Soil Erosion Risk Assessment in Europe

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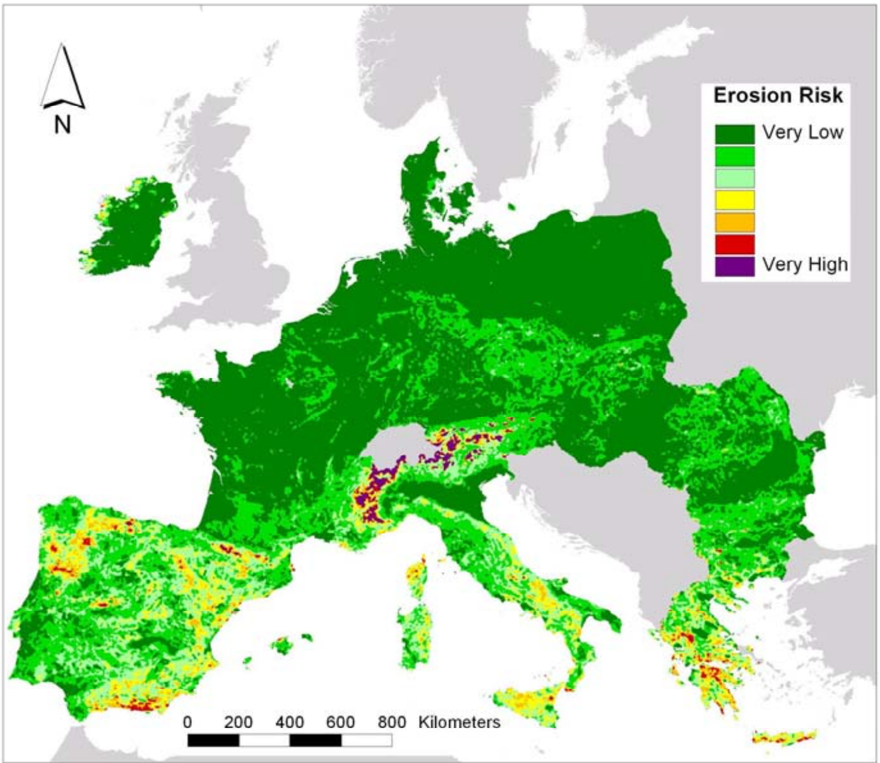

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Contents

1. Introduction

Soil erosion by water is a widespread problem throughout Europe. A report for the Council of Europe, using revised GLASOD data (Oldeman *et al.*, 1991; Van Lynden, 1995), provides an overview of the extent of soil degradation in Europe. Some of the findings are shown in the Table 1.1, but these figures shown are only a rough approximation of the area affected by soil degradation.

Table 1.1. Human-induced Soil Degradation in Europe[1] (M ha)

WATER EROSION	**Light**	**Moderate**	**Strong**	**Extreme**	**Total**
Loss of Topsoil	18.9	64.7	9.2	-	92.8
Terrain Deformation	2.5	16.3	0.6	2.4	21.8
Total:	21.4	81.0	9.8	2.4	114.5 (52.3%)

[1] Includes the European part of the former Soviet Union.

However, Table 1.1 indicates the importance of water erosion in Europe in terms of area affected. The most dominant effect is the loss of topsoil, which is often not conspicuous but nevertheless potentially very damaging. Physical factors like climate, topography and soil characteristics are important in the process of soil erosion. In part, this explains the difference between the severe water erosion problem in Iceland but the much less severe erosion in Scandinavia where the climate is less harsh and the soils are less erodible (Fournier, 1972).

The Mediterranean region is particularly prone to erosion. This is because it is subject to long dry periods followed by heavy bursts of erosive rainfall, falling on steep slopes with fragile soils, resulting in considerable amounts of erosion. This contrasts with NW Europe where soil erosion is slight because rain falling on mainly gentle slopes is evenly distributed throughout the year. Consequently, the area affected by erosion in northern Europe is much more restricted in its extent than in southern Europe.

In parts of the Mediterranean region, erosion has reached a stage of irreversibility and in some places erosion has practically ceased because there is no more soil left. With a very slow rate of soil formation, any soil loss of more than 1 t/ha/yr can be considered as irreversible within a time span of 50-100 years. Losses of 20 to 40 t/ha in individual storms, that may happen once every two or three years, are measured regularly in Europe with losses of more than 100 t/ha in extreme events (Morgan, 1992). It may take some time before the effects of such erosion become noticeable, especially in areas with the deepest and most fertile soils or on heavily fertilised land. However, this is all the more dangerous because, once the effects have become obvious, it is usually be too late to do anything about it.

The main causes of soil erosion are still inappropriate agricultural practices, deforestation, overgrazing and construction activities (Yassoglou *et al.*, 1998). Increasing the awareness amongst scientists and policy makers about the soil degradation problem in Europe is now an urgent requirement. The identification of areas that are vulnerable to soil erosion can be helpful for improving our knowledge about the extent of the areas affected and, ultimately, for developing measures to keep the problem under control whenever possible.

1.1 Europe Project

In an attempt to quantify erosion in Europe using modern digital techniques, a project was initiated that aims to assess erosion risk at continental level. The end product is a set of maps that can be used as an aid to identifying regions that are prone to erosion. This study addresses rill- and inter-rill erosion only. Other forms of erosion are also important, for example gully erosion, landslides and, to a lesser extent, wind erosion. However, these types of erosion are not addressed in this study, but they should be addressed some time in the future.

This report gives a detailed technical description of the project. An overview of the various methods that can be used to assess soil erosion risk at the regional scale and larger is given in Chapter 2. Chapter 3 gives a general description of the methods used in this study. A more detailed description can be found in Chapters 4-7, along with an explanation of the data sources and processing. A discussion of the results can be found in Chapter 8.

2. Assessing soil erosion risk

For assessing soil erosion risk, various approaches can be adopted. A distinction can be made here between *expert*-based and *model*-based approaches.

2.1 Expert based methods

An example of an expert-based approach is the soil erosion risk map of Western Europe by De Ploey (1989). The map was produced by various experts who delineated areas where, according to their judgement, erosion processes are important. A limitation of this approach is that the author does not give a clear-cut definition of the criteria according to which areas were delineated (Yassoglou *et al.*, 1998).

Factorial scoring is another approach that can be used to assess erosion risk (Morgan, 1995). An example is the CORINE soil erosion risk assessment of the Mediterranean region (CORINE, 1992). The analysis is based on factorial scores for soil erodibility (4 classes), erosivity (3 classes) and slope angle (4 classes). The scores are multiplied, giving a combined score that represents potential erosion risk. To assess actual soil erosion risk, the potential erosion risk map is combined with a land cover factor (2 classes).

Montier *et al.* (1998) developed an expert-based method for the whole of France. As with CORINE, the method is based on scores that are assigned to factors related to land cover (9 classes), the soil's susceptibility to surface crusting (4 classes), slope angle (8 classes) and erodibility (3 classes). An interesting feature of their method is that it takes into account the different types of erosion that occur on cultivated areas, vineyards, mountainous areas and the Mediterranean. This way, the interaction between soil, vegetation, slope and climate is accounted for to some extent.

A problem with most methods based on scoring is that the results are affected by the way the scores are defined. In addition to this, classifying the source data in e.g. slope classes results in information loss, and the results of the analyses may depend strongly on the class limits and the number of classes used. Moreover, unless some kind of weighting is used each factor is given equal weight, which is not realistic. If one decides to use some weighting, choosing realistic values for the weights may be difficult. The way in which the various factors are combined into classes that are functional with respect to erosion risk (addition, multiplication) may pose problems also (Morgan, 1995). Finally, as factorial scoring produces qualitative erosion classes, the interpretation of these classes can be difficult.

2.2 Model based methods

A wide variety of models are available for assessing soil erosion risk. Erosion models can be classified in a number of ways. One may make a subdivision based on the time scale for which a model can be used: some models are designed to predict long-term annual soil losses, while others predict single storm losses (event-based). Alternatively, a distinction can be made between lumped models that predict erosion at a single point, and spatially distributed models. Another useful division is the one between empirical and physically-based models. The choice for a particular model largely depends on the purpose for which it is intended and the available data, time and money.

Jäger (1994) used the empirical Universal Soil Loss Equation (USLE) to assess soil erosion risk in Baden-Württemberg (Germany). De Jong (1994) used the Morgan, Morgan and Finney model (Morgan *et al.*, 1984) as a basis for his SEMMED model. Input variables are derived from standard meteorological data, soil maps, multi-temporal satellite imagery, digital elevation models and a limited amount of field data. This way, erosion risk can be assessed over large, spatially diverse areas without the need for extensive field surveys. So far the SEMMED model has been used to produce regional erosion risk maps of parts of the Ardêche region and the Peyne catchment in Southern France (De Jong, 1994, De Jong *et al.*, 1998).

Kirkby and King (1998) assessed soil erosion risk for the whole of France using a model-based approach. Their model provides a simplified representation of erosion in an individual storm. The model contains terms for soil erodibility, topography and climate. All storm rainfall above a critical threshold (whose value depends on soil properties and land cover) is assumed to contribute to runoff, and erosion is assumed to be proportional to runoff. Monthly and annual erosion estimates are obtained by integrating over the frequency distribution of rainstorms.

Several problems arise when applying quantitative models at regional or larger scale. First, most erosion models were developed on a plot or field scale, which means that they are designed to provide point estimates of soil loss. When these models are applied over large areas the model output has to be interpreted carefully. One cannot expect that a model that was designed to predict soil loss on a single agricultural field produces accurate erosion estimates when applied to the regional scale on a grid of say 50 meter pixels or coarser. One should also be aware of which processes are actually being modeled. For example, the well-known Universal Soil Loss Equation was developed to predict rill- and inter-rill erosion only. Therefore, one cannot expect this model to perform well in areas where gully erosion is the dominant erosion type, let alone mass-movements like landslides and rockfalls.

Also, at the regional scale it is usually impossible to determine the model's input data (like soil and vegetation parameters) directly in the field. Usually, the model parameters are approximated by assigning values to mapping units on a soil or vegetation map, or through regression equations between e.g. vegetation cover and some satellite-derived spectral index. In general however, this will yield parameter values that are far less accurate than the results of a field survey. Because of all this, the relative soil loss values produced by models at this scale are generally more reliable than the absolute values. This is not necessarily a problem, as long as one is aware that the model results give a broad overview of the general pattern of the relative differences, rather than providing accurate absolute erosion rates. Because of this, the availability of input data is probably the most important consideration when selecting an erosion model at the regional/national scale. It would not make sense to use a sophisticated model if sufficient input data are not available. In the latter case, the only way to run the model would be to assume certain variables and model parameters to be constant. However, the results would probably be less reliable than the results that would have been obtained with a simpler model that requires less input data (De Roo, 1993). Also, uncertainties in the model's input propagate throughout the model, so one should be careful not to use an 'over-parameterised' model when the quality of the input data is poor.

Perhaps the biggest problem with erosion modelling is the difficulty of validating the estimates produced. At the regional and larger scale, virtually no reliable data exist for comparing estimates with actual soil losses. King *et al.* (1999) attempted to validate an erosion risk assessment for France by correlating soil loss with the occurrence of mudflows. However, other processes are involved here and such comparisons do not substitute for 'real' measurements.

3. Universal Soil Loss Equation

3.1 Introduction

For this study a model-based approach was used to assess soil erosion risk. As explained in Chapter 2, the availability of input data is a critical selection criterion when assessing soil erosion risk at the regional, national or continental scale. Even though a wide variety of models are available for assessing soil erosion risk, most of them simply require so much input data that applying them at these scales becomes problematic. The well-known Universal Soil Loss Equation (USLE) (Wischmeier & Smith, 1978) was used because it is one of the least data demanding erosion models that has been developed and it has been applied widely at different scales. The USLE is a simple empirical model, based on regression analyses of soil loss rates on erosion plots in the USA. The model is designed to estimate long-term annual erosion rates on agricultural fields. Although the equation has many shortcomings and limitations, it is widely used because of its relative simplicity and robustness (Desmet & Govers, 1996). It also represents a standardised approach.

Soil erosion is estimated using the following empirical equation:

$$A = R \cdot K \cdot L \cdot S \cdot C \tag{3.1}$$

Where:

A : Mean (annual) soil loss
R : Rainfall erosivity factor
K : Soil erodibility factor
L : Slope factor
S : Slope length factor
C : Cover management factor

The data sources that were used to estimate the various USLE factors are summarised in Figure 3.1. The procedures used to estimate the factors are explained in detail in the following chapters. The next paragraphs give a brief overview.

3.2 Rainfall erosivity factor

The USLE rainfall erosivity factor (*R*) for any given period is obtained by summing – for each rainstorm- the product of total storm energy (E) and the maximum 30-minute intensity (I_{30}). Unfortunately, these figures are rarely available at standard meteorological stations. Moreover, the workload involved would be rather high for any national or continental assessment. Fortunately, long-term average R-values are often correlated with more readily available rainfall figures like annual rainfall or the modified Fournier's index (Arnoldus, 1978). A similar approach was used to estimate *R* for the whole of Europe (Chapter 4).

3.3 Soil erodibility factor

The *K* factor is defined as the rate of soil loss per unit of *R* as measured on a unit plot ('Wischmeier plot'). It accounts for the influence of soil properties on soil loss during storm events (Renard *et al.*, 1997). The estimation of this factor is discussed in Chapter 5.

3.4 Slope- and slope length factors

The slope- and slope length factors (*S* and *L*, respectively) account for the effect of topography on soil erosion. It can be estimated from a digital elevation model (DEM), which is described in Chapter 6.

3.5 Cover management factor

The C-factor is defined as the ratio of soil loss from land with a specific vegetation to the corresponding soil loss from continuous fallow (Wischmeier & Smith, 1978). Its value depends on vegetation cover and management practices. For this study, *C* was estimated using a combination of satellite imagery and a land cover database, which is explained in Chapter 7.

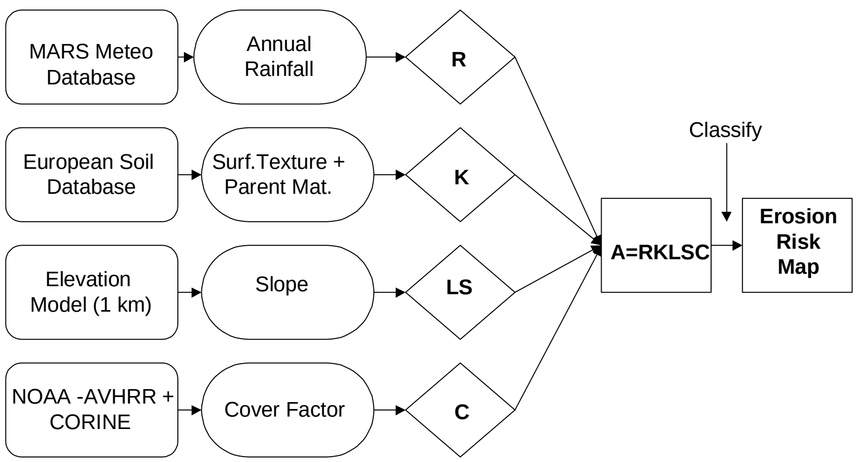

Figure 3.1 *Flowchart for creating a USLE-based erosion risk map.*

4. Rainfall Erosivity Factor

4.1 Introduction

As stated in Chapter 3, detailed information on both rainfall and rainfall intensity are needed for a direct estimation of the *R*-factor. As these data are usually unavailable for standard meteorological stations, a simplified approach to estimate *R* had to be used. A common solution is to use correlations between the *R*-factor and readily available meteorological parameters.

For Bavaria (Germany), Rogler & Schwertmann (1981) established the following regression equation:

$$R = 10 \cdot (-1.48 + 1.48 \cdot Ns) \qquad (4.1)$$

Where :

R : Mean annual erosivity ($MJ.mm.ha^{-1}h^{-1}y^{-1}$)
Ns : Mean rainfall amount in summer (May-October) (mm)

Zanchi (personal communication) found that for Tuscany (Italy) *R* is related to annual rainfall, so that *R* can be approximated by:

$$R = a \cdot P_j \qquad (4.2)$$

Where:

P_j : Annual rainfall (mm)

a ranges from 1.1 to 1.5, and a value of 1.3 was used for this study. The 'Tuscan equation' is based on rainfall data from 25 locations, with P_j ranging from 600 to 1200 mm.

As a first approximation, R-factor values for the whole of Europe were estimated using these two regression formulae. In the absence of any additional data it was assumed that the 'Bavarian' equation is more or less representative of northern-European conditions, and the 'Tuscan' equation of southern-European conditions, and that the transition between north and south is fuzzy. Although it is realised that this is a gross over-simplification of the actual situation, the approach may be 'fit for purpose' as the current project aims at giving a broad overview of regional patterns of erosion risk, rather than making a detailed quantitative assessment.

4.2 Processing of meteorological data

Long-term monthly and annual rainfall totals were computed using daily rainfall values that are stored in the MARS meteorological database (Rijks *et al.*, 1998). The locations of the meteorological stations are shown in Figure 4.1. For each station, mean annual rainfall was estimated for the period between the start of 1989 and the end of 1998. The monthly and annual station data were interpolated using an inverse distance interpolation.

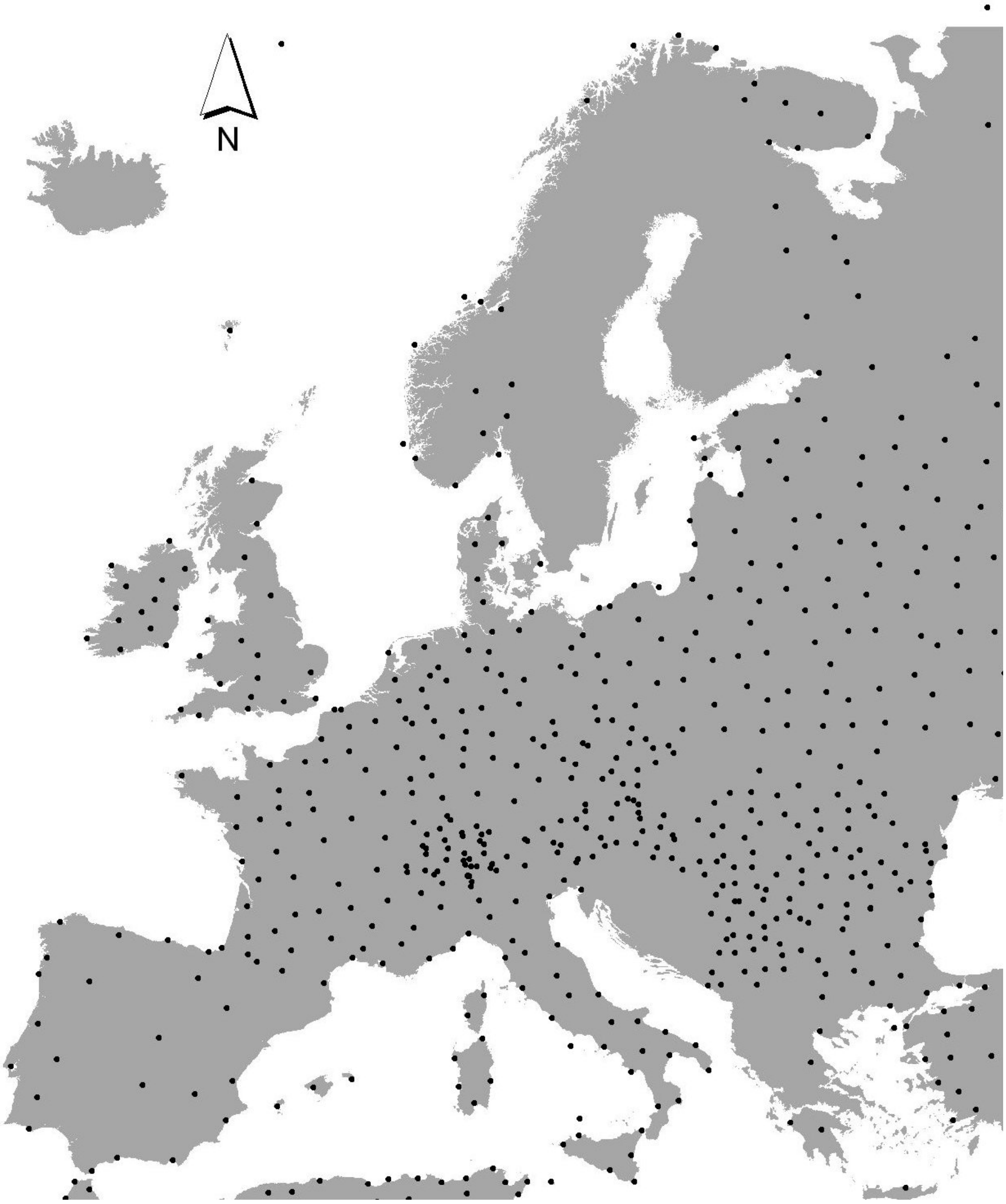

Figure 4.1 *Meteorological stations used*

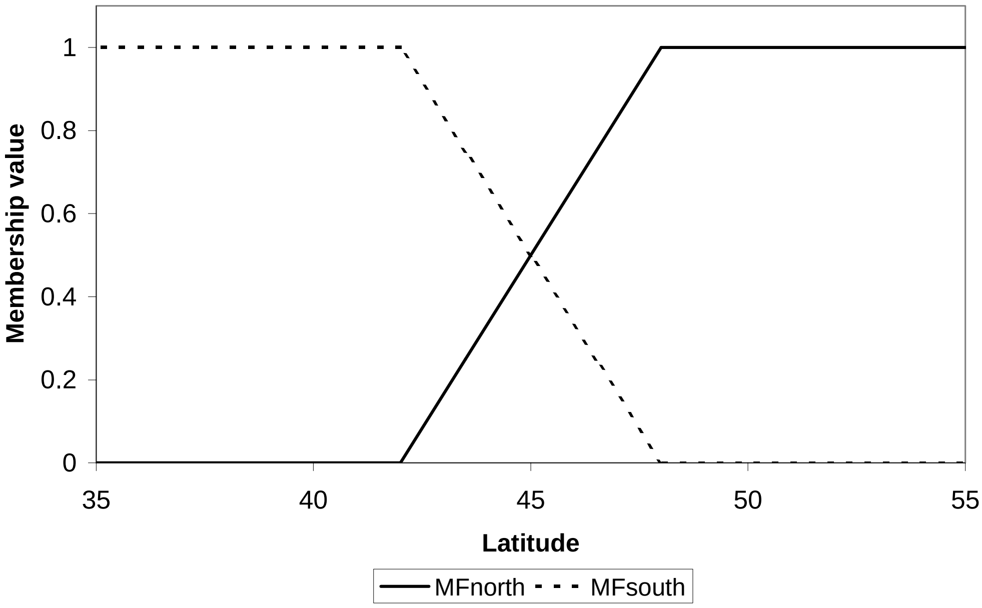

Figure 4.2 *Fuzzy rainfall erosivity factor.*

4.3 Rainfall erosivity map

Annual maps of rainfall erosivity were created by applying Equation 4.1 to total summer rain and Equation 4.2 to total annual rain. This results in two maps that are assumed to be more or less representative of northern and southern Europe respectively. After careful consideration it was decided to use the 'northern' map for latitudes greater than 48 degrees and the 'southern' map for areas below 42 degrees. A continuous, fuzzy transition between north and south was assumed for the area in between. This is achieved by defining fuzzy membership functions for 'northern-ness' and 'southern-ness' that are linear functions of latitude. This is shown in Figure 4.2. Looking at the membership function for 'southern-ness' (MF_{south}), it can be seen that it yields membership values that are equal to unity for latitudes below 42 degrees, then decreases linearly to zero between 42 and 48 degrees and remains zero for latitudes above 48 degrees. A fuzzy erosivity value can then be estimated using:

$$R_{fuzz} = MF_{north} \cdot R_{north} + MF_{south} \cdot R_{south}$$

which yields a smooth transition between 'North' and 'South'.

Figure 4.3 shows the resulting rainfall erosivity map.

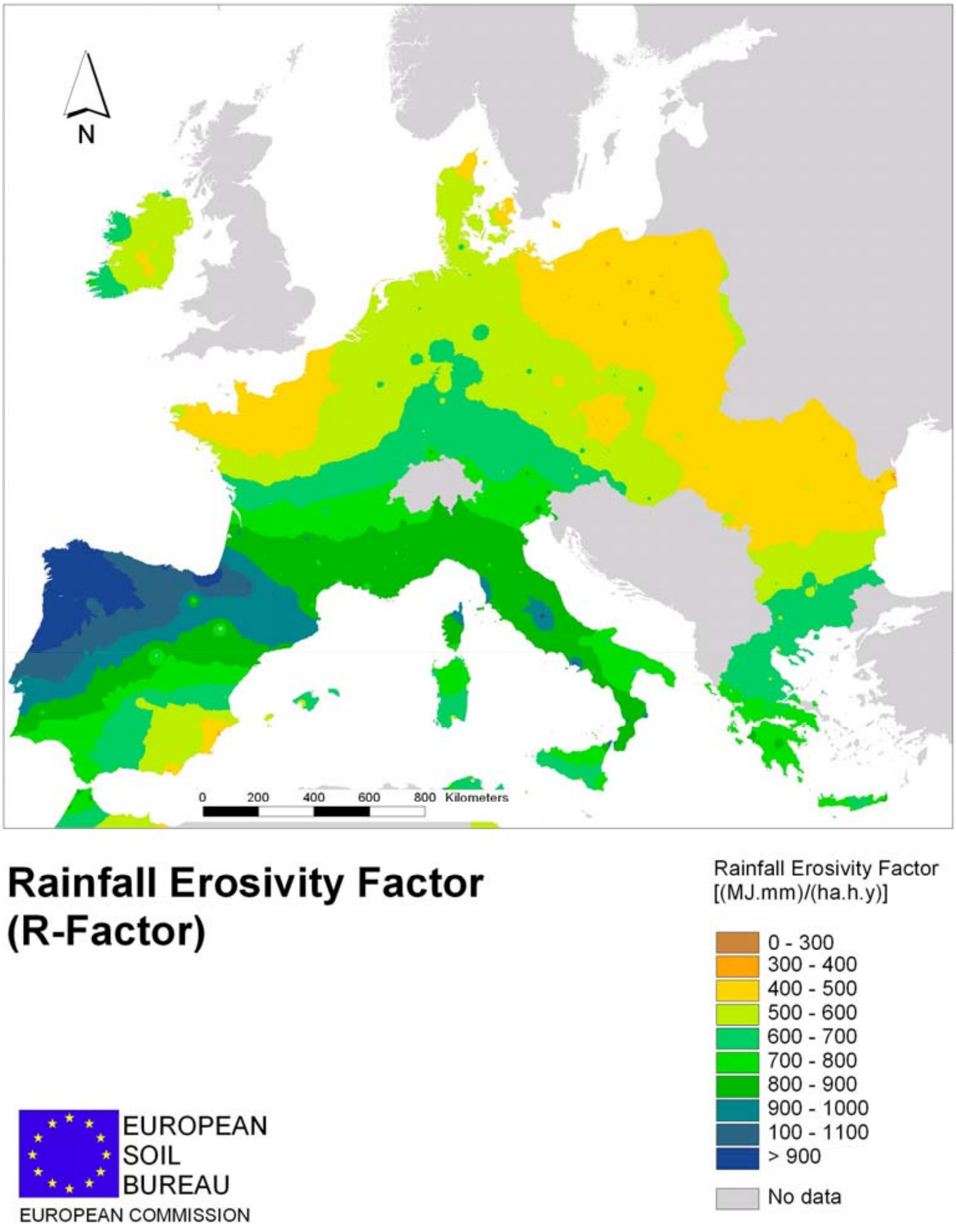

Figure 4.3 *Rainfall erosivity factor (R-factor) (MJ mm ha^{-1} h^{-1}y^{-1}).*

5. Soil erodibility factor

5.1 Introduction

The soil erodibility factor (K) is usually estimated using the nomographs and formulae that are published in for example Wischmeier & Smith (1978). While these equations are suitable for large parts of the USA (for which the USLE was originally developed), they produce unreliable results when applied to soils with textural extremes as well as well-aggregated soils (Römkens et. al., 1986). Therefore, they are not ideally suited for use under European conditions.

Römkens et. al. (1986) performed a regression analysis on a world-wide dataset of all measured K-values, which yielded the following equation (revised in Renard *et al.*, 1997):

$$K = 0.0034 + 0.0405 \cdot \exp[-0.5(\frac{\log D_g + 1.659}{0.7101})^2] \qquad (5.1)$$

Where:

K : Soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1})
D_g : Geometric mean weight diameter of the primary soil particles (mm)

D_g is a function of surface texture, and its value can be calculated using:

$$D_g = \exp(\sum f_i \cdot \ln(\frac{d_i + d_{i-1}}{2})) \qquad (5.2)$$

For each particle size class (clay, silt, sand), d_I is the maximum diameter (mm), d_{I-1} is the minimum diameter and f_I is the corresponding mass fraction.

For some soils erodibility is determined largely by soil properties other than texture. This is especially true for volcanic soils. The physical and chemical properties of these soils makes them extremely vulnerable to soil erosion, and the associated erodibility values are beyond the range that is predicted by Equation 5.1 (Torri, personal communication). After careful consideration it was decided to assign an erodibility value of 0.08 t ha h ha^{-1} MJ^{-1} mm^{-1} to all volcanic soils.

5.2 Soil data and processing

Information on soil surface texture was derived from the 1:1,000,000 Soil Geographical Database of Europe (ESGDB, see Heineke et al (1998) for more details). Texture information in the database is stored at the so-called ‘soil typological unit’ (STU) level. Each soil *mapping* unit (SMU) is made up of one or more STUs. This is shown in Figure 5.1.

For each texture class, ‘representative’ percentages of clay, silt and sand were estimated based on the class descriptions. The positions of these ‘representative’ values in the texture triangle are shown in Figure 5.2. These values were then used to estimate D_g for each texture class. Finally, the erodibility factor (*K*) was estimated using Equation (5.1). Table 5.1 gives an overview of the texture parameters and estimated K-values for each texture class.

For each Soil Mapping Unit (SMU), a K-value was estimated for all its underlying STUs. Then a weighted average was computed, where the weights are proportional to the area of each STU within a SMU. Volcanic soils were treated in a different way: first, all volcanic soils were identified using the parent material code in the soil database. Then, a fixed K-value of 0.08 was assigned to all volcanic soils, irrespective of surface texture. The resulting erodibility map is shown in Figure 5.3.

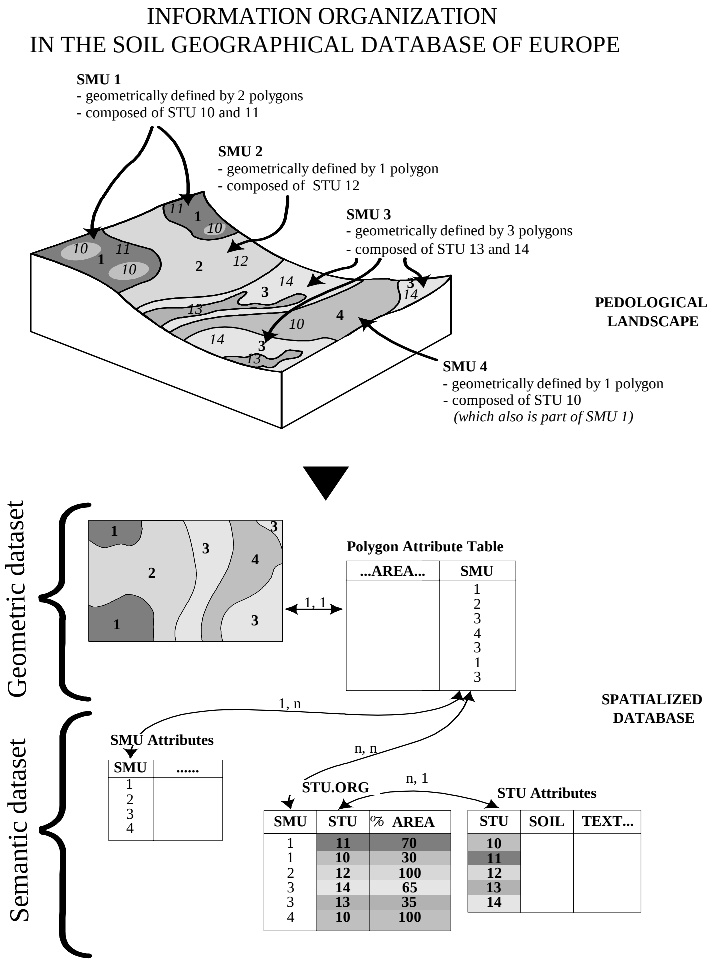

Figure 5.1 *Information organisation in the Soil Geographical Database of Europe.*

Table 5.1 *'Representative' texture parameters for each texture class*

TEXT	Dominant surface textural class.				
	(Present in: STU)	**% clay**	**% silt**	**% sand**	**K**
0	No information	-	-	-	
9	No texture (histosols, ...)	-	-	-	
1	Coarse (clay < 18 % and sand > 65 %)	9	8	83	0.0115
2	Medium (18% < clay < 35% and sand > 15%, or clay < 18% and 15% < sand < 65%)	27	15	58	0.0311
3	Medium fine (clay < 35 % and sand < 15 %)	18	74	8	0.0438
4	Fine (35 % < clay < 60 %)	48	48	4	0.0339
5	Very fine (clay > 60 %)	80	20	0	0.0170

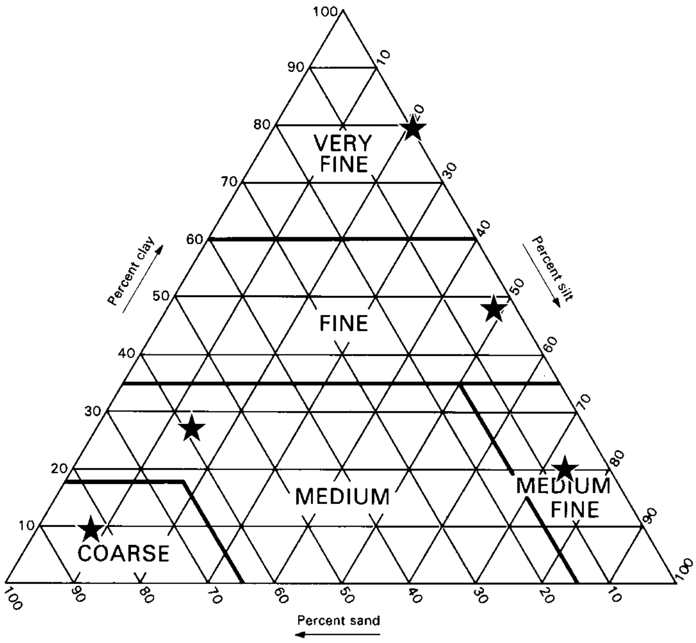

Figure 5.2 *Position of 'representative' texture parameters within the texture triangle*

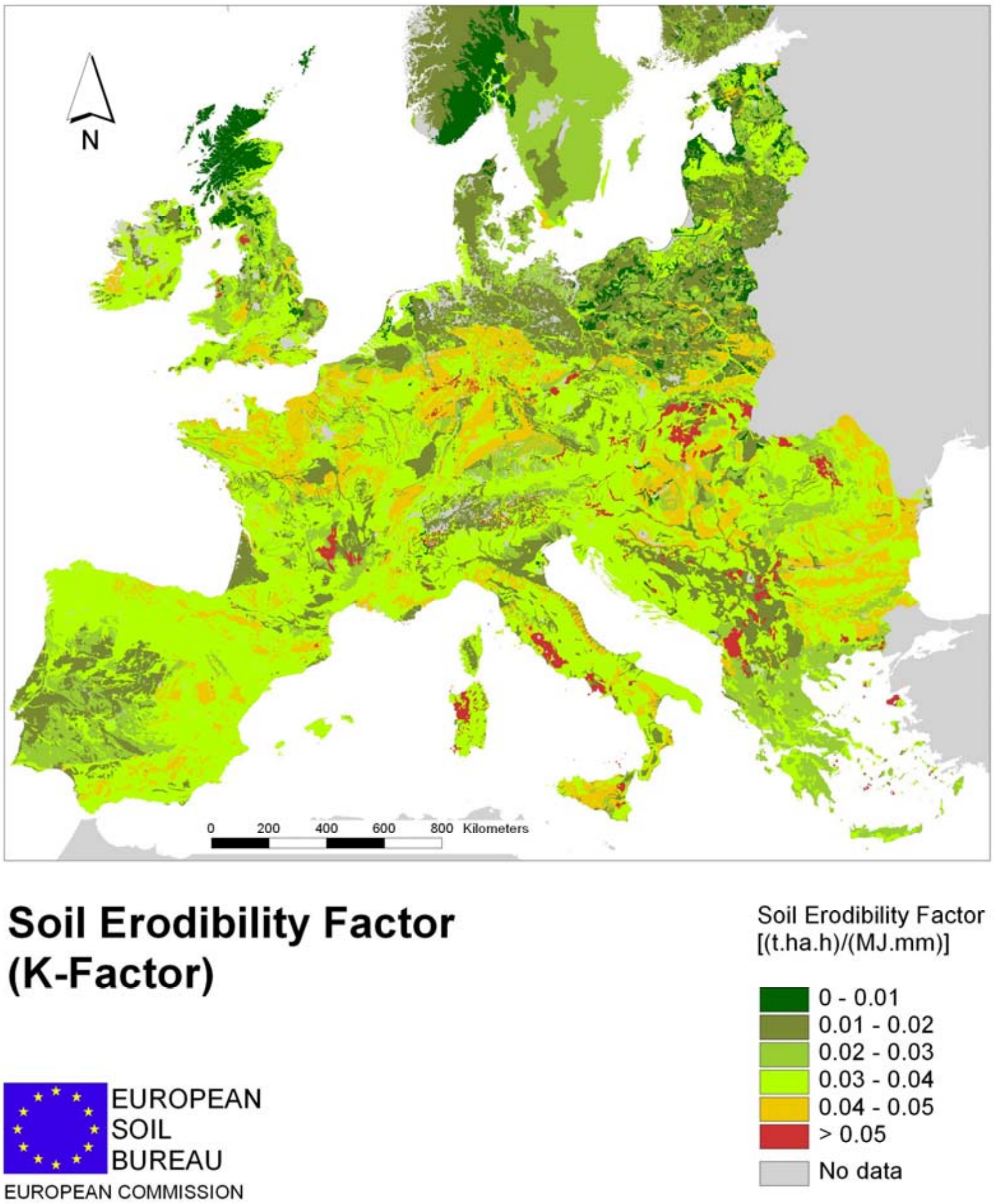

Figure 5.3 *Soil erodibility map (K-factor) (t ha h ha^{-1} MJ^{-1} mm^{-1}).*

6. Cover management factor

6.1 Introduction

Vegetation cover is – after topography – the second most important factor that controls soil erosion risk. In the (Revised) Universal Soil Loss Equation, the effect of vegetation cover is incorporated in the cover management factor (hereafter called C-factor). It is defined as the ratio of soil loss from land cropped under specific conditions to the corresponding loss from clean-tilled, continuous fallow (Wischmeier & Smith, 1978). The value of C mainly depends on the vegetation's cover percentage and growth stage. The effect of mulch cover, crop residues and tillage operations should also be accounted for in the C-factor. In the Revised Universal Soil Loss Equation (Renard et al, 1997) the C-factor is subdivided into 5 separate sub-factors that account for the effects of prior land use, canopy cover, surface cover, surface roughness and soil moisture respectively.

Up to the regional scale it would be fairly easy to assign monthly or annual C-values to classes in the CORINE land cover database by means of a lookup-table. However, at the scale used here this approach would be problematic, as Europe encompasses a wide variety of climatic conditions, which results in large spatial and temporal variations in growing season and crop vigour. This would be extremely difficult to incorporate using a table-based approach.

Instead, NOAA AVHRR imagery was used to obtain approximate C-factor values.

6.2 NOAA AVHRR

AVHRR is an acronym for 'Advanced Very High Resolution Radiometer'. It is a four (AVHRR/1) or five (AVHRR/2) channel radiometer with channels in the visible, near infrared, middle infrared and far infrared parts of the electromagnetic spectrum (http://www.belspo.be/telsat/avhrr/avts_001.htm). It has a ground resolution of approximately 1.1 km. The satellite orbits the earth 14 times each day, resulting in a daily global (pole-to pole) coverage. An automatic algorithm is used to geometrically correct the images, resulting in a geometric accuracy of about 4 km.

6.3 Normalised-Difference Vegetation Index

The most widely used remote-sensing derived indicator of vegetation growth is the Normalised Difference Vegetation Index (NDVI):

$$NDVI = \frac{(AVHRR2 - AVHRR1)}{(AVHRR2 + AVHRR1)} \qquad (6.1)$$

Where:

AVHRR1 : Reflectance value Channel 1 (visible)

AVHRR2 : Reflectance value Channel 2 (near infrared)

Its value varies between –1 and 1, where low values can be found at water bodies, bare soil and built-up areas. NDVI is positively correlated with the amount of green biomass, so it can be used to give an indication for differences in green vegetation coverage.

NDVI-values were scaled to approximate C-values using the following provisional formula:

$$C = \exp\left[-\alpha \cdot \frac{NDVI}{(\beta - NDVI)}\right] \qquad (6.2)$$

Where:

α, β : Parameters that determine the shape of the NDVI-C curve

An α-value of 2 and a β-value of 1 seem to give reasonable results (Van der Knijff *et al*., 1999). Figure 6.1 shows the corresponding (hypothetical) relationship between NDVI and C.

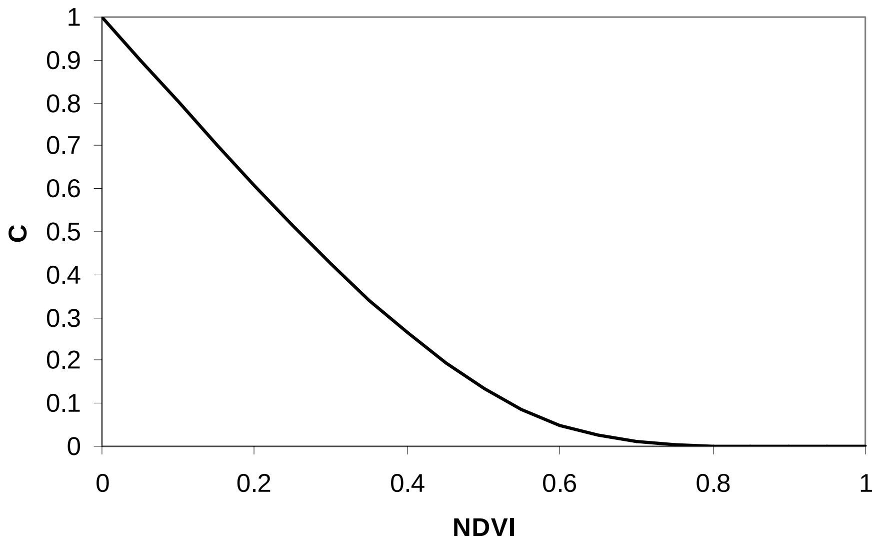

Figure 6.1 *Relationship between NDVI and USLE-C using exponential scaling formula*

Van der Knijff *et al*. (1999) aggregated 10-daily NDVI composites into average monthly images, and used these to make monthly estimates of *C* for the whole of Italy. Combined with monthly erosivity maps this allows the interaction between rainfall and vegetation growth to be taken into acount. However, for the whole of Europe this approach appeared problematic because of the large proportion of cloud pixels in the winter images. This is especially a problem for northern Europe. Therefore, it was decided to compute an annual average NDVI-value using all cloud-free pixels, and to estimate average C-values from this image and Equation 6.2.

Earlier work on Italy showed that estimating the C-factor from NDVI-images can result in unrealistically high C-values for especially woodland and grassland (van der Knijff *et al*., 1999). Therefore, the CORINE landcover database was used to assign maximum C-values for these classes. The maximum values used are 0.01 for woodland and pasture and 0.05 for natural grassland.

The resulting USLE-C map is shown in Figure 6.2.

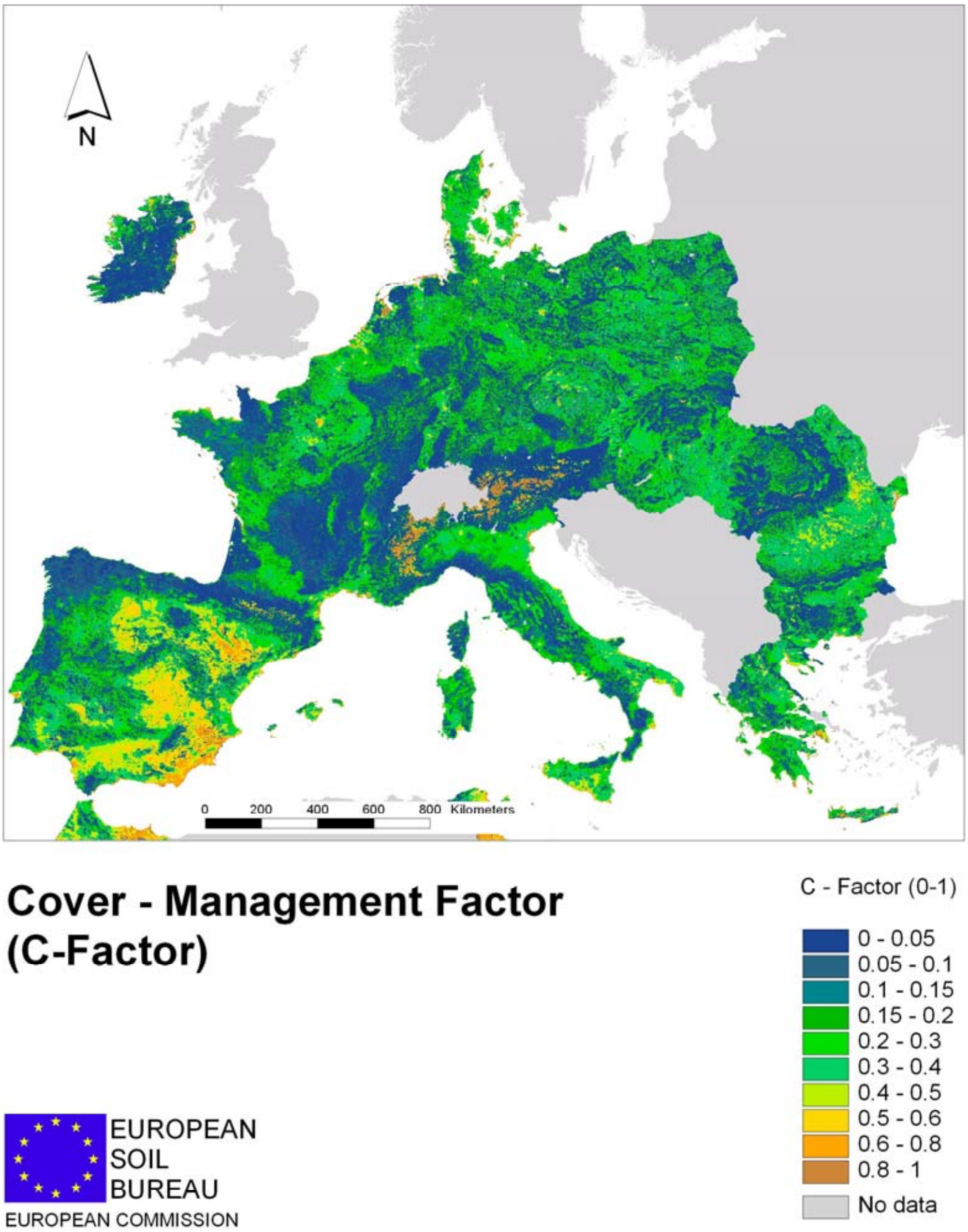

Figure 6.2 *Cover management factor map (C-factor) (dimensionless).*

7. Slope- and slope length factors

The slope- and slope length factors were estimated using the equations of Moore *et al.* (1993):

$$L = 1.4\left(\frac{A_s}{22.13}\right)^{0.4} \quad (7.1)$$

$$S = \left(\frac{\sin\beta}{0.0896}\right)^{1.3} \quad (7.1)$$

Where:

A_s : Specific contributing area (m^2/m)
β : Slope angle (degrees)

Slope was estimated using a 1-km resolution digital elevation model (DEM) of Europe. A_s was set to a constant value of 50 metres, as the 1-km DEM is simply too coarse for assessing this variable. It must be said though that the value used is completely arbitrary, and it is by no means representative or 'typical'.

Figure 7.1 shows the resulting LS map.

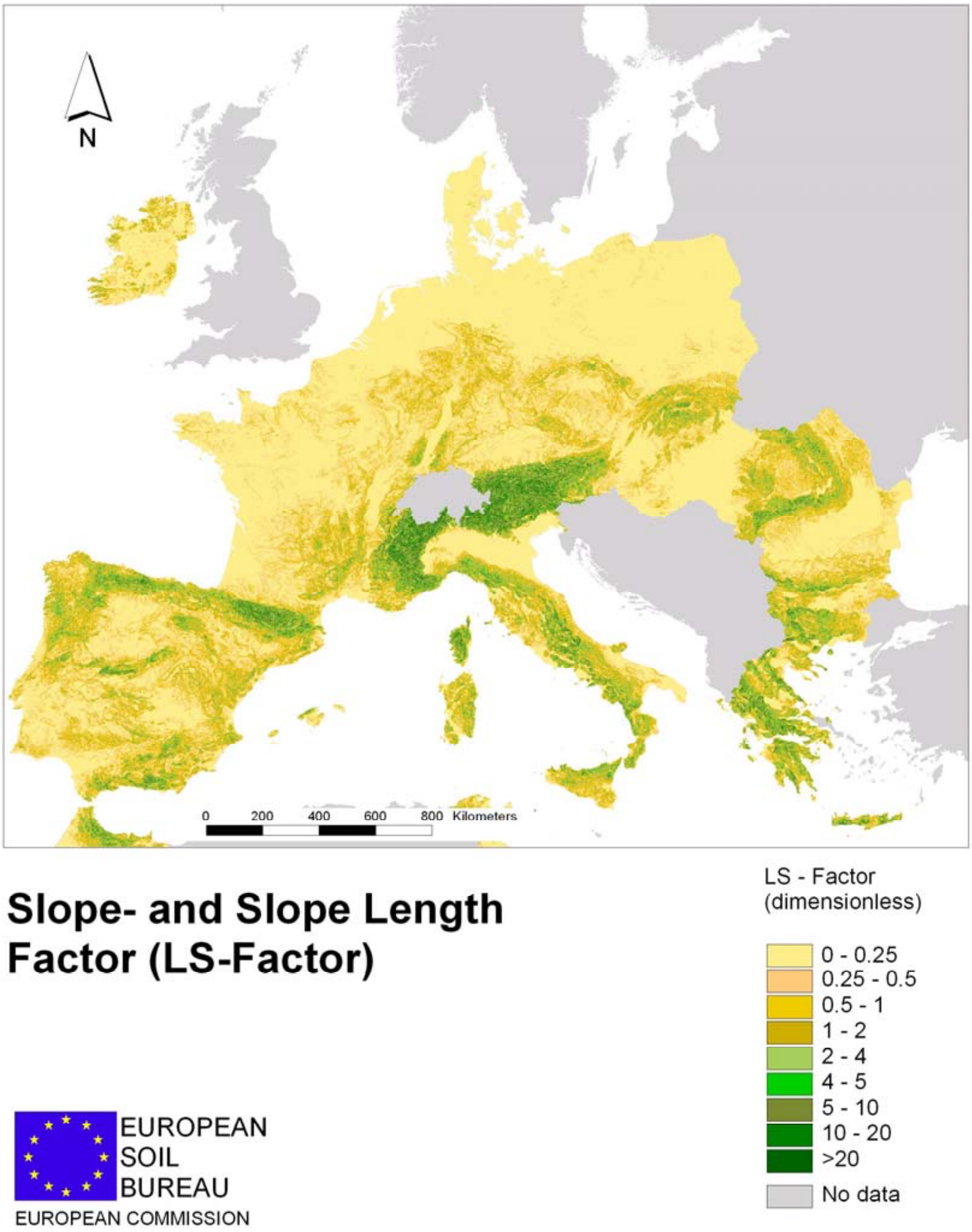

Figure 7.1 *Slope / Slope length factor map (LS) (dimensionless).*

8. Results and discussion

8.1 Results

The map of estimated annual soil erosion risk is shown in Figure 8.1. To improve its visual appearance, the map was smoothed using a median filter that replaces the actual pixel values by the median of all pixel values within a 5 pixel (5 km) search radius. *Potential* erosion risk was assessed by running the USLE on the assumption that there is a total absence of soil cover (i.e. $C = 1$). The resulting potential erosion risk map is shown in Figure 8.2. As can be seen erosion risk is expressed in qualitative terms rather than in actual rates of soil loss. The reason for this is that a quantitative assessment is not really appropriate with the available data, even though the model produces quantitative output.

8.2 Discussion

A proper validation of the results is hardly possible at the scale used. Nevertheless, it is possible to make some comments on the general pattern shown on the maps.

The maps' most apparent feature is the contrast between North and South. In general, soil erosion risk seems to be underestimated for most of northern Europe. This is mainly caused by the rainfall erosivity factor, whose predicted values are generally much lower for northern Europe than for the south. Even though rainfall in the north is less 'agressive' compared to the south, the differences shown on the map appear to be too extreme. The problem is probably caused by the fact that the 'Bavarian equation' which is used to estimate *R*, is only based on summer rainfall. This seems to result in underestimated R-values most of northern Europe. Extrapolating the 'Bavarian equation' to the whole of northern Europe thus does not seem to be appropriate.

Apart from this, many other limitations and shortcomings can be pinpointed. First of all, the Universal Soil Loss Equation only gives a very crude estimate of long-term expected soil loss. It only predicts rill- and interrill erosion: gully erosion is not taken into account. Deposition is not included, only gross erosion is predicted. As long as any of the factors in the equation is greater than zero, some erosion will be predicted, even if the actual erosion is nil.

Furthermore, some important factors influencing soil erosion are not taken into account at all. First, the effect of stones and rock fragments in the soil is not included. Römkens (1985) suggests that the effect of stones is best considered in the C-factor of the USLE, because stones protect the soil surface in a similar way as a surface mulch. Although the European Soil Geographical Database provides a way to estimate stone volume through a pedotransfer rule (Daroussin & King, 1996), only two stone volume classes are distinguished which is too crude for assessing erosion risk.

Second, the effect of management practice is not directly included in the model. This includes practices such as of contouring, stripcropping, terracing and subsurface drainage (Renard et al, 1997). Although these operations can be included in a so-called 'support practice factor' (P-factor), the effect of management practice is nearly impossible to assess at the scale used here. However, it should be realised that management practice may be one of the most important factors affecting erosion in many cases. Third, erosion by melting snow is not taken into account, even though this may be important in mountainous areas.

Probably even more important than the problems mentioned above are the uncertainties associated with the various data sources. Some of the main sources of uncertainty are:

- The estimate of the rainfall erosivity factor (R), which is based on approximate relationships with annual and summer rainfall in Tuscany and Bavaria. Extrapolating these equations to the whole of Europe is potentially inappropriate because of the wide variety of climatic conditions throughout the continent, leading to significant deviations from the conditions for which the equations were established. A possible improvement may be obtained by including similar regression equations obtained from other parts of Europe.
- The soil erodibility factor (K) is estimated from surface texture (except for volcanic soils). However, the actual correlation between K and the soil texture parameters is rather weak. Moreover, the soil units in the Soil Geographical Database of Europe have an unknown (but probably large) within-unit variance.
- The C-factor was estimated using a rather arbitrary scaling procedure of annual NDVI-images. Similar work that was carried out for the whole of Italy already showed some weaknesses of this approach. One major problem is that NDVI is only sensitive to photosynthetically active, healthy vegetation. Regarding the protective properties of vegetation against soil erosion however, the condition of the vegetation is relatively unimportant. As a workaround, maximum C-values were assigned to forest and grassland classes in the CORINE land cover database. Also, seasonal effects are not taken into account because average annual values were used. Because of all this, the resulting C-factor values are only crude estimates.
- For the LS factor, slope angle was derived from a 1-km resolution elevation model. This resolution is far too coarse for predicting soil erosion, and it results in slope angle estimates that are generally too low. In order to get more reliable results a more detailed elevation model is urgently needed. Also, as it was not considered feasible to estimate slope length (or specific contributing area) from the current DEM, an arbitrary constant slope length value was assumed, so the effect of slope length is not included at all.

These and many other uncertainties propagate throughout the model, resulting in an uncertainty in the estimated erosion rate. In theory it is possible to quantify this uncertainty using either an analytical approach or Monte Carlo simulation (Burrough & McDonnell, 1998). In practice, it is difficult to make even crude estimates of the errors associated with each of the individual factors in the USLE. Also, some of the individual factors are inter-correlated, which results in an even greater impact on the model results.

This study is a first attempt to produce a map of soil erosion risk by rill and interrill erosion for the whole of Europe. Its value lies in the fact that the estimates of erosion risk are based on standardised, harmonised data sets for the whole of Europe. However, at this stage it is hard to judge whether the results of the analysis are very helpful for e.g. agricultural and urban planning. The most critical problem seems to be the lack of suitable digital elevation data and the poor representation of rainfall erosivity.

Secondly, better ways to estimate vegetation cover from e.g. satellite imagery are urgently needed. A modelling approach that is better suited to the available data (especially rainfall erosivity) would be feasible, although the problems with the actual data probably pose bigger limitations. The interpretation of the maps is complicated by the fact that Europe encompasses a variety of hydrologic regimes, between which the processes influencing soil erosion by water are essentially different. Furthermore, a scientifically sound validation of the results is extremely difficult at this scale. Asking national experts to judge the results would probably be the most practical way to validate an erosion risk map at the scale used.

Finally, it is emphasised that the results of the analysis should be used with caution. For example, it would not be appropriate to use the maps to predict soil losses on any individual agricultural parcel, nor to predict soil loss for any individual year. Only soil erosion by water flow is taken into account, and then only rill- and inter-rill erosion. Thus, the maps should not be used to predict the occurrence of mass movements like landslides.

In conclusion, the results of this study may be considered as a first step towards a harmonised soil erosion risk map of Europe. Some major improvements could be achieved by using a more detailed digital elevation model, a better representation of rainfall erosivity, and satellite data that have better spectral and geometric characteristics than the NOAA AVHRR data that are currently used. Ideally, multi-temporal satellite imagery should be used in order to account for the interaction between vegetation growth and senescence over the year, and rainfall. Finally, more detailed soil data is required (especially soil depth, stone volume and surface texture).

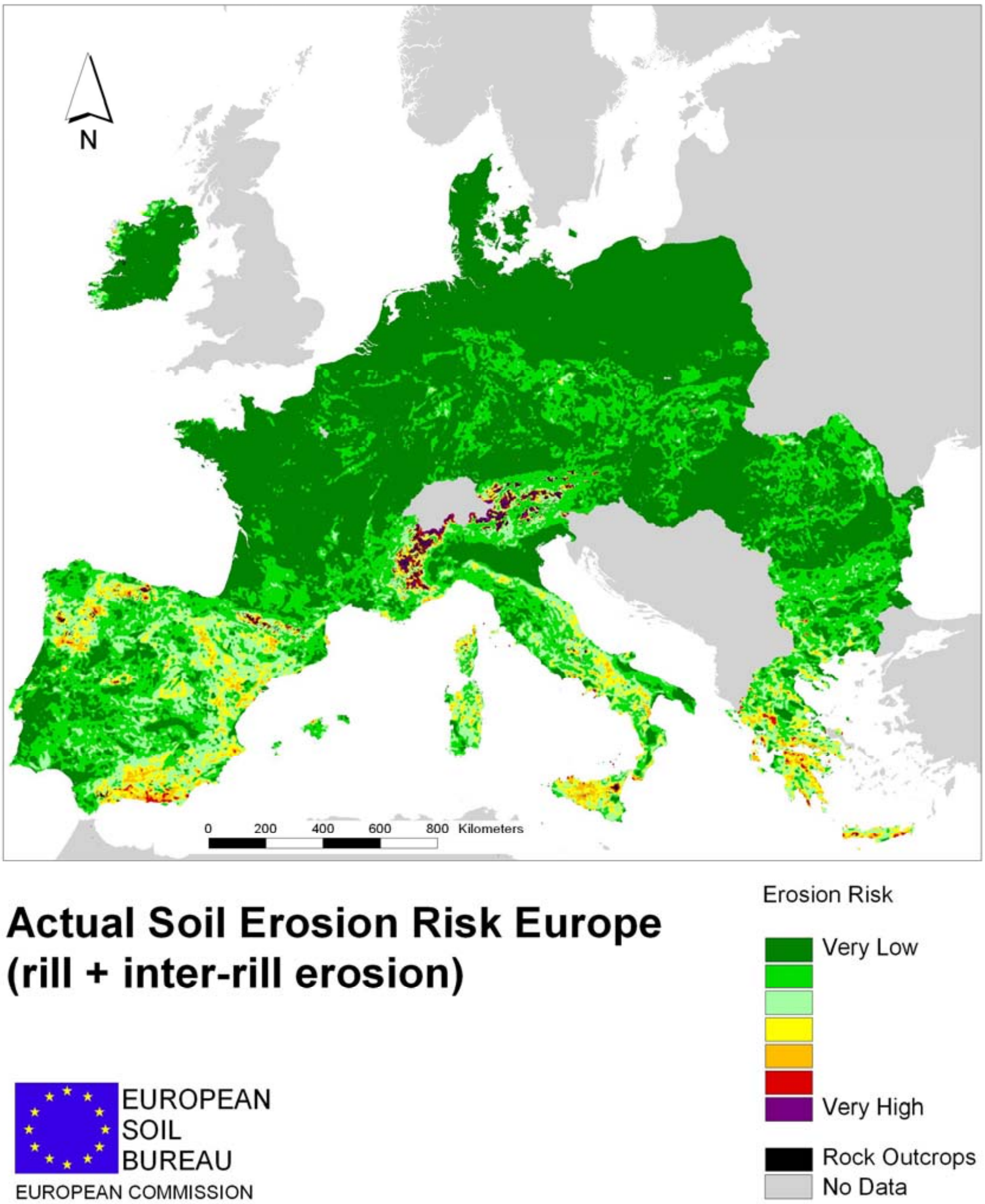

Figure 8.1 *Soil erosion risk map of Europe: Actual erosion risk.*

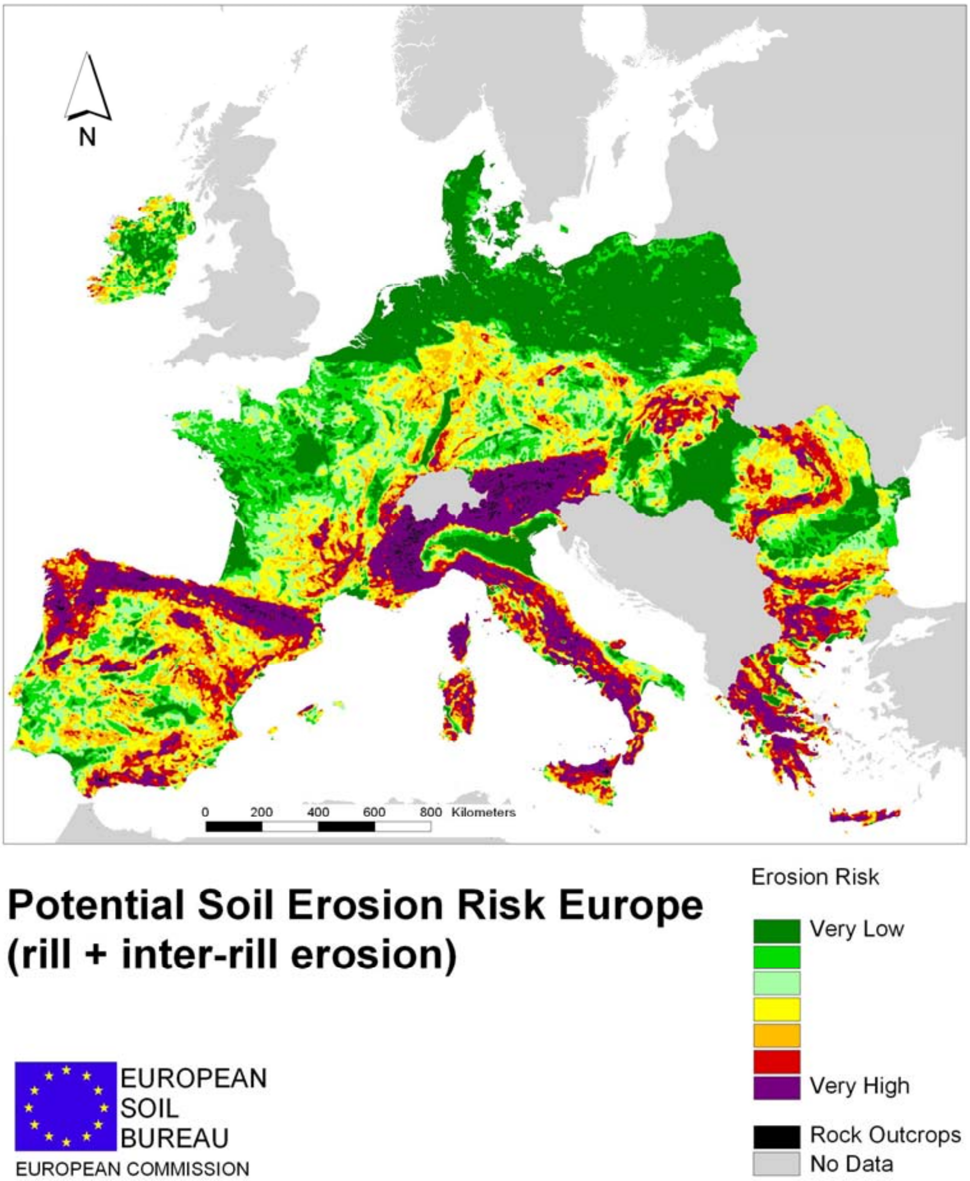

Figure 8.2 *Potential soil erosion risk.*

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