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Risk Assessment Methodologies of Soil Threats in Europe

Status and options for harmonization for risks by erosion, compaction, salinization, organic matter decline and landslides

Christy van Beek and Gergely Tóth (eds.)

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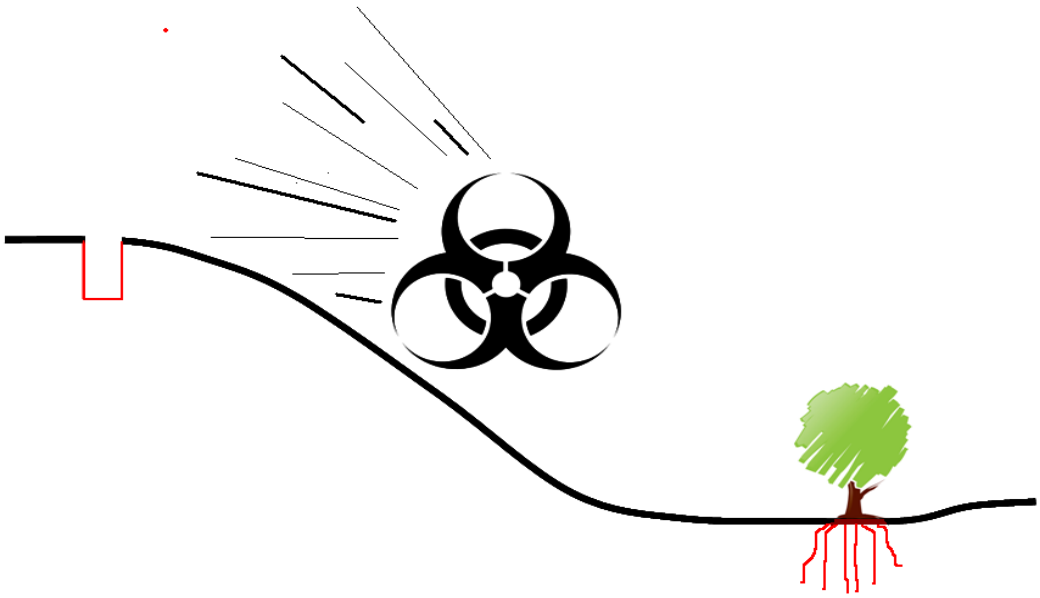

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1. Introduction to RAMSOIL

In 2006 the EU launched the Thematic Strategy for Soil Protection (COM (2006) 231), in response to assessment reports showing that soil degradation is a serious problem in Europe. The overall objective of the Strategy is (i) to prevent further soil degradation and to preserve its functions, and (ii) to restore degraded soils. The Strategy recognizes that certain soil threats, such as erosion, soil organic matter decline, compaction, salinization, erosion and landslides (and at a national level: acidification, contamination and decline of biodiversity), occur in specific risk areas, which must be identified. The identification of risk areas occurs on the basis of Risk Assessment Methodologies (RAMs) for erosion, soil organic matter decline, compaction, salinization, erosion and landslides.

However, at present many different soil RAMs are used in Europe which may potentially hinder unequivocal identification of areas at risk. This is, for instance, demonstrated in two case-studies on i) risks for soil erosion in Romania and ii) risks for subsoil compaction in The Netherlands (Figure 1.3). To overcome different interpretation of the current rate or state of a soil threat, soil RAMs in EU-27 may be harmonized, i.e. results may be made comparable or compatible between different RAMs. Therefore, in 2007 the RAMSOIL project was launched to study the possibilities of harmonization of currently used soil RAMs in EU-27.

In the RAMSOIL project RAMs for soil threats were assessed using the so called risk assessment chain of Figure 1. The risk assessment chain shows the subsequent steps that are taken to assess the risk of a soil threat, i.e. going from the establishment of a notion (definition) of the soil threat, data collection, data processing, data interpretation and risk perception and assessment. Data collection may refer to data derived from field measurements, remote sensing images and/or data statistics on land use, climate, etc. Data processing involves the quantification of a rate or state of the soil threat, using simulation modeling, empirical modeling, factorial assessment or expert evaluation of the data. Data interpretation refers to the comparison of the rate or state of the soil threat with previously defined threshold values. In the final step, the risk perception and assessment step, the risk of the soil threat is assessed in terms of the sense of urgency of actions and remedial measures.

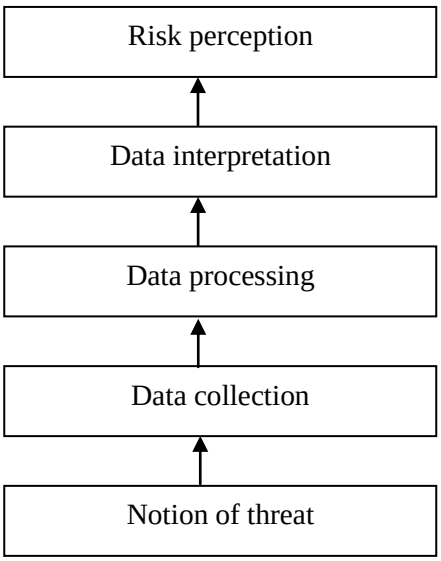


Figure 1.1. The risk assessment chain from data collection (bottom) towards risk perception (top).

To facilitate unequivocal and unambiguous understanding of the rate or state of a soil threat within Europe, soil RAMs may be harmonized, i.e. results may be made comparable or compatible between different RAMs. The EU defines harmonization as 'working towards a convergence of approaches' (Kamrin 1997). There are basically two procedures to overcome difficulties of using different methods and procedures, viz., standardization and harmonization. Standardization is the process of establishing uniform definitions, standards, specifications, methods and procedures. Standardization ultimately results in one uniform assessment procedure for each threat in EU-27. Such standardized methods and procedures may lack the site specificity and cultural identity of regionally-developed RAMs. Harmonization is commonly defined as making (intermediate) results compatible or comparable, hence consistent, and thereby minimizes the differences between standards or measures with similar scope. Harmonization emphasizes 'the combination of two or more things so that they go together, without loss of individual identities yet constitutes a frictionless or pleasing whole' (Webster's New Dictionary of Synonyms).

Possible consequences of *not* harmonizing soil RAMs in Europe

The use of different RAMs to assess the risk of a soil threat in different EU member states may have negative consequences with regard to unambiguous interpretation of the severity of the soil threat, when the results of these RAMs are different. To assess the impact of using different RAMs on the identification of areas at risk, two case studies were performed, testing in total 5 different RAMs:

- Case study 1: erosion in Romania using the SIDASS WEPP approach and the PESERA approach.
- Case study 2: compaction in the Netherlands using the method of Jones et al. (2004), SOCOMO (van den Akker et al., 2006) and bulk density measurements.

The case studies demonstrated the wide discrepancies in outcomes when using different RAMs. Results differed in spatial patterns as demonstrated for soil erosion in Figure, and also in acreage of affected areas using a specific threshold. Figure 1.2 shows that using different soil erosion RAMs may result of a discrepancy in affected area of up to 30%.

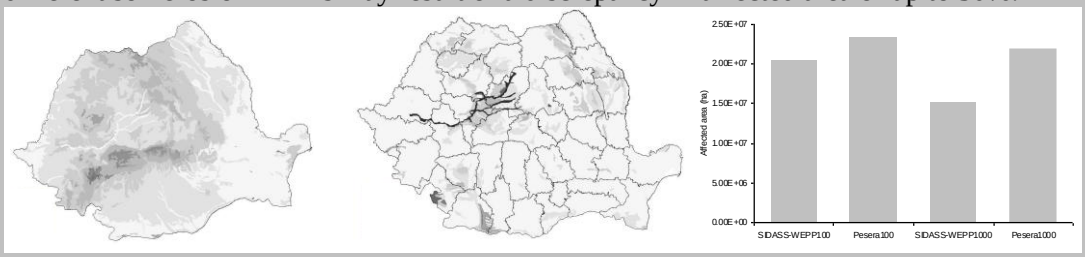


Figure 1.2. Soil loss (t $ha^{-1}y^{-1}$) in Romania evaluated using SIDASS-WEPP model and map of Europe scale 1:1,000,000 (left) and using PESERA model at 1 km grid (right) and affected area in Romania using a threshold value of 1 tonne $ha^{-1}y^{-1}$ using different RAMs and using different spatial grid sizes (100 and 1000 m).

Comparable discrepancies between spatial patterns and affected areas were observed when using different RAMs for compaction in The Netherlands (Figure 1.3).

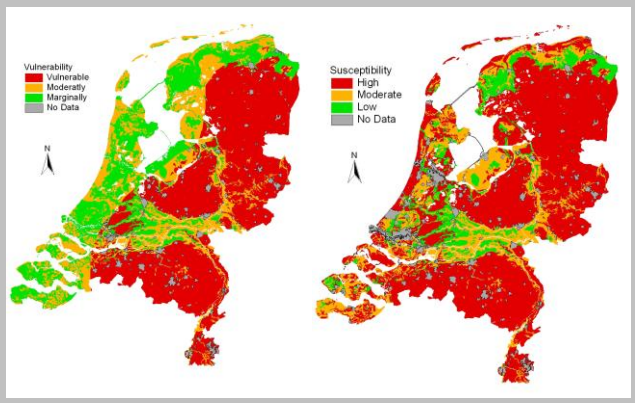


Figure 1.3. The susceptibility (or vulnerability) of soil for compaction according to compaction RAMs reported by Jones et al. (2003) (left) and SIDASS (right).

The interpretation of the terms harmonization and standardization in environmental assessments often differ depending on the field of reference. In the field of soil contamination harmonization may refer to uniforming parameters and toxicological data in simulation models (Theelen 1997). In the same field of study Provoost et al. (2006) recommended that model algorithms should be harmonized, but that critical levels will remain different. Although the authors indicate that harmonization of critical levels would be beneficial, they realize that differences in geography, ethnology and political situation may complicate the implementation of harmonized standards. Wagner et al. (2001) and Theocharopoulos et al. (2001) conclude that harmonization can be beneficial, but they focus on the physical environment of – in this case- sampling and sample treatment, whereas Green et al. (2000) also discusses risk-communication and risk-perception in the light of harmonizing environmental protection strategies.

Hence, harmonization may cover quite a range of issues, going from choosing sampling points to finally perceiving the actual risks and often includes elements of standardization, although this conflicts with the official definition. In the RAMSOIL project the term harmonization is used in a generic way, in line with the common use of the term, i.e. harmonization is considered as all activities leading towards unequivocal understanding of results, of which standardization is one possibility. When harmonizing RAMs a backward procedure is followed, i.e. harmonization is preferably performed at the highest possible level of the risk assessment chain going from notion of the threat, to data collection, data processing, data interpretation and finally risk perception. Moreover, harmonization focuses on the process of making results compatible or comparable, while standardization focuses on prescribing the assessment itself, thereby assuming that prescribed assessment will automatically result in comparable results. The differences, and similarities, between harmonization and standardization are conceptually visualized in Figure 1.4.

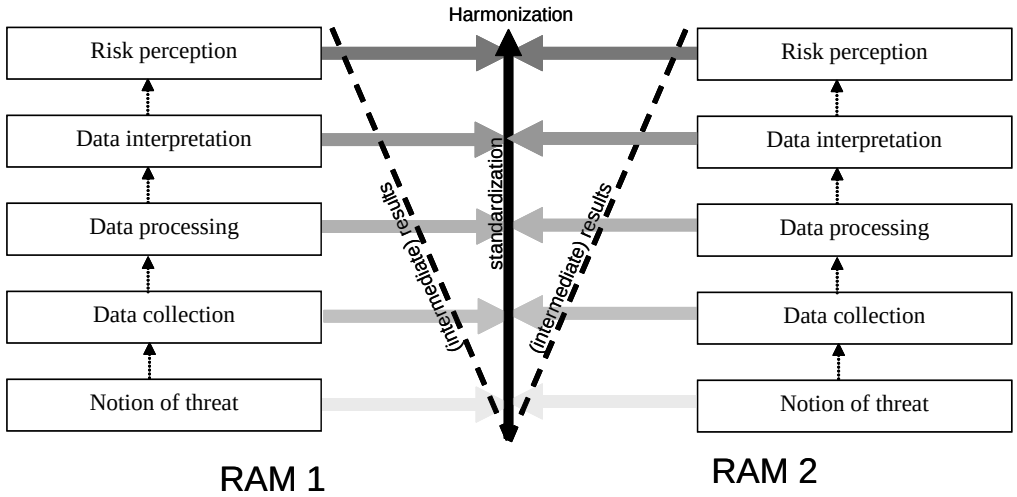


Figure 1.4 Conceptual representation of the meanings of harmonization and standardization of RAMs as used in this paper. The triangle in between the two risk assessment chains represents the increasing divergence of (intermediate) results of the two RAMs, from bottom to top. Standardization (vertical arrow) applies to prescribed procedures and activities in each step of the risk assessment chain, whereas harmonization (horizontal arrows) implies the use of conversion factors at the highest possible level (most direct way, indicated by dark color) and possibly at other steps. Ultimately, both standardization and harmonization should result in comparable risk perception and assessment.

In the RAMSOIL project the current status of soil RAMs in Europe was assessed and needs and options for harmonization were provided. Therefore, questionnaires were sent out to national and regional

scientists and policy makers. More than 100 questionnaires (out of about 200) were returned and provided the basis of the RAMSOIL project.

In this book the results for each threat are presented and options for harmonization are presented. In Chapter 7 an integrated assessment of needs and options for all soil threats is presented and in Chapter 8 a the concept of the soil threat index is discussed in the light of harmonization. Finally, in Chapter 9 the main conclusions of the RAMSOIL project are listed.

All work described is this book originates from underlying project reports which were produced by a consortium of scientists. The project consortium is described in Annex 1. The RAMSOIL report series is provided in Annex 2. Reports can be obtained by sending an e-mail to info@ramsoil.nl, or can be downloaded from www.ramsoil.eu. The RAMSOIL project was co-funded by the European Commission, DG Research, within the 6th Framework Programme of RTD, (Priority 8 - Specific Support to Policies, contract n° 44240). The views and opinions expressed in this publication are purely those of the writers and may not in any circumstances be regarded as stating an official position of the European Commission.

2. Soil erosion

S. Verzandvoort, L. Recatalá-Boix and C. Año-Vidal

2.1 Notion of threat, definition and terminology

Soil erosion is understood to be 'a physical phenomenon that results in the displacement of soil particles by water, wind, ice and gravity' (Eckelmann et al., 2006). It is a natural process that can be exacerbated by human activities. These are essentially reflected in the land cover, where land use changes and land management – such as tillage and implementation of conservation strategies – determine the susceptibility to erosion (Gobin et al., 2004; Boardman, 2006). Soil erosion is perceived as one of the major and most widespread forms of soil degradation and it has large environmental and economic impacts, especially in agricultural areas.

Soil erosion by water and to a lesser extent by wind continues to be a problem in Europe, despite the current trend of increasing shares of agricultural land with tolerable erosion rates due to prolonged efforts in soil conservation and innovations that increased land productivity (Gobin et al., 2004; Kirkby et al., 2004; Boardman and Poesen, 2006; EEA, 2006; Pimentel, 2006; EEA, 2007a; ESBN, 2005; OECD, 2008a). Focal areas include the SEE countries, Eastern Europe and the Mediterranean region (SOVEUR, 2000; EEA, 2007a). Political evidence of the attention to soil erosion problems in the Member States is found in the proposed Soil Framework Directive (COM(2006) 232),and in the increased share of agricultural area subject to instruments or activities employed in the Common Agricultural Policy (SMRs, GAEC and Agri Environmental Measures), EU environmental directives, and international conventions (UNCCD, UNFCCC) (SoCo, 2009; ahu/Ecologic, 2007; EEA, 2006; EC, 2005).

Soil erosion problems mostly originate on agricultural land, though recreational areas, roads and (abandoned) industrial zones are receiving growing attention as source areas for soil erosion (Höke and Burghardt, 2001; Morgan, 2005; Cao et al., 2009). The assessment of soil erosion is very difficult, costly and in most cases the average value of a catchment is considered ignoring the internal spatial losses and depositions. On-site effects of soil erosion are refuted and include soil and nutrient loss and breakdown of soil structure through wind erosion, gullying, piping, rill-interrill, snowmelt, harvesting and tillage erosion on the same hillslope or field as where the soil erosion phenomena originated. Off-site effects refer to the effects of sediment dropped at a distance from the source areas, causing pollution of surface waters, siltation of channels and reservoirs, and damage to infrastructure and buildings. The economic impacts of the off-site effects appear to be much larger than those of on-site effects (EC, 2006b; Schuler et al., 2006).

In most European countries, there is increasing awareness of the unfavorable effects of soil erosion, and governments and agencies are responding to these effects (EEA, 2001; Fullen et al., 2006). Several countries employ risk assessment methods for soil erosion to be able to focus agricultural, environmental and spatial planning policies on vulnerable areas.
The continuing importance of soil erosion and the expected changes in the use and spatial distribution of agricultural land in Europe call for a European-wide look at risk assessment of soil erosion, in addition to the national assessment methods. The proposal for a EU framework Directive for soil protection identified the need to compare levels of erosion risk between member states, to set minimum levels of protection, to evaluate measures for soil conservation and restoration in a similar way between states, and to identify new areas at risk (EC, 2006a).

Several European wide risk assessment methods for soil erosion were developed and applied to comply with these needs (e.g. CORINE, 1992; Van der Knijff et al., 2000; Le Bissonnais et al., 2002; Grimm et al., 2002; Kirkby et al., 2004; Gobin et al., 2006). However, these assessment methods are not related to the variety of national erosion risk assessment methods, neither in the technical phases of data collection and processing, nor in the more socio-economical phases of data interpretation and risk perception, if the latter two phases are covered at all. It may be difficult to use European wide assessments for the design of agricultural and environmental policy at the level of the European Community, because such policies need to be accepted and implemented by stakeholders in the member states.

If national risk assessment methods for soil erosion are to be used in response to the needs formulated in the proposal for the EU framework Directive for soil protection, this requires that results are made compatible or comparable. This activity is called harmonization (chapter 1). It can refer to all five phases in the risk assessment process (chapter 1).

In the case of soil erosion risk assessment, the data collection phase consists of the direct measurement of actual erosion rates (for direct assessment or calibration and validation of erosion models) or the collection of input data for expert analysis or modeling (e.g. rainfall regime, soil types, topography, land cover). In the data processing phase, the intensity of soil erosion processes is assessed based on the collected data, either qualitatively using expert judgement or factorial approaches, or quantitatively, using empirical or process-based erosion models. In order to determine which levels of erosion intensity create risks for society, the expressions of erosion intensity must be compared to thresholds. The output of this phase is often in the form of erosion risk maps. Finally, the risk perception phase assesses to which degree the land use system is affected once soil erosion intensity exceeds the established thresholds, and defines consequences corresponding to the sense of urgency of this degree (e.g. 'arable farming no longer sustainable').

Soil erosion hazard: the properties of erosion having the potential to cause unfavorable effects (after OECD, 2003)

Soil erosion risk: the probability of an unfavorable effect in a land or ecosystem system by exposure to erosion (after OECD, 2003)

Soil erosion risk assessment: the process to calculate or estimate the risk to a system, following exposure to erosion (after OECD, 2003)

Potential and actual soil erosion risk
The potential soil erosion risk reflects the local conditions of soil, climate and slope. Actual erosion risk takes account of land cover in addition to the local conditions (Morgan, 2005).

Box 2.1 Terminology in the soil erosion literature applied in the review of risk assessment methods for soil erosion.

The aim of this chapter is to review risk assessment methods for soil erosion currently applied in EU member states, and to outline options for harmonization. The review is structured along the four phases of the risk assessment process.

Table 2.1 Responses to political and thematic questionnaires in the framework of the RAMSOIL project.

Country	Political questionnaire	Thematic questionnaire for soil erosion	RAM for soil erosion with acknowledged status
Belgium	2	3	1
Denmark	2	1	
Cyprus		1	
Czech Republic	1		
Estonia	1		
Finland		1	1
France			1
Germany	2	3	1
Greece	1	2	
Hungary	1	2	1
Lithuania		1	
Italy		1	
Netherlands	1		
Norway		1	1
Poland		1	1
Serbia	1		
Spain		3	1

The data used include responses to questionnaires collected in the framework of the EU RAMSOIL Project. This yielded 13 responses to a political questionnaire, and 17 responses to a thematic questionnaire on soil erosion (Table 2.1). Of the latter, only the questionnaires from countries having a risk assessment method (RAM) with an officially acknowledged status were reviewed. In order to offer a framework of comparison, two pan-European risk assessment methods for soil erosion were also included in the review: CORINE (1992) and PESERA (Kirkby et al., 2004).

The review of the data collection and data processing phases is based on responses to the thematic questionnaire. The review of the data interpretation and risk perception phases is based on responses from the political and the thematic questionnaires, supplemented by literature search.

Some terms used in this chapter are explained in Box 2.1.

2.2 Data collection

The types of erosion addressed in the inventoried risk assessment methods (RAMs) include mostly water erosion, including rill-interrill erosion, channel erosion and snowmelt, and in a few cases wind erosion and tillage erosion.

Data used in risk assessment methods (RAMs) for soil erosion include direct measurements or field observations of soil erosion, soil loss or sediment yield, input data for erosion models or knowledge systems, and data used for the calibration and/or validation of these. Responses from four countries indicated to use direct measurements of soil erosion in their risk assessment methods (Poland, Spain, Finland, Hungary). However, the responses did not elicit the type of measurements or observations used in these RAMs.

Although most risk assessment methods use erosion models or knowledge systems requiring or providing data of a dynamic nature (e.g. crusting stage, rainfall), or providing quantitative estimates of soil erosion, for only 5 out of the 11 RAMs monitoring is performed for model input, calibration or validation (i.e. the RAMs from Poland, Spain, Finland and Hungary). Calibration and validation of erosion models is commonly accepted as a prerequisite to use results for societal applications (e.g. Jetten et al., 1999; Van Camp et al., 2004; Jetten, 2007). This implies that the investigated RAMs, even though some use the same basic model, provide erosion risk outputs of varying reliability.

Most RAMs use collected or existing data on the common criteria for risk area identification for soil erosion, as selected by Eckelmann et al. (2006) (figure 2.1). Strikingly, most RAMS also use information on conservation practices in the risk assessment of soil erosion, but the responses to the questionnaire give no insight in whether this information is used to evaluate the performance of these conservation practices with regard to the estimated risk.

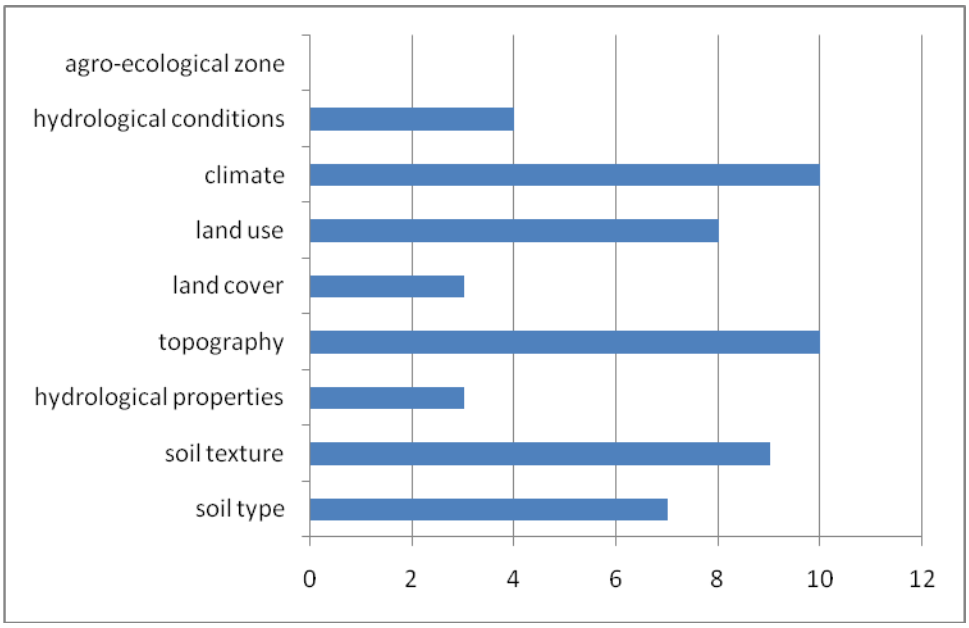


Figure 2.1 Use of data resorting under the common criteria for risk area identification of soil erosion according to Eckelmann et al. (2006).

In case existing spatial data are used, these vary widely in scale: from 1:50.000-1:250.000 for the RAM used in Spain, to 1:00.000-1:5.000.000 to the RAM used in France. Use of the CORINE database for land cover is mentioned for 3 RAMs.

2.3 Data processing

Risk assessment methods for soil erosion used in European countries or in European –wide assessments can be divided in three categories: based on field measurement and monitoring of actual soil erosion or soil erosion indicators, based on modeling, and based on expert analysis (Grimm et al., 2002; Gobin et al., 2006).

Methods for erosion risk assessment based on field measurement and monitoring provide measured actual erosion rates. However, actual soil loss currently is measured at few sites of soil monitoring networks in the EU Member States (Morvan et al., 2008). This type of risk assessment method is limited by the difficulty to obtain representative values for larger areas (catchments) due to the spatial and temporal variability of soil erosion processes (Gobin et al., 2004).

Risk assessment methods based on modeling employ knowledge systems, empirical or process-based models. Examples of knowledge systems are hierarchical multifactorial classifications, which provide qualitative estimates of the severity of soil erosion based on decision rules using indicators of soil erosion (also termed factors). Empirical models are based on empirical studies for their representation and governing equations of erosion processes (Parsons and Wainwright, 2006). They include well known erosion models like the Universal Soil Loss Equation (Wischmeier and Smith, 1978). Process-based erosion models describe processes leading to soil erosion mainly based on physical laws. Erosion rates estimated using these models generally do not assess degradation up to the present time (Gobin et al., 2006). Finally, in expert-based RAMs, soil erosion risk is based on expert judgement for pre-defined areas (e.g. GLASOD, Oldeman et al., 1991).

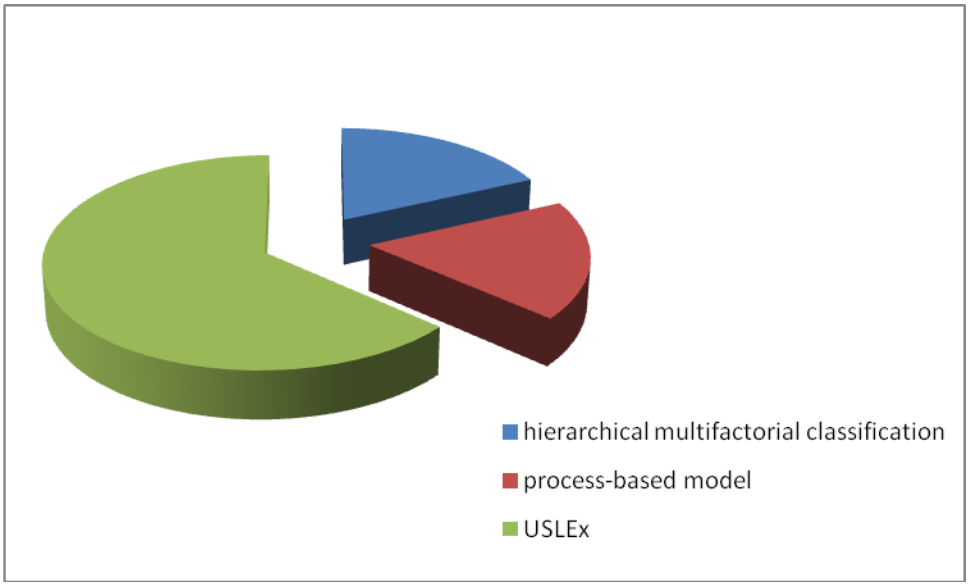


Figure 2.2 Categories of risk assessment methods for soil erosion.

The majority (7 out of 11) of the inventoried RAMs for soil erosion uses the USLE or modified forms of this model (figure 2.2). Three RAMs use factorial approaches in the form of a hierarchical multifactorial classification (INRA Approach in France, PWER[1] and AWER[2] indicator methods in Poland, and CORINE). Three RAMs use process-based models (EROSION 3D and SWAT in Poland; ICECREAMS in Finland, and PESERA).

As a consequence of the prevalent application of the USLE, most RAMs only consider water erosion, and then limited to sheet and rill erosion. One exception is the RAM of Belgium, which uses separate model components for channel and tillage erosion in conjunction with the RUSLE. RAMs which use process-based models also consider channel erosion, deposition, transport of chemicals, effects of snow/ice and and/or wind erosion.
Only in one RAM (Norway), the erosion model is calibrated. Only two RAMs validate the method used for erosion risk assessment, by expert judgement in combination with field visits (the INRA Approach-France) and field measurements at the field and catchment scale (PESERA). This implies that the quality of the model output used in the majority of RAMs is unknown, nor in terms of soil loss estimates, nor in terms of erosion risk classes.

2.4 Data interpretation

The responses to the questionnaires and literature (Boardman and Poesen, 2006; OECD, 2008a,b) reveal that classification systems for soil erosion risk vary widely between European countries. Classifications may employ ordinal or scalar scales, depending on whether the erosion rate is expressed in qualitative (e.g. 'low erosion risk', 'medium erosion risk', 'high erosion risk', 'very high erosion risk', as in the system of Norway) or quantitative terms (t/ha/y). Of the 18 countries for which risk classification systems were found, 9 use quantitative classes. Both qualitative and quantitative systems usually have 4 to 6 classes. The documentation of risk classifications often does not provide descriptions of classes or class limits, which makes it difficult to interpret (qualitative or quantitative) erosion rates in terms of risk (Box 2.1). This makes the comparison of risk classification systems between countries difficult.

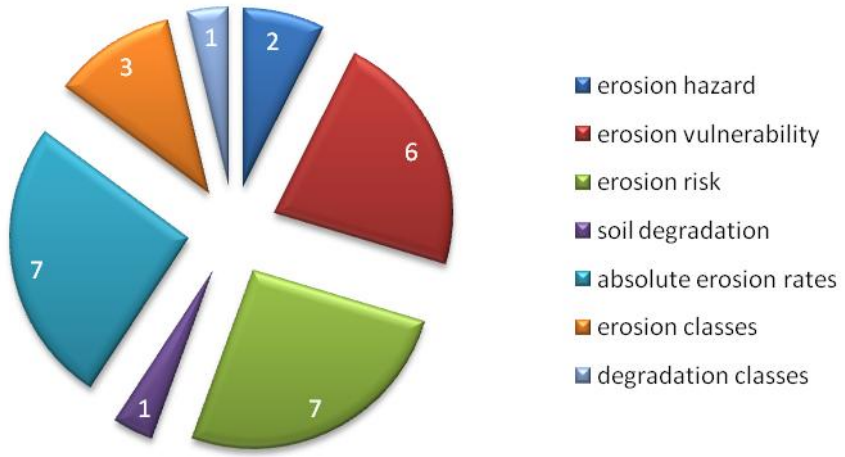


Figure 2.3 Output variables of risk assessment methods for soil erosion in EU member states. source: RAMSOIL thematic questionnaires.

[1] Potential Water Erosion Risk (Józe Faciuk and Józe Faciuk, 1995)
[2] Actual Water Erosion Risk (Józe Faciuk and Józe Faciuk, 1996)

Another difficulty with interpreting soil erosion assessments in terms of risk is that output variables of methods vary widely (figure 2.3). In most cases the vulnerability of land to erosion, erosion risk and/or absolute erosion rates are produced, mostly in the form of maps. The national reports on soil erosion in the review of Boardman & Poesen (2006) on soil erosion in European countries use expressions of 'soil erosion risk' for 13 countries. Often, in the maps or the covering text is not explained what the term 'erosion risk' means. In fact, many of these maps display erosion rates in t/ha/y, severity, or other expressions of erosion process rate, rather than a risk to the land use systems concerned in terms of value lost.

The questionnaires and literature report few countries to use threshold values to assess to which extent certain levels of erosion intensity may be considered tolerable or acceptable by the stakeholders in the area concerned . Some examples are given in figure 2.4. The threshold values are below 10 t/ha/y for most countries, and well below the maximum erosion rate figuring in the risk classification systems (figure 2.4). This implies that tolerable erosion rates are well below the maximum rates encountered in these countries, and that erosion is regarded as a process causing unfavourable effects to land use systems in these countries, or regions thereof.

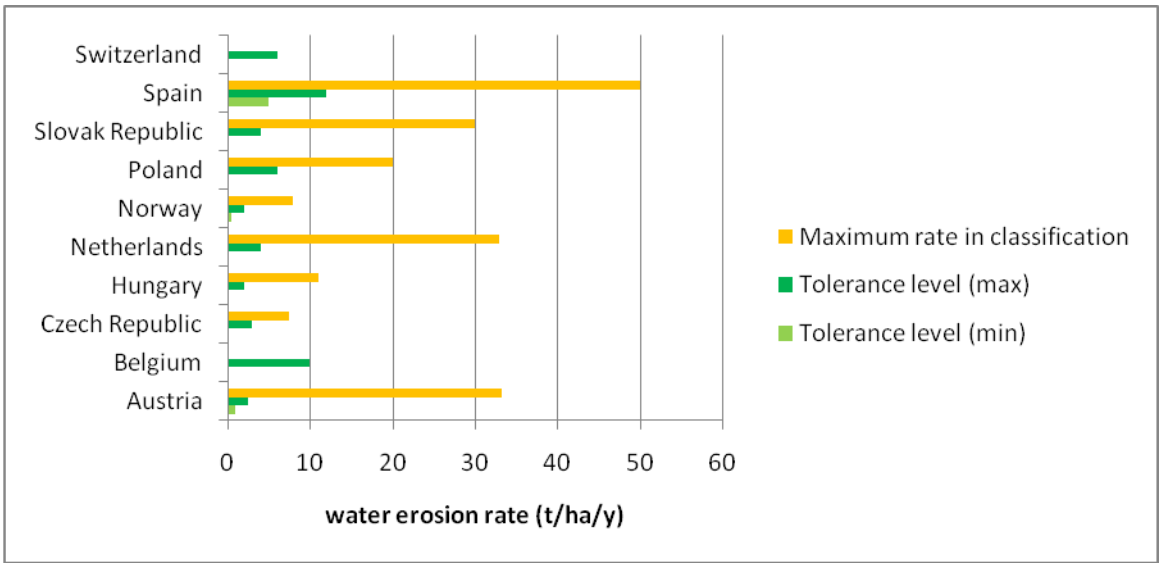


Figure 2.4. Tolerance levels for soil erosion by water in some European countries.
Sources: Boardman & Poesen (2006), OECD (2008a) and RAMSOIL thematic questionnaires.

2.5 Risk perception

For 11 out of the 21 inventoried RAMs, outputs are used for land use planning, management or conservation strategies, or for fine tuning policy regulations. 8 RAMs are directly linked to community policy (e.g. in SMRs, or as a basis for legal normatives). The responses to the questionnaires do not show how stakeholders are involved in the development and application of the RAMs. Also, the dissemination of results from risk assessments to the general public appears to be limited (mentioned for 5 RAMs only). This is emphasized by the observation that the majority of RAMs indicates 'science' as the reason for the development of the RAM instead of 'legislation'.

Information on risk levels and/or thresholds resulting from risk assessments of soil erosion is mostly used in recommendations for to use soil conservation and land management strategies on farmland (see the examples listed in Table 2.1).

Table 2.1 Use of risk levels and/or thresholds for response to soil erosion risk in several European countries. Source: Boardman and Poesen (2006).

European country	**Recommendations for response to soil erosion risk based on risk assessment**	**Intended implementing party for response**	**Specification**
Bulgaria	Recommended erosion prevention measures based on land capability evaluation and estimated average soil loss rates	(governmental) National Long-term Erosion Control Programme	
Estonia	None; no restrictions at national level imposed on the use of land at risk of erosion		
France	Delineate areas of erosion risk where farmers and land owners will be obliged to apply soil protection measures against erosion	Departmental Commissions on Natural Risks (Law on Natural Risks, 2003)	
Hungary	Recommendations for agricultural use		Soil loss 0-2 t/ha/y : average rate of soil formation according to Hungarian estimates
			2-11: agricultural production can be considered sustainable
			>11: areas where arable farming should not be allowed, or with strict regulations
Lithuania	Recommended erosion-resisting crop rotations for slope classes	Policy makers	Slopes 2-10°: erosion-resisting crop rotations
			Slopes >10°: no arable crops, only perennial grassland
Netherlands	Farm conservation plans	Water board, provincial and municipal authorities, farmers	Erosion events with recurrence intervals of 10 y for rural areas: prevent by implementing conservation measures according to farm conservation plan (min 40-100 points/ha)
			Erosion events with recurrence intervals of 25 y for residential and built-up areas: prevent by implementing conservation measures
Poland	Recommendations for good agricultural practice	Ministry of Agriculture and Rural Areas Development and Ministry of Environment (2002)	Arable land susceptible to erosion (no further definition) on slopes >20% Arable land susceptible to erosion (no further definition) on slopes 10-120%
Slovakia	Introduction of conservation tillage at national level	Ministry of Agriculture	Specifically on fields with wide row crops and highest erosion risk
United Kingdom	Advice how to avoid erosion and runoff, not linked to risk levels or thresholds	Environment Agency	
	No legislative or legal constraints on farmers in Britain who allow erosion to occur on their land; for off-site damage to property, laws of negligence or nuisance or the highways act may be invoked, but this requires risk assessment. There is a Code for Good Agricultural Practice		

Table 2.1 continued.

European country	Recommendations for response to soil erosion risk based on risk assessment	Intended implementing party for response	Specification
Iceland	Land use policy statements	Agricultural Research Institute & SCS(government institutes)	0 no erosion: no suggestion 1 little: no suggestion 2 slight: care needed 3 considerable: reduce or manage grazing 4 severe: protect – no grazing 5 very severe: protect – no grazing
Norway	Planning soil conservation measures, subsidies; 90% of subsidies to areas with medium to very high erosion risk	Farmers, advisory services, authorities	
Switzerland	Ordinance on Soil Protection	Cantonal governments	Cantons must compare erosion rates to thresholds using the USLE-based 'Key for the assessment of erosion risk". This method provides rough estimates of erosion risk for farmers and advisory services based on fixed standards
Ukraine	None		No land value appraisal; soil as a means of production has no price

2.6 Options for harmonization

Data collection phase

Even though in different countries the same types of data are being collected for soil erosion RAMs (figure 1), the data collection differs with regard to measurement methods and units, the spatial and temporal support used, the accuracy of the data and classification systems used. For example, a DEM can be derived using LIDAR (Light Detection and Ranging), or from digitised elevation contours (type of method); hydraulic conductivity can be determined on soil samples in the laboratory or on pedons by in situ methods in the field (spatial support); sediment concentration can be expressed in M/V or V/V, and different classifications are used for soil texture, soil type and land cover and use.

Only standardization can overcome the differences in risk assessments due to these differences in the data collection phase. The establishment and maintenance of data collection standards would require a large effort from both authorities and member states, certainly in the current situation, in which data collection for soil erosion risk assessments at the national level and below is the responsibility of member states.

Data processing phase

For the data processing phase, harmonization of erosion modelling based on the USLE may seem attractive, as the model is the most used in the investigated RAMs, and offers the benefit of transferring distributed data into simple factord (Gobin et al., 2006). Yet, harmonization of RAMs using this model would require notion of the following aspects: 1) the model factors are parameterized differently between RAMs to meet the limitations of available input data, 2) the lack of calibration and validation, as a result of which the quality of model outputs cannot be assessed between RAMs, and 3) the questionable application in settings for which the USLE was not designed (e.g. low-intensity rainfall, large slope lengths, stony soils, areas with gully erosion or predominant deposition – Jetten and Favis-Mortlock, 2006).

The qualitative, factorial approaches to erosion risk assessment based on expert judgement, applied in the INRA, CORINE and GLASOD approaches, have the advantage of being able to roughly delineate soil erosion risk at a European level in the absence of accurate and extensive information for input, calibration and validation, which are required by model-based RAMs. Difficulties for harmonization relate to the lack of objectivity in comparing the standards applied by different experts for different areas. Also, because no quantitative estimates of soil erosion are made, it is difficult to evaluate the effect of changes in land use and/or climate on the erosion risk. Scientific objections against the method refer to how factors are combined into a single scale, how individual weightings are being justified for separate factors, and to assumptions of linearity and statistical independence (Gobin et al., 2006). Results depend on the class limits and the number of classes used. These methods might be used to identify extremes in erosion in Europe-wide assessments, but not to compare assessments at the national scale.

In the framework of the RAMSOIL project, Geraedts et al. (2007) recommend the PESERA model as an option for harmonization based on modelling. This model has the advantages of a strong physical basis and the possibilities of validation at high resolutions and Europe-wide forecasting at a coarse resolution. These advantages cause the model to be scientifically sound and flexible, as the review from WP2 points out, but on the other hand to require enormous amounts of temporally and spatially distributed input data. Apart from that, the accuracy of the model cannot be guaranteed without frequent calibration and validation for changes in environmental settings (e.g. land use change, climate change). The high data demand and the requirement for calibration and validation may hamper the use of PESERA as a harmonized RAM for soil erosion by EU member states. In addition, if erosion risk assessment based on PESERA are to be comparable between member states, also data collection, processing and procedures for calibration and validation would need to be harmonized.

Part of the damage due to soil erosion in Europe concerns off-site damage, where the generation and transport of soil and water at places in the landscape cause nuisance in other places (e.g. sedimentation in reservoirs, water pollution). This implies that RAMs for soil erosion, whether based on expert judgement, model or factorial approaches, should somehow account for spatial connectivity in the landscape between source areas of soil loss and areas affected by the transport of water, sediment and nutrients or pollutants, or by the deposition of these.

Apart from the differences in risk assessment outputs due to different methods used in the data processing phase, differences may also arise from the spatial extent of the outputs. Within countries, results of erosion (hazard) assessment at different administrative levels (local, regional, international) do not always agree, as is the case for Spain (Sanchez Diaz et al., 2001; cited by Solé Benet, 2006).

Data interpretation phase

Classification systems for erosion risk are tailored to the hazard, vulnerability and value properties of a country's territory, and therefore differ between countries. Especially the delimitation of the tolerable erosion risk class expresses different perceptions of erosion risk between countries.In several countries (Austria, Czech Republic, Hungary, Norway) water erosion is no longer tolerable at smaller rates than those proposed by generally accepted standards like the expert-based rate of 2 t/ha/yr (ESBN, 2005) and the classification system of the OECD (2008). This illustrates the difficulty of harmonizing frameworks for the translation of erosion hazard into risk assessment.

In the literature on soil erosion, soil erosion is commonly considered 'tolerable' (i.e. at low or zero risk) if it does not impact on long-term productivity by changing properties like depth, nutrient status and texture (e.g. Gobin et al., 2004; Jones et al., 2004; OECD, 2008). This concept considers only the direct soil value for agricultural production. Morgan (2005) suggests that the severity (or risk) of soil erosion is better judged in relation to the damage caused and the costs of its amelioration. This would fit better into generally accepted definitions of environmental risks in terms of the consequences of the environmental process occurring, in this case soil erosion (damage caused relates to vulnerability ; costs of its amelioration relates to value).

The variety in outputs of RAMs for soil erosion indicates that there is no clearly defined target variable for soil erosion risk assessments performed in European countries. This is a requirement for harmonization of RAMs.

Risk perception phase

The inventory of RAMs for soil erosion show that the risk perception phase is not formalized or documented in the risk assessment methods. This may be explained by the fact that this phase essentially deals with the understanding of the significance of soil erosion risk to individual and societal values (Pollard et al., 2008), and therefore require the intricate involvement of stakeholders who benefit from the reduction of soil erosion risk. They also require a distinction between actual soil erosion risk, and potential (future) risk due to driving forces like climate change, land use and management, agro-ecosystem management and human population dynamics (EEA, 2002; Gobin et al., 2004). This distinction is not always provided in the preceding data processing phase. Due to the patchy nature of soil erosion impacts in time and space values and stakeholders are difficult to identify. The large number of farmers and other owners involved complicate the involvement of stakeholders compared to risk assessment processes for other environmental media (air, water). Also, there is a sense that private land owners are or should be sufficiently self interested and farsighted to manage their land (IEEP, 2008; Pesonen, 2008). Finally, soil erosion risk, like many other soil threats, is not readily noticed by the media and the general public, and there have not been dramatic triggers to initiate risk assessment compared to other threats like catastrophic flood and landslide events (IEEP, 2008; Van Beek et al., submitted).

These observations explain why current risk assessment methods for soil erosion in European countries, if existent at all, are not or marginally used in processes with stakeholders to judge erosion risk against acceptable levels of residual risk (because soil erosion will always occur), and to establish a 'common operational picture' of the sense of urgency to act against the gamut of soil erosion risks.

Ultimately, the weak incorporation of the data interpretation and risk perception phases in the risk assessment chain for soil erosion may be one of the reasons why European or national policy measures to directly or indirectly protect soil against erosion have only been adopted or implemented in a few member states, despite the huge amount of available knowledge on the principles and technologies of soil conservation developed in the US and Europe since the 1930s (Kwaad, 2008).

3. Subsoil compaction

Jan J.H. van den Akker and Catalin Simota

3.1 Notion of threat, definition and terminology

Problems of compaction are widely distributed throughout the world, but tend to be most prevalent where heavy machinery is used in agriculture or forestry, in both temperate and tropical areas. Soils that are naturally fragile in structure, such as soils of the humid tropical forest and light-textured soils in areas of low but erosive rainfall, are particularly prone to problems arising from compaction and subsequent high risks of erosion due to reduction of permeability.

Fraters (1996) estimates that about 32% of the subsoils in Europe are highly vulnerable to subsoil compaction and another 18% are moderately vulnerable, but no precise data are available. Oldeman et al., (1991) estimate that the area of degradation attributable to soil compaction is equal or exceeds 33 Mha in Europe. Batjes (2001) states that compaction is the most widespread kind of soil physical soil degradation in Central and Eastern Europe: about 25 Mha proved to be lightly and about 36 Mha moderately compacted.

Compaction is defined by Van den Akker and Soane (2005) as a process of densification and distortion in which total and air-filled porosity and permeability are reduced, strength is increased, soil structure partly destroyed, and many changes induced in the soil fabric and in various characteristics. The term "compaction" is used to identify a process and should be distinguished from the term "compactness", which indicates for a given time and position the state of packing of the solid soil constituent. Van den Akker and Soane (2005) define the subsoil as the soil below the loosened layer (about 20 - 35 cm thick). This definition of the subsoil includes the panlayer as the upper part of the subsoil. This panlayer is, in many cases, less permeable for roots, water and oxygen than the soil below it and is the bottleneck for the function of the subsoil. In contrast to the topsoil, the subsoil is not loosened annually, compaction is cumulative, and, in the long run, a more or less homogeneous compacted layer is created. The resilience of the subsoil for compaction is low and subsoil compaction is at least partly persistent. From the short-term economic and environmental point of view topsoil compaction has more impact than subsoil compaction. However, from the sustainable point of view subsoil compaction is the most serious threat. This was also the conclusion of the EU Soil Strategy Working Groups (Van Camp et al., 2004), and this is the reason why subsoil compaction is addressed in the European Soil Strategy and not topsoil compaction.

3.2 Data collection

In total 17 questionnaires on compaction were returned. Figure 3.1 shows the countries that returned the questionnaires on compaction. Figure 3.2 presents to which extent the RAMs have an official status. It should be noted that all RAMs in Germany, Denmark, France, Romania and Spain and the proposed RAM in Finland are based on more or less the same deterministic approach (Horn et al., 2005, Simota et al., 2005). The two questionnaires from Belgium are in fact a questionnaire from Wallonia and a questionnaire from Flanders. Wallonia has no RAM and the RAM of Flanders is in development. The four responses of Germany are from the federal government, an advisor on soil compaction RAMs, the state of Thüringen and the state of North Rhine Westfalia. Each state of Germany has its own policy how to deal with the implementation of the federal laws and European directives.

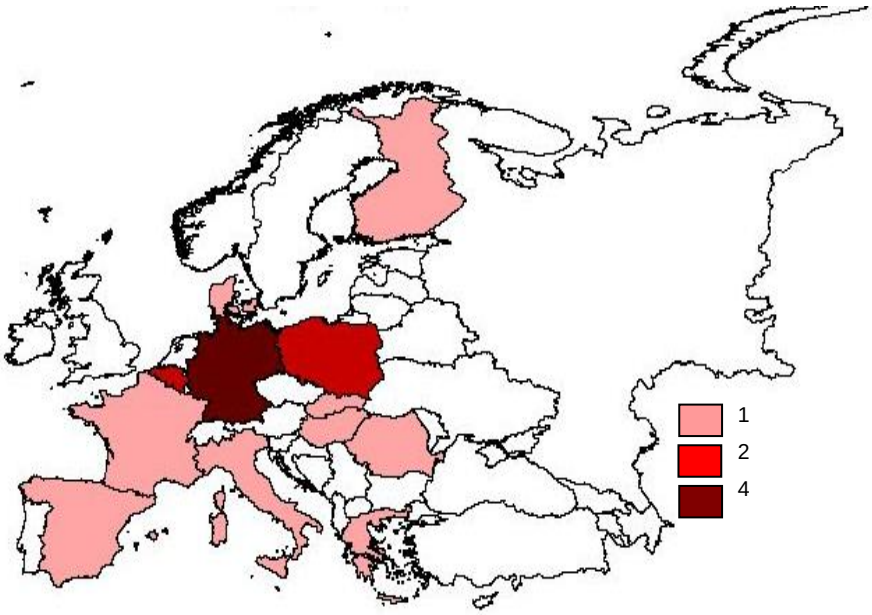


Figure 3.1. Return number of compaction questionnaires.

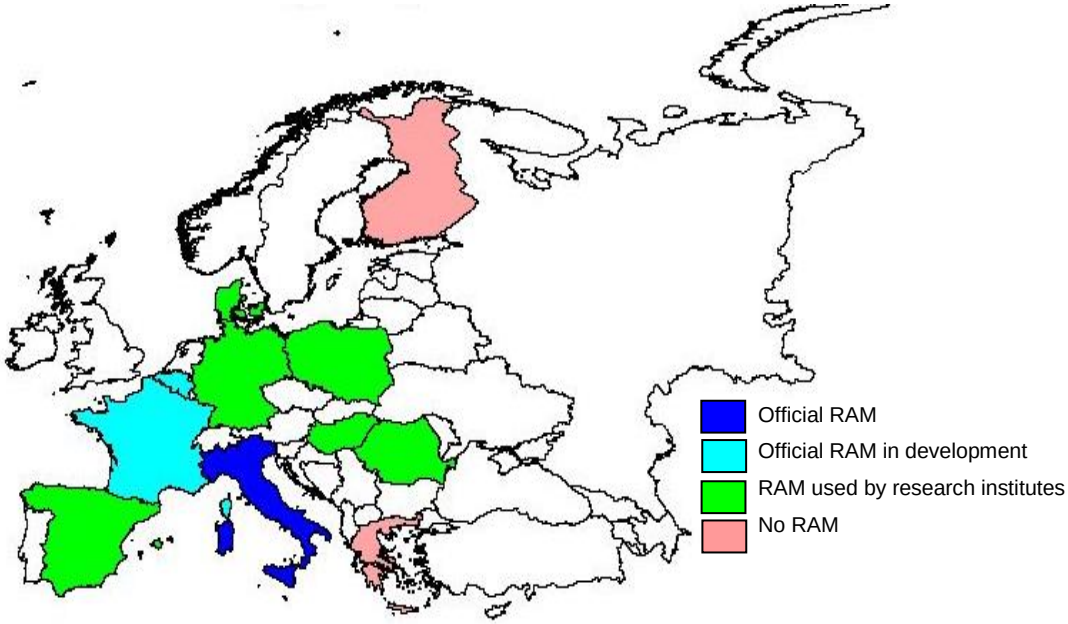


Figure 3.2. Status of the compaction Risk Assessment Methodologies (RAMs) in the European Union.

Table 3.1. Information used in the risk assessment of compaction.

CRITERIA																		
Country		Romania	Germany	Germany	Germany	Germany	Poland	Poland	Denmark	France	Spain	Greece	Italy	Finland	Slovakia	Hungary	Belgium	Belgium
Quest number		3A	4A	4B	4C	4D	5A	5B	6A	9A	11A	12A	18A	20A	23A	24A	25A	25B
RAM available?	Yes, Official, Development, Institute	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, D	Y, I	No	Y, O	No	Y, I	Y, I	No	Y, D
STU	Soil typological unit	X	X	X				X	X	X	X	X			X	X	X	X
Land use	e.g. LUCas	X		X					X	X		X		X	X	X	X	X
Equipment use	Weight, Wheel Load, Inflation Pressure , Tyre type	WL, IP	W, WL, IP, Ty	W, WL, IP, Ty			IP		W, WL, IP, Ty	W, WL, IP, Tyre	WL, IP	W		W, WL, IP				
Land cover	e.g. Corine	X							X	X							X	X
Topography	Digital elevation model			X							X	X						
PTF	Pedotransfer functions used:	X	X		X	X		X		X	X						X	
PTF +	Land Cover, Land Use, Spatial Soil Info, Model, GIS	GIS, LC, SSI, Model	GIS	Model + GIS	GIS	LC		Model + GIS	Model + GIS	Model, GIS	Model, GIS	LC, LU					GIS + SSI	
Texture		X	X	X	X	X		X	X	X	X	X		X	X	X	X	X
OM		X		X	X	X		X	X	X	X	X		X		X	X	
Density	Bulkdensity dry, Bulkdensity at fc, Packing Density, Porosity, Degree of	Bd, Bfc	Bd, PD	Bd, PD	Bd, PD	Bd, Bfc, PD	Bd, DegComp			Bd,	Bd, Bfc	Bd, Bfc, Por			Bd	Bd, Bfc		
Moisture	Field Capacity, Wilting Point, Water content sat, Workability Limit, Infiltration cap. sat	FC, WP, Wsat, WL, Ksat	FC, WP, Ksat	FC, WP, Ksat, Infil_sat	FC	FC, WP, Wsat, Ksat	FC, WP, Wsat, Ksat	FC, WP, Wsat, WorkL, Ksat	FC, pFcurve	FC, Wsat		FC, WP, Wsat		Ksat	FC, Wsat	FC, WorkL, Ksat		
Drainage class		X								X		X					X	X
Air	Air capacity, Air conductivity, Diffusion		Acap	Acap, Diff	Acap	Acap, Acond	Acap, Acond, Diff	Acap						Acond, Diff	Acap			
Mechanical	PreCompression stress, Shear Strength, Penetration resistance	PreC, ShearS	PreC, ShearS	PreC	PreC, ShearS	PreC	Pen	Pen, PreC	Pen, PreC	PreC, Compress-ibility	PreC	Pen				Pen		
Climate	Precipitation, Temperature, Radiation, Potential Evapotranspiration, yearly, seasonal, monthly, 10 days, daily	P-m, T-m		P-s		P-s, PE-s	P-y, P-s, T-j, T-s		R-d, PE-d	P-d, T-d, PE-s	PE-m	P-y, P-m, T m, R-y, R-m, PE-y, PE-s		R-10, PE-s		P-y, P-s, P-m, P-d, T-y, T-s	P-y	
Climate +	Land Cover, Land Use, Spatial Soil Info, Model, GIS	GIS, LC, SSI, Model				LC			GIS + Model	Model	Model, GIS	LC, LU						

Table 3.2. Thresholds and threshold values used in the risk assessment of compaction

THRESHOLDS																	
Country	Romania	Germany	Germany	Germany	Germany	Poland	Poland	Denmark	France	Spain	Greece	Italy	Finland	Slovakia	Hungary	Belgium	Belgium
Quest number	3A	4A	4B	4C	4D	5A	5B	6A	9A	11A	12A	18A	20A	23A	24A	25A	25B
RAM available?	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, I	Y, D	Y, I	No	Y, O	No		Y, I	No	Y, D
Water content	Workability limits					FC			X	Workability limits	X			35 vol%			
Saturated hydraulic conductivity		10 cm/d	10 cm/d		10cm/d								24 cm/d				
Air capacity		5 vol%	5 vol%		5 vol%	10 vol %								10 vol%			
Oxygen diffusion rate			1			<30 µg m^{-2} s^{-1}											
Penetrometer values						2-3 Mpa					X			2,8-6,0 MPa	2,8-3,0 MPa		X
Precompression stress	> Load	> Load	> Load					> Load	> Load	> Load							
Dry Bulk Density						1.4-1.5 Mg m^{-3}			X		X			1,35 - 1.7 g cm^{-3}	1,5 g cm^{-3}		
Bulk Density at Field Capacity					Klassen 4/ 5 (dicht/ sehr dicht)						X						
Packing Density		class 4 and 5 (DIN 19682-10, Germany)	X		Klassen 4/ 5 (dicht/ sehr dicht)	X											

3.3 Data processing and data interpretation

In a compaction risk assessment three factors affecting soil compaction should be considered: (1) climate, (2) soil and (3) soil and water management (Canarache, 1991, Van Ouwerkerk and Soane, 1994, Van den Akker, 2002).

Ad 1. Climate: The higher the rainfall and the lower the evapotranspiration, the higher the water infiltration capacity must be. Wet soils are weak and can be compacted easily. Shrinkage by drying is a very important mechanism for self-loosening and restructuring of soils. The same accounts for freezing, although the impact on subsoil compaction is disappointing (Håkansson and Petelkau, 1994). Soils that stay wet have a low resilience to compaction because they do not shrink and biological loosening activities are hindered.

Ad. 2 Soil: Strength, compactability and resilience depend on texture, clay mineralogy, humus content, soil moisture content, structure and soil fauna. Essential soil physical properties such as hydraulic conductivity, diffusivity of oxygen, penetration resistance for roots, air porosity are reduced by compaction. Soils are considered overcompacted if some of these soil physical properties are below certain threshold values. Compactability depends on texture, structure, density and moisture content, which determine the strength of a soil. Resilience of overcompacted soils depends on among others on shrinkage capacity, biological activity and loosening by tillage. The shrinkage capacity depends on clay and humus content. Biological activity depends strongly on humus content, structure, density and moisture content. Loosening by tillage can be very effective, however, destroys the structure, is detrimental for soil biota and makes the soil very susceptible to recompaction. Especially recompacted subsoils can have a degraded structure and poor soil physical properties (Dexter et al., 2004, Kooistra et al., 1984).

Ad 3. Soil and water management: A poor drainage situation or irrigation at the wrong moment can make the soil wet and in that way vulnerable for compaction. Soil and crop management is an important factor in the compaction risk assessment. Wheel load and tyre inflation pressure must be in agreement with the strength of the soil. In many cases this is not possible for the topsoil and in too many cases also the subsoil is compacted. The resilience and possibilities for loosening of the topsoil are good, however, this is not the case for the subsoil. Manuring and harvesting during wet periods can be very harmful. Ploughing with two wheels in the open furrow causes subsoil compaction. Intensive crop rotations with a high percentage of root crops that is harvested late in autumn (e.g. sugar beet) with heavy machinery is a high risk factor. On the other hand a high percentage crops such as cereals, that have the capability to improve the soil structure are beneficial. Also the ownership is important. Owned land is treated better than hired land.

Evaluation RAMs found in literature

Canarache (1987), Canarache et al., (2000) and Petelkau et al., (2000) included most of the factors mentioned in the preceding paragraph by using a semi-empirical approach in a compaction risk assessment for Romania respectively Eastern Germany. These assessments are based on many long term experiments, additional measurements of e.g. compactability and a lot of information in well developed databases and a good interpretation of the data including development of pedotransfer functions. The developed assessments are adjusted to the national situations and available data and are not directly suitable for other countries.

Horn et al., (2005), Simota et al., (2005) and Van den Akker (2004) used a more deterministic approach in compaction risk assessment by comparing calculated strengths of a series of soils with stresses excerted by a wheel load. A recent development of this family of compaction risk assessment methods is the Alcor model (www.microleis.com). Compaction risks at two moisture ratios were classified for topsoil and subsoil according the soil strength, maximum allowable wheel load/inflation pressure or the effect on soil physical properties. Pedotransfer functions including a structure classification based on mainly German measurements were used to calculate soil strengths. This assessment is used to construct maps from farm scale up to European scale (Horn et al., 2005). An advantage of this deterministic method is that is can be used in every country, although it is doubtful whether pedotransfer functions for German soils can be used in a country with a completely different climate and/or soils. Another disadvantage is that the used pedotransfer functions are not all based on readily available data.

Jones et al., (2003) used readily available data from the European Soil Database and climatic data stored in the agrometeorological database of the MARS project. This data was combined with a classification of vulnerability based on expert judgment derived in profile pit observations on a wide range of soils in mainly intensive farmed areas. The analyses resulted in a provisional map of inherent susceptibility of subsoils in Europe to compaction. Jones et al., (2003) concluded that better and actual climatic data and quantitative results of soil mechanical research should be incorporated in their approach. Another large scale approach to assess mainly the existence but also the risk for compaction concerning Eastern Europe is the SOVEUR project (Nachtergaele et al., 2002). In this project experts of the respective countries mainly base the assessment approach on expert judgment. This expert judgment can be based on sophisticated methods as indicated above, but also on subjective impressions. This possible difference in quality of the assessment is a weak point in this method.

It can be concluded that most compaction risk assessment methods are mainly based on a determination of the vulnerability of soil for compaction and only partly include other factors as climate and water- and soil use management. Also the resilience of soil for compaction is only partly included. In this respect, semi-empirical methods have the advantage that they inherently include these aspects. However, their disadvantage is that they are rather region dependent and are based on experiences in the (recent) past. The consequence is the use of these semi-empirical methods is doubtful in case changes in land management or climate occur.

Deterministic RAMs

Most RAMs presented have mainly a deterministic approach. Note that not all needed information used in the institutional RAMs in Germany, Denmark, France, Romania, Spain and the proposed RAM in Finland is the same, although all are based on more or less the same deterministic approach (Horn et al., 2005, Simota et al., 2005). The Alcor model (http://www.irnase.csic.es/users/microleis/microlei/manual1/alcor/alcor3.htm) and the SIDASS model (Horn et al., 2005, Simota et al., 2005) are the latest versions of this family of RAMs based on a deterministic approach. Of these two the SIDASS model is the most complete one considering wheel loads, strength of the subsoil, climatic conditions, drainage conditions, land cover and soil properties, and using GIS and databases to derive maps presenting risk areas for compaction. An usual way to show which areas are at risk for subsoil compaction is to present the strength of the subsoil. The stronger the subsoil is, the lower the risk of subsoil compaction. An example of such a presentation is shown in Figure 3.3

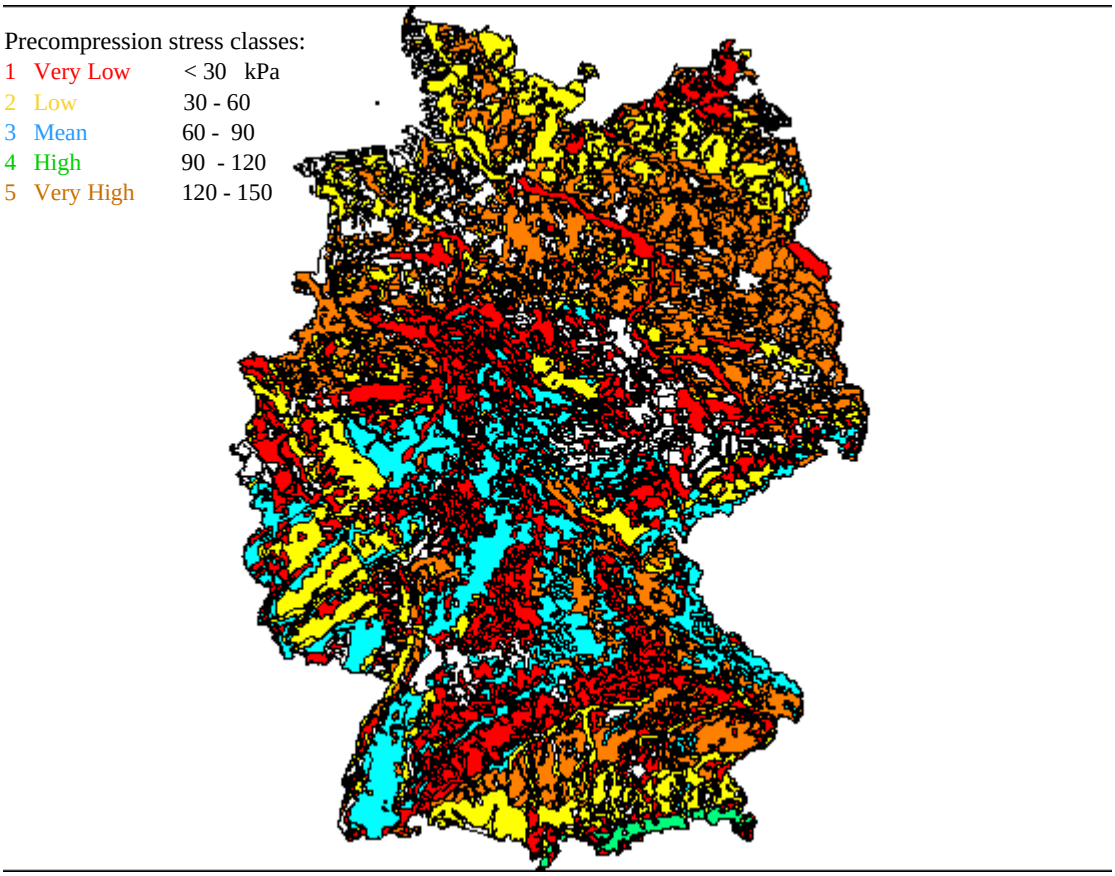


Figure 3.3 Example of the use of the SISASS model for Germany. The strength of a wet subsoil (soil water suction 6 kPa) is expressed as precompression stress.

Next step in the procedure is to compare the strength of the subsoil with the stresses on the subsoil exerted by a wheel load with a certain tire with a certain inflation pressure. In this way for a particular combination of tire and tire inflation pressure the maximum wheel load that just does not compact the subsoil can be calculated for each subsoil in a country. This has been done by e.g. Van den Akker (2004) for the Netherlands. The concept that the stresses in the subsoil should not exceed the strength of that subsoil means that no additional compaction is allowed, even if some compaction is not harmful to the most important soil qualities. Also the resilience of soil to compaction, so the natural regeneration of soil qualities by among others shrinkage, soil biota and rooting, is neglected. On the other hand an already highly compacted subsoil will be much stronger than a not overcompacted subsoil with a still satisfactory soil structure and soil (physical) properties. In fact this would mean that the farmer that created highly overcompacted subsoil is “rewarded” because now he can use much higher wheel loads than his neighbor with a healthy, however, weaker subsoil. The neglecting of the resilience of subsoils and the “rewarding” of the farmer with overcompacted subsoils was reason for Lebert et al. (2007) to propose a procedure in addition to the calculation of maximum allowable wheel loads, in which the soil qualities are checked of soils that are overloaded and compacted. Beside the soil physical soil qualities presented in column 4A in Table 2 also the visual inspection of the soil structure and the determination of the packing density in this way is part of the procedure. In the French RAM in development also the increase in bulk density by compaction will be calculated and compared with threshold values (G. Richard, personal communication 2007). These threshold values for the dry bulk density are in development and will be calibrated with the aid of a monitoring system in development in which several soil

physical properties are measured and also soil biodiversity is monitored. Together with available data this monitoring will produce relations between increased bulk density and important soil physical such as saturated hydraulic conductivity and air capacity.
Altogether the latest RAMs and the French RAM are very complete considering the cause of subsoil compaction, the reaction of the soil, the influence of climatic conditions and drainage conditions, the impact on important soil physical properties and to a certain extent the resilience of subsoils to compaction. A very weak point of the existing RAMs is that there has been no good validation of them up to now. The calculated allowable maximum wheel loads are in general rather low, and are in many cases much lower than the wheel loads used in praxis nowadays. For sure several subsoils are overloaded and overcompacted with degraded soil qualities, however, on the other hand in many cases the harm done to the soil seems to be acceptable or not noticeable. Probably the strength of subsoils is underestimated, because in general the static strength of the soil is measured and used in the calculations, while a wheel load is dynamic. In most cases the dynamic strength of a material is higher than the static strength. Another weak point is the lack of data on soil strength. Up to now the major part of data on strength of agricultural soils is collected by the research group of Rainer Horn in Kiel, Germany, and they mainly measured the strength properties of German soils. Positive developments are an increasing amount of measurements of soil strength in many European countries and an increasing amount of countries using and developing RAMs based on more or less the same deterministic approach.

The Italian RAM
The Italian RAM is evaluated separately because it is the only official accepted RAM and it only considers the cause of compaction. This means that the soil is not considered at all. The RAM is described at:
http://www.apat.gov.it/site/_contentfiles/00140000/140076_Annuario_2004_Versione2.pdf

The risk of compaction is evaluated by using a proxy indicator derived from the number and power of tractors (and harvesting machinery) and number of passes on agricultural soil. The indicator is calculated as follows:

$$Sp = kW*P*N*5 / S$$

Where:
Sp = Sum of weights
kW= kilowatt
P = average weight of the machinery = 102 kg·kW (Assuming a linear increase of the weight vs. power : 1kW=102 kg)
N = number of tractors and harvesting machinery
5 =mean number of passes on the field per year
S =hectares of arable land and orchards

The data on machineries is derived from official sources (ISTAT database). Results are comparable in time and space due to the homogeneity of the data sources. The indicator Sp is calculated for each region in Italy. The results Sp are represented over 8 classes (from values in the range 1 - 5, to high values > 141.
The strong point of this RAM is it simplicity and the availability of data. A very weak point is that the RAM completely ignores the impact on the soil. It is also not clear what the limiting value for Sp is and how such a limit can be determined.

RAMs based on measurements and experience.
The Polish RAM presented in column 5A of the Tables 3.1 and 3.2 is based on the determination of the degree of compactness of a soil. In short: the actual bulk density of a soil is compared with the bulk density of the same loosened soil artificially compacted by a pressure of 1 bar (100 kPa). By measuring also several important soil physical properties of the artificially compacted soil and comparing these with threshold values, it is possible to derive a value for the degree of compactness which implies a soil status with good or reasonable soil physical qualities (Lipiec et al., 1991, Lipiec and Håkansson, 2000, Håkansson and Lipiec, 2000, Lipiec and Hatano, 2003). The method is developed for topsoils and it is not clear and probably doubtful whether this method can be used for subsoils, because subsoils are generally not loosened. The method is only used on parcels and not to determine risk areas. These aspects make this RAM less useful for the determination of risk areas.

The second Polish RAM presented in column 5A of the Tables 3.1 and 3.2 is an institutional RAM in development and information is only in Polish available. The outcome of this RAM will be a compaction risk map 1 : 100 000 of Poland.

It was difficult to get a good impression of the RAM of the Slovak Republic, because beside the information in the questionnaire only some general information could be derived via a website (www.vupu.sk). Probably the RAM is based on a soil monitoring since 1993. This can be a good basis for the identification of already compacted subsoils and subsoils at risk. The RAM is expert based and requires analyses by an expert. These aspects make that this RAM can only be used in the Slovak Republic although several aspects, such as methods used and some threshold values, can be rather universal. An evaluation of e.g. the effect of changing climate or land use management is difficult because the RAM is mainly based on experience.

The RAM of Hungary is better documented (Birkás et al., 2000 and Birkás et al., 2004). The RAM is based on the evaluation of a soil monitoring since 1976 of parcels in arable use throughout Hungary. In this way a lot of experience is gained about the vulnerability and the actual compaction of Hungarian topsoils and subsoils. This RAM is comparable with the RAM of the Slovak Republik and has the same advantages and disadvantages.

The Flemish Belgian RAM (column 25B in Tables 3.1 and 3.2) is in development and no documentation is available yet. The idea up to now is that it will be mainly based on an inventory of the existence of compacted Flemish soils based on an assessment of the penetrometer resistance of representative soils followed by an evaluation.

The results of the questionnaires derived from Belgium (Wallonia, column 25A) and Greece are not further evaluated because they described desired RAMs and were lacking additional documentation.

3.4 Risk perception

The inventory of RAMs for soil compaction shows that the risk perception is not well defined and in fact is in most cases limited to the vulnerability to compaction. In the RAMs based on experience, expert judgment and measurements, the risk perception is focused on the effect on crop production. There is a general feeling that compaction will result in limitation of the infiltration capacity and storage of water in the soil and so will increased the risk of flooding, erosion and transport of nutrients and (agro-)chemicals to open waters, however, this risk

perception is mainly qualitative and not quantitative. Most RAMs based on experience address mainly topsoil compaction and in a lesser extent subsoil compaction, because severe topsoil compaction has more effect on crop production than subsoil compaction. The focus of the traditional (experimental) RAMs on compaction on crop production, means also that up to now compaction is mainly considered as a problem for the farmer and to a lesser extent as a problem for society. Moreover it is thought that the general interest of society to protect soil as a natural resource for the economic viable production of good food coincidences with the individual interest of the farmer. So it is generally believed that the farmer will take care of the soil and will produce food in a sustainable way. To a great extent this will be true, however, an economic viable agriculture does not mean by definition a sustainable agriculture: agriculture with a non-sustainable use of the soil can be economically profitable in a short- or mid-term time horizon, without being profitable in the long-term. Moreover, the interest of society in other aspects than food production, such as high infiltration capacity, biodiversity, prevention of erosion, storage of water, protection of open waters and groundwater, etc do not always coincidence with the (economical) interest of the individual farmer. The growing interest of society in these environmental aspects is in fact the basis for including soil compaction in the European Soil Strategy. This societal interest in the environmental aspects will require a focus of the risk perception, not only on soil as a natural resource for food production, but also on sustainability and environmental aspects. This also requires much more focus on potential (future) risk due to driving forces like climate change, land use and management, agro-ecosystem management and human population dynamics.

3.5 Options for harmonization

Most countries in the EU are using or developing a RAM based on a more or less similar deterministic approach. The most developed RAMs are very complete and include the cause of subsoil compaction, the reaction of the soil, the influence of climatic conditions and drainage conditions, the impact on important soil physical properties and to a certain extent the resilience of subsoils to compaction. The deterministic basis of these RAMs makes it easy to use them in GIS applications and to make use of soil data bases and climate data bases. Harmonization throughout Europe is in one way rather easy, because the structure of the RAMs is in essence the same. However, probably the harmonization of the data and measurement methods and interpretation of measurement results will be much more difficult. A major problem is that none of the RAMs is validated and that input data is lacking. Nevertheless a harmonized RAM based on a deterministic approach will be easier accepted EU wide and easier developed than a RAM based on experience.

The only official RAM from Italy does not consider soil at all and will probably not acceptable for the soil scientific community in the EU.

National RAMs based on experience and in most cases on large and long lasting monitoring systems are probably very useful for the determination of risk areas in that particular country, however, can not be used in other countries. A problem is also that these traditional experimental are focusing on the risk of decreased crop production, while the effect on environmental aspects is mainly neglected, while society and the European Soil Strategy requires ever more a focus on these environmental aspects.

A major problem is that the modern, deterministic RAMs mainly focus on the risk that a soil will compact and in this way these RAMs only address in an <u>indirect</u> way the risk that this

compaction leads to decreased crop production and a detrimental impact on environmental aspects such as flooding, erosion and biodiversity. However, on the other hand the deterministic approach of these RAMs is a very good basis for further development into the direction of RAMs that indeed include risk perceptions concerning the effect of compaction on crop production and environment.

4. Soil salinisation

E. Bloem, S.E.A.T.M. van der Zee, T. Tóth and A. Hagyó

4.1 Notion of threat, definitions and terminology

Soil (and groundwater) salinity is often used as a comprehensive term to refer to several different salinity forms. These forms are known under the names of, respectively, (1) saline soil, that have elevated salt concentrations, (2) sodic (or alkali) soil, with a disturbed monovalent/divalent cation ratio in favour of the monovalent alkali cations (Na, K), and (3) alkaline soil, for which the chemical composition is disturbed towards alkaline (high pH) compositions and often due to a dominance of (bi)carbonate anions in solution. These three salinity issues may be related, but this needs not be the case (Bolt and Bruggenwert, 1976).

The adverse consequences of salinity generally vary, depending on which form of salinity occurs. For saline soil, the impeded plant transpiration due to large osmotic values of soil water (Koorevaar et al., 1983) that render soil water poorly available for plants often dominates, whereas for sodic soil, the structural degradation caused by too large concentrations of sodium (Na) is generally most important. For alkaline soil, toxicity and deficiency effects due to altered plant availability of elements is the main problem, although such effects have also been observed for saline and sodic soils.

Of the three salinity forms, saline soils can be regarded as rapidly developing and easily cured whereas sodic soils develop slowly but may be very difficult to remediate. Alkaline soils are left out of consideration in this chapter as a sort of soil pollution case. For brevity, we do not discuss in detail the various involved processes of formation and remediation, but refer to handbooks (Bolt, 1982, Bresler et al., 1982).

Soil salinity, in its various manifestations, is an old problem in (semi)arid regions and has therefore become the focus of attention much earlier than (other) soil pollution problems. A major milestone towards managing salinity is the famous Handbook 60 (Richards et al., 1954), that compiled the hazards, measurable soil characteristics, and approaches for sustainable management of salinity. This handbook was both timely and appropriate for conditions where good laboratory facilities were limited or scarce, and has in this way become the reference worldwide.

Soil salinity is a widespread problem worldwide. Besides many semi-arid countries in irrigated regions, regions with shallow groundwater, and low-lying coastal areas and deltas (Salama et al., 1999) are confronted with it. An impression for Europe is given in Figure 4.1. It is only an impression as e.g. coastal area of The Netherlands that are troubled by salinity do not appear so on the map. Older maps such as provided by Szabolcs (1981) may over-estimate the problem. Within the EU, the major areas with SAS are found in various regions of the Iberic Peninsula, Italy and Greece, and Hungary and Romania.

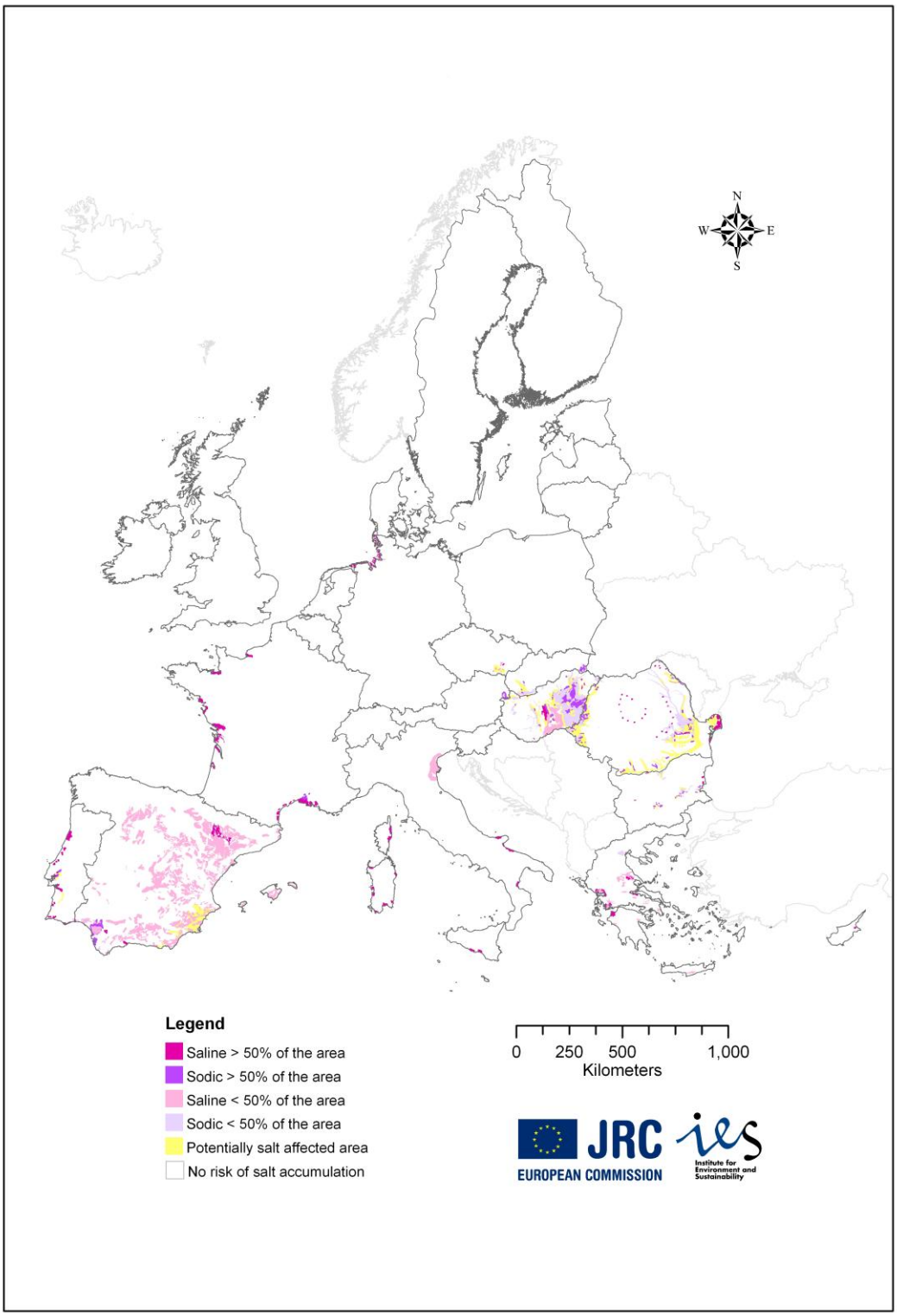


Figure 4.1. Map of saline and sodic soils in Europe according to Tóth et al., 2008.

The distribution of saline and sodic soils is shown by a recently compiled map of Tóth et al., 2008. Fig 1 shows that this maps is based largely on the 35 years old map of "Salt-affected soils in Europe" developed in the collaboration of FAO and the Subcommisson A of the International Union of Soil Science under the leadership of I Szabolcs, and on current databases.

4.2 Data collection

Salinity is quantified by a number of variables, that to some degree may be related. For instance, the concentration of ions in the soil or groundwater solution has been quantified by the molar (total) concentration (mol/L) which represents all cations and anions in solution, the electrical conductivity of solution (dS/m), and TDS or Total Dissolved Solids (mg/L). Such variables are related (Richards et al., 1954) and hence, we assume they are different operational quantities of the same property and in fact an early form of salinity-related harmonization. They differ usually for practical reasons (ease of measurement in field conditions). Total concentration, as we refer to it, represents the osmotic aspects of water availability for the biosphere, as ion-specific aspects are left out of consideration.

Besides non-specific aspects, also ion-specific parameters are of importance. Related to sodicity, the ratio of monovalent (Na, K) over divalent (Ca, Mg) cations is of importance. The underlying reason is that monovalent cations are less able to counter the negative electric field of clay colloids in soil and if dominant, make soils more susceptible for unrestricted swelling and shrinking. This behavior is causing soil structure deterioration, which is often poorly reversible. The major data pertaining to sodicity are, respectively, the Sodium Adsorption Ratio, SAR, and the Exchangeable Sodium Percentage, ESP, defined as:

$$SAR=[Cat+]/\sqrt{[Cat2+]}; \quad ESP=100\% \times \gamma(Cat+)/CEC$$

Where [Cat+] refers to monovalent cation concentration in solution (usually in $mmol_c/m^3$), γ refers to the adsorbed species in brackets (here monovalent cations), and CEC to the cation exchange complex.

Besides monovalent and divalent cations, also the composition of the anionic part is of importance, as the tendency of various cations to form insoluble salts differs for each cation. The chemical interactions between cations and anions determines which of the cations dominate (divalent cations are more susceptible to form insoluble salts) and the resulting pH-values that develop (alkalinity). Chemical analyses to determine with more detail the composition of the soil solution and the soil exchange complex are quite standard in soil chemistry (Bolt 1976).

Other data that need to be collected differ, depending on the complexity of the RAM. For sodicity, the mineralogy is of importance, whereas for salinity that is much less the case. The underlying reason for the various data are the factors that affect salt hazards:

- Sources and quality of rainfall and irrigation water
- The evaportranspiration demand of crops and vegetation
- The quality and proximity to the soil surface of ground water
- Soil textural and mineral composition
- Temporal and seasonal variations in soil dessication
- Managed or natural leaching of salts towards drainage infrastructure or groundwater.

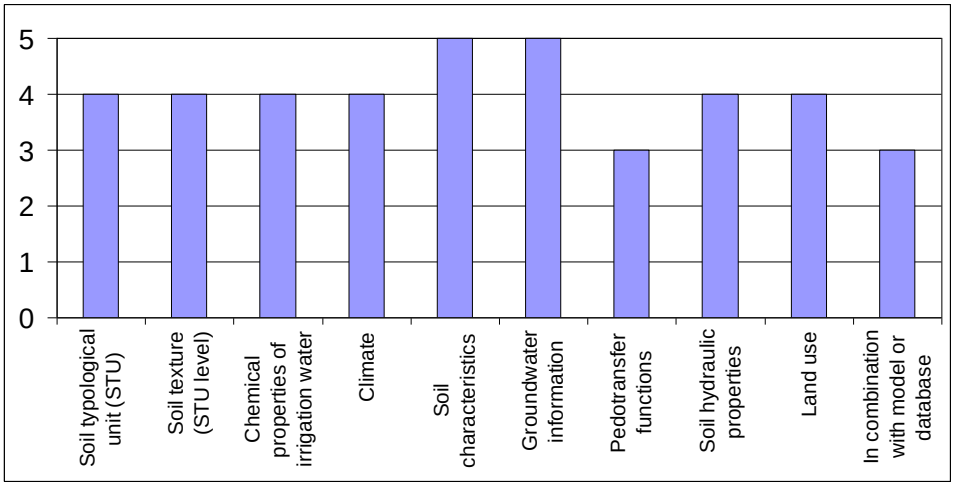


Figure 4.2. Common data used by risk assessments for salinity problems in 5 EU countries.

The return rate of the questionnaires was 21% for salinization. This relatively low response may reflect the fact that salinization is a regional and local phenomenon in Europe, related to poor drainage and seasonally dry weather conditions. Salinization is most severe in Hungary and only Hungary and the Czech Republic have an official assessment methodology. RAMs for salinization mainly differed in the indicators used to evaluate the risk, which is in part related to the specific objective of the RAM.

All RAMs (of 5 questionnaires returned in this project) use soil characteristics and groundwater information in their assessment (Figure 4.2). Soil typological, soil texture, chemical properties of irrigation water, climate, soil hydraulic properties, and land use are used in 80 % of the RAMs and pedotransfer function and combinations with models are used in 60 % of the RAMs. Case studies prevail although Hungary and Slovakia have RAMs which are used at the national or regional scales. Four of the five countries use field observations in combination with laboratory analysis. Two of them use also GIS and Slovakia is the only country with a different approach, they use remote sensing.

4.3 Data processing

The information provided reveal that all RAMs are based on quantitative based methods (measurement of water and soil properties) except for Slovakia. Usually, a combination of methodologies (Figure 4.3) is applied that is partly quantitative and partly qualitative, e.g. quantitative expert analysis or process based modelling.

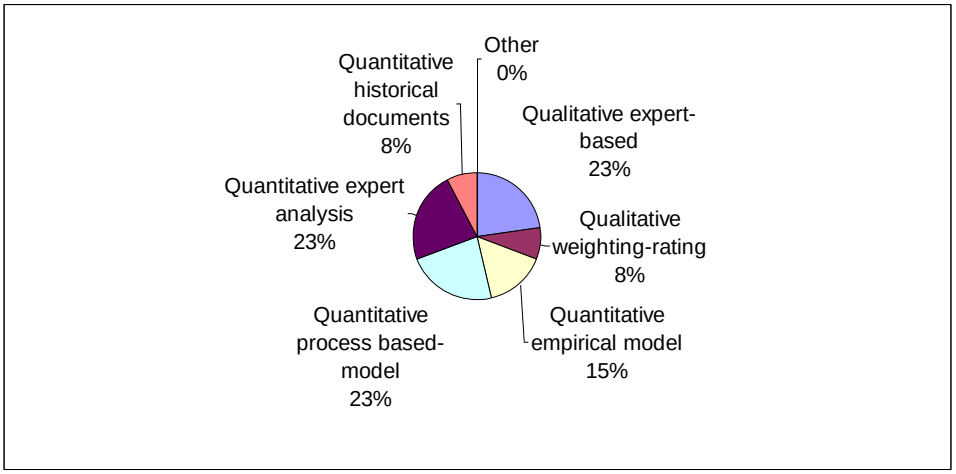


Figure 4.3. Type of methodology

The most common way to present results is by producing maps. Three countries have only one output document. Greece has a vulnerability map, Spain a risk map and Slovakia elements at risk map. Cyprus has an output of risk and vulnerability mapping, and Hungary has an output of risk, vulnerability, and hazard mapping. Risk and vulnerability zone mapping are both used for 30 %; other output maps are less used.

4.4 Data interpretation

Commonly, a first step in risk assessment involves a general identification of the threat and areas at risk, derived from existing data. The used data are mainly the factors given in part 4.2, in particular the determined soil and groundwater salinity, irrigation water quality and quantity, and the ionic composition of the different water sources. In the terminology of Eckelmann et al, (2006) this is the second stage of a tiered approach. The first approach identified by Eckelmann et al., (2006), i.e., expert knowledge, is generally an integral part of the interpretation of the mentioned data, except for sodicity which is a more complicated feature that is understood less broadly in a geographic sense (awareness is more restricted to countries where sodicity is a recognized problem). To some degree, also the third, model approach is used but as a national risk assessment tool, models are usually still unproven technology.

Because of their relevance for the first approaches of risk assessment, we can identify the following simple, experimental RAMs: salt concentration, making use of the 1:1 relationship between salinity concentration and electrical conductivity EC (in mS/cm) often quantified as the latter. To this purpose, the EC is classified in different classes with regard to salinity hazard. An important classification is that of USDA Salinity Laboratory (Richards, 1954; Table 4.1), that is still commonly used.

Table 4.1. Classification of the electrical conductivity or irrigation water (EC_{iw}) with regard to the hazard of adverse salinity effects

EC_{iw} (mS/cm)	**SALINITY HAZARD**
0-0.25	Low; water use is safe
0.25-0.75	Medium; water quality is marginal
0.75-2.25	High; water unsuitable for use
>2.25	Very high

An experimental RAM that directly refers to whether a soil must be considered to be saline or not, is based on the electrical conductivity of the saturated soil paste. The procedure is comparable to the RAM 1 approach, but involves a soil paste at water-lubrication level and intrinsically involves an expert judgement regarding crop vulnerability.

Table 4.2. Classification of electrical conductivity of soil saturation extract (EC_e) with regard to salinity effects on crops (Richards, 1954)

EC_e (mS/cm)	**Class**	**Effect**
0-2	Non saline	Negligible
2-4	Mildly saline	Yield reduction of sensitive crops
4-8	Medium saline	Yield reduction for many crops
8-12	Very saline	Normal yields for salt tolerant crops only
>16	Extremely saline	Reasonable crop yield for very tolerant crops only

It has been recognized early that crops have a different vulnerability for soil salinity. For this purpose, the RAM 2 has been related to a classification for different crops.

Table 4.3. Vulnerability of different crops for salt damage.

ECe (mS/cm)	**CROP**
2-4	Clover
3-4	Bean, sellery, radish
4-10	Flax, maize/corn, oats, wheat, rye, cucumber, peas, onions, carrots, potato, lettuce, cauliflower, cabbage, tomato
10-12	Spinach, asparagus, cabbage flower, red beet
10-16	Rape, sugar beet, barley

Whereas the above refers to elevated salt concentrations, the basis for risk assessment (experimentally) of sodicity is ESP, $ESP = \frac{\gamma_{Na}}{\gamma_T}.100\%$, where γ refers to the exchangeable quantity of cations (subscripts are *Na* for sodium, and *T* for the total cation exchange capacity). If ESP exceeds 15%, a soil is called sodic (in Australia, this is already the case if ESP exceeds 6%).

The USDA Soil Salinity Laboratory (Richards, 1954) has developed a widely adopted salinity classification system that considers the total salt level estimated from the electrical conductivity of the saturation extract (ECe), expressed in dS/cm at 25 degrees C temperature, and the exchangeable sodium present (ESP) or sodium adsorption ratio (SAR) to classify among saline, saline-alkaline and alkaline soils, and different degrees of them.

Table 4.4 Salinity/alkalinity/sodicity classification schemes (Richards, 1954)

Soil type	**Soil property**			
	SAR	**ESP**	**pH**	**ECe (mS/cm)**
Non saline, non alkaline	< 13	< 15	< 8.5	< 4
Saline	< 13	< 15	< 8.5	> 4
Alkaline	> 13	> 15	> 8.5	< 4
Saline - alkaline	> 13	> 15	> 8.5	> 4

The above scheme makes no distinction between ion types that enable to differentiate harmful from harmless salts, unlike the classification system based on anion types developed by Russian soil scientists (Plyusnin, 1964) (Table 4.5). In this approach, salt-affected soils are classified on the basis of salt types, in terms of chloride, sulphate and carbonate anion ratios present in the soil saturation extract. As not all salts are equally harmful, and so require different reclamation and management measures, it is of value to know the spatial distribution of salt-affected soils and their composition. The World Reference Base for Soil Resources also follows an approach based upon anion assemblages, distinguishing in Table 4.6 six facies of salt affected soils (Spaargaren, 1994). (Source: Metternicht, 2003)

Table 4.5. Harmful (above the line) and harmless (below the line) salts (Plyusnin, 1964)

$NaCl$	Na_2SO_4	Na_2CO_3	$NaHCO_3$
$MgCl_2$	$MgSO_4$	$MgCO_3$	$Mg(HCO_3)_2$
$CaCl_2$	$CaSO_4$	$CaCO_3$	$Ca(HCO3)_2$

Table 4.6. The World Reference Base for Soil Resources salinity approach upon anion assemblages (Spaargaren, 1994)

Soil type	**Facie**	**Characteristics**
Chloride soils	Acid chloride soils	Cl >> SO_4 > HCO_3, and Na >> Ca
	Neutral chloride-sulphate soils	Nearly neutral pH
Sulphate soils	Neutral sulphate soils	Nearly neutral pH, Na >> Ca, and SO_4 >> HCO_3 > Cl
	Acid sulphate soils	Very low pH (< 3.5)
Carbonate soils	Alkaline bicarbonate-sulphate soils	pH > 8.5, HCO_3 > SO_4 >> Cl, and Na > Ca
	Strongly alkaline soils	pH > 10, HCO_3 >> SO_4 >> Cl, and Na >> Ca

4.5 Risk perception

Whereas the quality of natural resources is important for a first assessment, modeling is usually required for the analysis of risks that are also dependent of management. In principle, it is quite easy to develop an equation that relate the RAMs called water resource salinity (EC) and soil salinity (ECe), and this relationship is commonly known as the Leaching Requirement (LR). Leaching requirement can be defined as the fraction of infiltrated water that must pass through the root zone to keep soil salinity from exceeding levels that would significantly reduce crop yield under steady-state conditions with associated good management and uniformity of leaching.

This concept can be formulated in terms of easily measurable properties (Richards, 1954, Rhoades, 1974), such as the water content of soil at field capacity and in the saturated paste, which are quite robust measures. Hence, also LR is quite a robust RAM for soil salinization:

$$LR = \frac{D_{DW}}{D_{IW}} \approx \frac{w_{FC}}{w_{SP}} . \frac{EC_{IW}}{EC_e^*}$$

Where *D* denotes an amount of water (mm/year), *w* stands for water content by weight, and *EC* is the electrical conductivity. Subscript *DW*, *IW*, *FC*, and *SP* denote drainage water, irrigation water, field capacity of soil, and saturation extract, respectively. Finally, the asterisk denotes that the electrical conductivity of the saturated paste may not exceed this particular value. This LR concept has been fine-tuned to account for nonideal flow of water through soil and other causes for deviation from the relatively simple LR-approach (Corwin et al., 2007, Letey et al., 1985)

Despite its simplicity, the LR concept is a robust way to convince stakeholders of the need for drainage and has motivated research on drainage improvements (Bos, Ritzema, 2009). In some cases, for instance if more factors of interest need to be taken into account, the complexity of required modeling is higher and for that purpose, various models have been developed, though scarcely used in national/regional RAM. Strongly focussed towards salinity/sodicity type of problems is UNSATCHEM (Simunek et al., 1996). This advanced code has been used succesfully to understand both salinity and sodicity process dynamics at a very local scale (Kaledhonkar et al., 2001, Jalali et al, 2007). Advantage of this code is that boundary conditions can be variant in time, whereas flow and transport are both transient. As was mentioned, Corwin et al. (2007) considered the leaching requirement as defined above, with more complicated models such as WATSUIT, TETrans, and UNSATCHEM. They found that transient modeling with the mentioned three models may lead to leaching requirements that are smaller than the steady state LR concept given above. Obviously, the demands regarding computation, model parameterization, and expertise of the modelers are much larger than for applying the LR concept. However, to save water in semi-arid regions, such an effort may appear to be economically sound.

Alternative software, such as LEACHM, (Wagenet, and Hutson, 1989) PHREEQC (Parkhurst and Appelo, 1999) and HYDRUS (Simunek et al., 1998, in Bastinaanssen), are less focussed to soil salinity issues . A relatively new code, ORCHESTRA, is object oriented and has recently been extended to incorporate transient flow and transport (Meeussen et al., 2005, Van der Broek et al., 2008). However, it has not yet been applied to salinity or sodicity issues.

The limitations of above software are commonly the same: they are complex, much a priori knowledge is required from the modeler, and important feed backs have been ignored. Possibly for these reasons, a less formal but potentially more focussed and appropriate involvement of expert knowledge is favoured and sufficient. In a recent overview of the state-of-the-art of modeling (Bastiaanssen et al., 2007), a selection of deterministic models was described, categorized as Bucket, Richards equation, SVAT, multi-D, and crop production models. They also observe the high qualifications needed of the modelers, which may impede the use of such models for routine risk assessments. A serious gap is identified between model complexity and the demands for application by the irrigation and drainage community.

In practice, despite the availability of complex models, the limitation of RAM for sodicity seems to be the gradual, stealthy way of development of sodicity, which makes the problem owners (often farmers and local authorities with limited academic education) unaware and and not easily convincible for its hazards.

4.6 Options for harmonization

Different for other soil quality threats, salinity has been recognized and partly undergone harmonization as early as 1954. The basic properties to consider and even the way of measurement in the field or the laboratory and classification schemes have been developed and become accepted. Despite that the framework of Richards (1954) has been challenged in about every aspect, and improvements for the first tiers of Eckelmann et al.'s (2006) scheme are feasible, it can be debated whether further harmonization or perhaps even standardization is necessary and urgent. For instance, considering the way of measuring the ESP of soil, it can be shown that the involved errors of e.g. different measurement protocols leads to negligible errors.

Figure 4 shows that if the ESP is measured by extracting a certain mass of soil with a designated volume of extractant solution (as all measurements are based on exchange between a solid and a liquid phase) the measured ESP^* after adding the extracting water differs from the ESP that corresponds to the initial soil sample, that is usually air dry.

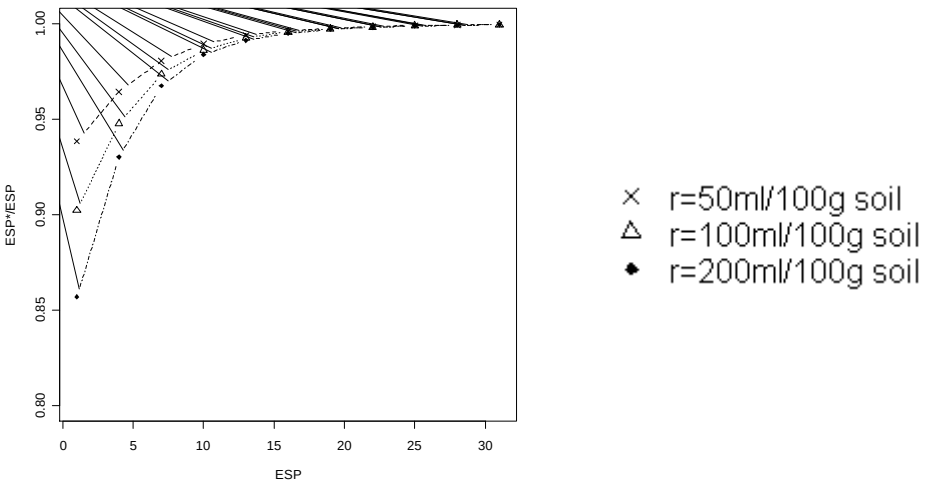


Figure 4.4. Relationship between the ESP/ESP where ESP* is the value after suspending soil in water and ESP is the value of the solid phase before suspension. Results for constant C_{tot}=0.01mmol$_c$/ml, CEC=30mmol$_c$/100g$_{soil}$ and w=25 ml/100g$_{soil}$*

It appears that the assessment of ESP for solution:solid ratios ranging from r=0.5-2, a systematic error may occur that ranges from neglible to 15% due to the shift in the solid/liquid equilibrium. This error is largest for relatively small ESP-values, where a correct assessment is the most important for a good anticipation of changes.

Obviously, the systematic bias, that can not be prevented completely if soil samples are stored under room-dry conditions, increase if the volume of water used for suspending the soil increases. For two relevant cases regarding initial ESP (before adding water), the bias is shown in Figure 4.2 to be limited to within 5-10%. Such a limited bias seems acceptable, as it will not affect the anticipation of the situation concerned and therefore we conclude that the liquid:solid ratio does not much affect our risk assessment.

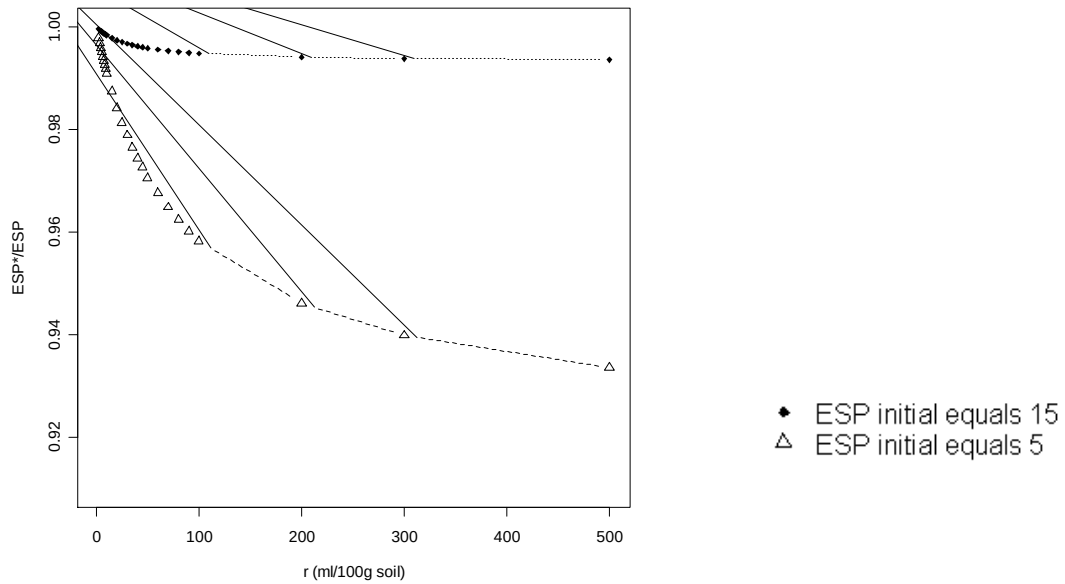


Figure 4.5. Relationship between ESP/ESP and r for different initial ESP keeping constant constant the amount of water w=20ml/100g$_{soil}$,and CEC=30mmol$_c$/100g$_{soil}$ and C$_{tot}$=0.01 mmol$_c$/ml*

In this research, we have varied several other parameters besides the solid: solution ratio, and found that systematic errors are limited to tens of %. Although clearly scientifically considered to be significant, it is unlikely that such biass would lead to a different anticipation of the situation (regarding hazards of sodicity). Hence, these results are not presented here, as for the present purpose of policy support they are not important enough.

For reasons given above, it can be concluded that *further* harmonization/standardization of present, predominantly experimental RAMs is not urgent. Whereas this may be the conclusion for the current, strongly experimentally inclined RAM of soil salinity, it is debatable whether it holds for further developments. Risk assessment in the GIS context, with more advanced numerical or analytical modeling is likely to be an unstoppable trend. Since this strongly model based larger-scale approach has received limited attention in the soil salinity context, is seems to have an excellent potential for harmonization before different authorities and scientific institutes have made their choices and have become less flexible.

So, logically, a further harmonization involves RAMs that use modeling as a dominant tool. The harmonization may involve different aspects, such as (i) the used model concepts, or even (ii) the used numerical software, (iii) the extent of data of diverse types to be integrated in modeling, (iv) the proper calibration and validation approaches, (v) scenario development, and (vi) more technological methods of 'good modeling practise' such as keeping a blog. Such aspects always have to be considered in a common sense context of gains and costs, which may affect the level of detail of model concepts to be considered, the available data, etcetera (Shah et al., 2009, Van der Zee et al., in pres). As is the case with other fields of decision making, we deal with growing knowledge and awareness, and for this reason, strong top-down forcing is less attractive than iterative learning processes by all stakeholders. Such a learning and development experience may significantly benefit from advances made in the dialogue regarding e.g. pesticide admission and evaluation policies, which dialogue is quite prominent in the EU. A similar dialogue between stakeholders (e.g. EU, different land use sectors, science, etcetera) with clear and increasingly focused Terms of References to avoid adverse effects of salinity, might be the best way of harmonization.

5. Soil organic matter decline

P.J. Kuikman, P.A.I. Ehlert, W.J. Chardon, C.L. van Beeek, G. Tóth and O. Oenema

5.1 Notion of threat, definition and terminology

Soils contain vast amounts of organic carbon (C). On a global scale, about 1500 Pg (1 Pg = 10^{15} g) is stored in the upper meter of the soil, which is about three times the amount of C in the aboveground biomass and twice the amount of C as CO_2 in the atmosphere (Batjes, 1996; Janzen, 2004). Most of soil C is found in the upper 10 to 20 cm of the soil, and the amount and quality of C in the topsoil is often used as indicator of soil quality and productivity (Allison, 1973; Bauer and Black, 1994; Davidson, 2000). In agriculture, increasing soil organic C (SOC) content is often seen as a desirable objective, especially in organic farming (Mader et al., 2002; Lovelock and Webb, 2003; Lal et al., 2004), though the benefits of organic C in soil in terms of fertility arise in part from its decay and not from its accumulation (Janzen; 2004; 2006). Sequestration of C in soils has also been promoted as strategy to mitigate the effects of increasing emissions of greenhouse gases in the atmosphere (Lal et al., 1998; 2001; Janzen, 2004).

Some recent studies suggest that SOC contents of European agricultural land is decreasing (Vleeshouwers and Verhagen, 2002; Sleutel et al., 2003; Bellamy et al., 2005). Such decreases are ascribed to changes in land use, soil cultivation and, possibly, climate change (Davidson and Janssens, 2006). Jones et al. (2005) calculated that 0.6% of soil carbon in European terrestrial ecosystems is lost annually. Farmers have concern that decreases in SOC compromises the production capacity of the soil by deterioration of soil physical properties and by impairment of nutrient cycling mechanisms (e.g., Loveland and Webb, 2003).

In EU-25, most soils are out of equilibrium as regards soil organic matter contents, as they have been affected by land management practices and land use (Smith et al., 2005). JRC-IES has compiled a soil organic matter map for Europe (Figure 5.1). Whether the information in the European soil map is up to date remains uncertain as very few national monitoring programs of soil organic matter exist and the 65 existing monitoring networks in the EU on soil quality are not set up to provide information for the soil map. Assessments of changes in soil organic matter suggest that in cropland soil carbon stocks in general continue to decline perhaps as a result of recent land use change or agricultural management, e.g. tillage practices or manure use (Smith et al., 2005; Vleeshouwers and Verhagen, 2002; Freibauer et al., 2004). The values for cropland soil carbon loss however are highly uncertain (Janssens et al., 2003).

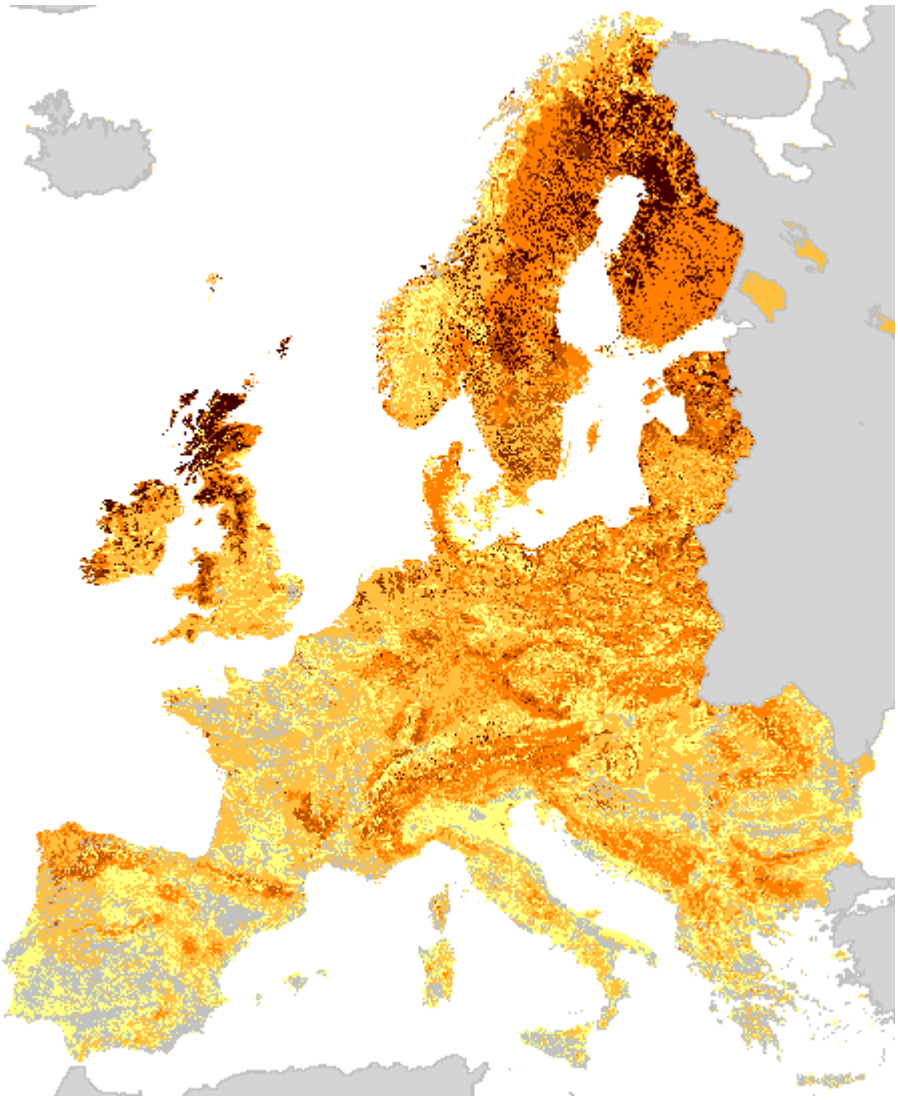


Figure 5.1. Soil organic matter in Europe (Jones et. al, 2005).

Considering the abovementioned the European Commission has identified soil organic matter decline as a threat for sustainable soil management. Soil organic matter decline is therefore one of the threats of the soil thematic strategy. Although no definition of soil organic matter decline is adopted in the communication on the directive COM(2006)231 it is generally defined as a decrease of soil organic matter contents in the topsoil over a given period of time. It thereby distinguishes from loss of soil organic matter stocks.

In this chapter an overview is given on currently used RAMs on soil organic matter decline. Currently, there are no official SOM decline RAMs in the EU.

5.2 Data collection

Most RAMs on SOM decline heavily depend on field measurements with a certain density and frequency. However, the field measurements often serve different purposes than determining

the SOM content, e.g. fertiliser recommendations and consequently methods of data collection are often not optimized for identifying SOM declines.

Sampling schemes differ between countries and even within a country. Several forms of sampling schemes can be distinguished; amongst others:

1. None systematic schemes for characterizing a soil type;
2. Systematic schemes;
3. Data collected from soil sample analysis for establishing fertilizer recommendations based on soil testing.
4. Soil chronosequences.

The use of different monitoring network is extensively revised in the ENVASSO project. They concluded that georeferenced monitoring is currently performed in Belgium (Wallonia) and Poland, but that the use of georeferenced soil profiles is not common throughout EU (Jones et al. 2004, 2005). Grid methods are much more common and are currently used and/or developed in Austria, France, Denmark, United Kingdom (England, Wales, and Scotland), Northern Ireland, Ireland, Germany, Hungary and Poland). These are national grids which were established for national soil surveys. Grids can be complete systematic or heterogeneous and based on a spatially irregular selection of sampling locations using expert judgement (Morvan et al., 2008). The differences in scale lead to differences in resolution. Within the ENVASSO project 65 monitoring networks were identified with in total 36104 locations where soil quality is measured. Of these 33334 locations provide information on SOM.

Table 5.1. Scales reported in questionnaires on sampling schemes of soil.

Country	Scale
Belgium (Flanders)	1:1,000,000
Belgium (Wallonia)	1:20,000, 1:25,0000
France	1:250,000 to 1:1,000,000
Greece	1:5,000
Poland	1:10,000
Slovak republic	1:400,000
Slovenia	1:10,000, 1:20,000, 1:25,000
Spain	1:50,000
United Kingdom	1:250,000

Soil sample analysis for establishing fertiliser recommendations provides large databases on soil characteristics. These samples are not send in by cooperation's, advisers or farmers for guidance on the use of fertilisers and therefore may be biased. However, they can be used to detect SOM changes, as e.g. in the case of The Netherlands (Reijneveld et al, 2009).

Soil chronosequences are genetically related suites of soils evolved under similar conditions of vegetation, topography, and climate (Harden, 1982). They translate spatial differences between soils into temporal differences (Huggett, 1998). Soil chronosequences (space for time substitutions with confounding of space and time) are instruments of pedological investigations. They have been used to determine carbon sequestration is soil and biomass following

afforestation (Vesterdal et al., 2007). As such it is a different technique in determining changes in a soil parameter such as SOM/SOC. Although a revisiting of sites can take place, often comparison of sites with comparable characteristics but differences in management (tree species, fertilisation, forest age) pinpoints chronosequences.

The period of assessment differs between countries that have an unofficial RAM. United Kingdom and Slovak Republic can assess a decline in SOM over a period of at least 15 years. The RAM of Belgium (Wallonia) is effective since four years while Spain and Poland just recently started the use of their RAM's. Other countries (Denmark, Germany, Netherlands) are currently investigating RAM's that can (or should) be used for assessing a decline in SOM.
The determination of SOM or SOC is a standard procedure. However despite the ubiquitous measurement there is no consensus on its definition (Carter, 2001). Discussion focuses on the fraction of organic matter that should or should not be included (fresh plant material versus decomposed organic matter, biomass or no biomass etc.).

SOM or SOC can be determined by destructive or non destructive methods. Depending on the nature of the method a quantitative of semi-quantitative result is obtained. Destructive methods use chemicals and/or heath to covert SOM or SOC in CO2. There are multiple methods. Titrimetric, gravimetric, volumetric, spectophotometric or chromatographic techniques are currently used for carbon quantification (Schumacher, 2002).
The determination of SOM or SOC contents is performed by either:
Semi quantitative:

- Loss on ignition (LOI), most often (but not always) corrected for inorganic carbonates and clay percentage.
- Peroxide destruction

Quantitative

- wet oxidation followed by titration (dichromate Walkey and Black, 1934)
- wet oxidation followed by measurement of CO2 evolution
- dry combustion and spectophotometric measurement (infrared) or thermal conductivity of evolved CO2

Within the ENVASSO project the different methods were compared (Spiegel, 2007; Hegymegi et al., 2007). The only comprehensive source of data using a standardized classification is the FAO database at 1:1000000 (Morvan et al., 2008). At smaller scale national datasets are available, but are often weakly embedded in structural monitoring schemes (Morvan et al., 2008).

5.3 Data processing

Data processing follows in general descriptive statistical analyses (means, medians, skewness and kurtosis) to enable a visualisation of possible trends. Descriptive statistical analyses are most often used but there is no general rule of thumb as design, sampling schemes and frequencies of sampling are not standardised. As data are often not normally distributed, non-parametric tests (example given Kruskal-Wallis) have to be used. Additionally, modelling approaches may be used. Models that can predict the decline in SOM (or SOC) are numerous. Within the SOMNET-framework 37 models have been identified of which 20 are developed by European research institutions (http://www.rothamsted.ac.uk/aen/somnet/intro.html). It is beyond the scope of this deliverable to discuss these models. The reader is referred to Smith et al, 1997; Pansu et al., 2007; Willigen et al., 2008.

Models differ in their description of SOM or SOC in soil, but all are dynamic equilibrium models with a number of carbon pools. Simple models describe one quality of SOM or SOC (mono component). More complex models divide SOM or SOC in pools with different quality (multi-component) to which different rates of decomposition, mineralisation, assimilation and alteration are ascribed (Willigen and Neeteson, 1985, McGill, 1996, Willigen, 1991, Diekkrüger et al, 1995, Kersebaum et al (2005), Falloon et al., 2006; Willigen et al, 2008).

Table 5.2. First inventory of datasets of SOC or SOM in cultivated agricultural land (arable land and grassland) and non- cultivated land for the assessments of a decline in SOM or SOC collected within the RAMSOIL framework.

Country	*Depth (cm)*	*Method*[1]	*Frequency*	*Spatial coverage*	*Reference*
Belgium	24		Annually[2]	21000 samples y^{-1}	
Belgium, Flanders	0-24 cm	WB (modified)	1990, 1993, 1996, 1999	190000	Sleutel et al., 2003
Belgium, Flanders	ploughlayer	WB	1952,1990, 2003	116 locations	Sleutel et al., 2006
Belgium, South	Variable 7 databases	Variable (LOI, DC)	1990, 2000	Variable (16-11977)	Lettens et al., 2005
Belgium, Wallonia, southern part	variable	WB, 4/3	1955 (1950-1970) resampled in 2005	295	Goidts and Wesemael, 2007
Finland	Ploughlayer (~0-20 cm)	WB (1974)/DC	1974, 1987, 1998	Farmplots (2000, 1320, 705)	Sippola & Yli Halla, 2005
Germany	40	WB	Irregular, 1983, 1989, 1998	Farmplots	Nieder & Richter, 2000
Germany	0-120 cm (8 soil profile layers)	WB (modified)/DC	1969, 1996	Farmplots	Rinklebe & Makeschin, 2002
Ireland	10 (grassland)	WB	1964 a second sampling 1995-1996	678/220	Zhang et al., 2004
Netherlands	5 (grassland) 20 or 25 (arable land)	SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	1984-2004 Intervals 4-5 years	2-50 ha	Reijneveld et al., (accepted)
Netherlands	5 (grassland) 20 (maize land)	LOI/DC SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	1984-2004 Intervals 4-5 years	2-50 ha	Hanegraaf et al., 2009
Netherlands	Variable	SOC≤12,%:KU (≤1994); DC (1994) DC (>1995) SOC>12.5%: LOI	Irregular[2]	2 -50 ha	Smit et al., 2007
Norway	Variable topsoil depth (1952)/0-20 cm.	Visual assessment (1952)/LOI	1952, 1976, 1986 and 2002	Farm, 25 ha	Riley & Bakkengard, 2006
Norway	0-25 cm	LOI	1991, 2001	291 Farmplots	Riley & Bakkengard, 2006
Sweden	0-25/25-60 cm(1956, 1984); 0-25, 25-35, 35-60 cm (2001)	WC (1956/1984 DC (2001)	1956, 1984, 2001	124 (1956), 65 (1984)124 (2001)	Kätterer et al., 2004
UK, England & Wales	0-15 cm	WB modified (%C<15 LOI (%C≥15)	1978-1983 first sampling; Second sampling 1994-1995 arable land; 1995-1996 grassland; 2003 non agricultural land	5661 (1st sampling); 853/971/535	Bellamy et al., 2005

[1] DC: dry combution followed by measuring CO_2, KU: Kumies, WB: Walkley & Black, LOI: Loss of ignition, [2] not each year at same place.

5.4 Data interpretation

Data interpretation of the decline of SOM or SOC uses a variety of factors. Soil typological unit (STU or soil type), soil texture (clay content), soil organic carbon (total and humus concentration) climate, topography and land cover. Table 5.3 gives an overview for these factors for the EU countries. Table 5.2 is based on the questionnaires.

Table 5.3. Factors used for interpretation a decline in SOM or SOC.

Country	Soil typological unit	Soil characteristics			Climate	Topography	Land cover
		texture/clay content	organic carbon (total and humus concentration)	organic carbon stock			
United Kingdom	x	x	x		x	x	x
France	x	x	x	x		x	x
Denmark	x	x	x	x	x	x	x
Spain	x	x	x	x	x	x	x
Greece	x	x	x		x	x	x
Slovenia	x	x	x		x	x	x
Finland	x	x	x	x	x	x	x
Slovakia	x	x	x	x		x	x
Belgium, Wallonia	x	x	x	x			x
Belgium, Flanders	x	x	x				x
German		x	x	x	x		x
Poland		x	x				

The soil map of each EU country provides information on baselines of SOM or SOC contents. The information of these soil maps have been integrated for a European soil map on the scale 1:1,000,000. There is no overall coverage of the EU. Estimates of soil properties have been derived by use of pedotransfer rules (PTR). The Soil Profile Analytical Database for Europe SPADE 1 and SPADE-2 contains (estimated) data on SOC in the topsoil (0-30 cm) for important soil types (Van Camp et al., 2004). SOC have been calculated using PTR and classified in 4 classes (table 5.4). These SOC classes and estimates for SOC can only be used on a continental level through there scale and derivation by use of PTR.

Comparable tables also exist for Slovakia and Greece (RAMSOIL report 2.5). The most common used threshold value for proper contents of soil organic matter is 2% SOC (~3.4 % SOM) and this value was also adopted by the EU as a starting value (Eckelmann et al., 2006). However, the first to come up with 2% were Greenland et al. (1975) and they developed this value as a threshold for soil structural stability, and not soil functioning, soil quality or other broader defined soil properties. Loveland and Webb (2003) made an extensive review on this –

now widely accepted- threshold value of 2% and found that much ‘evidence’ was based on qualitative or –even worse- anecdotal data.

Table 5.4. Classes of SOC of the toplayer 0-30 cm (Van Camp et al., 2004).

Class	SOC range
High	> 6.0%
Medium	2.1-6.0%
Low	1.1-2.0%
Very low	< 1.0%

5.5 Risk perception

There are within the 27 EU countries no official RAM’s on the soil threat on the decline of organic matter. RAM’s as such are – clearly – not recognized yet. Nevertheless the debate on climate change and on soil degradation and loss of soil quality initiated scientific studies to changes in SOM or SOC in time and factors that act upon these changes. These changes which can be positive, negative or zero (steady state) and are depending on the system (climate - soil - landuse – management).

Comprehensive and comparable data for EU27 on SOM or SOC content are not available. In stead, pedo tranfer functions are used to estimate geographical distributions of SOM contents, like the EU soil map of 1:1,000,000. Soil monitoring systems (SMN’s) are currently used or are implemented in two third of the EU-countries. Methodologies for assessing a change in SOM or SOC content or stock in the EU differ in datagathering, dataprocessing, data interpretation and risk perception. Differences are brought about by the national or region systems of soil mapping, scale, sampling method, physico-chemical analytical methods, dataprocessing and use of biophysical models. There is no uniform standarised accepted method for assessing changes in SOM or SOC contents and stocks yet but Eckelmann et al. (2006) and Stolbovoy et al (2005, 2006, 2007a, 2007b) proposed protocols for respectively datarequirement and data gathering and dataprocessing to arrive to comprehensive and comparable data on SOM or SOC contents and stock in European soils. However SMN’s are questioned as tools for measuring annual changes in C sinks in national accounting of greenhouse gasses (Saby et al, 2008).

Next, with soil biophysical models threshold values can be derived for impact factors and forecasts of trends in SOM or SOC can be made (Eckelmann et al., 2006). Thresholds will depend on the target: i.e. soil health, sustainable crop production, prevention of erosion, prevention of landslides. There is debate on these tresholdvalues for risk assessment and baseline values for identication of characteristic enties (example given a specific soil type).

Although the assessment methodologies are in general simple (resampling georeferenced soil profiles, soil fertility surveys, chronosequences), the scale and the intensity (resolution) and dataprocessing restrain SMN’s. Especially when high spatial and temporal variability lays a constrain on the assessment of a decline in SOM or SOC as this requires intensive and expensive research. The lack of referenced thresholds for specific effects of impact factors inhibits an (cost) efficiency development of SMN’s. A known threshold value can lead to a

more cost-efficient RAM through adaptation of sampling schemes, sampling size, sampling frequency and choice of the analytical method.

5.6 Options for harmonization

Biophysical models contribute greatly to the insight on the decline of SOM or SOC and are essential for deriving treshold values. There are still major challenges to combat. Current SOM or SOC contents partly reflect current land-use. When deriving threshold values or forecasting trends in SOM or SOC a distinction between the portion that is not related to the present land-use should be identified. Also, it is not clear how this portion is affected by current land-use. Complex models like CENTURY, Roth-C or CANDY may be used to quantify those portions for a variety of soil types, management and climate, through it's soundness and predictions. The high flexibility provides a sound basis for analyzing current conditions and future scenario's for a dynamic land use.
Past long-term experimental studies have shown that SOM or SOC is highly sensitive to changes in land use, with changes from native ecosystems such as forest to agricultural systems almost always resulting in a loss of SOC (Jenkinson 1977, Paul et al. 1997). Likewise, the way in which land is managed following land use change has also been shown to affect SOM or SOC contents and stocks. We therefore have the opportunity in the future to change to land use and land management strategies that lead to C storage in the soil, thereby mitigating effects of greenhouse gasses (GHGs) and improving soil fertility.

Jones et al. (2005) wrote 'There is an urgent need for harmonization of soil organic carbon monitoring networks'. This statement refers to the first step (data gathering) of the risk chain. From the results presented above, however, there seems to be as much an urgent need for the development of threshold values and the perception of risks. For data gathering and data processing we see little obstacles for harmonization and, as threshold values and risk perception, still largely need to be developed, harmonization can be easily introduced. Hence, for soil organic matter there are good options for harmonization, when the developments in threshold values and risk perception are properly coordinated (e.g. performed in EU actions).

6. Landslides

JP Malet and O. Maquaire

6.1 Notion of threat, definition and terminology

Landslides are classified according to their mechanisms (movement types) and the nature of the displaced material (material type), as well as information on their activity (state, distribution, style), ie.the rate of development over a period of time (Varnes, 1978; Dikau et al. 1996; Cruden and Varnes 1996). Five principal types of movements are distinguished according to the geomorphological classification proposed by Cruden and Varnes (1996) and Dikau et al. (1996).

1 Fall
A slope of movement for which the mass in motion travels most of the distance through the air, and includes free fall movement by leaps and bounds and rolling of fragments of material. A fall starts with the detachment of material from a steep slope along a surface in which little or no shear displacement takes place.

2 Topple
A slope movement that occurs due to forces that cause an over-turning moment about a pivot point below the centre of gravity of the slope. A topple is very similar to a fall in many aspects, but do not involve a complete separation at the base of the failure.

3 Lateral spreading
A slope movement characterized by the lateral extension of a more rigid mass over a deforming one of softer underlying material in which the controlling basal shear surface is often not well-defined.

4 Slide
A slope movement by which the material is displaced more or less coherently along a recognisable or less well-defined shear surface or band. Slide could be rotational (the sliding surface is curved) or translational (the sliding surface is more or less straight). In some cases a slide can change into a mudslide or slump-earthflow, especially on steep slopes, in highly tectonized clays or silty formations (Picarelli, 2001).

- Rotational slide: more or less rotational movement, about an axis that is parallel to the slope contours, involving shear displacement (sliding) along a concavely upward-curving failure surface, which isvisible or may reasonably be inferred' (Varnes, 1978).
- Translational slide: The material displaces along a planar or undulating surface of rupture, sliding out over the original ground surface.

5 Flows
A slope of movement characterized by internal differential movements that are distributed throughout the mass and in which the individual parcticles travel separately within the mass.
Debris flow and debris avalanche: Debris flow is a very rapid to extremely rapid flow (> 1 m.s-1) of saturated non-plastic debris in a steep channel. Characteristic of a debris flow of a debris flow is the presence of an established channel or regular confined path, unlike debris avalanches which are thin, partly or totally saturated and which occur on hillslopes (Hungr et al. 2001).

These five types may sometimes be combined or may succeed each other, forming a sixth type: a composite and complex movement, which consists of more than one type (e.g. a rotational-translational slide) or those where one type of failure develops into a second type (e.g. slumpearthflow).

The development of landslide RAMs followed always the occurrence of large landslide catastrophes:

In France, in 1970, the mudslide of Plateau d'Assy caused the death of 40 persons in Haute-Savoie (the most dramatic case of the century), strengthening in France the preoccupations in this domain and bringing the establishment of the first ZERMOS maps (*Zones Exposées à des Risques liés aux MOuvements du Sol et du sous-sol*; Besson, 2005).

In Italy, the catastrophic event of May 1998, which caused very large damage and deaths in the municipalities of Sarno and Quindici (Campania) urged the government to provide answers for development regulation. According to a decree named "the Sarno Decree" the government detailed legislative measures at the national level, including the procedure to define landslide risk areas (Bonnard et al., 2005).

In Switzerland, the flooding of 1987 obliged the federal authorities to update the criteria governing natural hazard protection. The "Federal Flood Protection Law" and the "Federal Forest Law" came into force in 1991. Their purpose is to protect the environment, human lives and property from the damage caused by water, mass movements, snow avalanches and forest fires (Raetzo et al., 2002).

Sweden also knew some large catastrophic landslides n the 1970s which have influenced the development of land use planning regulations. The Tuve landslide of 30 November 1977 engendered the death of 9 persons and 65 houses were ruined, 500 persons became homeless and fatalities occurred. Subsequent to this landslide the Ministry of Housing and Physical Planning commissioned the Swedish Geological Institute to carry out a survey of the unstable slopes. The government decided also that municipalities should be mapped generally regarding the presence of unstable slopes in built-up areas (Edwards, 2004).

In Spain, the disasters of Biescas (August, 1996) and of Alicante (September, 1997) conducted the Spanish Senate to exhort the constitutional court to recognize the necessity of including risk prevention measures to reduce the vulnerability of the slopes to natural risks.

Hence, at present 5 RAMs (French, Italian, Swiss, Swedish and Spanish) are used to estimate risks related to landslides.

6.2 Data collection and data processing

Data collection consists of 4 consecutive activities.

Risk prevention plans (*PPR: Plan de Prévention des Risques*) collects informative documents (a note of presentation, a localization map of the phenomena, a hazard map and some tatutory documents (risk zoning map at a scale of 1:10,000 or at 1:5,000 for the urban zones, and a regulation).

Inventory of processes
The RAM consists first in the elaboration of an informative map of the natural phenomena. It represents on a topographic map at 1:25,000, the observed and known phenomena inventoried from archives, aerial photographs and field work.

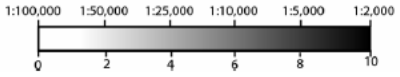
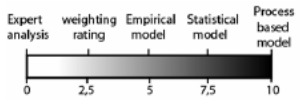
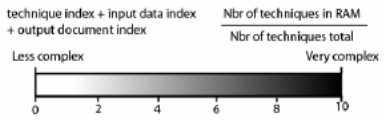
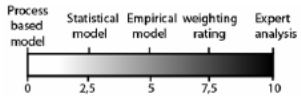
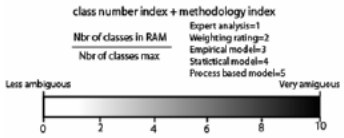
Hazard map
The hazard map is established by a forwardlooking approach where areas where any phenomena has been observed can be classified in hazard zone. The map is constructed through the combination of predisposing factors. The susceptibility of the site to landslide is estimated by a qualitative approach and is considered maximal where all the unfavourable factors (slope, lithology, …) are present.

Map of major asset
The inventory of the stakes consists in analyzing the landuse characteristics considering both the existent and the future developments. This analysis allows to identify the major assets such as establishments receiving public (hospital, schools, campsites, etc), strategic buildings (fireman's barracks, water drinkable tanks, etc), areas of major economic activities (industrial buildings, etc) as well as the communication capabilities (roads, railways, power roads, etc) which threatening may aggravate the risks during a major event. The cross-correlation of the hazard map and the map of major assets allows to identify qualitatively the main risk areas to be protected. The risk zoning consists in three risk classes and delineates zones in which prevention measures have to be taken.

6.3 Options for harmonisation

The 4 official landslide RAMs have been scientifically compared in terms of documentation, robustness, consistency, ambiguity, applicability and validity. The RAMs in development or the RAMs used by local research institutes or private engineering offices have not been analysed because some procedures were not totally detailed in the questionnaires, or were assumed to be modified. The RAMs have been compared according to 5 indicators (RAM Scale, RAM Transparency, RAM Complexity, RAM Cost efficiency and RAM Ambiguousness) which are defined in Table 6.1.

Table 6.1 Definition of indicators to compare RAMs.

Indicators	Definition	Coding value / indicator
Scale	This indicator is linked to the availability of documents and the scale of the maps to be produced.	
Transparency	It corresponds to the transparency of the human thought and so it depends of the experience of the expert in charge of the assessment. This indicator reveals the applicability of the methodology.	
Complexity	The complexity of the methodology is linked to the processing of the input data and the number of output information. The more input data are used, the more complex is the methodology.	
Cost efficiency	This indicator presents the profitability of the methodology in terms of means and costs to achieve the objective.	
Ambiguousness	This indicator represents the uncertainty in the delineation of hazard and risk zones.	

The indicators of the 4 official RAMs are represented with spider graphs. There are as many axes in the spider graph as indicators. Each indicator is coded through an index according to the maximal value observed for each indicator. The value of the index is evaluated according to our interpretation of the questionnaires.

Spider graphs were elaborated only for countries where an official risk assessment methodology is established for landslides. It seemed to us not pertinent to compare the RAMs in development because they may be subjected to some modifications or are not complete. Concerning methodologies employed by research institutes, they are most of the time developed for a specific objective, which sometimes is not for regulatory measures of risk zoning.

The shape of the spider graph are relatively similar, indicating that the RAMs are basically based on the same approach. They indicate however that the French and Italian RAMs are slightly more complex than the Sweden and Swiss RAMs. In European countries, several mapping scales can be used for risk assessment. In the official methodologies, the 1:25,000 scale or 1:10,000 scale are generally used: some local detailed zooms can also be mapped at a 1:5,000 scale. For example in France, a topographic map at 1:25,000 scale enlarged at a 1:10,000 scale and cadastral plans at 1:5,000 scale are used to delineate preventions areas.

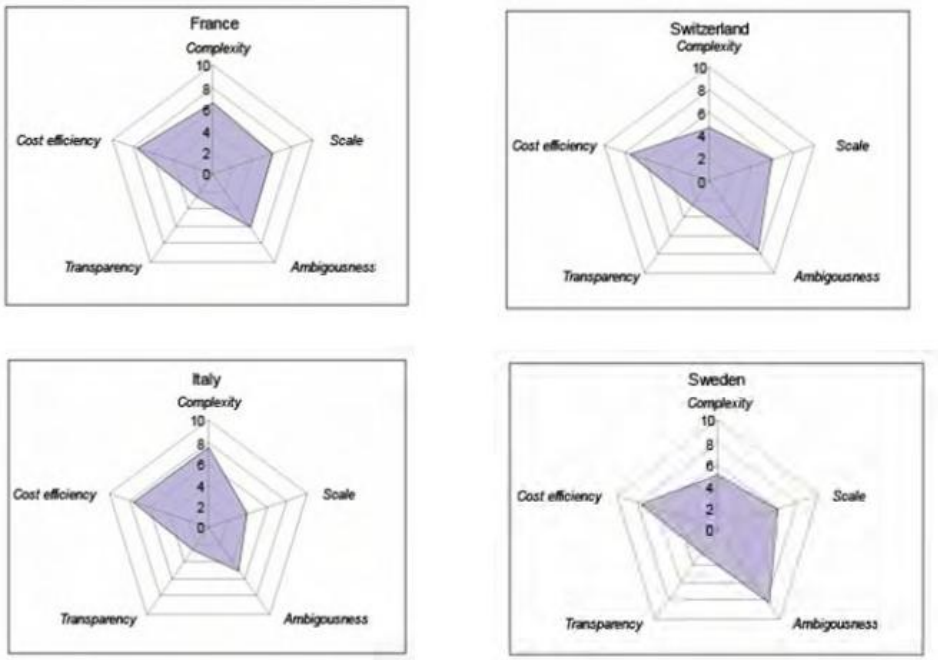


Figure 6.1 Spider graphs of the official landslide RAMs of France, Italy, Sweden and Switzerland.

The ambiguousness indicator reveals the precision of the method. All RAMs have mean values of ambiguity. The transparency indicator indicates low value of transparency because all RAMs are based on expert and heuristic analysis, and are thus subject to the thoughts of the experts.
All the RAMs present a high value of cost efficiency. For example, iIn France the elaboration of the PPR is very simple and is only based on available and existing data; the objective is to build documents taking into account the known risks rather than to focus on high accuracy.

Official landslide RAMs are similar. They are built on a qualitative approach, at medium scale and where necessary a more detailed local study can be realized with more complex (e.g. deterministic) techniques. They conduct all to plans of priority measures and thus lean on local urban plans. Their main advantages and disadvantages are detailed in Table 6.2.

Table 6.2 Main advantages and disadvantages of the official RAMs

Country	Methods	Advantages	Disadvantages
France Italy Sweden Switzerland	Field geomorphologic analysis	Allow a rapid assessment taking into account a large number of factors	Totally subjective methodology based on the use of implicit rules that hinder the critical analysis of the results
	Combination of index maps	• No hidden rules • Automation of the procedure • Standardisation of data management	Subjectivity in attributing weighted values to the single classes of each parameter

7. Towards harmonization of risk assessment methodologies of soil threats in Europe

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There are various possible consequences of harmonization, positive and negative, which are listed in Table 7.1.

Table 7.1. Possible consequences of harmonization of soil RAMs.

Advantages of harmonization	*Disadvantages of harmonization*
- unequivocal understanding of soil's state - possibility to exchange data and information - in case of action plans: equal efforts by all member states, hence equal market access and production constraints. - efforts made by member states to mitigate the soil threats are comparable - Possibility to define a minimum level of protection in EU-27 - equivocal understanding of RAMs stimulates public support of policies and legislations	- Loss of national or regional detail and information - Loss of public support due to loss of cultural identity of soil RAM. - large efforts are needed to harmonize soil RAMs. Harmonization may be regarded as a top-down approach. - possible conflict with subsidiarity of legislation

Although Table 7.1 provides more advantages than disadvantages of harmonization, there may still be considerable resistance with regard to harmonization of soil RAMs. Notably, in December 2007 the soil thematic directive was rejected by a blocking minority by Germany, the Netherlands and Austria, for reasons of subsidiarity, and by France and the UK for reasons of proportionality. The term 'subsidiarity' refers to an organizing principle that matters ought to be handled by the smallest, lowest or least centralized competent authority. Hence, whenever possible, actions should be performed at the lowest possible organizing level. Harmonization may conflict with this subsidiarity. Hence, at the end, the advantages of harmonization should outweigh the disadvantages of harmonization, considering regional variability in the bioclimatic and biophysical conditions. Although, this is a highly arbitrarily discussion, with many policy related aspects, an attempt was made to provide more objective and quantitative information about the needs and options for harmonization.

The need for harmonization was derived from 'the variation in outcomes of the different RAMs for a specific soil threat'. The larger the variation in outcomes, the larger the need for harmonization. Basically, this would require applying all RAMs for a specific threat to various areas, and to compare and assess the outcome per threat and area for all RAMs. This was done at a small scale in two case-studies, but it was practically impossible to compare all RAMs given the large number of RAMs and the complexities involved when applying RAMS in practice. In stead the need for harmonization was assessed on the basis of the number of different concepts used in the data processing step of the risk assessment chain, while assuming that the notion/definitions of the threats in each RAM were similar. In this step four different

(combinations of) concepts can be used, namely (i) process modeling, (ii) expert judgment, (iii) factorial approach and (iv) empirical modeling (Eckelmann et al. 2006). We assumed that RAMs using the same concept show less differences in outcomes compared to RAMs that use different concepts. The need for harmonization was derived from the cumulative frequency distribution (CFD) of the total number of concepts for data processing (4) on the x-axis and the cumulative use of these methods on the y-axis. The highest need for harmonization was expected for the least steep CFD, while relative little need for harmonization was expected for soil threats with a steep CFD.

The efforts for harmonization refer to the endeavours that are needed to harmonize the RAMs. Efforts for harmonization were assessed per step in the risk assessment chain, using the so called 'matching index' (MI). The matching index was defined as the fraction of common elements within different RAMs.

$$MI = \frac{Common\,elements\ per\ step\ in\ risk\ assessment\ chain}{Total\ elements\ per\ step\ in\ risk\ assessment\ chain} \qquad eq.6.1$$

The MI provides a number between 0 and 1 and was interpreted as (i) relatively high efforts are needed for harmonization (MI < 0.25), (ii) intermediate efforts for harmonization (0.25 < MI < 0.75) and (iii) little efforts for harmonization (MI > 0.75). Because of the different nature of activities in each step of the risk assessment chain, the definition of MI was adjusted for each step in the risk assessment chain. For data collection the MI was defined as the shared coverage of common criteria as provided in Annex 1 of the soil thematic strategy. For data processing, MI refers to the common use of a specific approach, choosing from process modelling, factorial assessments, empirical modelling and expert judgement. The definition of MI for data interpretation refers to the reciprocal of the number of different threshold values that are used, so that a limited number of dynamic or fixed thresholds coincides with a high MI, i.e. harmonization requires relatively little efforts. We could not quantify the MI for risk perception because of poor information about this final step of the risk assessment chain. These threshold values should consider regional climatic and biophysical conditions throughout EU.

To assess the need for harmonization all RAMs with similar notion of the threat were classified according to their main concept in data processing. Subsequently, the cumulative frequency distribution (CFD) of the concepts used per threat was calculated for each soil threat. Figure 6.1 demonstrates that the steepest CFDs were observed for landslides, SOM decline and salinization, which suggest relatively little need for the harmonization of RAMS for these threats. The more flat curves for erosion and compaction suggest a relatively high need for harmonization of these RAMs.

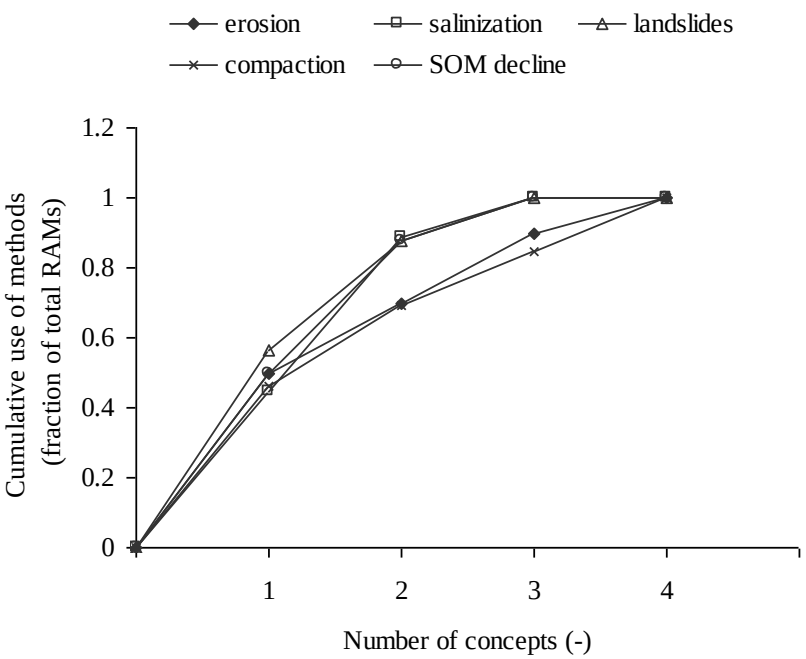

Figure 7.1. Need for harmonization for different soil threats in EU27, as derived from the cumulative frequency distribution (CFD; see text).

The common criteria of Annex 1 of the soil thematic strategy are listed in Table 7.2 for each threat. The Matching Indices (MIs) for data collection were calculated for each soil threat and ranged from 0.58 for compaction to 0.88 for SOM decline (Table 6.3). This suggests that the least consensus about required data exists for compaction and most for SOM decline. The MI for erosion equalled 0.62. This relatively low value was mainly related to the absence of agro-ecological zones in all RAMs and to the absence of land cover in most of the RAMs. For salinization a relative high coverage (81%) of the common criteria was observed and several RAMs took all criteria into account (Table 7.2). For compaction, topography and to a lesser extent land cover was frequently missing in the RAMs, yielding a MI of 0.58. For landslides a MI of 0.77 was observed, based on the facts that climate and seismic risks were commonly missing (Table 7.2).

For data processing MIs were highest for landslides and salinization. Most used methods were empirical modeling (erosion), expert judgment (salinization), process modeling (compaction and landslides). For SOM decline both expert judgment, factorial approaches and process modeling are currently applied.

For data interpretation, most different threshold values were observed for compaction. For this soil threat both different indicators (e.g. saturated hydraulic conductivity, air capacity and penetrometer values) and different values were used per indicator. Also for salinization the differences in threshold values originate from the use of different indicators (e.g. ESP, EC, LR). For erosion the number of reported thresholds is under debate, but 6 out of 11 RAMs reported the use of a threshold value. Most scientists agree that thresholds should be related to baseline (or ‘natural’) erosion rates using ‘benchmark’ sites. This is, however, not yet practice and reported tolerable erosion rates range from 1 to 2 t $ha^{-1}y^{-1}$ (Huber et al. 2007).

Table 7.2. Inclusion of common criteria in RAMs per soil threat. Cells in grey are not part of the criteria for the specific soil threat, x = included in RAM, - = not included in RAM. Brief descriptions of the common criteria are given in the column headings, more elaborate descriptions can be found in Annex 1 of the proposal for a framework directive (COM(2006)232) and in Eckelmann et al. (2006).

Soil threat	Country (RAM)	Soil type	Soil texture	Soil density, hydraulic properties	Topography	Land cover	Land use	Climate	Hydrological conditions	Agro-ecological zone	Soil hydraulic properties	Irrigation	Groundwater	Bulkdensity	Soil organic matter	Occurrence/density of existing landslides
Erosion	Germany	x	x	x	x	-	x	x	x							
	Finland	x	x	x	x	-	x	x	x							
	Spain	x	x	x	x	x	-	x	x							
	Hungary	x	x	-	-	-	x	x	x							
	Belgium	x	x	-	x	-	x	x	-							
	Norway	x	x	x	x	-	-	x	-							
	Poland	x	-		x	-	x	x	-							
	France	x	x	x	x	-	x	x	-							
	CORINE	-	x		x	-	x	x	-							
	PESERA	-	x	x	x	-	x	x	x							
	GLASOD	-	-	-	-	-	-	-	-							
Salinization	Cyprus		x					x			x	x	x			
	Hungary 1	x	x					x			x	x	x			
	Hungary 2	x	x					x			x	x	x			
	Hungary (TIM)	x	-					x			-	-	-			
	Romania	-	x					x			-	x	x			
	Slovakia	x	x					x			x	-	x			
	Spain	-	x					x			x	x	x			
	Greece	x	x					x			x	x	x			
Compaction	Romania	x	x			-	x	x						x	x	
	Germany	x	x			-	-	-						x		
	Germany	x	x			x	x	x						x	x	
	Germany	-	x			-	-	-						x	x	
	Germany	-	x			-	-	x						x	x	
	Poland	-	-			-	-	x						x		
	Poland	x	x			-	-	-						-	x	
	Denmark	x	x			-	x	x						-	x	
	France	x	x			-	x	x						x	x	
	Spain	x	x			x		x						x	x	
	Greece	x	x			x	x	x						x	x	
	Italy	-	-			-	-	-						-	-	
	Finland	-	x			-	x	x						-	x	
	Slovakia	x	x			-	x	-						x	-	
	Hungary	x	x			-	x	x						x	x	
	Belgium	x	x			-	x	x						-	-	
	Belgium	x	x			-	x	-						-	-	
Landslides	France	x			x	-	x	-								x
	Italy	x			x	x	x	x								x
	Sweden	x			x	x	x	x								x
	Switzerland	x				x	x	-								x
	Belgium	x			x	-	-	-								x
	cyprus	-			x	x	x	x								x
	Czech republic	-			x	-	x	x								x
	Ireland	x			x	-	x	-								x
	Hungaria	x			x	x	x	x								x
	Slovenia	x			x	x	x	x								x
	Slovakia	-			x	x	x	x								x
	spain	x			x		x	x								x
	United Kingdom	x			x	x	-	-								x
	Portugal	-			x	x	x	-								x
	Greece	x			x	x	x	-								x
	Poland	-			x	x	-	x								x
SOM decline	Belgium	x	x		x	x	x	x								x
	France	x	x		x	x	x	x								
	Slovak Republic	x	x		x	x	x	x								
	United Kingdom	x	x		x	x	x	x								
	Slovenia	x	x		x	x	x	x								
	Denmark	x	x		x	x	x	x								
	Greece	x	x		x	x	x	x								
	Germany	x	x		x	x	x	x								

There was little information available for risk perception, which was presumably caused by the debates about thresholds values. The steps in the risk assessment chain are in successive order and hence incomplete information in a previous step will hamper the execution of the next step. The average results of the MI of each step in the risk assessment chain suggested that best options for harmonization were expected for SOM decline and landslides (Table 7.3).

Table 7.3. Summary of matching indices (MI) per soil threat and per step in risk assessment chain. MIs are a measure for the relative common elements of different soil RAMs. n.c.= non conclusive.

	Data collection	*Data processing*	*Data interpretation*	*Risk perception*	*Average*
Erosion	0.62	0.55	0.17	n.c.	0.45
Salinization	0.81	0.62	0.13	n.c.	0.52
Compaction	0.58	0.35	0.09	n.c.	0.34
Landslides	0.77	0.63	0.55	0.50	0.61
SOM decline	0.88	0.50	n.c.	n.c.	0.69

Based on the results in Table 7.3 least efforts for harmonization were found for the RAMs for landslides and SOM decline. The reasons however for these two threats to provide good options were different: the landslide scientific community is already in a process towards harmonization of risk assessment and hence at present provides the best basis for harmonization. At the same time RAMs for SOM decline provide relative good possibilities for harmonization because RAMs for SOM decline are incomplete and least developed. We argue that the best time to harmonize guidelines and procedures for RAMs is when they are being developed, provided that this development is properly coordinated.

In general, least efforts for harmonization were needed for data collection for each soil threat. In our approach we focused on the inclusion of common criteria in the RAMs as a basis for assessing the efforts needed for harmonization. However, even when common elements of data collection methods are similar, still considerable differences in outcomes may occur due to differences in sampling schemes, laboratory protocols, etc. With regard to harmonization of sampling schemes Morvan et al. (2008) concluded that an additional 4100 sampling sites are needed to achieve a harmonized, i.e. comparable, scheme across EU-27. Likewise, the MI for data processing refers to the common use of data processing methodologies, but even when similar methodologies are used results may be quite different because of different parameterization, scaling, etc. However, to our view, the use of similar methodologies demonstrate a common understanding of how the data should be processed and hence indicate that relatively little efforts are needed for harmonization of the data processing step of soil RAMs.

8. Soil Threat Index: an option for harmonized presentation of RAM result

G. Tóth

Options for harmonization of risk assessment methodologies are diverse. This diversity originates from the differences in the objectives, data availability, historical and terminological differences etc. While in the cases of salinization, compaction and landslides the harmonization of the assessment processes which are applied in different parts of Europe is feasible, for erosion and organic matter loss it is less so. However, the common ground for risk perception can be the decline in the functioning ability of soil with regards to all the soil degradation threats. This approach is articulated in the Thematic Strategy for Soil Protection of the EU (EC 2006) and reflects the principles of most RAMs in Europe.

The changed functional ability of soils is caused by the alteration of soil characteristics by anthropogenic impact. Long-term human impact (sealing, deforestation etc.), as well as seasonal soil management (drainage, cultivation, irrigation, nutrient management etc.) modifies material and energy flows, that result transformation of the soil processes to smaller or greater extent. When these processes are controllable, soil-use and soil quality remains sustainable on the long run.

The degree of loss in functional capacity due to soil degradation (of different kinds) is an interim reaction of different soil types. On the basis of quantitative soil quality evaluation the effects of various kinds of soil degradation (erosion, acidification, compaction, etc.) measurements, an integrated method becomes available to express the soil quality - soil degradation relationship, thus, soil sustainability (Tóth et al 2007).

Risk of soil degradation depends on soil and terrain properties which make the soil inherently receptive of degradation. Van Camp et al. (2004) provide substantial knowledge towards identifying and describing hazards (threats) to soil. Eckelman et al. (2006) summarizes the risk assessment methodologies applicable for soil degradation studies and offers the concept of threats to represent the hazards endangering the functioning of soils.

Within the context of the soil protection strategy of the EU, the Soil Threat Index has been proposed for an indicator of soil characteristics and processes and degradation-related hazards (Tóth et al 2007, Tóth 2008). According to the definition, Soil Threat Index (STI) is a composite indicator of degradation-related Soil Response Properties and external factors (climate, land use) expressing the level of risk on which the soil is exposed to the main degradation threats (Tóth et al. 2007). For applications in the EU, STI refers to the comparative risk of the major threats (erosion, salinization, compaction, loss of organic matter, landslides) identified in the Thematic Strategy for Soil Protection (EC 2006).

The two component of Soil Threat Index can be matched to the elements in the framework of soil threat assessment identified by by Eckelmann et al. (2006) as indicated in Table 8.1.

Table 8.1. Components and assessment of the Soil Threat Index

	Components of Soil Threat Index	**Procedure of assessment**
1	Soil response properties (soil attributes that identify vulnerability)	Characterization of receptor (degradation-specific classification)
2	External factors of degradation (climate, land use)	Identification of factors/hazards (quantification of impact/ exposure)

The risk level on which the soil is exposed to degradation is considered within the general approach of risk assessment, where risk is "the combination of probability or frequency of occurrence of a defined hazard and the magnitude of consequences of occurrence" as defined by the European Environmental Agency (EEA1999).

In this framework the Soil Threat Index can be regarded as the probability and magnitude of degradation. The magnitude of degradation can be defined by the level of vulnerability and the force of degrading impact. The probability is the likelihood of the occurrence of the degrading impact and as such can be identified on the time perspective.

Therefore the Soil Threat Index reflects the magnitude of degradation and the number and duration of occurrence of the degradation, in time. The formula to define the degradation risk (modified after Tóth et al. 2007) is presented below:

$$STI = SRP * P(DI_{i,n}) \qquad \textit{eq.8.1}$$

(Soil threat index = soil response properties * degrading impacts * probability of occurrence)

Where:

a) Soil Response Properties (SRP) can be defined as:

$SRP = \Sigma f_{i,n} (\Sigma SC)$

Where:

f is a (non linear) function describing the response (both its direction and magnitude) to an impact, determined by, ΣSC that represents soil characteristics.

b) $P(DI_{i,n})$ is the probability and magnitude of the Degrading Impacts corresponding to the external factors of degradation (e.g. soil management, climate change) from i to n.

The STI is presented as a relative number without a dimension, on any scale appropriate for the purpose. For European applications a 10 grade scale is suggested.

The STI can be interpreted as the magnitude of degradation, which depends on the vulnerability and stress. Vulnerability is an inherent soil attribute that can be modified by external factors. For example in the case of erosion, vulnerability is marked by the erodibility of soil and conditioned by soil cover and slope. Climate and land use as external factors represent the degradation pressure on soil.

The erosion threat index (the specific STI for erosion) is proportional to the two components. In case if any of the two constituents is missing, there is no erosion threat. Even the most sensitive soil will have no erosion threat if there is no external degrading effect (stress from water). On the other hand a strong stress does not necessarily lead to erosion, if the soil is resistant.

However, the combination of individual external factors (soil management, precipitation) in most cases provides an effect strong enough to lead to erosion.

If we consider the duration of the degradation stress, the calculation of the cumulative degradation effect becomes possible.

Although the indication of the magnitude of threat can be degradation-specific, the STI can be applied as a complex indicator, summing the degradation risk of all threats relevant to the area concerned. In this respect, the Soil Threat Index might provide an adequate tool to make result of risk assessment methodologies in Europe comparable.

9. Conclusions

The soils in Europe are under increasing stress. The release of the European Strategy on Soil Protection in 2006 has put the protection of this natural resource high on the political agenda of many countries. However, as soil is an unmoving mass and many processes leading to soil degradation occur before the degradation becomes visible, the evaluation of soil degradation is complex. Therefore, many risk assessment methodologies (RAMs) were developed, of which many are (very) complex in the sense of data requirements, software capacities and expert interpretation. From the RAMSOIL project the following conclusions were drawn:

1. Many different soil RAMs are used by only a limited number of countries within EU-27.
2. Most of these RAMs are incomplete, i.e., they quantify process rates or states of the soil threat under consideration rather than risks, i.e. the probability of occurrence of unwanted soil degradation. This complicates the evaluation of options for harmonization of soil RAMs.
3. Many different RAMs have common elements, yet differ in comprehensiveness, complexity and in spatial and temporal scales.
4. Using different RAMs may have large consequences with regard to affected area, spatial distribution and patchiness of the threat occurring. A case-study on soil erosion in Romania showed differences in affected areas up to 30%.
5. There are various interpretations of the term 'harmonization' ranging from the strict definition of making results comparable or compatible to standardizing protocols. The ambivalent use of the term harmonization in international literature complicates the assessment of options for harmonization.
6. Harmonization of RAMs, i.e. making results comparable or compatible is complicated by differences in the notion of the threat, data collection, data processing, data interpretation and risk perception.

Although, given the common (scientific) grounds of many soil RAMs, our impression is that in practice harmonization of soil RAMs will be very laborious and most likely in the end will result in some kind of compromise. To maintain the site specificity and cultural identity of regionally developed RAMs we suggest two options that may facilitate unequivocal identification of risk (or priority) areas for soil threats:

1. a two-tier approach where tier 1 is at a relatively low spatial resolution, and tier 2 is a more detailed assessment.
2. generic harmonization, i.e. combining standardization and harmonization.

Ad. 1. A tiered approach for the identification of risk areas for soil threats was also suggested by Eckelmann et al. (2006). They suggest to firstly provide broadly defined zones at Tier 1 level, within which specific measures have to be planned at Tier 2 level. Also, the landslides scientific community already uses a Tiered approach, which paves the way for the other soil threats to use a comparable approach. However, in the landslide community a slightly different version of the Tiered approach is followed compared to the suggestion of Eckelmann et al. (2006). In the landslides scientific community Tier 1 is a generic landslide susceptibility map using a heuristic weighting-rating model, Tier 2 is a landslide susceptibility map (by types) using a multivariate statistical model and Tier 3 is a landslide susceptibility/hazard map using a process-based model. Recently, an overview of soil threats in Europe including maps at

European level was released by Tóth et al. (2008). Although these maps may have shortcomings, as was, among others, discussed during the EUROSOIL conference 2008, they may provide a starting point for a Tier 1 assessment. At Tier 2 level regionally developed RAMs may be applied at a smaller scale on the condition that the Tier 2 approach is harmonized (i.e. compatible) to the Tier 1 approach.

Ad. 2. Generic harmonization implies that some part of the risk assessment chain are standardized whereas others are harmonized. This approach is in line with the activities performed under the ENVASSO project on procedures and protocols for soil assessments throughout Europe (Kibblewhite et al., 2008). For instance, data collection and data processing are standardized, i.e. prescribed in procedures and protocols, whereas data interpretation and risk perception is harmonized. This option would leave the member states free to choose most appropriate threshold values and risk categories as long as they are mutually harmonized.

Both of the abovementioned options for harmonization result in unequivocal understanding of the severity of risk assessment in each member state of EU-27. Future research should focus on the identification of the most suitable option for the assignment of risk (or priority) areas for soil threats in Europe. It is foreseen that this identification is not only about technical details but also about political willingness and public support.

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Annexes

Annex 1. Project consortium

RAMSOIL is a consortium of:

Alterra, The Netherlands, Co-ordinating institution

Wageningen University (WU), The Netherlands

Research Institute for Soil Science and Agricultural Chemistry of the Hungarian Academy of Sciences (RISSAC), Hungary

Centre National de la Recherche Scientifique (CNRS), France

Institutul de Cercetari pentru Pedologie si Agrochimie (ICPA), Romania

Centro de Investigaciones sobre Desertificación (CIDE), Spain

Joint Research Centre, Institute for Environment and Sustainability (JRC), Italy

Annex 2. RAMSOIL report series

Reports

PR = Project Report, MR = Management Report. First numbers refer to WP.

No.	Leading participant	Author(s)	Title
PR 1.1	Alterra	H. Heesmans	Bibliography on current risk assessment methods
PR 1.2	Alterra	H. Heesmans	Questionnaires
PR 1.3	JRC-IES	G. Tóth	Development and assessment of RAMSOIL prototype Access database
PR 2.1	Alterra	L. Geraedts, L. Recatala-Boix, C. Ano-Vidal, C.J. Ritsema	Risk assessment methods of soil erosion by water
PR 2.2	CNRS	J.P. Malet and O. Maquaire	Risk assessment methods of landslides
PR 2.3	Alterra	J.J.H. van den Akker and C. Simota	Risk assessment methods of compaction
PR 2.4	Alterra	E. Bloem, S.E.A.T.M. van der Zee, T. Toth, and A. Hagyó	Risk assessment methods of salinization
PR 2.5	Alterra	P.J. Kuikman, P.A.I. Ehlert, W.J. Chardon, C.L. van Beek, G. Tóth and O. Oenema	Current status of risk assessment methodologies for soil organic matter decline
PR 4.1	RISSAC	T. Tóth, C. Simota, C. van Beek, L. Recatalá-Boix, C. Añó-Vidal and A. Hagyó	Case-study Report for the Work package No 4. of Project RAMSOIL. „Identification of geographical risk area”
PR 4.2	Alterra	T. Hoogland and J.J.H. van den Akker	Comparison of two RAMs for compaction: a case study for The Netherlands

Presentations and conference proceedings

Date	Conference	Presenting author(s)	Title
Apr. 2007	ESBN meeting, Hannover	C.L. van Beek and H. Heesmans	The objectives of RAMSOIL; request to contribute to questionnaires
May 2008	EGU	J.-P. Malet, A. Thénot, O. Maquaire, C.L. van Beek, O. Oenema	Inventory of landslide risk assessment methodologies in Europe
Aug. 2008	Eurosoil	T. Tóth	Risk Assessment Methods of Salinization in Europe: a quick look.
Aug. 2008	Eurosoil	C. Simota	Evaluation of an Index for Spatial Assessment of Environmental Sensible Areas to Desertification at a National Scale.
Aug. 2008	Eurosoil	J. van den Akker	Soil Quality Indicators and Risk Assessment Methodologies for Subsoil Compaction
Aug. 2008	Eurosoil	C.L. van Beek	Harmonization of Risk Assessment Methodologies for Soil Threats in Europe.
Sep. 2008	5ICLD (Bari, Italy)	C.L. van Beek	Moving ahead from assessment to action using harmonized risk assessment methodologies for soil threats in Europe (*keynote lecture).*
Dec. 2008	Bodembreed (Dutch soil conference)	C.L. van Beek	Soil degradation in the EU: Harmonization of risk assessment methods
Dec. 2008	Bodembreed (Dutch soil conference)	J.J.H. van den Akker	Soil compaction: an underestimated problem
Feb. 2009	Int. conf. soil degradation (Riga, Lativa)	C.L. van Beek	Harmonization of Risk Assessment Methodologies for Soil Threats in Europe

Posters

Date	Author(s)	Title	Comment
Aug. 2008	L. Geraedts, L. Recatalá Boix, C. Añó Vidal, C.J. Ritsema & J. Sánchez	Risk assessment methods of soil erosion by water in the European Union	Poster presented at Eurosoil 2008

Papers

Author(s)	Title
C.L. van Beek, T. Tóth, A. Hagyo, G. Tóth, L. Recatalá Boix, C Añó, J.P. Malet, O. Maquire, J.H.H. van den Akker, S.E.A.T.M. van der Zee, S. Verzandvoort-van Dijk, C. Simota, P.J. Kuikman, O.Oenema	Towards harmonization off risk assessment methodologies for soil threats in Europe. Soil Use and Management 2010, 26: 299-309
J.J.H. van den Akker, T. Hoogland and C. Simota	Differences in perception and outcomes of two risk assessment methodologies for compaction in The Netherlands. Submitted to Geoderma
S. H. H. Shah, R. W. Vervoort, N. Dal ferro & S. E. A. T. M. van der Zee	Water and salinity interactions between groundwater, rootzone, vegetation and climate. In: Hydrology and Ecology: Ecosystems Interfacing with Groundwater and Surface Water (ed. by J. Bruthans, K. Kovar & P. Nachtnebel), Proc. HydroEco2009 Conf. (20-23 April 2009, Vienna, Austria). Published by Universität für Bodenkultur (BOKU), Vienna, 41-44.

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Abstract

The EU thematic strategy for soil protection recognizes that soil degradation through erosion, soil organic matter decline, compaction, salinization and landslides occurs in specific areas, and that these areas must be identified in an unequivocal way. Currently, there are various risk assessment methodologies (RAMs) and the question has risen to what extent these RAMs yield similar outcome and, if not, whether the outcome can be harmonized, i.e. whether the results of the various RAMs can be made compatible or comparable.

In this study i) the current status of RAMs for erosion, soil organic matter decline, compaction, and salinization in the European Union (EU27) is reviewed, and ii) the need and the options for harmonization are assessed. The need for harmonization was defined as the likelihood of achieving different outcomes when using different RAMs, whereas the options for harmonization refer to the efforts that are required to harmonize soil RAMs. The current status of RAMSs in EU-27 was assessed on the basis of questionnaires, which were sent out to soil specialists and policy officers in all Member States. We received more than 100 (response rate >50%) completed questionnaires. It turned out that many of the so called RAMs are still incomplete; they are 'process (or threat) quantifications' rather than methodologies that assess the risk of a soil threat. Moreover, there were significant differences between RAMs for a soil threat in terms of (i) the notion of the threat, (ii) data collection, (iii) data processing, (iv) data interpretation, and (v) risk perception.

The need for harmonization appeared highest for erosion and salinization, whereas the options for harmonization were best for SOM decline. Harmonization of soil RAMs may be very complex and for that reason not always feasible. We suggest two options that may facilitate unequivocal identification of risk (or priority) areas for soil threats, i) a two Tiered approach based on data availability and spatial scale and ii) generic harmonization, i.e. combining standardization and harmonization in a rather pragmatic way.

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