Author(s): M. Tähtikarhu, T.A. Räsänen, J. Uusi-Kämppä, J. Hyväluoma

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How to site grassed areas to reduce agricultural erosion efficiently? A computational analysis in Finland

M. Tähtikarhu *, T.A. Räsänen, J. Uusi-Kämppä, J. Hyväluoma

Natural Resources Institute Finland, Tietotie 4, FI-31600 Jokioinen, Finland

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ABSTRACT

Spatial patterns of land-cover and agricultural operations have clear impacts on soil erosion. Allocating a portion of cultivated area for grass is a widely applied strategy to control erosion. However, it is still unclear how much and where grassed area should be spatially targeted in different landscapes to control erosion efficiently. To address this challenge, we estimate the potential of high-resolution RUSLE-based spatial targeting of grassed areas to improve erosion mitigation in two topographically different catchments in southern Finland. Erosion reductions of 1) policy-based targeting (buffer strips along main streams according to current CAP strategy) were compared with 2) RUSLE-targeted grassed areas (based on the highest computed erosion values within field parcels and sub-catchments). Furthermore, we computationally explored 3) how different rates of optimally located grass areas affected erosion and 4) how the areas could be computationally processed to continuous entities. The erosion reductions were estimated with 2×2 m^2 resolution RUSLE computations in all the scenarios. The RUSLE-targeted grassed areas demonstrated greater erosion reductions compared to the policy-based siting of grass areas along riparian fields. With optimal targeting, erosion risks could potentially be reduced up to 24 percentage points (up to 46 % erosion reduction), compared to the buffer strips. Increasing optimally targeted grassed area gradually from 0 to 100 % decreased erosion non-linearly. The largest share of erosion was generated in disproportionally small land areas (~20 % of the land area). The location of the hotspots in relation to the streams varied between the sub-catchments and field parcels. These quantifications demonstrate the potential value of models for targeted landscape scale spatial erosion management. A more comprehensive assessment of erosion mitigation could benefit of improved empirical validation and consideration of other aspects of erosion and sediment transport, such as local drainage efficiency and reduction of erosion during flooding of rivers.

1. Introduction

Agricultural operations, land cover and their spatial patterns have clear implications on soil erosion (e.g. Zhang et al., 2014; El Kateb et al., 2013; Zhou and Shangguan, 2007). Cultivated agricultural land areas are particularly prone to erosion (García-Ruiz et al., 2015) which has consequences on agricultural soils and on the locations where the eroded sediment is transported. The sediments can degrade downstream water quality and block drainage infrastructure (e.g. Boardman et al., 2019). The processes leading to erosion are spatiotemporally variable due to local soil properties, topography, and weather conditions (e.g. Remund et al., 2021; Ulén et al., 2012) and challenging to accurately estimate in different locations (García-Ruiz et al., 2015). Several erosion mitigation measures have been developed and their impacts have been demonstrated in different agricultural land areas (e.g. Remund et al., 2021; Stutter et al., 2012; Turtola et al., 2007; Istok and Kling, 1983). These measures include for example grassed buffer strips, grassed water ways, no-till and conservation tillage measures and drainage procedures. It can be argued that erosion control should start by preventing detachment of soil particles at the source area with crop and soil management, and continue by trapping soil particles on field, field edges, and outside fields (Uusi-Kämppä, 2020). While the overall erosion impacts of the measures have been documented, it is still unclear how the mitigation measures should be spatially targeted to different landscapes to optimally control the erosion generation. Allocating a portion of cultivated fields for grassed areas is a widely applied strategy to control erosion (e.g. Stutter et al., 2012), which makes their improved targeting a worthwhile research topic and a concrete example of improved

* Corresponding author.
E-mail address: mika.tahtikarhu@luke.fi (M. Tähtikarhu).

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efficiency of spatially targeted mitigation measures.

Impacts of efficient targeting of grassed areas have been previously demonstrated in laboratory- and hillslope-scale studies (Zhang et al., 2024; Pan and Ma, 2020; Zhang et al., 2018). Grass areas can efficiently reduce erosion in the particular area of the field where they are located, and may also affect sediment transport processes by filtering sediment suspended in overland flow (e.g. Deeks et al., 2012; Dillaha et al., 1988). Grassed areas can be spatially implemented in different ways, such as buffer strips, grassed water ways or larger grassed areas. The challenge of optimal targeting of erosion reduction measures at the landscape scale is the identification of those locations where the erosion rate is high and where the potential of the measures to control erosion is also high. Clear differences in erosion rates have been previously noted between and within catchments, landscapes and field parcels (e.g. Räsänen et al., 2023; Remund et al., 2021; Ulén et al., 2012). Relatively small land areas within catchments can contribute disproportionally to the total erosion seen in a catchment (Remund et al., 2021; Belayneh et al., 2019; Djodjic and Villa, 2015). However, currently the targeting of erosion control measures in Finland is done with simplistic means, and the current strategy to implement the common agricultural policy of the European Union (CAP) sets a minimum target of 3 m wide grassed strips along watercourses. According to CAP Strategic Plan 2023–2027 for Finland, 30–50 m wide buffer strips along watercourses can be established on erosion-prone fields, and in certain cases, small parcels (<1 ha) along watercourses may be left entirely as grassed areas. Siting the grassed areas to mitigate the impact of erosion on water quality from hotspots could potentially be a more efficient practice to control erosion, but spatial data on the locations of the hotspots are rare. Note also that in the local Finnish high-latitude conditions, agricultural fields are typically well drained using subsurface drains. Proper subdrainage induces relatively low surface runoff and consequently low sediment loads via surface runoff compared to loads via subsurface drains (Räsänen et al., 2023; Turunen et al., 2017; Turtola et al., 2007). The subsurface drains can convey sediment loads directly to adjacent surface water bodies. Thus, siting the grassed areas to reduce erosion at the source areas in those locations where they are placed can be considered a reasonable strategy to control erosion. This study focuses particularly on how differently located grassed areas can reduce erosion generation.

Computational models provide possibilities to identify potential erosion hotspots within landscapes (e.g. Belayneh et al., 2019; Djodjic and Villa, 2015; Hamel et al., 2015). There are considerable uncertainties in the models and consequently differences in their outputs, but the predicted long-term averages and simulated relative spatial differences can be useful and informative (Jetten et al., 1999), particularly for the identification of such areas where erosion control measures could have a high impact. Revised Universal Soil Loss equation (RUSLE) is among the most widely applied erosion models, especially for policy-relevant estimates (Schmaltz et al., 2024) and RUSLE-based models can have the capability to distinguish between areas of high and low erosion (e.g. Gashaw et al., 2021; Djodjic and Villa, 2015; Hamel et al., 2015, 2017). RUSLE aims to describe the impacts of different key factors on long-term erosion, including topography, meteorological conditions, soil properties and farming practices (Renard et al., 1997). Recent RUSLE-based developments in the high-resolution spatial estimates on erosion (e.g. Räsänen et al., 2023; Panagos et al., 2015) provide means to computationally evaluate the potential of targeting grassed areas in high resolution from field to region and country scales. The potential of such targeting has been previously discussed (Belayneh et al., 2019; Hamel et al., 2015) and different decision support tools have been developed (e.g. Deeks et al., 2012; Dosskey et al., 2008). However, the implications of improved targeting have been rarely quantitatively analyzed.

The aim of this study was to compare the erosion reduction potential of RUSLE-based spatial targeting of grassed areas with current policy-based targeting. The analysis was conducted in two different sub-catchments in southern Finland, and the implications of these targeting strategies were determined with RUSLE. The study used recent RUSLE erosion data of Räsänen et al. (2023) with 2×2 m^2 resolution, and the resolution was considered appropriate for considering spatial variability of erosion within field parcels. This study focused particularly on the role of grassed areas in reducing erosion.

2. Materials and methods

Methodologically the study combines spatial data from two study areas with spatial computations. The study outline is schematically shown in Fig. 1 and described in more detail below.

2.1. Study area and field parcels

Two coastal sub-catchments, Aurajoki (WGS84: 60.5325°N, 22.4371°E) and Mustionjoki (WGS84: 60.1232°N, 23.7409°E) in southern Finland (Fig. 2a), were chosen for the study to explore the impacts of spatial targeting of grass areas on erosion in topographically different areas with relatively similar climatic conditions. At the study locations, the mean annual temperature is around 6 °C, monthly means vary typically between −9 and 19 °C, and the mean annual precipitation is around 700 mm (e.g. Jokinen et al., 2021). The long-term annual precipitation exceeds the annual evapotranspiration in the local conditions (e.g. Koivusalo et al., 2017). The precipitation excess generates the need for proper land drainage in the agricultural areas. Thereby, the fields are commonly drained by subsurface drains and open ditches typically surround the field parcels (TIKE, 2011). In Aurajoki, the topographical variations were concentrated close to the mainstream while in Mustionjoki they were typically located further away from the streams (Tähtikarhu et al., 2022). Clay soils are the dominating soil type within both catchments. More information on the areas can be found from previous erosion and sediment transport studies (Räsänen et al., 2024; Tähtikarhu et al., 2022).

Since the current buffer strip policies of Finland provide subsidies only for such field parcels, which are located adjacent (≤10 m distance) to main water bodies, we selected those field parcels which were similarly located adjacent to Aurajoki and Mustionjoki rivers. The delineation of the field parcels was taken from the field parcel data of the Finnish Food Authority (e.g. Tähtikarhu et al., 2022) and the location of the main streams was from vectorized open data containing rivers and streams (Finnish Environment Institute and National Land Survey of Finland, 2021; e.g. Viinikka et al., 2023). This resulted into areas of 870 ha (2.2 million grid cells with the 2×2 m^2 grid) and 595 ha (1.5 million cells) in the Aurajoki and Mustionjoki sub-cathment, respectively. The parcel areas had medians of 4.1 and 6.7 ha in Aurajoki and Mustionjoki, respectively, and the distribution of their areas is shown in Fig. 2 (206 parcels in total). The mean slopes of the studied areas were 2.8 % (standard deviation $\sigma = 3.3$ %) in Aurajoki and 1.9 % ($\sigma = 2.0$ %) in Mustionjoki. At Aurajoki, the 20, 40, 60 and 80 % slope quantiles were 0.62, 1.17, 2.18 and 4.39, respectively, demonstrating relatively rare occurrence of steep slopes. At Mustionjoki, the corresponding slope quantiles were 0.35, 0.68, 1.4 and 3.0, respectively.

2.2. RUSLE data

The RUSLE (Renard et al., 1997) is an empirical model for estimating long-term average soil loss. It was originally developed for estimating soil loss at a field slope scale, but it is commonly used as a spatially distributed model where calculations are performed over a spatial grid. It is the most commonly used erosion model, it has been used in various regions around the world where it is often used to support policy making (Schmaltz et al., 2024; Borrelli et al., 2021; Alewell et al., 2019). The RUSLE equation is:

$$A = R \times K \times LS \times C \times P \tag{1}$$

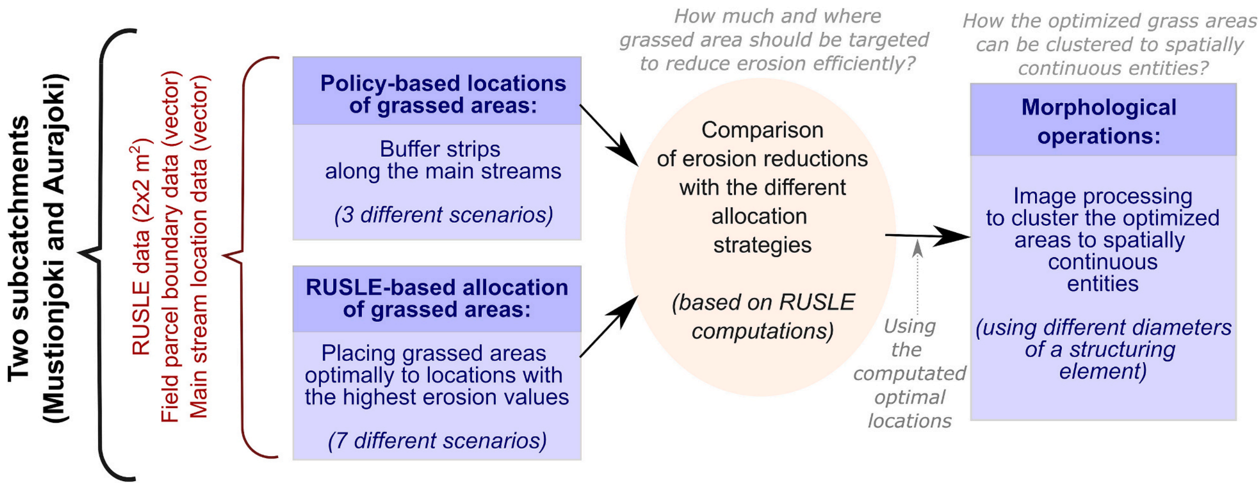

Fig. 1. Schematic overview of the utilized data and the conducted computations.

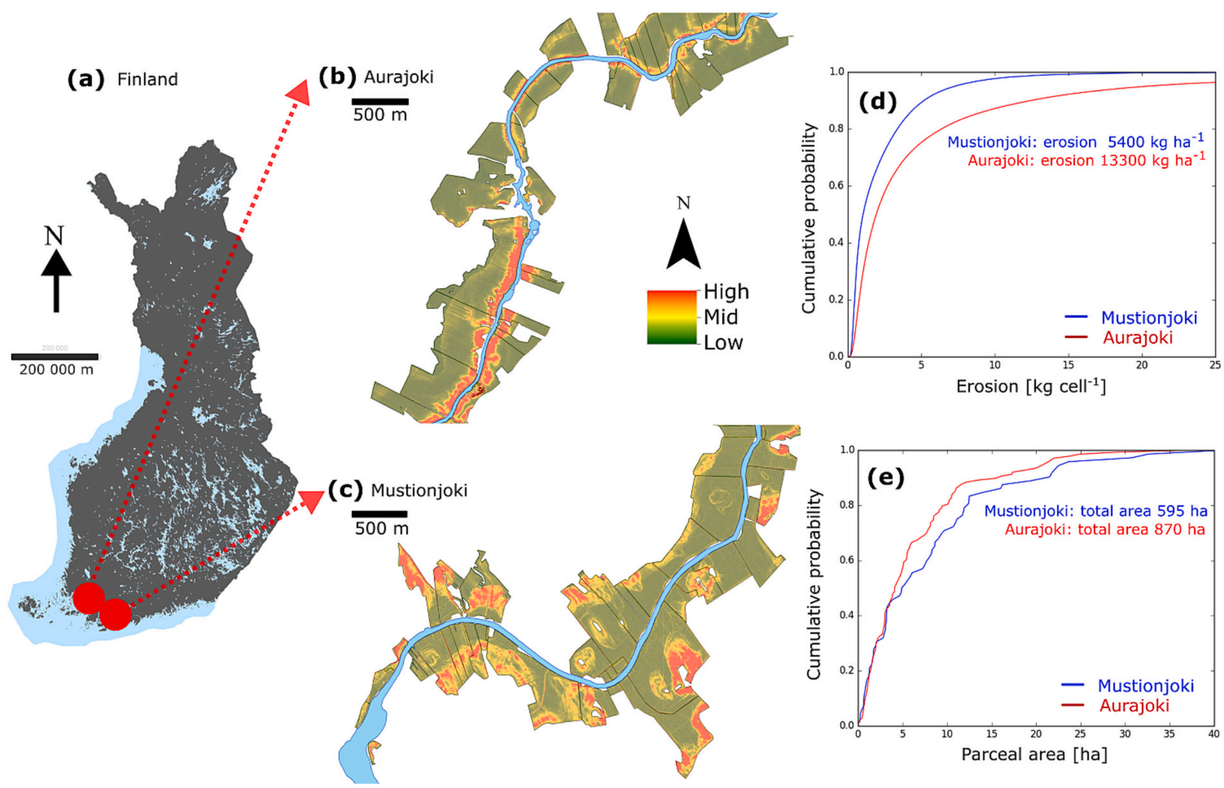

Fig. 2. (a) Location of the study areas in Finland and a visual example of the used data (Räsänen et al., 2023) showing the distribution of erosion within the field parcels located close to the mainstream in (b) Aurajoki and (c) Mustionjoki sub-catchments and (d) cumulative probability distribution of the erosion values (2 × 2 m^2 grid cell resolution) within the field parcels and (e) the areas of the field parcels within the study area. The blue areas in b-c show the main streams. Erosion corresponds to erosion under spring cereal cultivation with moldboard ploughing in the autumn, without subsurface drainage. The data in b-d are from Räsänen et al. (2023) and a contains data from the National Land Survey of Finland Database (administrative areas).

where A is the annual average soil loss (t ha^{-1} yr^{-1}), R is the rainfall-runoff erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), K is the soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}), LS is the slope length (L) and steepness (S) factor (−), C is the cover-management factor (−), and P is the support practice factor (−). In the original field slope scale use, A is the total sum of sediments transported to the end of the slope, but in the spatially distributed use it is the sum of sediments transported to the downslope edge of that cell where the erosion occurs. In other words, the spatially distributed RUSLE does not account for sediment transport between the grid cells and its estimates can be considered as gross erosion estimates.

In the current study, we used a gridded 2 × 2 m^2 resolution *R, K, LS* data from Räsänen et al. (2023). The R was calculated from gridded 1 × 1 km^2 resolution European R data with 64 precipitations measurement stations (on average 7 years of data per station) for Finland (Panagos et al., 2015). The K data was derived from vector-based 1:200000 scale Finnish Soil database (Lilja et al., 2017a) attributed with soil class specific K values (Lilja et al., 2017b, 2017c). The LS data was computed from 2 × 2 m^2 LiDAR-derived Digital Elevation Model (DEM) using Desmet and Govers (1996) method.

The average R values for the Aurajoki and Mustionjoki sub-catchments are 360 and 314 MJ mm ha^{-1} t^{-1} yr^{-1}, respectively. For the field areas of Aurajoki and Mustionjoki sub-catchments, the average K values are 0.040 and 0.057 t ha h ha^{-1} MJ^{-1} mm^{-1}, and the average LS values 0.470 and 0.830, respectively. At the study areas, LS varied clearly more (0.04–326.86) than K (0.008–0.057 t ha h ha^{-1} MJ^{-1} mm^{-1}) and R (279–432 MJ mm ha^{-1} t^{-1} yr^{-1}).

The 2 × 2 m^2 grid resolution was chosen to reasonably consider spatial variation within different field parcels. Using 2 × 2 m^2 DEM data and calculating the other factors from more coarse data can be consider plausible, since topographical variations are considered to be spatially more frequent than those of rainfall and soil class in the local conditions. The soil classes of the studied areas were mostly clay (K value 0.04 in 91 % of the area in both sub-catchments). Rest of the areas consisted of coarser soil materials (K values 0.008–0.057).

For cultivated field areas, we considered uniform management and crop cover for all field parcels (representing spring cereals with autumn ploughing without subsurface drainage) to facilitate straightforward comparability of the study areas. For these areas we used the C value of 0.211, which is the calibrated value for spring cereals with conventional autumn moldboard ploughing in Finland (Räsänen et al., 2023). For grass covered areas (perennial), we used the C value of 0.065, which is also the calibrated value in Finland (Räsänen et al., 2023). We did not consider the potential subsurface drainage in P. Subsurface drainage reduces the magnitude of erosion (Renard et al., 1997) and it is a common practice in Finland, but its implementation between and within the field parcels vary.

Thus, in our results, we use the term "erosion" to refer to RUSLE estimated long-term gross erosion under spring cereal cultivation with autumn moldboard ploughing, and with different implementation rates of perennial grass areas. The spatial variable can be considered to describe erosion risk rather than absolute erosion. The details and performance related to the data is described in Räsänen et al. (2023).

2.3. Computational scenarios

The conducted scenarios consisted of allocation of grassed areas to the two sub-catchment areas with different strategies. The allocation was conducted either 1) as buffer strips with different widths along the main streams (policy-based siting) or 2) by placing the grassed area optimally (sequentially from the computational grid cells with the largest erosion values towards the lowest values) based on the RUSLE data. The optimization was conducted separately for each parcel and also for the whole studied area, to explore which one results in larger erosion reduction. The resulting nine scenarios are listed in Table 1. In all computations, the grassed areas were described with the C-factor of RUSLE (see Section 2.2). Thus the approach described the erosion reduction at the area where the grassed area is located, and does not account for sediment transport and the trapping of suspended sediment in such overland flow fluxes which might flow through the grassed area.

The results of the scenarios were analyzed by comparing the differences in erosion reductions due to the scenarios. Due to modeling uncertainties (e.g. Jetten et al., 1999), the analysis is considered to be informative particularly regarding the relative potential of different allocation strategies to reduce erosion efficiently (rather than absolute reductions in erosion). The spatial computations were conducted using ArcGIS software.

In addition to these scenarios, we also conducted a scenario where the coverage of grassed area was gradually increased from 0 to 100 % (with steps of 5 %) sequentially from the cells with the largest erosion values towards the lowest values (scenario abbreviation C0–100,

Table 1
Computational scenarios with the different allocation strategies of grassed areas within the studied areas (field parcels in the vicinity of the main streams) of Aurajoki and Mustionjoki sub-catchments. Cells refer to computational grid cells (2 × 2 m^2).

	Strategy	Location	Grass coverage	Abbreviation
Policy- (buffer strip) and RUSLE-based (optimized) allocation strategies	Buffer strip	Along the main stream	Width 10 m	BF10
		Along the main stream	Width 30 m	BF30
		Along the main stream	Width 50 m	BF50
	Optimization, parcels	Cells with the largest erosion values within each parcel	Covered area corresponds to the parcel-specific area covered by the 10 m buffer strip (each parcel separately)	OP10
		Cells with the largest erosion values within each parcel	Covered area corresponds to the parcel-specific area covered by the 30 m buffer strip (each parcel separately)	OP30
		Cells with the largest erosion values within each parcel	Covered area corresponds to the parcel-specific area covered by the 50 m buffer strip (each parcel separately)	OP50
	Optimization, sub-catchment	Cells with the largest erosion values within the sub-catchment	Covered area corresponds to the total area covered by the 10 m buffer strips	OC10
		Cells with the largest erosion values within the sub-catchment	Covered area corresponds to the total area covered by the 30 m buffer strips	OC30
		Cells with the largest erosion values within the sub-catchment	Covered area corresponds to the total area covered by the 50 m buffer strips	OC50
	Increasing rate	From cells with the largest to the lowest erosion values within the sub-catchment	From 0 to 100 % of the sub-catchment area	C0–100
	Morphological operations on OC50	OC50 allocations subjected to morphological operations	Based on OC50 and morphological operations	M0–90

Table 1). This scenario was done separately for Aurajoki and Mustionjoki. The purpose of the scenario was to explore how large erosion reductions could be potentially achieved with different allocation rates of grassed area.

In addition to these scenarios, we conducted morphological image processing to assess how the optimized grass areas could be computationally clustered to spatially more continuous entities, instead of possibly spatially highly fragmented areas. The grassed areas were filtered with morphological closing and opening operations, which fill in holes and gaps in the grassed area and removes isolated grass areas, respectively. Morphological operations traverse the image with a small-sized structuring element, whose size defines the size of the features filtered. Here, the filtering was conducted using both opening and closing operations with a spherical structuring element diameter ranging from 6 m to 90 m (scenario abbreviation M0–90, Table 1). The morphological image processing was conducted using Python and the OpenCV library.

3. Results

The buffer strips with different widths reduced erosion more efficiently in Aurajoki (reduction range 6–34 %) than in Mustionjoki (1–9 %), as shown in Fig. 3. The optimization of the grass areas increased the reductions more in Mustionjoki (up to 24 percentage points, pp.) than in Aurajoki (up to 15 pp.), as compared to the buffer strips (Fig. 3). The optimization at the catchment-scale was more efficient in reducing erosion than the parcel-scale optimization in all of the scenarios, but the efficiency gain was higher in Mustionjoki (3–7 pp.) than Aurajoki (2–3 pp.) compared to optimization at field parcel scale (Fig. 3). Clearly, the optimal areas were generated by varying *LS*, as *LS* had a greater variability than the other RUSLE factors in the study conditions, as visually demonstrated in Fig. 1A. These results underline the role of topography in the generation of the optimal areas within field parcels.

The optimal grass locations differed from the buffer locations more in Mustionjoki than Aurajoki area as exemplified in Fig. 4a-b. Note that the optimal areas were scattered throughout the area and distributed among a wide range of field parcels. Regarding the erosion of different field parcels, the optimization brought a more efficient reduction per parcel, compared to the buffer strips along watercourses (Fig. 4c-d). The median reduction ratios (per total parcel-scale erosion risk) were 0.35–0.66 with the optimization scenarios and the parcel-scale reductions varied widely (Fig. 4c-f). Compared to parcel-scale optimization, the catchment scale optimization focused the distributions towards less efficient reduction per parcel and simultaneously left some parcels without grassed area (Fig. 4e-f).

Optimal increase of grassed area (sequentially from the areas with the largest erosion values towards the lowest values) reduced relative erosion (the erosion per total erosion) nonlinearly as shown in Fig. 5a (scenario C0–100). Allocation of grassed area reduced the relative erosion more efficiently in Aurajoki than Mustionjoki area (Fig. 5a). Compared to the relative reduction (Fig. 5a), grassed areas reduced the amount of erosion (in terms of kg ha^{-1}) clearly more efficiently in Aurajoki than Mustionjoki area (Fig. 5b), due to the higher total amount of erosion in Aurajoki than Mustionjoki (Fig. 2d).

Morphological operations with the different filter diameters further showed how the optimized areas can be spatially clustered by using the image processing method (Fig. 6). The areas were more tightly clustered when filters were used (Fig. 6). The 10 and 30 m filters increased the optimized areas by 0–1 % and 8–17 %, respectively (compared to the OC50 reference scenario). The consequent change in erosion reduction with the 10 and 30 m filters were -3…–4 % and -1…7 %, respectively. Filters with diameters 50–90 m increased the areas 7–58 % but increased erosion reduction only slightly (compared to the optimal areas) in Mustionjoki and decreased erosion reduction in Aurajoki (Fig. 5a).

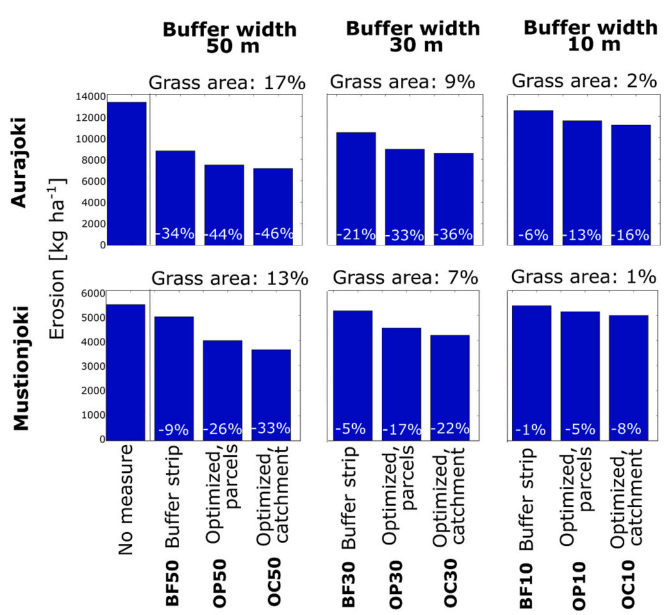

Fig. 3. Erosion within the studied areas with the different mitigation measures and three different buffer strip widths. The percentages overlapping the bars denote the computed erosion reduction with the measure. The percentage of grassed area describes how much of the total field area was covered by grass in the scenarios (with different buffer widths). The abbreviations on the x-axis refer to Table 1.

4. Discussion

4.1. Optimally sited grassed areas and implications for erosion control

Based on the results, the RUSLE-targeted grassed areas demonstrated greater erosion reductions compared to the policy-based siting of grass areas along riparian fields, as RUSLE-based targeting more efficiently considered local areas with high erosion. The policies direct the grass areas as buffer strips along water bodies, but a large share of erosion can be generated in disproportionally small land areas which are often located further from the streams (Fig. 4a-b). Moreover, much of the eroded material is transported via subsurface drains (e.g. Turunen et al., 2017; Turtola et al., 2007; Uusitalo et al., 2001; Øygarden et al., 1997) therefore by-passing grass buffer strips. This further highlights the importance of targeting grassed areas to erosion hotspots. Previous studies have also shown that hotspot erosion areas can markedly contribute to total erosion in different catchments (Remund et al., 2021; Belayneh et al., 2019; Djodjic and Villa, 2015). Our study demonstrates such occurrence within the modest Finnish topographical variations and sets the findings to the context of grass area allocation. The current study showed also how the location of the hotspots in relation to the streams varied between the sub-catchments and between the field parcels (Fig. 4a-b). Note also that typically the optimized grass area locations were scattered throughout the studied areas (Fig. 4) and did not concentrate for example to a few individual field parcels. Therefore, particularly location-specific erosion estimates (e.g. Räsänen et al., 2023) can be important in terms of efficient mitigation of erosion with targeted grass areas.

Increasing grassed area (with optimal locations) up to ~20 % of the total area induced large relative erosion reductions compared to further increases (Fig. 5). These results form a quantification on how much land area could be reasonable to allocate for grasses. Previously also Sharpley et al. (2009) noted that the majority of phosphorus load from a catchment can originate from 20 % of the catchment area. In our study, the erosion reductions were much larger in Aurajoki than Mustionjoki area, which points out the importance of considering differences between

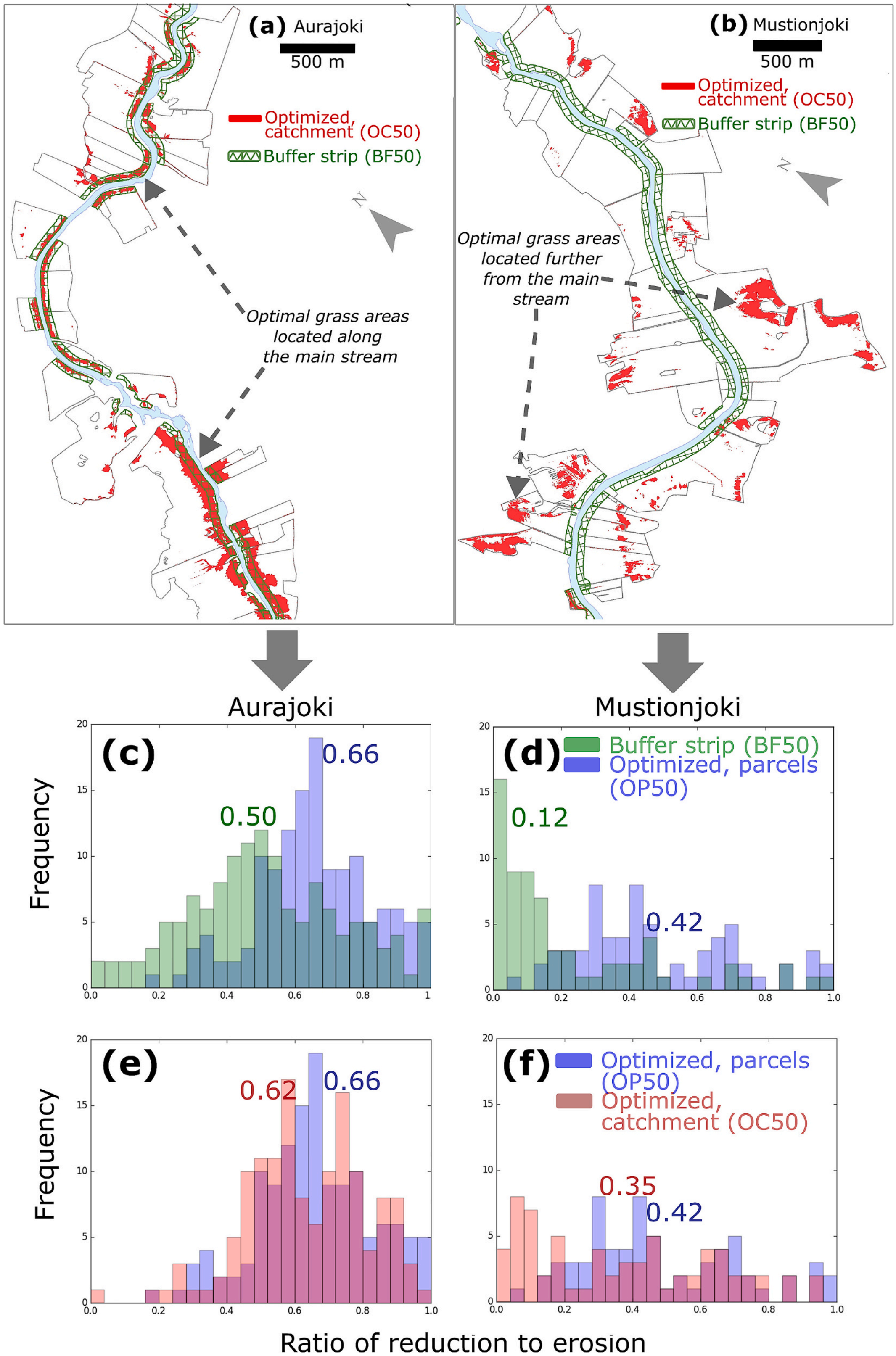

Fig. 4. (a-b) A representative snapshot of the spatial distribution of grassed areas within the Aurajoki and Mustionjoki area in the two different scenarios (optimal and policy-based areas), and (c-f) distribution of the erosion reduction within different field parcels in the areas (note that the overlapping areas of the bars are shown with the shaded colors). The blue areas in a-b show the main streams and the arrows highlight features of importance. The numbers in c-f denote the median values for each scenario. The scenario abbreviations refer to Table 1.

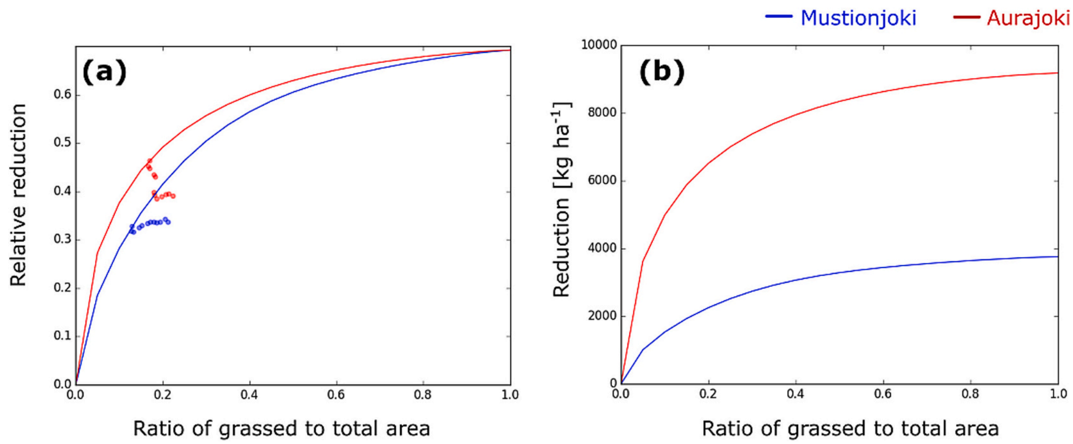

Fig. 5. (a) Ratio of reduced erosion to total erosion (with no measure) and (b) reduction of erosion as a function of grassed area in the studied area in Mustionjoki and Aurajoki sub-catchments (scenario C0–100). The dots show the reductions for the morphological analyses with the different radiuses and consequently different areas (scenario M0–90).

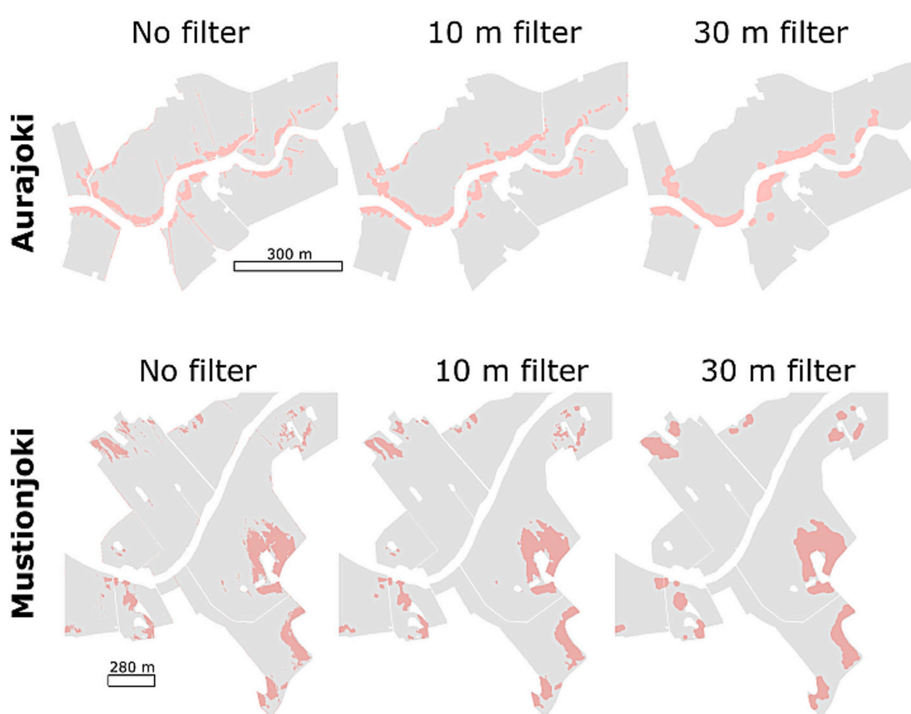

Fig. 6. Morphological operations conducted with different filter diameters (scenario M0–90).

catchments and landscapes when allocating grassed area within larger regions. Consideration of the erosion control efficiency of grassed areas in different locations has the potential to reduce the costs related to water quality improvements (e.g. Balana et al., 2012). Water quality issues and thereby ecological status of surface waters can be considered as the main concern of agricultural erosion in the Nordic context (Ulén et al., 2012).

The computed efficiency gains due to the improved targeting (4–24 pp., Fig. 3) can be weighed against other measures to reduce erosion. For example, perennial grass has been observed to have 54–73 % lower erosion rate than spring cereal cultivation with autumn moldboard ploughing (Puustinen et al., 2005). No-till in spring cereal cultivation has been found to reduce erosion by 56–62 % and reduced tillage by 16–32 %, as compared to moldboard ploughing in autumn (Honkanen et al., 2021; Puustinen et al., 2010; Kukkonen et al., 2004; Uusitalo et al., 2001). Subsurface drainage has been reported to reduce erosion by 17–55 % (Turtola et al., 2007; Istok and Kling, 1983). Remund et al. (2021) suggested that conservation tillage can reduce erosion by more than 80 %. The overall erosion reduction due to the targeted grass areas in the current study were up to 46 % (Fig. 3), which demonstrated that the grass areas may have an important role in erosion control. Note, however, that the reduction values above are not directly comparable due to methodological and site-specific differences but are roughly informative in terms of the relative erosion reduction efficiencies. Our analysis also pointed out that increasing the buffer strip width from 10 to 50 m increased erosion reduction more in Aurajoki (from 6 to 34 %, Fig. 3), than Mustionjoki (from 1 to 9 %). Therefore, decisions related to optimal buffer strip widths also need to consider local conditions.

Overall, our study provided an analysis and quantification of the potential of improved targeting of grassed areas, as compared to buffer strips along the main streams. The implications are that spatial differences in the amount of erosion from field to sub-catchment scale (for example identifying subcatchments with high erosion and hotspots within them) are worth consideration for improved and efficient erosion control with grassed areas. Spatially explicit approaches can provide considerable benefits for erosion control, but the current policies do not consider spatial differences in erosion risk in the allocation of grassed areas.

4.2. Computational aspects, limitations and development directions

Computationally, our study considered the cultivated areas to be spatially uniform in terms of C (value 0.211 for cereals and 0.065 for grass) and P (Eq. 1). This facilitated the straightforward spatial analysis focusing on erosion risk (describing spring cereal cultivation with autumn ploughing without subsurface drainage). For more accurate estimates of erosion in a particular area, the factors should be determined using local data on farming practices such as choice of crop, tillage method and drainage systems. However, farming practices are typically implemented uniformly for each field, and therefore changing farming practices would not affect the optimized parcel-specific grass areas (Fig. 3). Our computations point out the potential of improved targeting of grassed areas (Fig. 3) by considering key aspects of long-term erosion risk. For practical implementation of the findings, it is beneficial to relate the erosion risk results to the locally implemented farming practices. Generally, one of the strengths of model applications is their ability to disentangle different factors affecting erosion in a landscape or study area.

The practical implementation of the spatially targeted grass areas also needs to consider that farming operations would require reasonable shape for the grass areas. Instead of the occasionally scattered small fragments within field parcels shown in Fig. 4a-b, continuous grass areas within each field parcels would be necessary for reasonable trafficability. Our morphological analyses showed how such continuous areas can be reasonably computationally produced (Fig. 6) without major losses in the erosion reduction efficiencies, if the filter diameter is around 30 m. Larger diameters clearly reduced the gains achieved with RUSLE-based targeting (Fig. 5a). The practical applicability of the grass area shapes could be further improved at the field parcel scale by, e.g., considering the driving directions in the analysis.

While our analysis provided a first quantitative estimate on the potential of improved grass area targeting in the local conditions, note that the study focused particularly on potential in terms of erosion risk. When considering environmental loads to surface water, the theme links with several other aspects related to erosion. Overall catchment-scale erosion mitigation measure design could also consider flooding of rivers and streams in the agricultural areas. Also for example sediment connectivity within the fields and catchments (e.g. Schmaltz et al., 2024) can be an important aspect in many landscapes. On the other hand, consideration of surface runoff connectivity likely would not induce major differences to our analysis (Tähtikarhu et al., 2022). However, farming operations in the direct vicinity of surface water bodies would likely increase the risk of leaching of substances directly at the border of the field. Therefore, in Finland, 3 m wide buffer strips are required along all watercourses, especially where the slope steepness exceeds 10 %. Establishing of 30–50 m wide buffer zones or leaving of small parcels less than one hectare as buffer zones is also possible along watercourses. In certain conditions buffer strips can filter a significant amount of suspended sediment transported via overland flow (Uusi-Kämppä and Jauhiainen, 2010; Syversen and Borch, 2005; Dillaha et al., 1988), but the filtering efficiency is likely challenged by concentrated surface runoff fluxes (Ramler and Strauss, 2024; Tähtikarhu et al., 2022; Dosskey et al., 2002). While our analysis considered reduction of erosion in those locations where the grass areas are located, spatial analysis on the potentially optimal locations for sediment filtering could be beneficial when considering erosion mitigation measure design within landscapes (e.g. Ramler and Strauss, 2024).

Artificial subsurface drainpipes provide a sediment transport pathway for the eroded sediment from large field areas directly to surrounding ditches and surface waters (e.g. Turunen et al., 2017; Turtola et al., 2007; Øygarden et al., 1997), even when the erosion hotspot areas are not located in the vicinity of streams and lakes. This subsurface connectivity perspective supports the concept of siting grassed areas to areas which are not in the direct vicinity of streams and lakes. For example, Turtola et al. (2007), Uusitalo et al. (2001), Turunen et al. (2017) and Øygarden et al. (1997) showed how the majority of eroded material from the soil surface can be transported to subsurface drains via preferential flowpaths (such as cracks and biopores). While measurements at experimental fields in Finland show a 53–72 % reduction in sediment load via surface runoff by 10–15 m wide grass buffer strips (Puustinen et al., 2005; Uusi-Kämppä and Jauhiainen, 2010), the reduction can be a relatively small fraction of the total sediment load due to sediments bypassing the buffer strips and the load via subsurface drains. The transport processes in preferential flowpaths, in turn, are dependent on factors such as drainage efficiency, local hydrological conditions, soil type, and soil structure (e.g. Turunen et al., 2017; Turtola et al., 2007; Øygarden et al., 1997), which are challenging to determine locally. Together, these connectivity factors support the view, that erosion is best managed at locations where the erosion occurs, as the reduction of sediment transport can be challenging particularly at well-drained fields. For quantification of the absolute sediment loads via the different transport pathways under different erosion management scenarios and varying landscape conditions, more measurements and simulations are needed. Currently many simulation methods are, however, limited in their capacity to consider sediment connectivity at landscape scale, and particularly connectivity via subsurface drainage (Schmaltz et al., 2024).

Regarding uncertainties related to the assessment of the erosion risk, the application relies on the previous RUSLE-applications (Räsänen et al., 2023; Renard et al., 1997) and on the notions that RUSLE-based models can have capability to rank areas in terms of erosion or high erosion risk (e.g. Gashaw et al., 2021; Hamel et al., 2015, 2017; Djodjic and Villa, 2015). In Finland, the performance of RUSLE has been evaluated against measurements from eight experimental fields under different cultivation and management practices (Lilja et al., 2017b; Räsänen et al., 2023). The Nash-Sutcliffe efficiency between predictions and estimates was 0.72 (R2 = 0.76), which correspond to the typical performance of erosion models (Batista et al., 2019). The experimental fields used in the evaluation consisted of a range of different environmental conditions, including different soil types, organic matter contents (e.g. Turunen et al., 2017; Puustinen et al., 2005) slopes ranging from 0.8 to 7 % (see Table 1 in Räsänen et al., 2023 for details). It is, however, recognized that the outputs of erosion models are uncertain especially in terms of absolute values (e.g. Schmaltz et al., 2024; Batista et al., 2019; Jetten et al., 1999), but can be useful in terms of relative comparison between different areas and system response and scenario analyses,

Also applied data affects the outcome of erosion models (e.g. Schmaltz et al., 2024). Our study used erosion estimates with the resolution of 2×2 m^2. The erosion was computed based on a 2×2 m^2 DEM, while the other factors were derived from spatially more coarse data (Räsänen et al., 2023). Methodologically, this can be considered a reasonable approach since spatial variation in topography in the area is clearly greater than variations in soil type and rainfall patterns. It is however, recognized that data availability to further develop erosion models (including validation data and more comprehensive soil maps) is a common challenge which can partly limit the accuracy of erosion estimates (Schmaltz et al., 2024). Reasonable spatial data resolution of the factors is also dependent on region-specific conditions. For example, in Finland the spatial variability in R values can be considered small compared to mountainous regions and Italy (e.g. Panagos et al., 2015). Variability in various soil properties, such as soil structure, within soil types might be one considerable future challenge for improved erosion estimates. Altogether, the resolution of the factor data can have considerable influence on erosion estimates, but the appropriate resolution depends on local conditions and objectives of the performed simulations and analyses.

The applied computational approach seems promising and would further benefit of test sites for example at hillslope scale, or by empirical determination of erosion hotspots within landscapes for model validation purposes (e.g. Djodjic and Villa, 2015). Our computational approach did not include validation of the model outputs with erosion data from differently sited grassed areas, as such data is currently very rare. The long-term annual amount of erosion in Finland is relatively small (e.g. Räsänen et al., 2023; Panagos et al., 2015), and thus the spatial distribution of erosion can be partly challenging to identify empirically. Effects of efficient siting of grassed areas have been previously demonstrated in laboratory studies (Zhang et al., 2024; Pan and Ma, 2020; Zhang et al., 2018) and it would be likely challenging and costly to conduct siting experiments within experimental sites or catchments due to spatial heterogeneities and related dynamics. Furthermore, also empirical measurement campaigns have uncertainties (e.g. García-Ruiz et al., 2015) which highlights the value of combining data with models. While our study focused particularly on spatially analyzing erosion control at long-term erosion hotspot areas, considerations on erosion dynamics could also be useful in terms of exploring the potential of temporal targeting. For example, majority of erosion in Finland occurs outside the growing season in autumn and spring, when the soils have less vegetation cover and are more exposed.

Overall, conducting spatial erosion control assessment can be considered to require consideration of local aspects of erosion and sediment transport, which consequently guides requirements for a reasonable modeling approach and data, Models can function as computational frameworks, which combine different data and allow systematic assessment of different key aspects of erosion as well as knowledge gaps. Likely model applications with different degrees of complexity combined with comprehensive local data would further improve process knowledge and accuracy of erosion assessments (e.g. Schmaltz et al., 2024).

5. Conclusions

Our computational analysis demonstrated how RUSLE-based siting of grassed areas can result to greater erosion reductions compared to the policy-based placement of grass areas along riparian fields, as the computational targeting more efficiently considered areas with high erosion. A large share of erosion can be generated in disproportionally small land areas, which are often located further from the streams. The analysis showed how erosion risks could potentially be reduced (up to 24 percentage points) by improved spatial targeting of grassed areas within sub-catchments. Moreover, much of the eroded material can be transported via subsurface drains and thereby by-pass grass buffer strips, which highlights the importance of targeting grassed areas to erosion hotspots. This applies particularly to well-drained fields, where high erosion areas are well-connected to surface waters due to the subsurface drains, and thereby the reduction of sediment connectivity is challenging. The implementation of the improved spatial targeting of grassed areas would, however, require revision of current policies to support the tailoring of solutions according to local conditions, as well as the farmers willingness to implement the measures.

The analysis adds to the current knowledge of erosion management and comprehensive assessment of erosion mitigation would need to consider also other aspects of erosion and sediment transport, such as reducing erosion during flooding of the rivers. Furthermore, empirical validation of the approach could further improve the reliability of the RUSLE-based assessment. However, the study points out the potential value of models to improve landscape scale spatially targeted erosion management. Altogether the findings suggest that erosion management can be further improved through better consideration of local conditions.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. A visualisation of the RUSLE factors

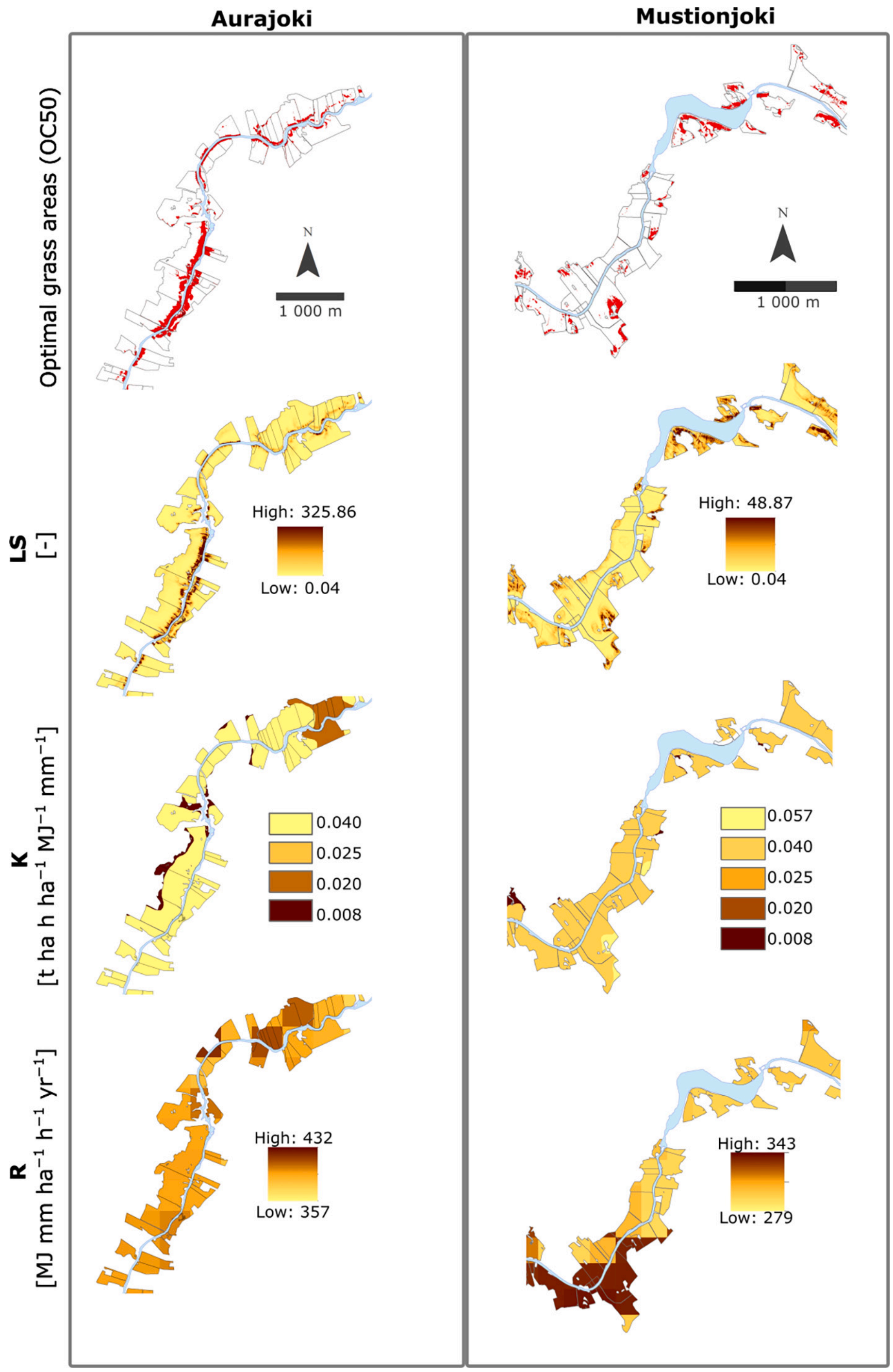

Fig. A.1. A visualization of the spatial distribution of the RUSLE factors within the studied areas

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