt, 1996), PESERA (Kirkby et al., 2008), EUROSEM (Morgan et al., 1998). These approaches often require a large amount of data, which may not be available for large modelled areas, e.g. at regional or national scales. Many models have also been developed as a combination of process-based and empirical elements. These are defined as 'conceptual models' (Hajigholizadeh et al., 2018) or 'semi-empirical models' (Borrelli et al., 2021). Widely-used examples are WaTEM/SEDEM (van Oost et al., 2000) or SWAT (Arnold et al., 1998) or MMF (Morgan et al., 1984).

The performance of a model strongly depends on the quality of the input data, the goodness of the model parameterisation, and the adequacy of the model structure (Batista et al., 2019; Fiener et al., 2020; Fischer et al., 2018). Fiener et al. (2020) showed that the parameterisation of models has a large impact on the modelling results. Particularly, the implementation of landscape elements and mitigation measures as part of the model structure varies between models and their applications, which can significantly influence the model results and understanding of landscape processes (Keesstra et al., 2018). However, despite the importance of off-site effects of erosion, the diversity of the modelling approaches regarding the input data and parameterisation, and particularly how the landscape elements and soil erosion mitigation measures are implemented within these models to account for connectivity, is not well understood. Furthermore, a broader understanding of how different modelling approaches are being used to support environmental planning and decision-making within the EU is limited. These limitations considerably hinder the comparability of soil erosion assessments and their practical implications between European countries.

Understanding how connectivity elements are currently implemented in models and how connectivity affects the applications can be considered to be a prerequisite for conducting more reliable and harmonised erosion assessments in agricultural landscapes. Therefore, our research aims to address the above limitations through the following research questions:

- Which soil erosion models are used in Europe to assess the risk of soil erosion and the effect of mitigation measures?
- Which environmental parameters are included in the models and which data sets are being used to estimate these parameters?
- How are connectivity elements and mitigation measures considered or parameterised in the models used?

To answer these questions, we conducted and evaluated a European-wide survey, where we sent a questionnaire to soil erosion modellers who produce policy-relevant soil erosion risk maps for supporting local, regional, or national authorities or decision-makers.

2. Material and methods

2.1. Survey strategy

Our study is underpinned by an exploratory survey that was conducted from April to July 2021, within the project SCALE which is a part of the European Joint Programme (EJP) on SOIL. The ambition of the survey was to identify which soil erosion models are in use across different European countries. We aimed at contacting respondents in all European countries, as well as to cover individual regions in countries with different soil erosion modelling and management approaches. The targeted respondents were soil erosion modellers that either produce decision support for policy makers (such as soil erosion risk maps), provide advice to policy-makers and farmers or conduct soil erosion research with implications for policy design. Intensive research was conducted to identify regional stakeholders responsible for soil erosion. Renowned scientists and modelers were then engaged based on a combination of published work, recommendations from ministries, and referrals from SCALE project partners' contact lists. Hence, we contacted 53 respondents. For the final data analysis, 46 fully completed survey responses from various model users with a background at research institutions, consultancy firms and national or regional authorities in 18 European countries were available. The respondents came from Austria (1), Belgium (3), Czechia (1), Denmark (1), Finland (4), France (5), Germany (6), Hungary (3), Italy (4), Latvia (1), Luxembourg (1), Netherlands (1), Norway (1), Poland (1), Portugal (2), Slovenia (1), Spain (3) and Switzerland (2).

The survey was made available for respondents to be filled out via the SurveyExact online platform or a spreadsheet file and covered four thematic categories (A-D):

A. Model type and use
B. Model output and scales
C. Model-specific data sets and parameters
D. Parameterisation of connectivity elements and mitigation measures

The survey included a combination of open and closed questions addressing model use and output, model-specific data sets and parameters, and parameterisation of connectivity elements and mitigation measures. The survey contained specific questions regarding data sets and parameters depending on which model type. The questions on model output, connectivity elements and mitigation measures were the same regardless of the chosen model type. The questionaire is provided in the Supplementary Material. The following open and closed questions were addressed: 1) open questions, where respondents were provided with the opportunity to broaden their description of model use. 2) closed questions, where one option was always “Other – please specify”. In most closed questions, it was possible to check several answers, which gave multiple answers per model application. This combination of questions provides different types of information, providing complementary insights on the modelling setup and use.

2.2. Questions related to specific topics

In the category ‘Model type and use’ (A), we requested answers about the region that is covered by the soil erosion risk map (A1) and the type of model that was used to produce it (A2). We further assigned the responses on whether the model output is used for policy-making (A3), who are the respective end-users of the output (A4) and what is the main purpose of the model application (A5).

Further, we asked which type of output (B1) the selected model produces (soil loss net and gross, runoff, sediment yield, etc.), as well as the spatial (B2) and temporal (B3) scales of the model output. At last, a question on whether and how the model applications are validated was posed (B4).

Regarding ‘Data sets and modelling parameters’ (C), we asked which data sets and parameters are used to represent rainfall, soil, topography, land use and management (C2, C3, C5, C7). To allow better comparability of the data sets used, we asked for the data source (C1, C4, C6).

Questions related to the category ‘Parameterisation of landscape elements and mitigation measures’ (D) addressed which landscape elements (D1) and mitigation measures (D2) that have an impact on connectivity are implemented in the respective modelling approach.

2.3. Data analysis

2.3.1. Model type and use

In presenting the results, we analyse quantitative elements using descriptive statistics and deliberately refrain from using advanced statistical methods, as the purpose of the study is to explore differences in the use and setup of models. Furthermore, analysis of the open replies was used to highlight recurrent themes and broaden perspectives of the closed questions.

The responses were ordered according to which model group the model application belonged to in the following way:

- Group 1: USLE and all its versions (USLE, RUSLE, RUSLE2, other USLE version).
- Group 2: Models based on or with USLE elements (WaTEM/SEDEM, SWAT, Epic-Grid, VEMALA).
- Group 3: Process-based models, decision tree models and qualitative models (EROSION 3D, MMF, MCST-C, (Open)LISEM, PESERA, FLUSH, WaterSed, MESALES, unnamed qualitative model).

Although process-based models, decision tree models and qualitative models appear to be fundamentally different, we summarised them into Group 3 according to the low number of replies for decision tree and qualitative models (see section 3.1). We analysed how many of the responded models are used for policy. If it was stated that the model output is used for policy, we differentiated between authorities that are using the model on different administrative levels (national authorities, such as Ministries or regional authorities on federal state or county level). We assigned the model output to different schemes of policy type, i.e. regulations, subsidies and local planning. Regulations imply minimum standards for direct payments, as they are defined in the EU’s CAP GAEC Standards of conditionality. Subsidies imply Rural Development Programmes and Agri-Environmental Programmes, which are primarily based on voluntary participation of farmers. Local planning implies schemes that are not connected to CAP’s regulations or subsidies.

If it was stated that the model has no policy relevance, we distinguished if the model was used by an authority, from an academic research institution or the private sector, with the purposes of either research or consulting. Further, we grouped the model applications according to the scale of the output. The scales were a) national scale, b) regional scale, c) catchment scale and d) field scale, including parcels and plots. The exact numerical scale of the different scale categories was not specified in the survey.

2.3.2. Data sets and modelling parameters

To be able to compare single model parameters, only USLE type model applications (Group 1) were analysed regarding the use of data sets and modelling parameters (questions C1-C7), since the parameters of the models in groups 2 and 3 largely differ from each other. The modelling data and parameters were divided into four main groups related to rainfall, soil, topography and land use and management data, respectively.

USLE-like models are composed of different environmental and agricultural management factors: rainfall and runoff erosivity (R-Factor), soil erodibility (K-Factor), slope length and steepness (LS-Factor), cover and management (C-Factor) and support practice (P-Factor). The product of these factors yields the long-term average soil erosion in t ha^{-1} yr^{-1} (Wischmeier and Smith, 1978; Renard et al., 1997).

Table 1
Classification of landscape elements into landscape element groups (LEG) with description on the connectivity impact and examples for implementation in soil erosion models.

LEG	Landscape element	Description	Examples for implementation in models
1	Land use change (Luc), parcel borders (Pb)	Luc and Pb have only indirect impacts on connectivity, e.g. through the restructuring of a landscape, reduction of flow paths and changes in infiltration (Bakker et al., 2008; Devátý et al., 2019; Wang et al., 2022).	Transport capacity estimation in SEDEM (Van Rompaey et al., 2002); hydrological parameters in SWAT (Gessesse et al., 2015); via C-Factor in RUSLE (Cebecauer and Hofierka, 2008; Borelli et al., 2017); flow direction patterns in LISEM (Takken et al., 2001); multiple flow routing as part of the RUSLE L-Factor (Fiener et al., 2020)
2	Ditches (D), streamlets (S)	D and S increase or enable connectivity to another geomorphic system (Streeter and Schilling, 2020). D also disconnect land units (e.g. field parcels) from each other (Tähtikarhu et al., 2022).	Adaption of the drainage network (Alder et al., 2015)
3	Roads (R), thalwegs (T)	R and T function both connective and disconnective and might have implications on certain erosion processes as gullying (Harden, 2001; Harden, 2013; Ploey, 1990; Vandaele and Poesen, 1995)	Implementation of linear structures in WaTEM/SEDEM (Batista et al., 2022);
4	Buffer strips (Bs), vegetated waterways (Vw)	Bs and Vw lower sediment connectivity by sediment retention while hydrological connectivity remains rather constant (Fiener and Auerswald, 2005)	Modification of C-Factor and transport capacity coefficient in WaTEM/SEDEM (Verstraeten et al., 2006); process-based infiltration/runoff model (Fiener and Auerswald, 2005);

2.3.3. Landscape elements and mitigation measures

We classify landscape elements into four groups based on their key implications for connectivity and increased modelling complexity (Table 1).

We are aware that some landscape elements have multiple impacts on connectivity and thus could be assigned to other groups as well.

Likewise, we divide mitigation measures into four groups depending on their impact on hydrological and sediment connectivity (Table 2). For example mulching, no-till or cover crops are on-site measures that mitigate soil particle detachment but also increase infiltration and thus reduce overland flow (Klik and Rosner, 2020). Off-site measures are designed to minimise sediment loss, hamper or even intercept runoff, such as grassed waterways, in-furrow micro dams and flood retention basins, respectively. In this regard, we consider off-site mitigation measures not only as measures that are situated outside the field (such as flood retention basins) but that are also situated in the field but with an effect to reduce off-site impacts (such as grassed waterways and in-furrow micro dams).

3. Results

3.1. Model types and use

Altogether, we received 46 answers from 18 countries with a broad spatial distribution across Europe (Fig. 1a). As the community of erosion modellers with a policy focus is rather limited, 46 responses can be considered a solid basis for analysis. We received responses for the following models (Fig. 1b): USLE, RUSLE, RUSLE2 and other USLE applications (Group 1), Epic-Grid, SWAT, VEMALA, WaTEM/SEDEM (Group 2), Erosion3D, MMF, FLUSH, LISEM, MESALES, PESERA, MCST-C, WaterSed and a qualitative model applied in Poland without name (Group 3). The survey responses indicate that USLE models (Group 1) are with 23 applications the most widely used model type among all responding countries, while semi-empirical models (Group 2) with 7 and process-based models (Group 3) with 16 applications are less used (Fig. 1b).

Table 2
Classification of mitigation measures into mitigation measures groups (MMG) with description on the connectivity impact and examples for implementation in soil erosion models.

MMG	Landscape element	Description	Examples for implementation in models
1	Mulching (M), mulch-till farming (Mtf), no-till farming (Ntf), cover crops (Cc), terracing (T)	M, Mtf, Ntf and Cc cause transport limiting conditions due to higher soil cover, increased soil aggregate stability, smaller runoff and lower flow velocity (Klik and Rosner, 2020; Bombino et al., 2023; Räsänen et al., 2023). T reduces slope, which leads to decrease of runoff and increase of infiltration (Widomski, 2011)	Adaption of C-Factor in RUSLE or related models (Fiener et al., 2020; Bombino et al., 2023; Schmaltz et al, 2023; Räsänen et al., 2023); adaption of Digital Elevation Models (DEM) for consideration of terracing (Pijl, 2020)
2	Vegetated waterways (Vw), riparian buffer strips (Rbs), plant cover between rows (Pcbr), hedgerows (H)	Vw, Rbs, Pcbr and H minimise sediment loss by filtering sediment from runoff water or hampering soil translocation (Fiener and Auerswald, 2005; Verstraeten et al., 2006; Salvador-Blanes et al., 2006).	Modification of C-Factor and transport capacity coefficient in WaTEM/SEDEM (Verstraeten et al., 2006); process-based infiltration/runoff model (Fiener and Auerswald, 2005);
3	Contour farming (Cf), strip-till farming (Stf), in-furrow micro dams (Im)	Cf, Stf and Im temporally derogate or hamper runoff and reduce transport capacity of the water flow (Didoné et al., 2021; Heinen et al., 2022).	Adaption of the RUSLE P-Factor for Cf (Panagos et al., 2015b); calibration of numerical model for implementing Im (Heinen et al., 2022);
4	Flood retention basins (Frb)	Frb intercept or block water and sediment flow (Evrard et al., 2007).	Sediment accumulation in Frb in WaTEM/SEDEM (Krasa et al., 2019).

Concerning the question of whether the model application is used for policy-making (A3), 31 model applications are policy relevant, while 15 model applications are not explicitly used for policy. Considering model groups, it is observable that models of group 1 and 3 are predominantly used in policy to a higher degree than group 2 models (Fig. 2a). Models for the implementation of regulations or policy are used on different administrative levels. Policy relevant models are mostly used by regional authorities at the level of federal states or counties (60 to 67 %), compared to national authorities such as Ministries (Fig. 2b). Mostly policy relevant models are used for regulations, with 60 % particularly those of group 2, while group 3 models are predominantly used for local planning. For models with no policy relevance, the main users of group 1 and 3 models are research institutions (67 and 55 %, respectively), whereas, for group 2 models, the type of users is equally distributed (Fig. 2c). While more than 60 % of group 2 models are used for research purposes only, group 1 and 3 models are mostly used for consulting purposes, when not having policy relevance (Fig. 2c).

Group 2 and group 3 models produce more diverse outputs than group 1 models, which is represented by the responses of our survey (Fig. 3a-c). It is observable that soil loss is the most important output for models in group 1, while soil organic carbon (SOC) load and nutrient loss are rather negligible outputs (i.e. these processes are not often modelled) in all model groups.

All spatial scales in group 1 are almost equally represented (Fig. 3d). Temporal scales in group 1 are concentrated on event, annual and multi-annual scales, whereas modelling on event scale is applied on parcels or plots. Daily, monthly and seasonal scales are not represented. Group 2 models are applied on catchment and regional scale with daily or annual data (Fig. 3e). Group 3 models find their main use on the event scale with numerous applications from plot to catchment scale (Fig. 3f).

Validation of the models or single model parameters (B4) was not performed in 30 % of the group 1 model applications. This was the case for 6 % of group 3 model applications, while all model applications in group 2 included validation. The outputs of the model applications at the national and regional scales were not validated in 22 % and 20 % of the cases, respectively. At the field scale validation was not included in 14 % of model applications, while this was only the case for 4 % of the model applications at the catchment scale. Overall, the most commonly used validation technique was expert knowledge. Especially when models were applied at the regional or catchment level, data on sediment load and runoff were used, probably obtained through gauges or measurement stations.

3.2. Underlying data and modelling parameters

The main source of rainfall input data was national meteorological institutes. The use of national and regional data sets also meant that the temporal and spatial resolution of rainfall data varied among the model applications. For the rainfall input data, sub-hourly rainfall data were the most used temporal resolution (8 out of 23 responses) followed by hourly (5 responses) and daily data (4 responses). Seasonal data were not used at all. Of the sub-hourly rainfall data 5 min and 10 min data were the most reported followed by 30 min data. A variation of R-factor calculation methods was applied e.g. using various kinetic energy-intensity relationships, mainly those of Brown & Foster (1987) and Wischmeier & Smith (1978), but also e.g. regional relationships derived

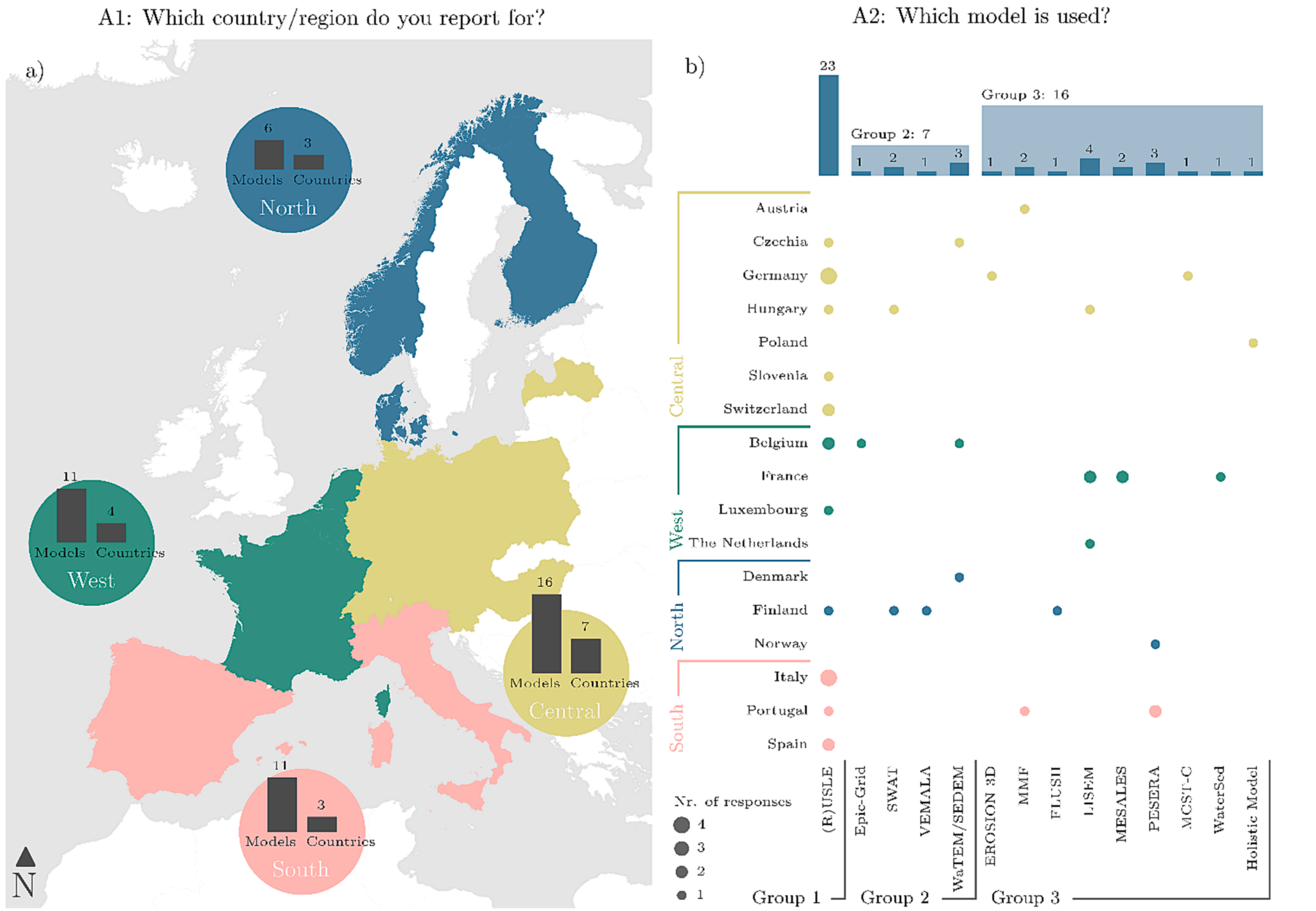

Fig. 1. Overview of survey responses by European Region (a) and model type (b). a: Responding countries are aggregated to European regions (North, West, South and Central). b: The number of responses is indicated by the size of the dot for each country and model. The bar plot on the top indicates the sum of the used models (and model groups) in all responding countries.

from field measurements (Petan, 2010). The majority used nationally or regionally available rainfall station data or spatially distributed rainfall erosivity grids based on empirically derived functions of measured rainfall station data. Two model applications (Finland and Luxembourg) used the European R-factor map produced by Panagos et al. (2015a) for their specific area, while two model applications used a fixed R-factor for their entire modelled area (Czechia and Flanders).

Regional or national soil databases (16 responses/59 %) are the most frequently used data sets, followed by “Other” (8 responses/30 %), which contained mostly field measurements and local soil surveys (Fig. 4a). EU data sets, such as LUCAS or the Soil Erodibility Data set (Orgiazzi et al., 2018; Panagos et al., 2014a) are scarcely used. Especially silt, sand, clay and organic matter content are widely used input soil data (Fig. 4b). Soil parameters were stated as derived from pedotransfer functions, expert-based methods, literature values, and directly assessed from measured field data or from soil (polygon) maps. These methods were also sometimes combined within one model application. While eight K-Factor maps are based on the approach of Wischmeier and Smith (1978) and four on the USLE nomograph, the majority (11) use various approaches to compute the K-Factor (Fig. 4d). The latter are mostly based on national adaptions, such as the DIN 19708 (2017) standard in Germany that uses empirical computation methods based on local soil data or experiments.

The resolution of the used DEM for modelling varied from 1 m to 25 m, with most applications using 5 m and 10 m resolutions. Hydrological corrections of the DEM (depression and sink filling) was used by 43 % of model applications via various methods (39 % No, 17 % no answer). A variety of single and multiple flow direction algorithms were used. None of the model applications considered the influence of tillage direction and oriented roughness on flow direction. The number of answers concerning the computation of the L-Factor divides into the two methods proposed by McCool et al. (1987) and Desmet & Govers (1996), with different combinations for using the slope-dependent exponent in both approaches. Similarly, S-Factors are computed using the methods of McCool et al. (1987), Nearing (1997), Govers (1991) and Wischmeier & Smith (1978).

The main land use data sources were regional and national databases (15). Field mapping (5) and Remote Sensing and GIS Mapping (6) were also applied in several model applications. The Corine Land Cover Database (2), LUCAS Land Cover Information (none) and Integrated Administration and Control System (IACS, 2) are scarcely used (Figs. 5-C4). The land use parameters used in the model applications (Figs. 5-C4) predominantly consist of canopy cover (8) and surface cover (9). Some model applications use information about prior land use (5) and rather specific parameters, such as rockiness or rock fraction of the surfaces (mentioned in ‘Others’, where detailed or special answers are possible). Only a few include surface roughness (2) and soil moisture (2). This is also reflected by the use of different methodological approaches to estimate C-Factors. C-Factors are mostly taken from a general or regional specific literature, as well as from laboratory analyses or field experiments. Only a few approaches estimate C-Factors from remote sensing data. Those approaches that compute C-Factors by the SLR-method (SLR: Soil Loss Ratio) proposed by Wischmeier and Smith (1978) also consider most of the land cover parameters for the calculation of the SLR-subfactors. The spatial allocation of C-Factors is almost equally distributed: while 11 approaches assign C-Factors for different land use

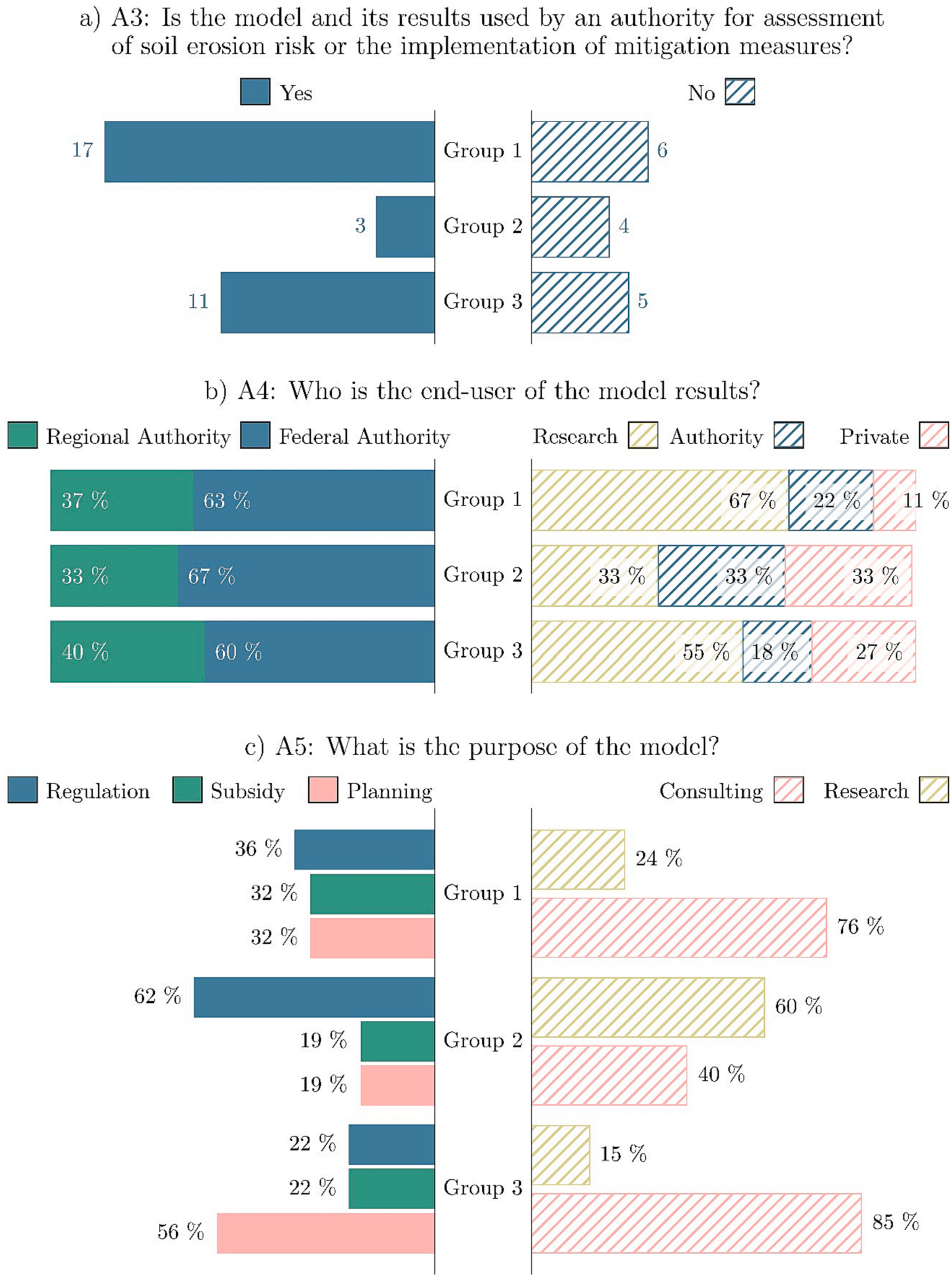

Fig. 2. Answers on a) model use of an authority (A3), b) end-user of the model results (A4) and c) purpose of the model (A5). Note that answers of all questions that are illustrated in filled colours indicate policy relevance (i.e. question A3 was answered with 'yes') and those with hatched colours indicate indirect policy relevance.

classes, 10 approaches assign them to individual fields. Most model applications made use of average annual values of land use or C-factor (18), instead of time varying values (4). Similar to the used land cover data sources, the most applied source of management data (Figs. 5-C4) was a regional or national database (10). The most modelled management practices (Figs. 5-C4) were tillage (11) and crop rotations (11). The consideration of management practices in the P-Factor is manifold, although most approaches do not consider P-Factors at all. Those who include P-Factors (9 out of 23 model applications) in their soil erosion computation use it for rather specific cases, such as for terraces in Italy or subsurface drainages in Finland.

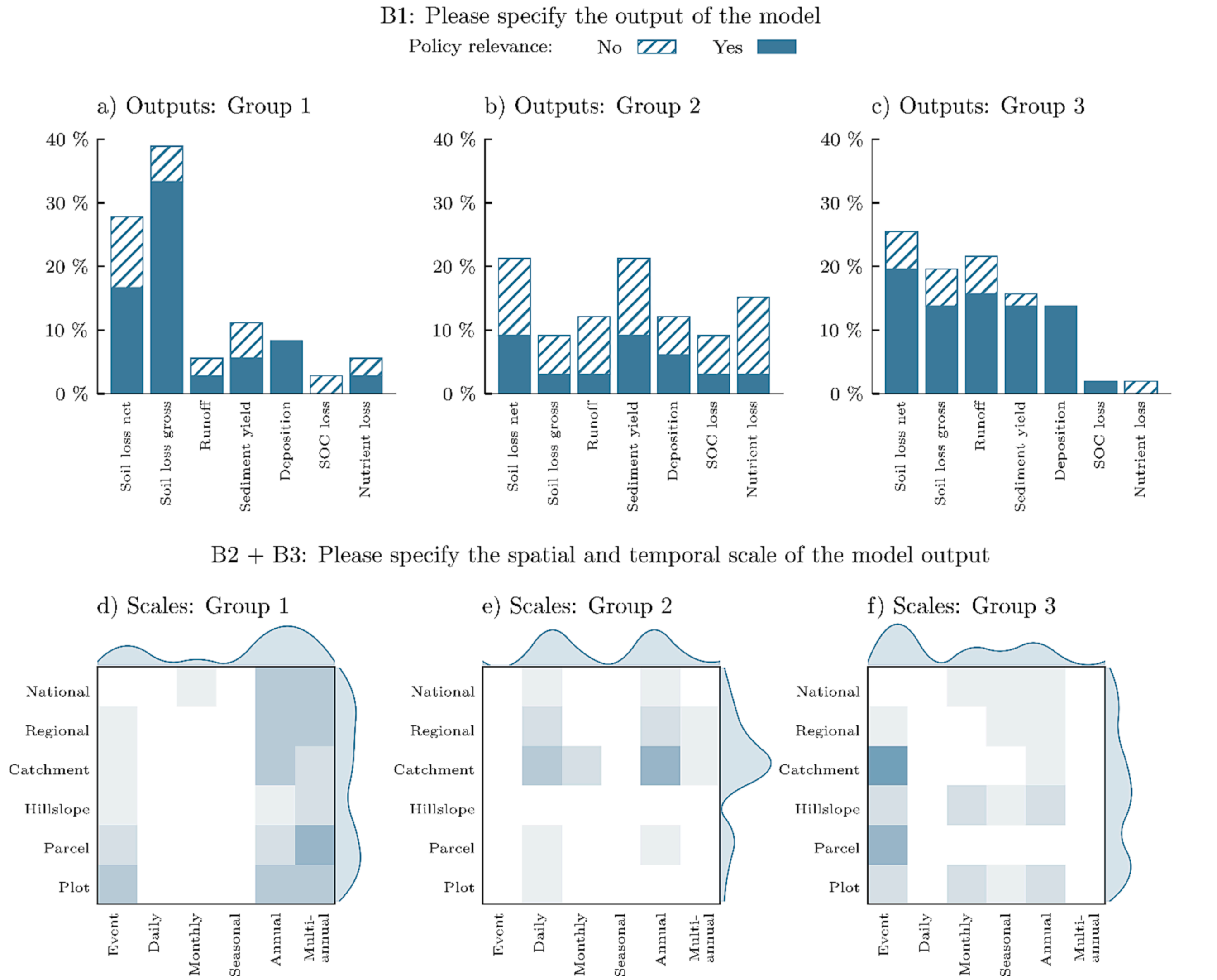

Fig. 3. Outputs of the model applications per model group and policy relevance (a-c), and spatial and temporal scales of the model output per model group (d-f).

3.3. Parameterisation of landscape elements and mitigation measures

Out of all 46 replies, 29 model applications include landscape elements, while 17 did not. The most included landscape elements were roads and land use change, which were both included in 21 model applications (Fig. 6 a). Ditches and parcel borders were also often included in the modelling (16 and 17 model applications, respectively). The majority of all model groups stated to include landscape elements (group 1: 57 %, group 2: 86 %, group 3: 63 %). Interestingly, the group 1 modelling applications included almost as many landscape elements as group 3 models (Fig. 6b). In group 2 the WaTEM/SEDEM model applications included the most landscape elements, while in group 3 this was done by the LISEM model applications.

The implementation of mitigation measures in the model applications was performed in 33 cases, while 13 model applications did not include mitigation measures. The most applied mitigation measures were cover crops (21), no-till farming (19), mulch-till farming (17) and mulching (15) (Fig. 7a). In all group 1 model applications, 59 mitigation measures were stated as included, while it was 33 for group 2 and 66 for group 3 model applications (Fig. 7b).

Looking at the spatial scale, we see that most landscape elements were included in model applications whose final application scale was regional or catchment scale (Table 3). Landscape elements with implications for connectivity were also included at a national scale (21), while only very few landscape elements were applied on parcel/plot scale (5). None of the model applications included a connectivity index. On a national scale, 8 % of the erosion models use a sediment transport model, while this applies to 20 % of the models on a regional scale, 54 % on catchment scale and 17 % on a plot/field scale. Thus, sediment transport models are mainly included in the models used at catchment scale. In contrast, none of the model group 1 use a sediment transport model, compared to 57 % of group 2 and 44 % of group 3. In case of mitigation measures, the most mitigation measures were included in the modelling applications that were applied at catchment scale (69), followed by regional (53), national (32) and field (4) scale (Table 3).

4. Discussion

4.1. Usage of models

The number of replies show that the USLE in its various forms and versions are widely applied throughout Europe and appears to be an important tool not only for research purposes (Borelli et al., 2021), but also for policy-making and decision support. The number of replies by model group should be considered with caution, since several models are used manifold by certain countries, which increases the total number of replies for the respective model (e.g. LISEM in France or PESERA in Portugal). The frequent use of the USLE model applications seems to

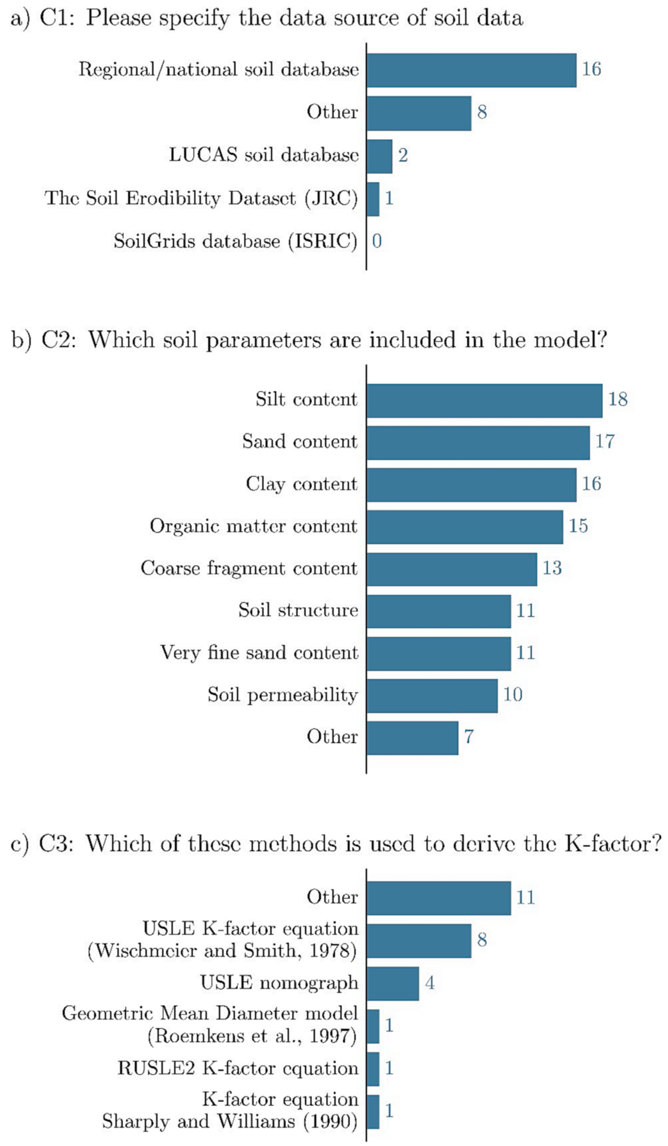

Fig. 4. Answers to a) soil data source (C1), b) soil parameters (C2) and c) the applied K-factor method (C3) included in group 1 model application.

result from their ease of use and straight-forward application routine with less data requirements compared to process-based models (Alewell et al., 2019; Benavidez et al., 2018). The high number of USLE model applications with policy-relevance show that this model is often applied nationally/regionally predominantly for identifying soil erosion hotspots (e.g. Prasuhn et al., 2013).

Semi-empirical models not only calculate soil erosion according to USLE, but also consider further erosion processes such as sediment transport and deposition. This implies that the model output allow for the interpretation of off-site effects and thus these models provide an additional perspective on erosion for policy makers. Initially we expected that semi-empirical models that are similarly easy to use, such as the WaTEM/SEDEM (Borrelli et al., 2021), would be more frequently represented in our data set.

The group of process-based models can provide a more comprehensive picture of erosion and sediment transport processes. However, they often include a large number of parameters and thus require a larger amount of input data, which may not be available for all areas (Hajigholizadeh et al., 2018). Furthermore, a lack of knowledge of certain erosion processes and their interactions can cause uncertainties in model predictions (Poesen, 2018). Increasing complexity may lead to increased uncertainty, but on the other hand, their explanatory capability is much higher than that of empirical models. Regarding sediment connectivity, process-based models can include detailed sediment transport descriptions (e.g. Turunen et al. 2017), which have potential to describe the degree and dynamics of connectivity more comperensively than other model types. However, for policy design, simpler models seem to be more feasible due to their capability to provide reasonable estimates of key variables with limited data and computational resources. This is also reflected by the answers in our survey, which show that process-based models are barely used for regional or national assessments, but rather on a local or catchment scale for planning and consulting purposes. The few decision-tree and qualitative models in group 3 make use of a completely different form of decision support allowing for the application on regional or national scales, such as MESALES for France or the unnamed model application for Poland.

Looking at the spatial distribution of model applications across Europe, our survey did not reveal any spatial tendency in model use according to the diversity of agricultural landscapes. Although, it could be hypothesised that in countries with predominantly on-site erosion problems, the use of models designed to account for on-site erosion, such as the USLE, would be dominant. In contrast, in countries with a rather flat terrain, we expected a focus on off-site effects such as pollution of water courses, and thus the application of modelling approaches that also account for sediment transport and deposition. This was also the case in Denmark and Finland, where modelling approaches that account for off-site effects, such as WaTEM/SEDEM and VEMALA, are used. Instead, it seemed like the scale of the model application could rather be the defining factor in the choice of model.

4.2. Representation of model data and parameters

Our survey documents that despite using the same model type (group 1 – (R)USLE), model applications are performed in various ways with different data sets and parameterisation. The single model parameters are calculated via a plethora of different equations, even though the same basic model is used. On this point, Wilken et al. (2018) showed that simulated sediment delivery had a high sensitivity (a range from 10 to 27 t ha^{-1}) to different kinetic energy-intensity relationships. Furthermore, the choice and quality of input data have a great impact on soil erosion prediction ability (Fischer et al., 2018; Panagos et al., 2014b). For example, the resolution of DEMs is crucial for reliably identifying erosion hotspots (Bircher et al., 2019). Fischer et al. (2018) found that parameterisation weaknesses were the most likely cause of differences between USLE predictions and field observations. This, together with the use of primarily national or regional data sets, which may not correspond to that of another country/region, hinders the comparability of soil erosion risk between countries/regions. This suggests that there is a need for harmonisation of data sets, parameterisation and modelling methods across Europe, as regional modelling applications differ and lead to substantial differences in erosion estimation, as also demonstrated by Fiener et al. (2020). On the other hand, it may also be argued that data that are available at national/regional scale are usually more representative and with more detail compared to data that are available only at the European level. For example, Räsänen et al. (2023) compared RUSLE-based European soil erosion map (resolution 100 m) by Panagos et al. (2015) and Finnish national soil erosion map (resolution 2 m) at agricultural areas and found that both identify roughly the same high erosion areas at large spatial scale (5 km), but at more local scale (100 m) the correlation between the maps was low (r = 0.32).

4.3. Implementation of connectivity elements and mitigation measures

Although models incorporating connectivity have been found to still be limited (Keesstra et al., 2018), our survey showed that several model applications included landscape elements and mitigation measures that are relevant for connectivity.

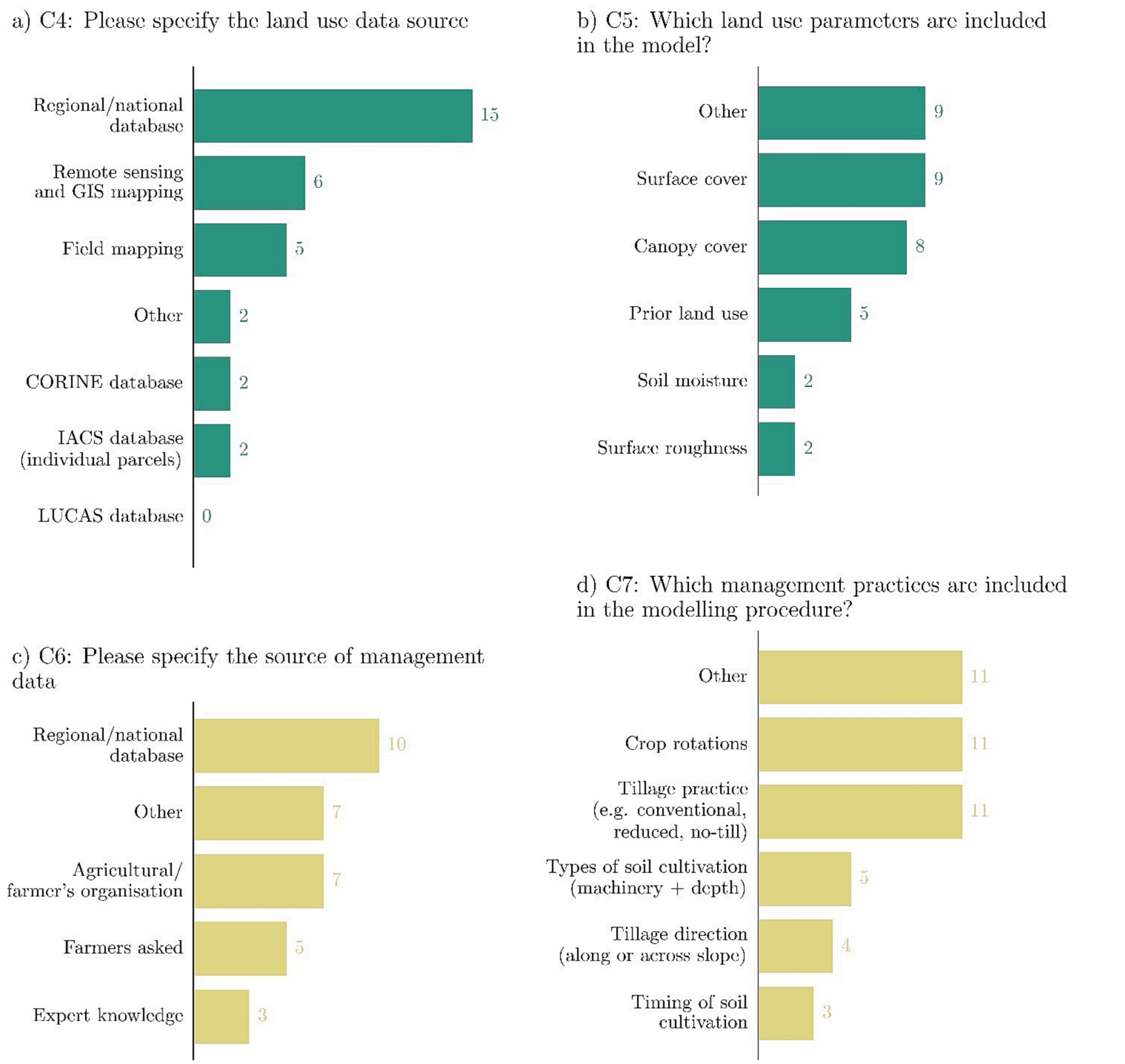

Fig. 5. Answers to a) land use data source (C4), b) land use parameters (C5), c) management data source (C6) and d) management practices (C7) included in group 1 model applications.

In our survey, group 1 model applications included several connectivity elements, which seems surprising, as none of the group 1 model applications included a sediment transport model. Models of group 2 and 3, which are generally more spatially and temporally distributed compared to USLE models, account for connectivity better, as they simulate the relevant processes over time and space. This may also be reflected in the high number of connectivity elements included in models used at the regional or catchment scale. Connectivity can be accounted for in modelling applications through the addition of various landscape elements, which affect runoff and sediment transport. How different models implement and model these elements depends on the model setup and parameterisation. Further, how the landscape elements are implemented in the model can affect the predicted sediment transfer and connectivity.

Land use change and parcel borders (LEG 1) are the most applied landscape elements in the model applications in our survey. This could be because these landscape elements are rather easy to implement in models, e.g. land use change can be directly included in the (R)USLE C-factor (Borrelli et al., 2017; Devátý et al., 2019). In addition, land use, land cover and field parcel organisation, as well as changes in these, has a strong effect on soil erosion risk and sediment fluxes (Wang et al., 2022) and are thus important to include in the modelling. In this way, land use change towards de-intensification can also be used as a mitigation measure for soil erosion (Bakker et al., 2008) and a strategy for future soil conservation (Borrelli et al., 2020). Parcel borders can act as a barrier to sediment connectivity due to differences in vegetation (Takken et al., 1999) but also produce higher erosion rates due to the concentration of waterflow at parcel borders (Takken et al., 2001). As such, parcel borders are a landscape element, which properties are not fully predictable from satellite images or DEMs, but needs observations and field mapping to make valid connectivity assessments (Boardman et al., 2019). Modelling efforts can introduce a measure of connectivity between parcels by changing the sediment transport e.g. in the parcel connectivity parameter of WaTEM/SEDEM (Batista et al., 2022) or the

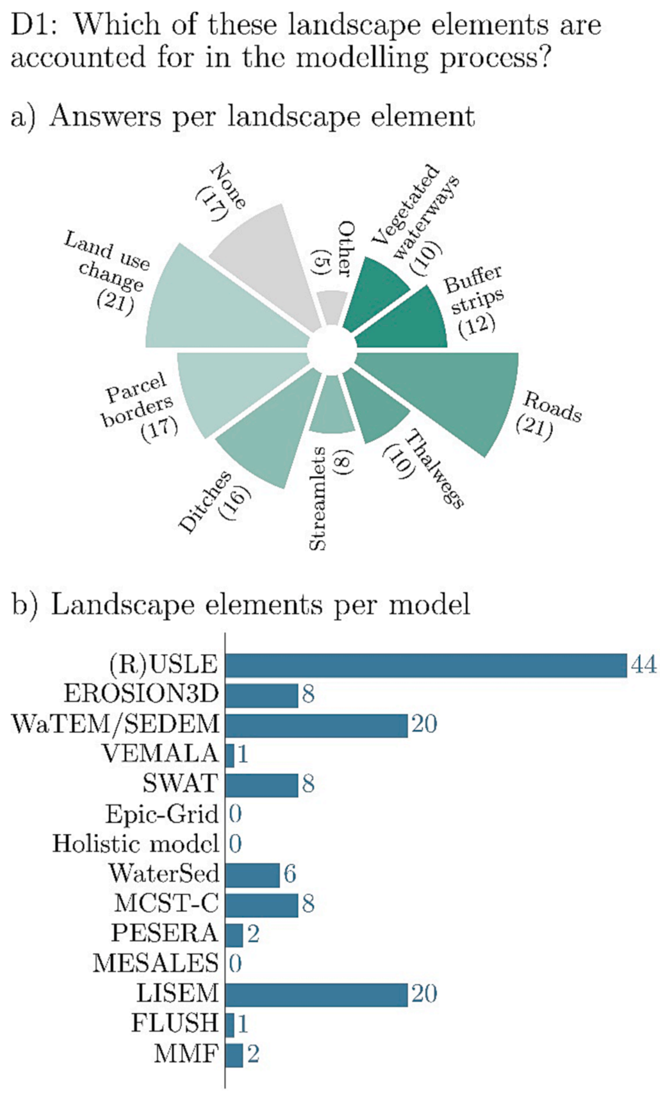

Fig. 6. Answers to the application of landscape elements (D1). a) Number of answers per landscape element and b) number of landscape elements implemented in each model.

LS-factor of (R)USLE (Devátý et al., 2019).

Landscape elements which connect one geomorphic system with another, such as ditches and streamlets, facilitate direct sediment transfer from fields to surface waters. Ditches and streamlets are thus often represented by so-called stream burning of the DEM, where the landscape elements are artificially decreased in the DEM (Bazzoffi et al., 2011; Hösl et al., 2012). This ensures that runoff is intercepted and drained via the hydrological network created by stream burning, but it is not without errors (Kenny and Matthews, 2005). Furthermore, it requires rather highly-resolved DEMs, which are not always available on a national/regional scale. Therefore ditches and streamlets are not so often considered compared to other landscape elements in the surveyed model applications. Further, for assessing soil erosion, both ditches and streamlets are not always necessary to be considered in the modelling procedure (e.g. on plot or parcel scale). In addition to ditches, subsurface drain pipes can form another drainage-related sediment connectivity element which is challenging to consider in models (e.g. Räsänen et al., 2023; Turunen et al., 2017).

Roads are well accounted for in the surveyed model applications, while their effect on connectivity is not one way (LEG 3 – both connective and disconnective). Correctly applying road connectivity in the modelling procedure can significantly influence the sediment transfer

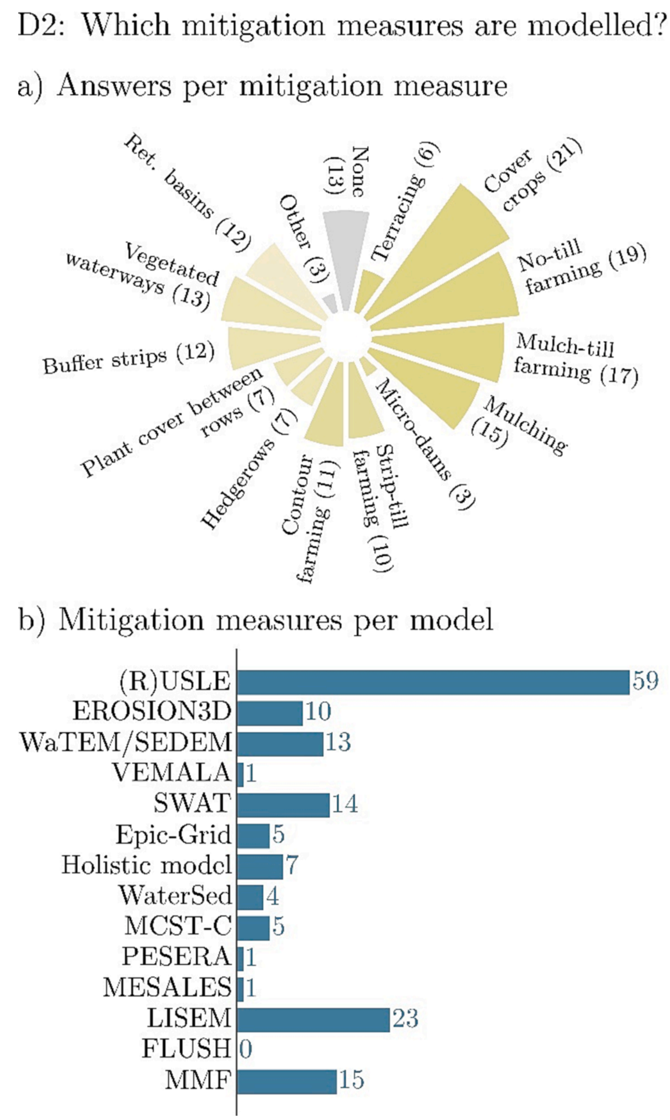

Fig. 7. Answers to the application of mitigation measures (D2). a) Number of answers per mitigation measure and b) number of mitigation measures implemented in each model.

Table 3
Number of landscape elements and mitigation measures applied in all modelling approaches distinguished for national, regional, catchment and parcel/plot scale.

	National	Regional	Catchment	Parcel/plot
Landscape elements	21	47	47	5
Mitigation measures	32	53	69	4

from field parcels to water courses (Batista et al., 2022). Runoff may be routed along linear features such as roads (Takken et al., 2001), causing deposition upslope of roads (Takken et al., 1999). On the other hand, roads can amplify connectivity by leading to concentrated runoff, which may be led into a drainage network, a stream or spill over onto downhill fields, thereby making them vulnerable to soil erosion (Bracken et al., 2013; Harden, 2013). The question is whether this dual role is observed in the model applications or if roads are rather considered as non-routing in the model and act as sediment sinks (e.g. Onnen et al., 2019). Thalwegs, roads and their drainage systems were specifically integrated with the soil erosion risk to produce a connectivity risk assessment map for Switzerland (Alder et al., 2015).

Buffer strips and vegetated waterways hinder sediment transfer and are thus an important landscape element to consider for connectivity. In the survey model applications, they were however the group (LEG 4) with fewest applications. These landscape elements work differently on water and sediment connectivity in that vegetated waterways and buffer strips may not hinder water flow but trap sediments (Boardman et al., 2019). This creates the need for model functions that simulate both deposition of sediment in the vegetation and the transport of water through it. Specialised models exist, which are able to account for the small-scale phenomena based on detailed physical descriptions at a local scale, although the parameterisation of these models is usually too demanding at a larger scale (Gumiere et al., 2011). Often models make use of the (R)USLE P-factor or the sediment delivery ratio (SDR), although these do not consider the effect of the spatial distribution of mitigation measures on the total sediment removal. Spatially distributed models may apply specific physical parameters such as sediment transport capacity (Verstraeten et al., 2006), sediment trapping efficiency (Lam et al., 2011), roughness or infiltrability parameters, which can be spatially parameterised for each mitigation measure (Gumiere et al., 2011).

None of the model applications in our survey on policy relevant modelling in Europe used a connectivity index, even though the use of such an index has been already proposed (Borselli et al., 2008; Mahoney et al., 2018; Michalek et al., 2021; Tähtikarhu et al., 2022). Connectivity indices are comprehensive measures that aim to describe landscape-scale connections and linkages that influence sediment flux. In contrast, the sediment delivery ratio, which is also used in some studies to address connectivity, is a metric that quantifies the efficiency of sediment transport, expressing the percentage of sediment yield reaching the stream relative to total soil erosion. Thus, the survey responses indicate that there is a potential for including connectivity indices in policy-relevant models.

The majority of model applications implemented some form of mitigation measures. Many measures were stated to be applied by changing the land use and C- or P-factor. The mitigation measures in MMG 1 have been shown to give a significant decrease in soil loss (Klik and Rosner, 2020; Prasuhn, 2020). These measures are also ones that can be relatively easily implemented in modelling through changes in the C-factor (Bombino et al., 2023; Räsänen et al., 2023) or by changing soil, land use and management practices in the already given model parameter catalogue (Ullrich and Volk, 2009; Vogel et al., 2016). This probably contributes to the high number of model applications including these measures (MMG 1) in their modelling.

The mitigation measures in MMG 2 are frequently not implemented in the model applications. These measures require very detailed information for modelling, such as the need to account for soil surface roughness in the runoff process (Heinen et al., 2022; Thomsen et al., 2015). On the other hand, in the USLE based models, mitigation measures such as contour farming may be simulated by altering the P-factor according to empirically estimated table values (Wischmeier and Smith 1978). However, many studies tend to ignore the P-factor due to difficulty in mapping or because the C-factor already considers some form of support practice (Benavidez et al., 2018). Mapping of e.g. contour farming can be done with a well-resolved DEM and the contouring efficiency used to derive the final P-factor (Didoné et al., 2021).

The measures in MMG 3 are the second most applied measures in the survey's model applications. These mitigation measures function and can be modelled similarly to the landscape elements in LEG 4, so we refer to the discussion above.

Flood retention basins (MMG 4) were integrated into several of the model applications, mostly in the process-based models. This is an off-site measure used to intercept and retain runoff and thus needs to be modelled with water flow information (Evrard et al., 2007). Thus, mainly models with a transport function would be able to model such a mitigation measure and its effect on connectivity.

The implementation of landscape elements or mitigation measures in a certain model does not necessarily mean that the function of this respective element or measure is adequately represented (or representable) in the chosen model. For example, a flood retention basin can be somehow considered a land use feature in the USLE but its function to retain water flow cannot be represented. Likewise, the implementation of a flood retention basin into a process-based model that may represent the key flow processes can still include uncertainties related to the parameterisation and model structure. In this manner, it would be wrong to simply conclude that 'good' models are those that incorporate a high number of landscape elements and mitigation measures with implications for connectivity and vice versa. A survey showed that modellers do not necessarily consider complex models as better models (Baartman et al., 2020). However, it has been argued that an improvement of soil erosion models should include a better representation of connectivity i.e. water and sediment fluxes through space (Nunes et al., 2018). It can be argued that comparison of model applications with different degrees of complexity could improve understanding on how much the complex processes can be simplified to produce reasonably accurate sediment connectivity descriptions and predictions in different spatial scales.

Overall, the majority of model applications in our survey included landscape elements and mitigation measures with impacts on connectivity in some form. However, detailed modelling of sediment connectivity was only the focus of a few model applications. In this regard, Favis-Mortlock et al. (2022) state that it is crucial to integrate more explicit representations of how landscape elements interact with runoff and that this would improve particularly catchment-scale erosion models to more accurately capture the characteristics of runoff, sediment transportation and size distribution. In addition, respondents may have had a different understanding of what constitutes a connectivity element. It has been found that literature holds no consensus on the connectivity concept due to the wide range of definitions and methods to describe and quantify connectivity (Najafi et al., 2021). Furthermore, the perception of the term connectivity by stakeholders was shown to differ (Smetanová et al., 2018), highlighting the need for knowledge transfer to enhance effective land and water management. This underpins the need for communication about the topic within both the soil erosion modelling community and among policy-makers.

4.4. Implications of modelling use on policy making

In general, our approach offered beneficial input from a varied group of stakeholders through published work, recommendations and referrals. However, it should be taken into account that more objective methodologies could be used for similar research, such as employing expert panels, random or stratified sampling or specialised online platforms. We opted for direct contact to stakeholders because the seemingly more objective approaches would have been either too inefficient or less transparent.

Our survey shows that multiple European countries use a variety of different modelling approaches to estimate soil erosion. However, the data used and parameterisation strategies of models vary substantially, which most likely leads to rather diverging modelling results. The application of models on a local scale rather serves planning purposes and is thus adapted to the individual study area, without the necessity for comparison to other study areas or modelling approaches. In contrast, the comparability of model results is particularly important on a regional, national or European scale. This underpins the need for harmonisation of data sets and parameterisation of models that are used as a basis to compile soil erosion risk maps which have policy implications on regional, national or European level. It could be argued that it is more important to adapt models to local conditions and to use locally available data sets than to strive for harmonisation of data sets and parameterisation and thereby possibly compromise the quality of the model results. However, multiple publications in the last decade have shown that there is a need and tendency for harmonised soil erosion risk maps with implications for national policy (German Industry Standard DIN 19708 for federal states in Germany, Panagos et al., 2015a for Europe, Borrelli et al., 2020 for the globe, etc.). However, a consistent implementation to produce harmonised erosion risk maps can be difficult, even if methodological standards exist. For example, Plambeck (2020) describes that there is no consensus on the use of the ABAG (German adaption of the USLE) in Germany. The federal state authorities are obliged to assess the soil erosion risk in accordance with the ABAG (as defined in the German Industry Standard DIN 19708), but the applied methodology differs considerably. This hinders comparison of erosion risk between states or regions and potentially leads to mismanagement and a diverging domestic implementation of mitigation measures in support of joint policies like the CAP or the WFD.

An important shortcoming of soil erosion risk maps on regional, national or European scale is that most are not validated by measured data. However, the possibilities of a plausible validation of large-scale erosion maps are limited. On the one hand, monitoring plots can be very costly and time-consuming, e.g. if soil erosion rates are to be quantified over longer periods of time in order to validate (R)USLE results. On the other hand, individual monitoring plots are only moderately suitable for validating large-scale maps. Thus, we agree with Batista et al. (2019) and Alewell et al. (2019) that more effort is required to validate soil erosion models and add that research should focus on the development of suitable validation strategies.

In the context of deriving legal regulations based on erosion risk maps, policy-makers usually require threshold values, which distinguish acceptable from unacceptable soil erosion, to define a certain legal regulation in a regionally meaningful way. For instance, in some European Member States, a soil erosion threshold based on USLE calculations is used to distinguish acceptable from unacceptable soil erosion in the GAEC 5 standards (e.g. Denmark, Belgium, Germany) or to evaluate measures from Agri-Environmental Programmes (cf. Schmaltz et al., 2023). However, these thresholds are based on arbitrary limits rather than on empirical evidence. The 'true' threshold between acceptable and unacceptable soil erosion varies largely from region to region or even parcel to parcel. This makes it necessary to define a threshold that is both flexible and evidence-based. Therefore, a threshold describing the ratio between soil erosion and soil formation would be suitable for defining the boundary between acceptable and unacceptable soil erosion (Verheijen et al., 2009). In this regard, also other criteria could play a role, such as crop yield, SOC-contents or agricultural economy, etc. This threshold applied to harmonised and considerably validated soil erosion risk maps would provide a better basis for policy-making and erosion risk assessment across national borders.

5. Conclusion

The current use of soil erosion models in Europe was investigated through an exploratory survey of 46 model applications covering 18 European countries. This novel analysis of erosion model applications revealed a variation in parameterisation, incorporation of landscape elements and mitigation measures with implications for connectivity and model use for decision-making in Europe.

The analysis showed that the majority of model applications (23 out of 46) were employing the USLE model or versions thereof. The models were applied at various scales (national, regional, catchment or field), and a variety of semi-empirical (7) and process-based + decision-tree (16) model applications were in use. The model applications were used for policy relevant purposes such as erosion risk assessment or mitigation measure implementation at a range of spatial scales in 31 cases. Other purposes mainly included research or consulting model applications.

The analysis highlighted a clear preference for using national or regional data sets and the use of differing parameterisations, even between model applications using the same model type (USLE). This leads to inconsistent soil erosion assessments, hinders comparison of model outcomes across Europe and potentially enforces inefficient management requirements. This implies that harmonisation may be beneficial in certain cases where a comparison of model predictions is necessary, e.g. large-scale model applications for use in European policy programs such as the CAP.

Most model applications included some form of a landscape element or mitigation measure that have implications for connectivity, such as roads, parcel borders, land use change or cover crops, mulch- and no-till farming, respectively. However, detailed modelling of sediment connectivity was rarely the focus of the model applications. The analysis of our survey model applications and literature documented that the implementation of landscape elements and mitigation measures depends on model setup and parameterisation. The increasing complexity in the effect on the connectivity of certain landscape elements introduces uncertainties in the modelling procedure and consequently affects the predicted sediment transfer and connectivity of the landscape. We believe an increased focus on sediment connectivity modelling in diverse agricultural landscapes across scales would improve erosion risk assessment and implementation of targeted mitigation measures through an improved system understanding of the erosion and sediment transport process. We recommend that policy-relevant erosion risk maps should be verified by empirical data and thresholds for policy guidelines derived from erosion risk maps should be adapted to regional conditions. Hence, comparability, comprehensibility and regional adaptation are essential qualities of policy-relevant erosion maps.

Declaration of competing interest

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

Data availability

Data will be made available on request.

Acknowledgements

This work has been funded by the European Union's Horizon 2020 research and innovation programme (Grant agreement No 862695).

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2023.107600.

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