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Threats to Soil Quality in Europe

Gergely Tóth, Luca Montanarella and Ezio Rusco (eds.)

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Threats to Soil Quality in Europe

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Land Management and Natural Hazards Unit

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Preface

During the recent years, there has been a surge of concern and attention in Europe to soil degradation processes. Initiated by the German Ministry for Environment in 1998 with the first European Soil Forum a process has been developing over the past ten years leading to the adoption by the European Commission in September 2006 of the Thematic Strategy for Soil Protection including a proposal for a Soil framework Directive aiming to the reduction of the exponentially growing soil degradation processes in Europe and to the establishment of a legislative framework allowing for the sustainable use of the limited, non-renewable, natural resource.

One of the most innovative aspects of the newly proposed Soil Thematic Strategy for the EU is the recognition of the multifunctionality of soils. Previous legislation, like the soil protection act of 1930' of the USA, has been essentially focusing on soil protection in relation to the single function of soils as the substrate for food and fiber production (agricultural function). Therefore, traditional definitions of soil quality have been related to the quality of soils for agriculture. Most of the soil science achievements of the past have been related to this mono-functional perspective on soils, starting with the National and International soil classification systems (USDA Soil taxonomy, FAO, WRB, etc…), all focusing mostly on the classification of soils under an agricultural perspective.

The adoption of the EU Soil thematic strategy opens new perspectives towards a new definition of soil quality taking into account the various functions of soils: food and fiber production, buffering and filtering of contaminants, biodiversity pool, archive of cultural heritage, source of raw materials, substrate for housing and infrastructure, etc…

The re-definition of soil quality will also have a major impact on the environmental reporting process, both at national and International level. Soil Quality is a recognized indicator by the OECD countries and is included in the list of agro-environmental indicators relevant to EUROSTAT as well as to EEA. A more robust and innovative definition of soil quality for Europe will allow more efficient reporting about the status of the environment and will allow to design appropriate monitoring systems for detecting changes in soil quality over time.

The special session during EUROSOIL 2008 dedicated to the threats to soil quality in Europe has allowed for an in-depth analysis of the status of research in this are and the identification of still existing research gaps for future action. The full coverage of the threats identified within the Soil Thematic Strategy will allow to further support the on-going process towards better soil protection in Europe.

L. Montanarella

Characterisation of soil degradation risk: an overview

W.E.H. Blum

1. Introduction

Soil degradation means loss of soil or soil quality for specific functions.

Risk of soil degradation can derive from extreme natural events, such as long-lasting torrential rainfalls, causing e.g. erosion, inundations, landslides and further adverse effects. – Those forms of degradation are rather rare, compared to risks caused by human interactions, e.g. by different forms of intensive land use. Therefore, human activities can be regarded as the main causes of soil degradation risk (Blum, 2002).

Risk of soil degradation is a result of the competition between the uses of different soil functions and the overuse of singular ones without sufficient control. Therefore, the questions arise: What are the main functions of soil and what are the risks of soil degradation?

2. The six main functions of soil

Soils have at least 6 different functions for the social and economic development of humankind, which can be distinguished into three more ecological functions and three others, directly linked to human activities defined as technical, industrial and socio-economic functions (Blum 2006; COM(2002)179 final).

The three ecological functions are:

1. production of biomass, ensuring food, fodder, renewable energy and raw materials. These well-known functions are the basis of human and animal life.

2. Filtering, buffering and transformation capacity between the atmosphere, the ground water and the plant cover, strongly influencing the water cycle at the earth surface, the gas exchange between terrestrial and atmospheric systems and protecting the environment, including human beings against the contamination of ground water and the food chain, see Fig. 1. These functions become increasingly important, because of the deposition of many solid, liquid or gaseous, organic and inorganic compounds, to which soils react through mechanical filtration, physical or physico-chemical adsorption and precipitation on its inner surfaces and microbiological and biochemical mineralisation and metabolisation processes, mainly of organic compounds. These biochemical reactions also contribute to global change through the emission of gases from the soil into the atmosphere, because globally the total pool of organic carbon in soils is three times higher than that in the atmosphere. Under this aspect, soils are a central link in the bio-transformation of organic carbon and continually play a role in releasing CO_2 and further trace gases into the atmosphere, especially N_2O and CH_4, all three of them known as "greenhouse gases". These gases are causing processes of global change, which in this case involve large-scale feedback of many localised small-scale processes.

   As long as these filtering, buffering and transformation capacities can be maintained, there is no danger for the ground water or for the food chain. However, capacities of soils are limited and vary according to specific soil conditions.

3. A biological habitat and gene reserve, with a large variety of organisms. Soils contain more species in number and quantity than all other above ground biota together. Therefore, soils are the main basis of biodiversity. Human life is extremely dependent on this biodiversity, because we do not know if we will need new genes from soils for maintaining human life in the near or the remote future. Moreover, genes from soil become increasingly important for many technological, especially biotechnological and bioengineering processes.

In addition to these three ecological functions soil has three other functions, more linked to technical, industrial and socio-economic uses:

4. Soils are the physical basis for technical, industrial and socio-economic structures and their development, e.g. industrial premises, housing, transport, sports, recreation, dumping of refuse etc. One of the main problems in this context is the exponential increase of urban and peri-urban areas, including transport facilities between them. This is not only true for Europe, but also for other continents, and especially for countries in development in Africa, Asia and Latin America.

5. Soils are a source of raw materials, e.g. of clay, sand, gravel and minerals in general, as well as a source of geogenic energy and water. These raw materials are the basis for technical and socio-economic development.
6. Last but not least, soils are a geogenic and cultural heritage, forming an essential part of the landscape in which we live, concealing and protecting palaeontological and archaeological remnants of high value for the understanding of our own history and that of the earth.

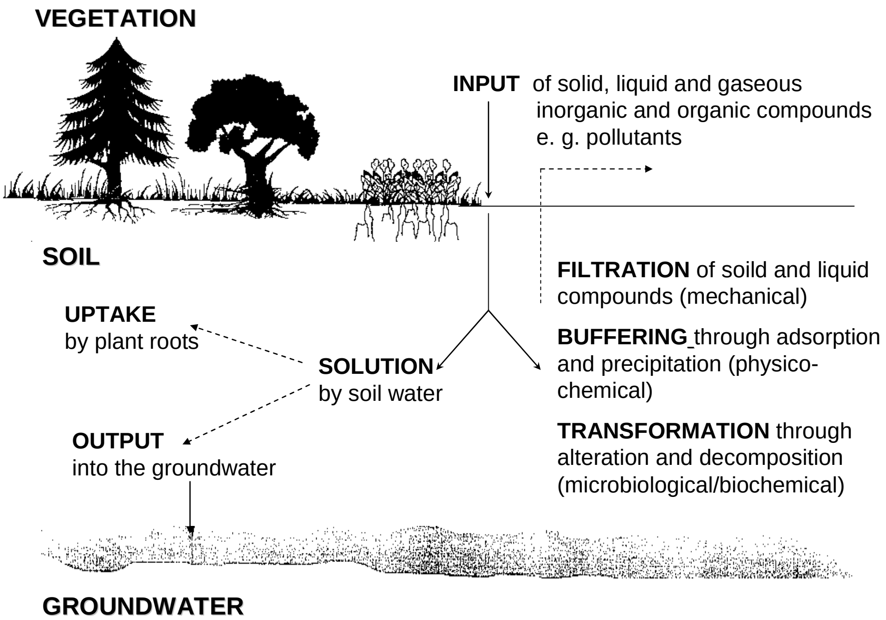


Figure 1. Filtering, buffering and transformation activities of soil

In view of the soil as an absolutely limited resource, which cannot be extended or enlarged, the use of these 6 main functions of soil and land, which is often concomitant in the same area, becomes a key issue for soil degradation. In this context, soil or land use can be defined as the temporarily or spatially simultaneous use of all six functions, although they are not always complementary in a given area.

3. Risk of soil degradation through spatial and/or temporal competition in the use of soil functions and their misuse.

In the following, three different categories of interactions and competitions can be distinguished:

1. Exclusive competition between the use of soil for infrastructure development, as a source of raw materials and as a geogenic and cultural heritage on the one hand and the use of soil for biomass production, filtering, buffering and transformation activities and as a gene reserve on the other hand.

   This becomes evident by sealing of soils through urban and industrial development, e.g. the construction of roads, houses, industrial premises, and sporting facilities or when soils are used for the dumping of refuse, all this being known as the process of urbanisation and industrialisation, thus excluding all other uses of soil and land, see Fig. 2 and 3.

   In these figures, the natural resources of Europe at daylight and the sealing of soils, visible by nightlights, is evident. A more detailed aspect of soil sealing is given in Fig. 4, showing the sealing of a landscape in southern Germany.

   At the moment, it is estimated that Europe loses between 8 and 10 km² of fertile soils per day through urbanisation and industrialisation, which means irreversible soil losses, with additional soil degradation processes in the surrounding environment, such as compaction, erosion, pollution, salinisation, loss of organic matter, loss of biodiversity, inundations and landslides.

Figure 2. Europe's natural resources at daylight

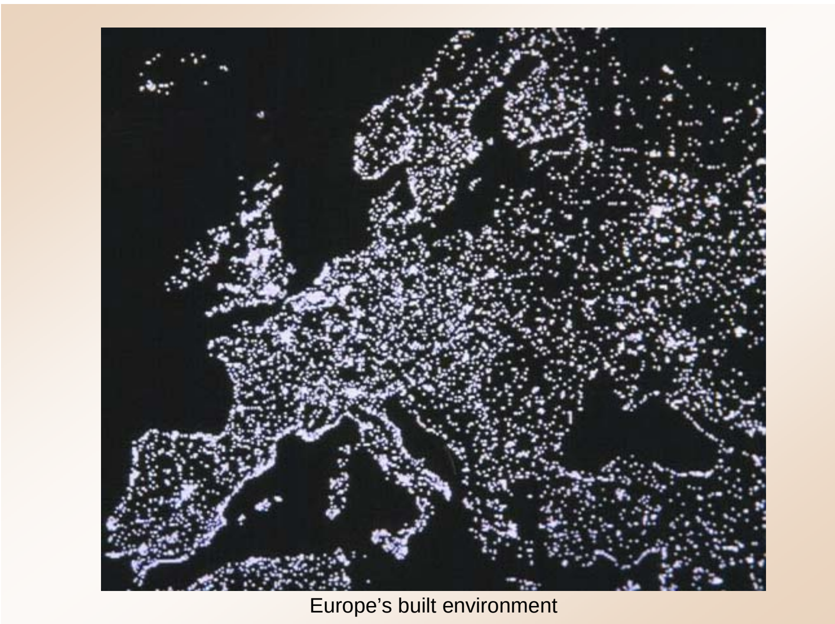
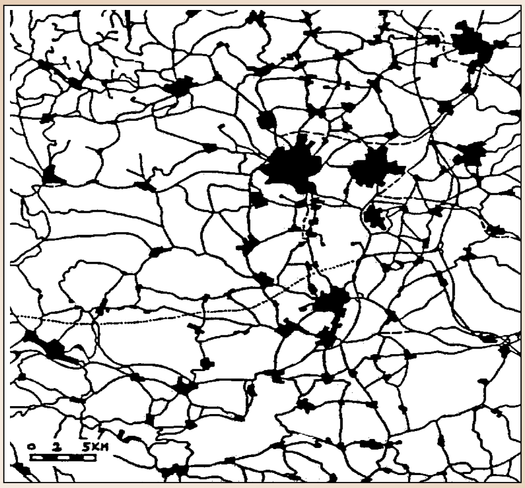


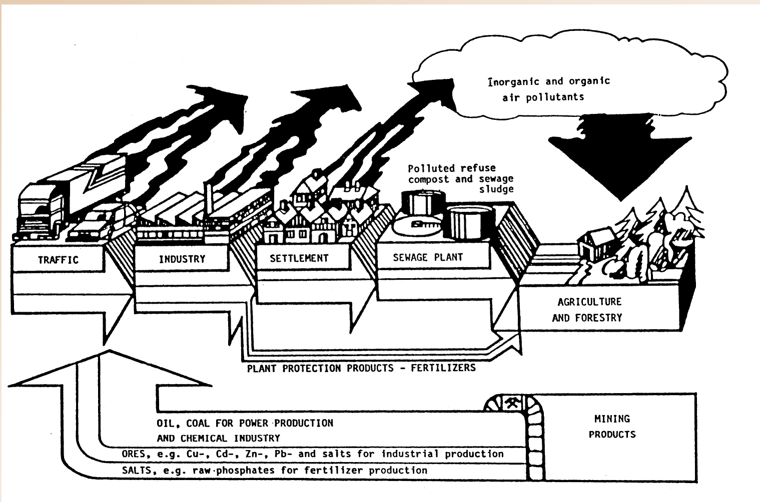
Figure 3. Europe's built environment at night



Sealing of soils and landscapes by settlements and roads
(Example: south-western part of Baden-Württemberg, Germany)

Figure 4. Sealing of soils and landscapes by urban and industrial development
(Baar region in south-western Germany, observe the scale)

2. A second category of competition exists through intensive interactions between the infrastructural soil and land uses on one side and agriculture and forestry on the other side, as shown in Fig. 5, where soil contamination by deposition, caused by intensive use of fossil energy and raw materials is shown. These depositions occur on the atmospheric pathway, the waterway and through terrestrial transport. This is especially true for densely populated areas like Europe. In this context, it seems necessary to point out that soils are the last but one sink for many inorganic and organic depositions, the last one being the bottom of the oceans. In Fig. 5, different forms of loads can be distinguished: Inorganic and organic depositions from traffic and transport and form industrial and urban activities.



Soil pollution through excessive use of fossil energy and raw materials (Blum 1988)

Figure 5. Soil contamination and pollution through intensive use of fossil energy and raw materials

Most of these loads, especially those causing severe acidification, pollution by inorganic compounds, such as heavy metals or by xenobiotic organic compounds and deposition of non-soil materials can cause irreversible soil degradation (Blum 1998). In this context, irreversibility is defined as the non-reversibility by natural forces or technical remediation within 100 years, a time span which corresponds to about four human generations.

Only few processes of soil degradation, such as contamination by biodegradable organics or by small amounts of heavy metals can be regarded as reversible (Blum, 2000). Some further adverse effects of transport, urbanisation and industrialisation on agricultural and forest soils are exemplified by Blum (1998).

3. A third form of competition also exists among the three ecological soil uses themselves, as shown in Fig. 6. Waste compost and sewage sludge deposition on soil as well as intensive use of fertilizers and pesticides, in addition to the deposition of air pollutants (cf. Fig. 5), may have a negative impact on soil quality and especially the ground water and the food chain, by surpassing the natural capacity of soils for mechanical filtering, chemical buffering and biochemical transformation (Blum, 2000). In this context, it should be remembered that agriculture and forestry not only produce biomass above the ground, but also influence the quality and quantity of ground water generation underneath, because each drop of rain falling on the land has to pass the soil before it becomes ground water or drinking water.

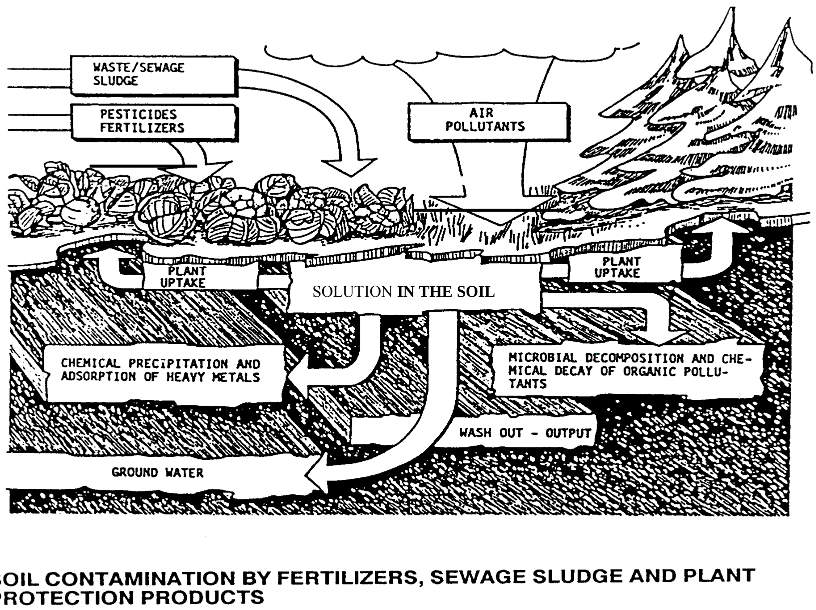


Figure 6. Competition between the production of biomass and groundwater and the maintenance of biodiversity, due to pollutive depositions and the use of fertilizers, sewage sludges and plant protection products

Such problems are well-known for many parts of Europe, where contamination of the ground water used as drinking water, through nitrate, pesticides and/or other chemical compounds, from industrial, agricultural and other activities are known. When the ground water is used as drinking water, the production of food and fibre on top of the soil and the production of groundwater underneath can become competing activities for the satisfaction of basic human needs. In many areas of Europe, conventional agricultural production systems are controlled by quality standards for surface and ground water resources. It is easier to transport and sell food and fodder over large distances than to do the same with the necessary amount of drinking and household water.

A general view on impacts of human activities, causing risks of soil degradation is given in Fig. 7, in which different human activities of land use and their impacts are shown.

4. Risks – threats – degradation processes

In the Communication of the European Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions, in April 2002 (COM(2002)179 final), entitled: Towards a Thematic Strategy for Soil Protection, 8 main threats to soil were identified: erosion, local and diffuse contamination, loss of organic matter, loss of biodiversity, compaction and other physical soil deterioration, salinisation, floods and landslides, and sealing. In the following contributions, these threats, endangering soil quality in Europe are discussed in detail.

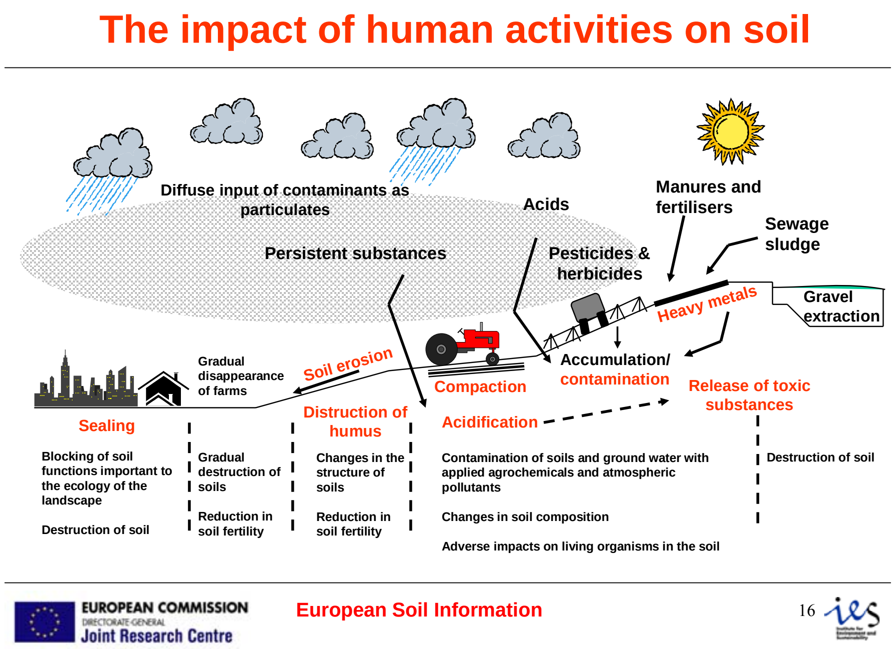


Figure 7. Impact of human activities on soil, causing risk of soil degradation
(EC, JRC; IES)

5. Conclusions

It could be shown that the risk of soil degradation is mainly caused by human activities through different forms of soil and land use. Those activities should be controlled by laws and regulations on a world level (WTO), regional level (EU), country and/or county level. These legal instruments are most important in relation to soil degradation risk. In the following chapters it will be shown, how risk can be minimised, especially through new legal regulations developed on a European level, such as the Thematic Strategy for Soil Protection and the proposal for a framework directive for soil protection in Europe (COM(2006)232 final).

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COM(2006)232 final: Proposal for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive 2004/35/EC.

Soil quality in the European Union

G. Tóth

Summary

The concept of soil quality and sustainability evaluation provides bridging between the utilization and protection aspects of soil-use planning. A framework and definitions for evaluating the quality and sustainable use of soil resources is developed for applications in the European Union in the support of the Thematic Strategy for Soil Protection. The method for evaluating soil quality is designed fully flexible in order to link it with the evaluation of degradation threats. Based on the evaluation procedure, three main indexes in the sustainable soil-use domain are calculated: 1) Soil Quality Index - to express the ability of soil to perform ecosystem and social services 2) Soil Threat Index - to express the level of risk on which the soil is exposed to degradation threats 3) Soil Sustainability Index - for the comparative measurement of soil quality across a gradient of a stress or disturbance

Sustainability analysis of soil-use can be performed for any individual soil function or groups of soil functions in defined land use systems in a comparative manner, taking the potential effects of degradation into account.

1. Introduction

Descriptions of soil (and land) quality through the multifunctional nature of soils (and land) appeared in the second half of the 20th century worldwide, giving frame for possible common scientific recognition of the problem. One of the first widely accepted definitions was published by FAO (1976) describing land quality as „a complex attribute of land which acts in a distinct manner in its influence on the suitability of land for a specific kind of use". As one replaces the word "land" with "soil" in this statement, an acceptable broad definition for soil quality appears[1]. The soil system however includes material and energy processes that may act independently from the primary purpose of soil use and performance of soil functions that are in secondary interest might be modified. Therefore a more comprehensive soil quality definition was desired, that takes this complexity into account.

Soil scientists in the last decades came up with a number of modern theories about soil quality (Bouma 1997a; Davidson 2000; de Han et al. 1990; Karlen et al. 1997. Loveland and Thompson 2001; Máté and Tóth 1996; Sojka and Upchurch 1999; Tóth et al. 2007). Discussion around definitions and underlying scientific concepts has been evolving in the 1990s. The scientific debate, however has always involved considerations of practical aspects as well. The Soil Science Society of America (SSSA) had proposed a definition (Allan et. al 1995), which presents an integration of scientific knowledge with practical approach. The SSSA's description of soil quality as „The capacity of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal production, maintain or enhance water and air quality and support human health and habitation" can be regarded as one of the most comprehensive definitions.

Although, some may criticize that this scientific concept is biased by our current perception of the services provided by soil (Sojka and Upchurch 1999), rather than being based solely and purely on theoretical ground (with possible windows towards applicability), it is generally accepted that apart from studying its natural phenomena, soil

[1] It is however important to make clear the difference between soil and land quality from the very beginning. Land comprises all elements of the physical environment, including climate, relief, soils, hydrology and vegetation, as well as includes the results of past and present human activity (FAO 1976). Soil is one compartment of the physical environment (land) and receives the influence of its other elements. In this respect, soil is examined as a subset of land and its characteristics are determined by other land forming factors. In the meanwhile soil is also regulating environmental processes, thus influencing other elements of the physical environment. Present paper follows the approach of classical soil science and considers soil as a medium that integrates, transforms, stores and filters material (and energy) relevant to its environmental and management conditions in the spatial context. Soil, on the other hand, is a medium that is challenged by changing environmental and management conditions, therefore variable in time as well.

resources should always be considered within the socio-economic context (FAO 1976, Karlen et al. 2001), therefore soil quality assessment within the framework of its functions is as well justified.

Soil scientists in Europe have common understanding on the fundamental meaning of soil quality with their colleagues worldwide, represented by the above approaches.(Bouma 1997b; *Blum 2005,* Davidson 2000; Loveland and Thompson 2001; Schjønning 2004, *Várallyay 1997*).

With the publication of the Thematic Strategy for Soil Protection (EC 2006a,b) of the European Union a framework has been put forward, which sets the way towards operational soil quality criteria in Europe. The scientific background documents prepared by working groups of researchers and different stakeholders from across Europe (Van-Camp et al. 2004) describe the components of sustainable soil use in Europe, including fundamental elements of soil quality.

Major soil functions have been identified and described in a concise form in the official EC communication on the Thematic Strategy (EC 2006a,b). According to the Strategy soil delivers its services through seven of its main functions: (1) food and other biomass production, (2) storing, filtering and transformation of materials, (3) habitat and gene pool of living organisms, (4) physical and cultural environment for humankind and, (5) source of raw materials, (6) acting as a carbon pool, (7) archive of geological and archeological heritage.

The Thematic Strategy, on the other hand declares that for sustainable development, soils (soil functions) need to be protected from degradation. Main threats to soils are identified as decline in organic matter, soil erosion, compaction, salinisation, floods, landslides, contamination, acidification and sealing.

Degradation alters the functioning capacity of soils; therefore soil quality evaluation linked to land use assessment can only be framed by a system approach, which requires the evaluation of the soil threat and soil quality within a common framework.

The overall goal of this paper is to introduce the soil quality evaluation framework for the European Union. The evaluation is based on the set of guidelines put by the Thematic Strategy for Soil Protection and covers considerations on:

(1) major soil functions that need to be maintained on the highest possible level and

(2) soil threats as the main boundary conditions for sustainable utilization of soil functions.

2. Soil quality concept

2.1 Principles

Soil quality is an account of the soil's ability to provide ecosystem and social services through its capacities to perform its functions under changing conditions (after Tóth et al. 2007.) The concept of soil quality expressed by this definition allows practical applications with regards to targeted social and/or ecosystem services. Targeted applications may be linked to special soil functions like in the case of soil productivity ranking, evaluation of carbon sequestration potential, in accounting peat stock etc. The simplest case of soil quality evaluation therefore is to assess the performing potential of soil by a single soil function. On higher levels of aggregation soil quality can express the sum of capacities.

The concept of soil quality recognizes that the comparative importance of soil functions may be spatially and temporally dynamic and it is up to the evaluator to define the conditions of evaluation in accordance with the goal of the assessment.

However, the evaluation scheme has to consider the two basic elements of soil quality: (1) functional ability and (2) response properties. (Figure 1.)

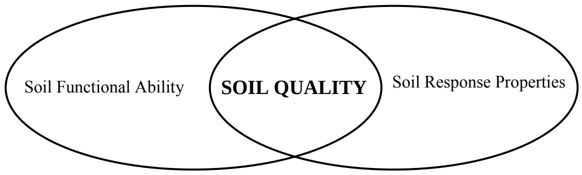

Figure 1. Constituents of soil quality

These two elements reveal the (1) capacity to perform a function under given conditions and the (2) range of the functioning capacity under changing conditions.

The relative role of any of these two components may vary according to the goal of the assessment. For example soil quality evaluation in flood risk assessment may be performed on the basis of the water holding capacity and unsaturated/saturated hydraulic conductivity of soil. In this case the functional ability alone describes the soil quality for flood prevention. In other situations the complex indicator (of functional ability and response properties) is needed. For example in soil bonitation the availability of water set the target of productivity (functional ability). However, productivity can be altered by fertilization. The degree of yield change to fertilization is a distinct response property of each soil. Therefore when evaluating productivity function, both the functional ability (reference level) and response property (elasticity of ability) has to be considered.

The above examples also underline that soil quality evaluation has to be performed with special regards to the goal of the assessment.

The ability of soil to perform any of the identified functions (on given levels) depends on its physical, biological and chemical attributes („internal" attributes), while the realization of the performance is conditioned by natural (e.g. slope steepness) and/or anthropogenic (e.g. drainage) factors ('external' factors). All these factors are time dependant. Humans, amongst the most influential players, indirectly alter the performance characteristics of soils, thus limit or enhance its capacity to function.

Different classification/evaluation schemes group soils based on diverse methods and data needs and none of the schemes can meet all purposes. The selection of the appropriate soil classification/evaluation scheme (model) becomes an important duty for soil quality assessment because by this operation a link between soil processes and our understanding of their meaning in a wider context can be established.

There are various ways to structure the soil quality evaluation system. However, it is worth structuring the model in such a way that the different aspects of soil quality could be included within the same categorization framework, in a clear and comprehensive manner.

The quality of various soil functions should be evaluated on a scale adequate for easy comparison. The evaluation could be carried out for soil of various (energy and material) input levels. The basic input level - which could be used as a standard for further comparisons - should be defined by a clearly described soil-use system.

2.2 Functional ability of soil

The functional ability of soil refers to the number and composition of functions a given soil is able to provide and the level on which functions are provided (Tóth et al. 2007).

This definition can be illustrated by examples from the major soil functions:

- soil productivity / capability (F1: food and other biomass production)
- nitrate retention (F2: storing, filtering and transformation of materials)
- soil biodiversity (F3: habitat and gene pool of living organisms)
- construction bearing capacity (F4: physical and cultural environment for humankind)
- peat stock (F5: source of raw materials)
- organic carbon stock (F6: acting as a carbon pool)
- preservation potential (F7: archive of geological and archeological heritage.)

Functional ability of soil is corresponding to well-defined (actual or desired) unique conditions. This prerequisite is necessary for the assessment, however the fact that conditions determining the performance of soils are in constant change in the nature is recognized (and considered with regards to response properties). The functional ability of soil marks the "baseline" information component of soil quality. Therefore the functional ability may be called as soil quality of stable external conditions.

The specification of influential soil characteristics, their interrelations and their importance for Functional Ability requires a complex approach. Functioning ability of a soil is determined by the number and internal dynamics of soil characteristics. In addition, external conditions for individual functions and other factors are also influential and therefore need to be considered.

Soil characteristics should be evaluated according to the conditions they provide for the specific function of interest. Actual characteristics might be in favor of or can limit the performance of the function. e.g. In a detailed functional ability analysis the assessment of soil characteristics can be carried out to identify those soil characteristics (and/or clusters of characteristics) within the selected soil classes that are most important determinants of the level of performance and to describe the soil-property-driven regulatory principles of material and energy exchange in soils.

The assessment of functional ability of soil requires the measurement of physical, chemical and biological soil characteristics and processes, together with the interaction between them according to specific purpose of the evaluation.

Soil characteristics are ranked according to their role in the performance of soil classes. Soil parameters should also be examined from the viewpoint whether their effect on functional ability could be expressed through some other, more easily measurable characteristics (pedotransfer rules) or, if their importance is increasing in combination with any other soil property. This practice of indicator development as well as the evaluation process could only be carried out by using information that is available in soil maps, databases of soil monitoring and other soil information registries. Conclusions for complex soil characteristics could only be drawn on the bases of appropriate information.

Relevant soil groupings may apply purpose-oriented methods concerning the classifications of soil functional characteristics (e.g. water and nutrient dynamics). Such soil classification schemes should provide a good framework for estimating the behavior of soil.

According to the role of different soil characteristics in the quality of soil classes, correction factors (weighting factors that accent the importance of the characteristics for the evaluated property) can be assigned to each soil parameter during the detailed evaluation process. These correction factors or weights - like in any classical quantitative land evaluation processes - modify the mean index of the soil class.

The weights express the relative role of the characteristic in the functional ability of the soil class. These weights (or factors) are the control parameters of the functional ability evaluation model. By knowing the dynamic properties of soil class, these factors can be used to evaluate the complexity of the soil sustainability system. With this classical land evaluation method, a continuous scale of functional ability of different soil varieties can be derived, which spans the lowest to the highest value of the soil type. If required for different scales, on different level of the taxonomic hierarchy.

2.3 Soil response properties

Soil response properties are particular characteristics that determine the soil's reactions to environmental (or human) influences (Tóth et al. 2007). Soil response properties mark different potentials of soil functional ability by determining both the direction and magnitude how soil reacts to a disturbance or change.

The above definition of soil response properties reveals the dynamic phenomena of soil. Whiles the functional ability of soil indicate soil quality under stable external conditions, the consideration of response properties extends the meaning of soil quality by the information on possible changes under varying external conditions.

For example in the productivity function domain (F1) response properties are characterized by water and nutrient dynamics. Nutrient reaction of the different soil types can be different, and this difference is reflected in the soil quality (in this case: productivity) indices. On soils where the effect of fertilization is higher this should be considered as positive response property. To give a concrete example: as opposed to the strong nitrogen response of Haplic Luvisols, the reaction of Calcic Chernozems do not increase significantly with higher doses of nitrogen fertilization (Tóth et al 2005) On the other hand, annual variations of yield depends on complex weather conditions, including precipitation and temperature. The soil water element of productivity acts in interaction with the dynamics of nutrient availability. However, from the annual variability of yields one can deduce the effect of water regime by applying climate-productivity models (eg. Szász 2002). The water regime of soil types is reflected in the variability of yields over the years (which may differ to a great extent among soil classes). In this case the stability of production is desirable, therefore soil characteristics resulting the higher variability should be considered as indicators of negative response property.

Soil response properties influence soil quality by individual soil functions.

The meaning of a soil response property might also be read as an indicator of the diversities of soil functional abilities within widened boundary conditions. With the extension of soil quality perception by this dynamic component, scenario-based performance potentials may be calculated with associated probabilities.

As we can see, response properties influence the soil quality by altering the functional ability of soil. The alteration may be either natural or human induced. Response properties are characterized by the direction, the degree and the speed of change. Since each soil type has a distinct path of reactions to any environmental change, response properties may also vary thus influencing the stability of the performance (or adaptability to external impacts) to a great extent.

3. Soil degradation threats: setting boundary conditions of soil quality

3.1 Soil degradation threats

Degradation deteriorates soil quality by partially or entirely damaging one or more of its functions (Blum 1988). Degradation processes occurring in Europe are widely studied (Batjes and Bridges 1993, EC 2006c, EEA 2000, Kirkby et al. 2004, van Lynden 1997, 2000) and incorporated to soil protection policies on national (Kraemer et al. 1999) and European levels (EC 2006a,b). The focus of policy actions is the reduction of soil degradation risk.

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. The Thematic Strategy for Soil Protection (EC 2006a) declares that for sustainable development, soils (soil functions) need to be protected from degradation, thus soil threats set the boundary condition for - sustainable - soil quality as well.

The main threats to soil functioning abilities are identified as

(1) decline in organic matter	(2) soil erosion	
(3) compaction	(4) salinisation	(5) landslides
(6) floods	(7) contamination	(8) sealing

Threats 1-5 are area (and soil) specific in their appearance, therefore, they require additional spatial consideration during soil conservation planning. Risk identification for these major soil threats – which have definite environmental and spatial dimensions – in the European Union is proposed by Eckelmann et al. (2006), in their “Common Criteria” document.

For each area-dependent threat, the following conditions have been examined in order to define common criteria of risk identification throughout Europe:

- - identification of factors/hazards related to threat (‘external’ factors),
- - characterization of receptor (‘internal’ attributes),
- - performance specification, model selection (with data requirements).

In order to identify and describe areas at risk to soil threats in the “Common Criteria” document, Eckelmann et al. (2006) proposes three types of approaches:

1) qualitative approach: land use in combination with “sensitive soils”, or other political boundaries using other combined criteria, e.g. nitrate pollution, intensive cropping areas, urban areas, etc.;

2) quantitative approach: thresholds;

3) model approach: in the absence of monitoring data, the potential for soil degradation can be assessed (in the presence of monitoring data and in combination with approach 1.: regionalization/upscaling of plot data).

For the application options the "Common Criteria" document provides explanations of the above approaches, articulating that thresholds initially require that reasonable values are available beyond which degradation of soil properties limits sustainable functioning of the soil. Data from soil inventories or monitoring must be available in a further step, in order to match observed values with thresholds. Even if thresholds, status and trends are based on models, soil inventory/monitoring data are still needed. The model approach needs to be eventually supplemented by a quantitative approach: not only for model validation and calibration, but also in order to detect the area where the degradation actually occurs, and to observe the trend after the implementation of measures. Models can also help in approach 1) and 2) to regionalize soil information, from the plot-level to the area/region. Ecklemann et al. (2006) proposes a list of requirements that should be fulfilled in order to have a common base for comparison the soil degradation risk in the member states of the European Union.

3.2 The Soil Threats Index

The Soil Threats 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 2006a).

The two component of Soil Threat Index can be matched to the elements in the framework of soil threat assessment by Eckelmann et al. (2006) as indicated in Table 1.

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.

Table 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)

Therefore the Soil Threat Index reflects the magnitude of degradation and the number and duration of occurrence of the degradation, in time.

The magnitude of degradation 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.

4 Soil quality in time

Alteration of soil characteristics by anthropogenic impact changes functional ability of the soil. Long-term human impact (sealing, forestation etc.), as well as seasonal soil management (drainage, cultivation, irrigation, nutrient management etc.) modifies material and energy flows. This modification result transformation of the soil processes to smaller or greater extent. When these processes are traceable, 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.

Matching soil quality and degradation characteristics with a time horizon is the basis to evaluate soil sustainability.

Within the context of the soil protection strategy of the EU, the Soil Sustainability Index has been proposed for an indicator of soil functional ability and degradation-related hazards taking the time perspective into account. According to the definition, Soil Sustainability Index (Tóth et al. 2007) is a comparative measure of Soil Quality across a gradient of stress or disturbance. The expression includes the stability of soil characteristics in time and the internal and/or external environmental interactions of soil, thus also relates to the degradation threats.

The sustainability of soil-use and preservation of soil resources depends on the capacity of soil to respond to impacts and maintain its function over time and under changing pressure of soil degradation threats.

The application of the soil sustainability evaluation can be illustrated by an example of soil productivity under degradation stress, with given land use. Soil can have high, medium or poor productivity, which may improve, stagnate or deplete under the given land use. The grade of productivity change indicates the sustainability or the non-sustainability of the soil use system. Stability of favourable soil processes secure the sustainability of soil utilization.

However, one of the great challenge remains for soil science is to characterize soil processes in relation to the socio-ecological (land use; human impact) and biophisical context, taking the time perspective into account.

5 Conclusions

The Thematic Strategy for Soil Protection of the European Union set the frame for a common soil quality approach in Europe by declaring the soil functions

The evaluation of the main soil functions is in the core of the European approach of soil quality assessment which is supplemented with the evaluation of the boundary condition of soil threats to allow for a complex description of the sustainability of the soil-use system.

Results of this evaluation process may provide an adequate answer to the society’s needs for a simple measurement to compare the options for utilizing soil functions and measuring the risk of any particular utilization.

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Main threats to soil quality in Europe

The natural susceptibility of european soils to compaction

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Summary

Compaction belongs to the significant threats to soil. This fact is declared also in the "Soil Thematic Strategy" where compaction is one of the five most frequent threats to soils in Europe. Compaction invades soil balance with the other parts of the environment and can therefore accelerate the other threats, e.g. erosion; both wind as well as water. Compaction itself is coming out from several factors. They can have different origin: some of them are natural, some are created by man. Very often the combination of these factors can occur. Compaction is not stable soil damage, besides that situation when the soil is destroyed completely (totally and irretrievable changes in characteristic soil functions) and because of this fact, it is more possibly on European level, to evaluate soil susceptibility to compaction rather than already existing compaction. In addition, it is always better to know possible damages and prevent them than just to comment already existing ones. From this point of view, the evaluation of soil susceptibility to compaction on European level was done. The evaluation comprises natural soil susceptibility coming out from the fact that some soils are more susceptible to compaction because of their basic properties like texture, water content, water regime kind and arrangement of soil horizons, etc. This evaluation results in a new actualized version of the Map of Soil Susceptibility to Compaction in Europe and demonstrates priority areas related with soil compaction at European scale.

1. Introduction

The status of European soils is mainly result of human activities over the past millennia. Only soils in some forests are not influenced by man. The whole area of agricultural soils is influenced by human activities [11]. Cultivation essentially changes soil environment. Although the aim of cultivation is always to improve soil properties to increase crops, not always the result is in accordance with the aim. Especially in case of intensively used agricultural soils the time needed for soil to come in balance with surrounding environment, either after cultivation or harvesting, is very often not enough long. In case of forest soils, intensive works can destroy soil balance as well. As consequence of intensive or not proper use, soil is very often expressed to many threats. Some of them are connected and one can accelerate another one, e.g. compaction can accelerate soil erosion. Almost all of soil threats are credits of human activities, or their intensity is influenced by man and it is now our task to reduce their occurrence and eliminate their influence on soil. The prevention is of course the best solution and can be realised via sustainable soil use in balance with surrounding environment. The most harmful is soil sealing, e.g. for building areas, because soil in this case passes up the majority of its functions and soil properties. Soil threats lead to soil degradation or even destroying. The "*EU Thematic Strategy for Soil protection*" aims to protect soil functions, both, natural coming out from pedogenesis and/or retrieved by human activities in the process of soil management and fertility improvement via cultivation. It is necessary to identify the areas that are at risk of irreversible or significant degradation by the major threats to soil: erosion, decline of organic matter, landslides, salinisation and *compaction.* The aim is to prevent further deterioration of soil properties as well as to stop the further spreading of the threats. Mentioned threats are soil and area specific.

1.1 Definition of Soil Compaction Threat and Soil Susceptibility to Compaction

Soil compaction is the rearrangement of soil aggregates and/or particles in the denser way when the voids and pores mainly between aggregates and particles are smaller, even missing in comparison with the arrangement of similar but not compacted soil. Orientation, size and shape of soil aggregates is evidence of compaction of the soil. Aggregates are arranged with longer side in horizontal way (platy structure), they do not have round shape but one side much longer than the other and depending on intensity of compaction they can be totally destroyed if the compaction is too big.

Soil susceptibility to compaction is the probability that soil becomes compacted when exposed to compaction risk. It can be low, medium, high and very high in dependence of soil properties and the set of external factors like, e.g. climate, soil use, etc.

Reasons for soil compaction and soil susceptibility to compaction can be different and are divided into two main groups: *natural* and *man induced* as well as one subgroup: *combined* which is result of two previous occurring simultaneously on the same place.

Natural reasons come out from specific soil properties, surrounding environment and climate. This type of soil compaction is called also *primary compaction*.

Man induced reasons for soil compaction [2, 3], *secondary compaction,* are coming out exclusively from the type of soil use – soil management. In many cases secondary compaction is induced in planned way, e.g. in civil engineering and it is not considered as negative effect, in the other types of soil use like in forestry and agriculture, soil compaction is considered as negative result of improper soil use [5] and there are efforts to remove it and soil compaction is considered in these cases as *soil threat*.

It is necessary to stress that the *prevention of soil compaction is the most important factor* in the combat this soil threat because the compaction itself can negatively influence the other parts of the environment and can create significant damage to soil functions. Compaction is also costly because decreases significantly crops and can influence even soil productivity. Both cases are money consuming which means the soil use is not economically positive. Also removal of secondary soil compaction can be very costly depending on the intensity of soil damage.

With prevention is connected evaluation of *soil susceptibility to compaction*. Soil susceptibility to compaction can be also divided, similarly to compaction, into two main parts: natural and man induced susceptibility. It is important to know which soil is susceptible to compaction to be able to apply proper soil use and cultivation, to prevent real compaction.

From environmental point of view soil compaction is soil threat and has negative effect not only on soil functions but also on the other parts of the environment [6]. It has typical feature of soil threats: it can induce or accelerate the other soil threats - mainly erosion and, when occurring in large scale as mass movement, also landslides.

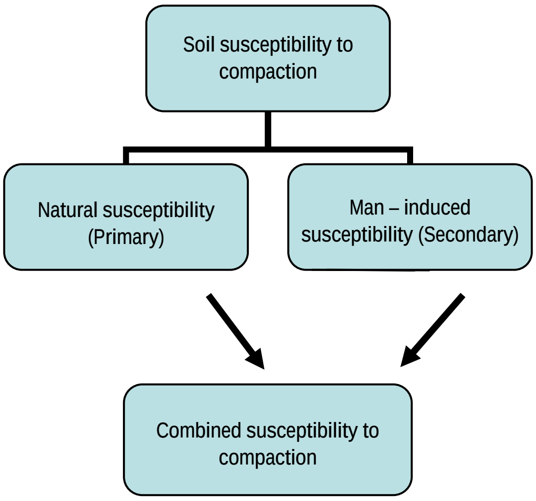

Figure 1. Three types of soil susceptibility to compaction

The new soil compaction map is dealing with natural soil susceptibility to compaction, which is a very important criteria for implementing effective prevention strategies but does not necessarily show the real status of compaction.

2. Material and methods

The proposed method for compilation of the map: 'The natural suspeptibility of soils to compaction' is based on the creation of logical connections between chosen parameters relevant in determining the soil susceptibility to compaction. These logical connections – *pedotransfer rules* – have been created by the use of parameters taken from the European soil database (ESDB), distribution version 2.0, which can be found on (http://eusoils.jrc.it/Website/eusoils/sg_attr.htm). The database itself is divided into four parts [4]:

1. The Soil Geographical Database of Eurasia at scale 1:1 M (SGDB)
2. The Pedotransfer Rules Database (PTRDB)
3. The Soil Profile Analytical Database of Europe (SPADE)
4. The Database of Hydraulic properties of European Soils (HYPRES).

Agricultural soils in Europe have been selected by use of CORINE land cover data [1]. For the purposes of the map compilation, the SGDB part of the database was used. The definition of SGDB is in Table 1 and explanations are as follows:

SMU – soil mapping units containing the geometrical part of information

STU – soil typological units having semantic type of information concerning soil type (WRB, 1998)

Table 1. Soil Geographical database of Eurasia (SGDB4_0.dbf)

COLUMN NAME	DESCRIPTION
SMU	SMU number
STU_DOM	STU number of the STU which is dominant in this SMU (i.e. the STU with highest PCAREA value for this SMU ; if there are equal maximum PCAREA values for two or more STU's, one of them is randomly chosen
PCAREA	How much area of SMU is occupied by the STU_DOM; expressed in percent

There are common rules between these attributes. On the map, each SMU has to be presented at least in one polygon. Each SMU has to contain at least one STU and on the contrary each STU has to be presented at least in one SMU. The relationship between SMU's and their STU's is described in the STU.ORG table.

Table 2. Soil Typological Units organization (STU.ORG)

NAME	DESCRIPTION	TYPE	SIZE
SMU	Soil Mapping Unit (SMU) identifier	Integer number	7
STU	Soil Typological Unit (STU) identifier	Integer number	7
PCAREA	Percentage of area of the Soil Mapping Unit (SMU) covered by the Soil Typological Unit (STU)	Integer number	3

Methods

The process of map construction was divided into three steps:

STEP 1: selection of relevant parameters from SGDB

STEP 2: evaluation of selected parameters

STEP 3: creation of Pedotransfer rules

STEP 1: Selection of relevant parameters from SGDB

Parameters have been selected according to the actual database content and their relevance for the natural soil susceptibility to compaction. The parameters selected for the development of *pedotransfer rules* are divided into two groups according to their relevance: main parameters and auxiliary parameters. In the final evaluation process the main parameters have priority but the role of auxiliary parameters is important mainly for evaluation in ambiguous cases. Soil type, texture and water regime are the most important parameters from the SGDB used in the evaluation process. Soil types are classified according to the international "World Reference Base for Soil Resources (WRB) classification from the year 1998. Reason for this is the fact that also the STU's in the database are evaluated according WRB (1998). Limitation to agricultural use is also applied as in the table as it is a direct indicator of the soil status and of a possible, very often natural, limitation for agricultural use.

The list of main parameters is in Table 3.

Table 3. Main Parameters of SGDB used in pedotransfer rules

CODE	PARAMETER
AGLIM1	The most important limitation to agricultural use of the STU
AGLIM2	Secondary limitation to agricultural use of the STU
WRB-ADJ1	First soil adjective code of the STU from the World Reference Base (1998)
WRB-FULL	Full soil code of the STU from the World Reference Base (1998)
TEXT-SRF-DOM	Dominant surface textural class of the STU
TEXT-SRF-SEC	Secondary surface textural class of the STU
TEXT-SUB-DOM	Dominant subsurface textural class of the STU
TEXT-SUB-SEC	Secondary subsurface textural class of the STU
TEXT-DEP_CHG	Depth class to a textural change of the dominant and/or secondary STU
WR	Dominant annual average soil water regime class of the soil profile of the STU

Auxiliary parameters are in Table 4:

Table 4. Auxiliary Parameters of SGDB used in Pedotransfer rules

CODE	PARAMETER
IL	The presence of an impermeable layer within the soil profile of the STU
ROO	Depth class of an obstacle to roots within the STU
WM1	Normal presence and purpose of an existing water management system
USE-DOM	Dominant land use of the STU
USE-SEC	Secondary land use of the STU

Some parameters which are auxiliary in the evaluation of the natural soil susceptibility to compaction can become main parameters in the process of a secondary (man induced) compaction evaluation.

STEP 2: Evaluation of selected parameters

Evaluation of selected parameters was done according to the expert knowledge of their properties and influence on soil susceptibility to compaction. Susceptibility of single parameters was divided into three categories:

1. L – low
2. M – medium
3. H – high

Not evaluated (*NE*) and not relevant (*NR*) categories are also in the dataset:

NE category is represented by soils in the towns and soils disturbed by man. These soils have not been evaluated because of lack of reliable information. It does not mean automatically that they can not have natural susceptibility to compaction, even if it is expected that secondary compaction processes will play a key role in their case. Also combined susceptibility to compaction (primary + secondary) will predominate here over natural one. In *the NR* category are the localities not relevant for such evaluation: water bodies, marshes, glaciers and rock outcrops.

The depth to impermeable layer and depth class of an obstacle to roots within the STU are auxiliary parameters used in combination with the main parameters, mainly texture and water regime. The ESDB database contains information concerning the location of the impermeable layer and of a layer being an obstacle for roots, however it is not mentioned the reason for this situation, e.g. rock, extreme values of pH, toxic environment, etc. Because of this, the information concerning these parameters can be used according to the assumption that decrease of soil depth decreases also its stability against external influences (e.g. weather in general, particularly heavy rain, floodings, etc.) and diminishes or excludes the area for plants roots development. Soil without continuous plant's cover is more susceptible to external factors influence, mainly in negative direction – deterioration of soil properties.

Evaluation of selected parameters is in following tables:

Table 5. Evaluation of susceptibility to compaction on the level of main reference soil groups

WRB (1998)-GRP CODES AND THEIR MEANING / SUSCEPTIBILITY TO COMPACTION								
AC	Acrisol	**M**	GL	Gleysol	**H**	SC	Solonchak	**H**
AB	Albeluvisol	**H**	GY	Gypsisol	**M**	SN	Solonetz	**H**
AN	Andosol	**L**	HS	Histosol	**L**	UM	Umbrisol	**L**
AT	Anthrosol	**NE**	KS	Kastanozem	**L**	VR	Vertisol	**H**
AR	Arenosol	**L**	LP	Leptosol	**L**	1	Town	**NE**
CL	Calcisol	**L/M***	LV	Luvisol	**H**	2	Soil disturbed by man	**NE**
CM	Cambisol	**M**	PH	Phaeozem	**L**	3	Water body	**NR**
CH	Chernozem	**L**	PL	Planosol	**H**	4	Marsh	**NR**
CR	Cryosol	**M**	PZ	Podzol	**M**	5	Glacier	**NR**
FL	Fluvisol	**M**	RG	Regosol	**L**	6	Rock outcrops	**NR**

*calcisols can have medium susceptibility in case of argic or vertic horizon presence

The main principles of the evaluation of soil susceptibility to compaction according to soil groups are coming out from the properties of group specific, profile forming horizons, from water regime and localization of soil groups in the terrain.

In principle, presence of horizon with high susceptibility to compaction increases inclination of soil to compaction even if according to the main group the susceptibility is low, e.g. Gleyic Umbrisol. On the contrary, high susceptibility of soil to compaction according to the main group decreases if in the profile of such soil is present horizon with low susceptibility, e.g. Mollic Gleysol.

Table 6. Evaluation of the susceptibility to compaction on the level of soil subunits (WRB_ADJ) present in the SGDB

WRB (1998)_ADJ CODES AND THEIR MEANING / SUSCEPTIBILITY TO COMPACTION								
ao	Acroxic	**L**	rz	Rendzic	**L**	rs	Rustic	**H**
ab	Albic	**M**	fr	Ferric	**H**	sz	Salic	**H**
an	Andic	**L**	fi	Fibric	**L**	sa	Sapric	**L**
ar	Arenic	**L**	ge	Gelic	**H**	so	Sodic	**H**
ad	Aridic	**NR**	gl	Gleyic	**H**	st	Stagnic	**H**
ca	Calcaric	**L**	gs	Glossic	**L**	ti	Thionic	**M**
cc	Calcic	**L/H***	ha	Haplic	**NR**	tu	Turbic	**M**
cb	Carbic	**H**	hi	Histic	**L**	um	Umbric	**L**
ch	Chernic	**L**	hu	Humic	**L**	vr	Vertic	**H**
cr	Chromic	**NR**	le	Leptic	**NR**	1	Town	**NE**
cy	Cryic	**H**	li	Lithic	**NR**	2	Soil disturbed by man	**NE**
dy	Dystric	**M**	Iv	Luvic	**H**	3	Water body	**NR**
et	Entic	**H**	mo	Mollic	**L**	4	Marsh	**NR**
eu	Eutric	**L**	pe	Pellic	**NR**	5	Glacier	**NR**
pr	Protic	**L**	pi	Placic	**H**	6	Rock outcrops	**NR**

*LOW- in case of diffuse form of secondary carbonates; *HIGH – in case of cutans and nodules

Soil texture plays key role in its susceptibility to compaction because influences directly ground bearing capacity [7]. Texture has also indirect influence through determination of total organic matter content according to generally valid rule stating that with increase of clay content increases also organic matter content. In the ESDB soil texture is presened extra for topsoil (TEXT_SRF), which can be dominant or secondary and extra for subsoil (TEXT_SUB), also dominant and secondary. In ambiguous cases the secondary texture can change final evaluation decision.

Table 7. Evaluation of soil susceptibility to compaction according to soil texture

TEXTURAL CODES IN ESDB	DESCRIPTION	EVALUATION
0	No information	**NR**
9	No mineral texture (Peat soils, rocks, etc.)	**L**
1	Coarse (clay <18 % and sand >65 %)	**L**
2	Medium (18 % < clay < 35 % and sand > 15 %, or clay <18 % and 15 % < sand <65 %)	**M**
3	Medium fine (clay <35 % and sand <15 %)	**M/H***
4	Fine (35 % < clay < 60 %)	**H**
5	Very fine (clay > 60 %)	**H**

*final evaluation can be influenced by amount of organic matter and sand

The depth to textural change (TEXT-DEP-CHG) was used as additional information in the process of soil texture evaluation because itself it does not give explicit result. In general, especially soils with shallow depth to textural change represent less stable environment in comparison with soils without textural change or when it occurs in deeper parts of the profile (80, 120 cm).

Table 8. Depth to the textural change in soil profile (ESDB)

TEXT-DEP-CHG CODES AND THEIR MEANING		EVALUATION
0	No information	**NR**
1	Textural change between 20 and 40 cm depth	**H**
2	Textural change between 40 and 60 cm depth	**H**
3	Textural change between 60 and 80 cm depth	**M**
4	Textural change between 80 and 120 cm depth	**L**
5	No textural change between 20 and 120 cm depth	**L**
6	Textural change between 20 and 60 cm depth	**H**
7	Textural change between 60 and 120 cm depth	**M**

Textural change in the profile influences many soil properties [7], mainly permeability, water regime and redistribution of moisture in both cases: coming from the top (source: irrigation, rain water) as well as coming from the bottom (source: ground water table). Usually deeper horizons have finer texture as horizons above them [8], but there are many cases when it is opposite.

Limitation to agricultural use gives also information of limitation for plant growth in general. It was the reason why also this parameter was included into evaluation of natural soil susceptibility. Categories connected with the use of agricultural machinery limitation have not been evaluated (2 and 3) as well as category 10: Soils disturbed by man. The codes are the same also in case of secondary limitation to agricultural use. Limitations to agricultural use with codes 16 and 17 (duripan and petrofferic horizon) are not present in the ESDB database.

Table 9. Evaluation of soil susceptibility to compaction according to primary and secondary limitation to agricultural use (AGLIM)

AGLIM CODES	DESCRIPTION	EVALUATION
0	No information	**NR**
1	No limitation to agricultural use	**L**
2	Gravelly (over 35 % gravel diameter < 7.5 cm)	**NE**
3	Stony (presence of stones diameter > 7.5 cm, impracticable mechanization)	**NE**
4	Lithic (coherent and hard rock within 50 cm)	**M/H***
5	Concretionary (over 35 % concretions diameter < 7.5 cm near the surface)	**H**
6	Petrocalcic (cemented or indurated calcic horizon within 100 cm)	**L/M/H***
7	Saline (electric conductivity > 4 mS.cm-1 within 100 cm)	**H**
8	Sodic (Na/T > 6 % within 100 cm)	**H**
9	Glaciers and snow-caps	**NR**
10	Soils disturbed by man (i.e. landfills, paved surfaces, mine spoils)	**NE**
11	Fragipans	**H**
12	Excessively drained	**L**
13	Almost always flooded	**H**
14	Eroded phase, erosion	**NR**
15	Phreatic phase (shallow water table)	**H**
18	Permafrost	**H**

*final evaluation of AGLIM depends on the texture of above lying soil horizons

Water regime (WR) [9, 10] is in the ESDB presented in classes as dominant annual average soil water regime class for given STU. In the Table 10 are classes of WR and their evaluation according to class influence on soil susceptibility to compaction.

Table 10. Dominant annual average soil water regime classes on STU level and their influence on soil susceptibility to compaction

CLASS	DESCRIPTION	EVALUATION
0	No information	**NR**
1	Not wet within 80 cm for over 3 months, nor wet within 40 cm for over 1 month	**L***
2	Wet within 80 cm for 3 to 6 months, but not wet within 40 cm for over 1 month	**M/H***
3	Wet within 80 cm for over 6 months, but not wet within 40 cm for over 11 months	**H**
4	Wet within 40 cm depth for over 11 months	**H**

*final evaluation depends on soil texture as well

Susceptibility of soil to compaction according to water regime class must be evaluated together with the other parameters, mainly soil texture and soil type, because it does not give explicit evaluation. Especially, evaluation of the class 2 depends on soil texture.

Auxiliary parameters

The depth to impermeable layer and depth class of an obstacle to roots within the STU are auxiliary parameters used in combination with main parameters, mainly texture and water regime. The ESDB database contains information concerning the location of impermeable layer and layer of obstacle for roots, however it is not mentioned the reason for this situation, e.g. rock, extreme values of pH, toxic environment, etc. Because of this, the information concerning these parameters can be used according to the assumption that decrease of soil depth decreases also its stability against external influences (e.g. weather in general, particularly heavy rain, floodings, etc.) and diminishes or excludes the area for plants roots development. Soil without continuous plant’s cover is more susceptible to external factors influence, mainly in negative direction – deterioration of soil properties.

Table 11. Depth of impermeable layer (IL) in soil profile of the STU in ESDB database

IL CODES AND THEIR MEANING		EVALUATION
0	No information	**NR**
1	No impermeable layer within 150 cm	**L**
2	Impermeable layer between 80 and 150 cm	**L**
3	Impermeable layer between 40 and 80 cm	**M/H***
4	Impermeable layer within 40 cm	**H**

*final evaluation depends on soil texture

Presence of impermeable layer in category 3 creates high susceptibility of soil to compaction when the soil texture for given STU's is in textural category 3, 4 or 5 (see Table 7).

Table 12. Depth class of an obstacle to roots of the STU

ROO CODES AND THEIR MEANING		EVALUATION
0	No information	**NR**
1	No obstacle to roots between 0 and 80 cm	**L**
2	Obstacle to roots between 60 and 80 cm depth	**M**
3	Obstacle to roots between 40 and 60 cm depth	**H**
4	Obstacle to roots between 20 and 40 cm depth	**H**
5	Obstacle to roots between 0 and 80 cm depth	**H**
6	Obstacle to roots between 0 and 20 cm depth	**H**

Depth class of an obstacle to roots was evaluated according to the same assumption as the depth to impermeable layer.

Soil use and presence of water management system were used as auxiliary premise parameters and have not been directly evaluated as the other parameters.

Soil use (code **USE-DOM**) describes the dominant and the most apparent land use for a STU. A second type of land use can be also taken into account (**USE-SEC).** Not all soil use categories from SGDB have been used in the process of soil susceptibility evaluation. There have been selected just those, which are directly connected with the natural environment, e.g. forest, halophyte grassland, grassland (see Table 13). The premise is that every selected land use (land cover) has specific properties which create also specific soil properties.

Table 13. Selected categories of land use from ESDB

USE-DOM AND USE-SEC CODES AND THEIR MEANING	
1	Pasture, grassland, grazing land
2	Poplars
4	Wasteland, shrub
5	Forest, coppice
9	Bush, macchia
10	Moor
11	Halophile grassland
22	Wildlife refuge, land above timberline

Information concerning existing water management was directed to the presence and purpose of an existing water management system (code **WM1**) and not to the type of the system. Such information includes the presence and purpose of an existing water management system in agricultural land on more than 50 % of the STU.

Table 14. Presence and purpose of an existing water management system

WM1 CODES AND THEIR MEANING	
0	No information
1	Not applicable (no agriculture)
2	No water management system
3	A water management system exists to alleviate waterlogging (drainage)
4	A water management system exists to alleviate drought stress (irrigation)
5	A water management system exists to alleviate salinity (drainage)
6	A water management system exists to alleviate both waterlogging and drought stress
7	A water management system exists to alleviate both waterlogging and salinity

Water management system was used together with information concerning soil water regime (Table 10).

STEP 3: Creation of Pedotransfer rules

Pedotransfer rules have been created for evaluation of the natural soil susceptibility to compaction within the STU's of given SMU's. The main premise is that every soil as a porous medium can be compacted. The soil susceptibility to compaction was divided into 4 categories. Two additional categories represent the data concerning places where this evaluation was either not relevant or could not been provided because of lack of information. In total there are 6 categories:

0. - no soil
1. - low susceptibility to compaction
1. - medium susceptibility to compaction
2. - high susceptibility to compaction
3. - very high susceptibility to compaction
9. – no evaluation possible

Category 0 – no soil; represents water bodies, glaciers and rock outcrops. Category 9 – no evaluation possible; was the case of towns including also soils, soils disturbed by man and marsh. For these situations, evaluation was not possible because of lack of relevant data.

Pedotransfer rules comprise the **basic assumption**:

IF the soil represents a given soil unit (WRB_GRP) with a given soil subunit (WRB_ADJ) and has a specific topsoil texture (TEXT_SRF) and a specific subsoil texture (TEXT_SUB) with a given depth to textural change (TEXT-DEPTH-CHG) and with a given water regime (WR)[1], and if there is/is not limitation to agricultural use (AGLIM)[2], THEN the soil has low, medium, high or very high natural susceptibility to compaction.

[1] WR can be possibly evident also from a given water management system (WM1)

[2] AGLIM can be possibly evident also from the depth to impermeable (IL) layer and the depth to the layer having obstacle for roots development (ROO)

Selected parameters of the ESDB have been evaluated according to this basic premise and natural soil susceptibility to compaction was set up for the purposes of the map construction. The greatest simplification was in two marginal situations:

1. -if all selected parameters show high susceptibility to compaction, than the soil has very high final susceptibility to compaction;
2. -if all selected parameters show low susceptibility to compaction, than the soil has low final susceptibility to compaction.

For the case 1, parameters with high susceptibility have been in the final evaluation considered with a higher value as it was in case 2, where parameters determining low susceptibility are represented. The reason for this assumption is that if a soil has all the properties relevant for susceptibility to compaction in a bad status (high susceptibility), its final susceptibility is very high. On the contrary, if a soil has all relevant properties in good status (low susceptibility), this fact will not influence the final already low susceptibility because every soil as a porous medium has some susceptibility to compaction, so always the susceptibility is present.

The rest of cases have been evaluated according to the **basic assumption reported above** and expert knowledge. Direct mathematical operations or pure combinatory analysis are not enough for any given evaluation due to the high complexity of the soil system.

3. Results and Discussion

The map of natural soil susceptibility to compaction was created from the evaluation of selected parameters, data of which are present in the ESDB.

Soils with naturally low (L) susceptibility to compaction

In the case of low susceptibility to compaction the following soil units and subunits are present:

AN: ao, dy, hi, hu, th; AR: ha, pr; CH: cc, ch, gs, ha; CL: ad, ha; CM: ca, cr, dy, eu, ge, hu, mo, CRan; FL: ca, dy, eu; HS: dy, eu; KS: cc, ha; LP: ca, dy, eu, ha, hu, li, mo, rz, um; LV: ar, cc, cr, ha; PH: ca, ha; PZ: et, ha, hi, le, um; RG: ca, dy, eu, ha; Um: ar.

These soil units and subunits have predominantly coarse or medium texture and water regime class 1 or 2. Soils with medium fine or fine texture can not have water regime other than 1. Soils with water regime class 3 or 4 are not present in this category.

Soils with naturally medium (M) susceptibility to compaction

Soil units and subunits having medium susceptibility to compaction:

AB: eu, st, ha, gl; AC: ha; AN: dy, hu; AR: ab; Ch: cc, ha; CM: ca, cr, dy, eu, ge, gl, mo, vr; CR: cc, ha, tu, um; FL: ca, dy, eu; GL: ca, dy, eu, ha, mo; HS: dy, eu; KS: ha; LP: ca, dy, eu, ha, mo, rz; LV: ab, cc, cr, dy, fr, gl, ha, vr; PH: ab, ca, gl, ha, lv; PL: dy, mo; PZ: cb, gl, ha, le, pi, rs; RG: ca, dy; VR: cr, pe.

These soil units and subunits have predominantly medium and medium fine texture and water regime class 1 or 2. Supplementary parameter – water management has in many cases code 3: “a water management system exists to alleviate waterlogging (drainage)” or code 6: “a water management system exists to alleviate both waterlogging and drought stress”.

Soils with naturally high (H) susceptibility to compaction

Soil units and subunits having high susceptibility to compaction:

AB: gl, ha, hi; AC: gl; AR: ha; CH: lv; CL: szn; CM: ca, dy, eu, gl, vr; CR: gl, hi; FL: ca, dy, eu, mo, um; GL: ca, dy, eu, hu, mo, so; GY: ad; HS: cy, dy, eu, fi, ge, sa; KS : cc; LP : ha, rz; LV : ab, cr, cc, gl, ha; PH ; gl, lv; PL : dy, eu, mo; PZ : gl, ha, pi, um; RG : ca; SC : gl, ha; SN : ha, mo; UM : gl; VR : cr, pe.

These soils are mainly in textural categories 3, 4 and 5. Main limitations for them are either excess of salts, shallow water table or they are flooded. Water regime class has all categories: 1, 2, 3 and 4. In case of category 1 there are other limitations, mainly excess of salts.

Soils with naturally very high (VH) susceptibility to compaction

These soils in general have several limitations. The main limitation is excess of salts, shallow water table, or combination of both, they might be flooded and water regime class is often 3 and 4. In the case that water regime class is 1 or 2, the soils are saline and heavy. In general, soils are mainly in textural categories 3, 4 and 5.

Soil units and subunits:

AB: ha; CH: cc; CM: cr; FL: ca, dy, eu, gl, hi, sz, ti, um; GL: eu, hi, hu, mo, ti; HS: cy, dy, eu, fi, ge, sa, sz; KS: lv; LV: cr, gl, ha; PH: gl, lv; PL: lv; PZ: gl, um; SC: gl, ha; SN: gl, mo, VR: ha, pe.

Two maps dealing with natural susceptibility of soil to compaction have been created: map with all soils (Figure 1) and a map showing agricultural soils (Figure 2).

4. Conclusion

In this study natural susceptibility of soils to compaction was evaluated. Susceptibility does not automatically mean that a soil is compacted. Real status of soil's compaction was not subject of this study because of the lack of actual data and because of non stable character of such threat. The study was based on the use of existing data from SGDB, Vers.2 from ESDAC database (JRC). Selected parameters, relevant to the evaluation of soil susceptibility to compaction have been set up and evaluated separately. Different combinations of selected parameters according to *Pedotransfer rules* were the basis of the evaluation process. Basic assumption for this study was that every soil, as a porous medium, could be potentially compacted. This implies that there are no soils without natural susceptibility to compaction.

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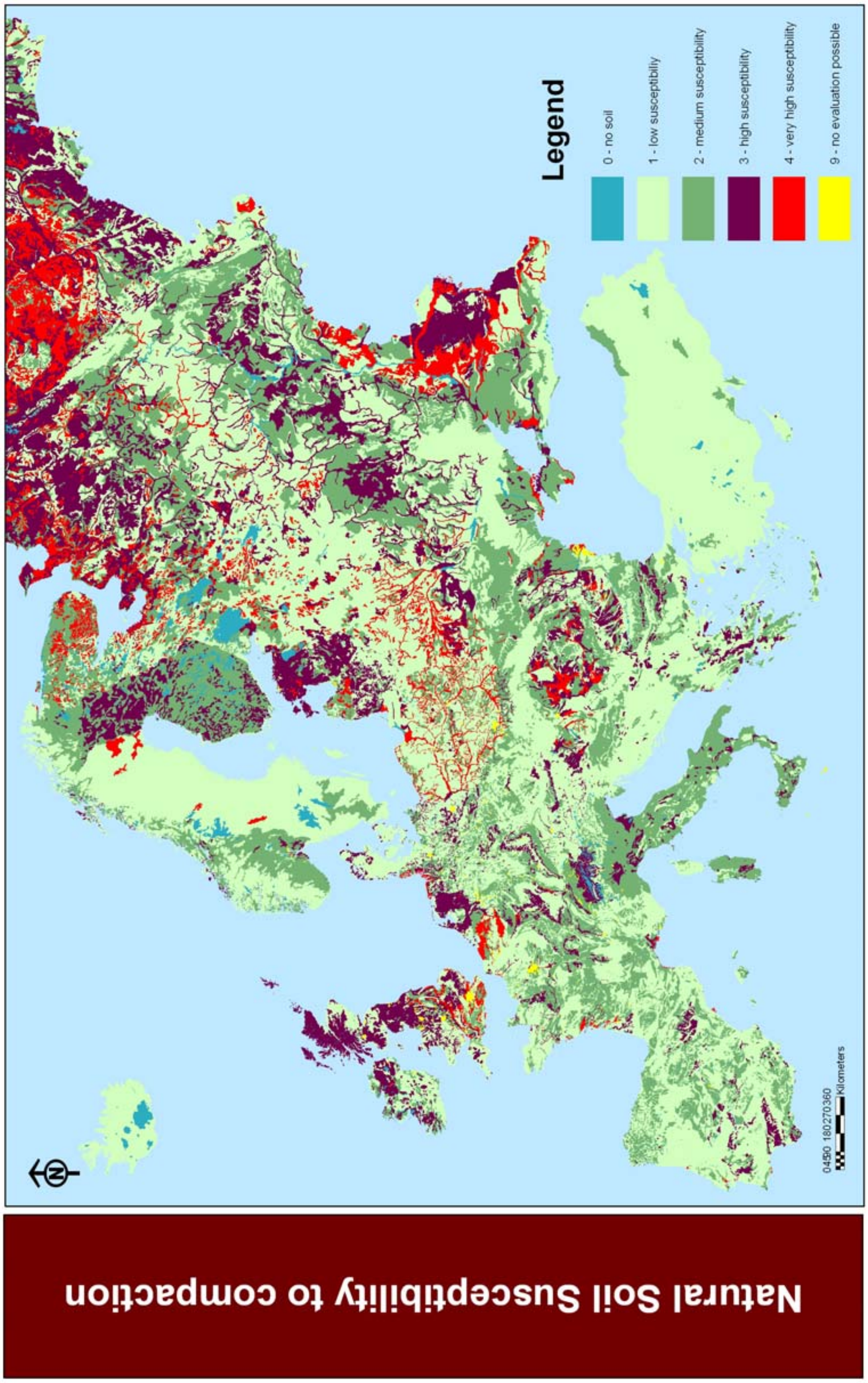
Natural Soil Susceptibility to compaction
Legend
0 - no soil
1 - low susceptibility
2 - medium susceptibility
3 - high susceptibility
4 - very high susceptibility
9 - no evaluation possible
Kilometers

Figure 1.

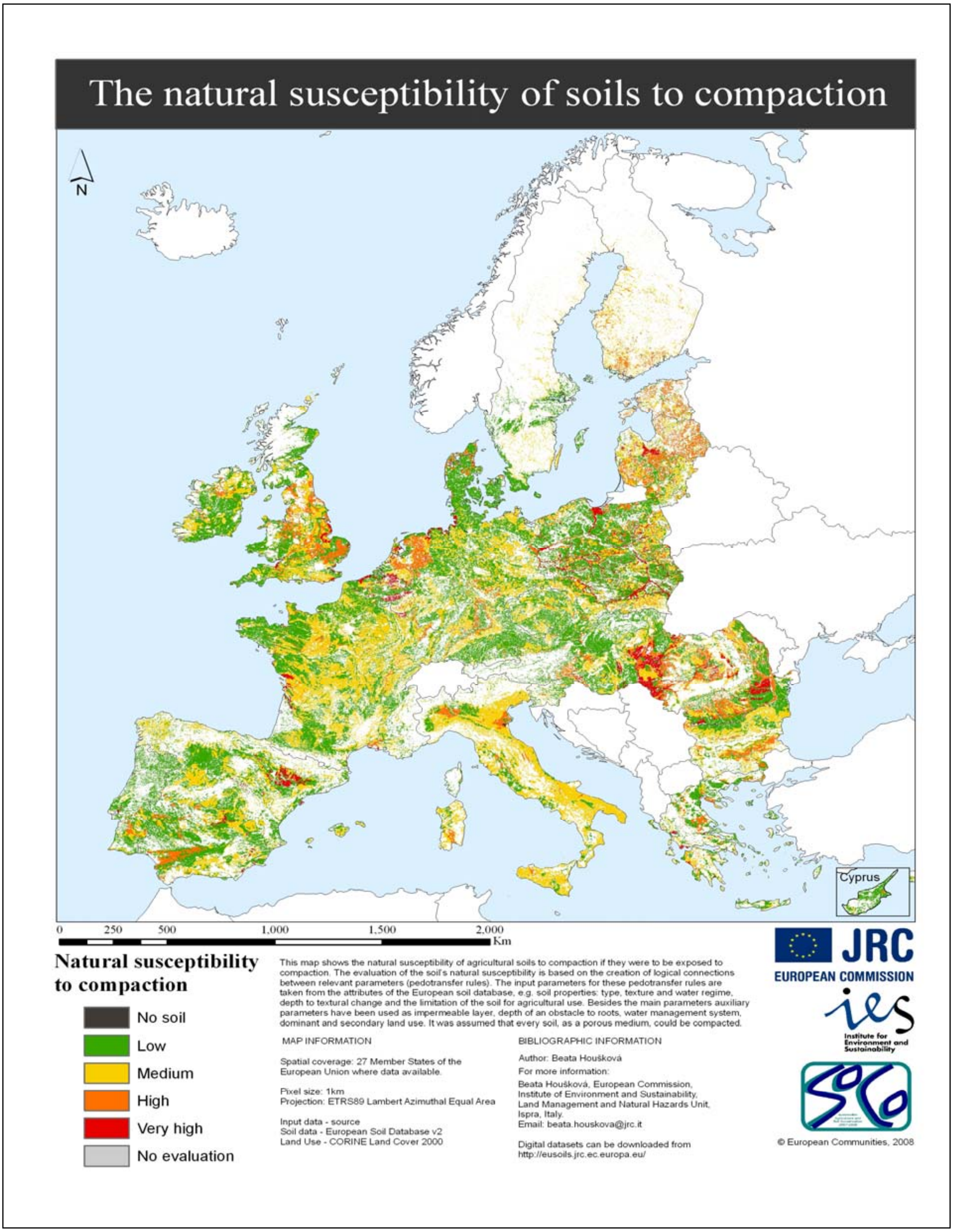

Figure 2. Natural Soil Susceptibility to Compaction in case of agricultural soils

Soil erosion: a main threats to the soils in Europe

E. Rusco, L. Montanarella, C. Bosco

Summary

Soil erosion is complex process with many influencing factors. In recent years the physical interpretation of the various factors and processes has intensified with the design and implementation of physically based models as PESERA (Pan-European Soil Erosion Risk Assessment) and WEPP (Water Erosion Prediction Project). Some major problems remaining are the level of detail of the input data for such models; the application of the model for large areas e.g. continental level; and the validation of the data output. Another fundamentally important aspect is the "static" output of the models. The Soil Erosion Risk maps of PESERA and WEPP represent a trend of average erosion risk based on historical data series. Changes in the political and environmental climate demonstrate a need to identify patterns of daily risk analysis, e.g. to include climate change in the analysis. In addition, soil erosion risk assessments directed at the implications for off-site damage should be predicted in order to prevent extreme events. The first steps have been made towards defining a model for assessing soil erosion risk based on meteorological forecast data. A proposed title of the model is DaFoSER, Daily Forecast Soil Erosion Risk. The issues, strengths and weaknesses of the DaFoSER design will be analysed below.

1. Background

Soil erosion is the wearing away of the land surface by physical forces such as rainfall, flowing water, wind, ice, temperature change, gravity or other natural or anthropogenic agents that abrade, detach and remove soil or geological material from one point on the earth's surface to be deposited elsewhere. Soil erosion is a natural process that can be exacerbated by human activities.

Soil erosion is increasing in Europe. Precise erosion estimates are not possible due to the lack of comparable data, therefore it is difficult to assess the total area of the EU affected by erosion [2].

In the 90's soil erosion research in a spatial context was conducted using several models. Here we will describe the CORINE EROSION model, RIVM model, GLASOD and a model by the EEA.

The CORINE EROSION model, based on the Universal Soil Loss Equation (USLE) approach estimates the soil erosion risk in Southern Europe. Figure 1 shows the framework methodology and the factors that have been taken into account.

One of the main problems of the CORINE erosion model is that the land cover is represented by just two classes: fully or not protected, and fully protected.

As part of a major report on strategies for the European Environment (RIVM, 1992), a baseline assessment of water erosion was prepared in 1990. This assessment of current risk (Figure 2.14) was combined with climate and economic projections within the framework of the IMAGE 2 model to generate scenario projections for 2010 and 2050. This approach, which is also expert-based, has the advantage of making explicit scenario projections, a feature lacking in the other approaches. Currently it is only available at a 50km resolution, so it cannot be readily interpreted at sub-national scales (Van Camp et Al., 2004).

The GLASOD project (Global Assessment of Soil Degradation) produced a worldwide soil erosion risk map based on expert judgment. The European part of the GLASOD map was afterwards updated through questionnaires that were sent to experts in all European countries. The feedback from the countries was not homogeneous and the related information of GLASOD soil erosion risk map varies greatly. The scale of the map is comparable to 1:10.000.000.

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[2] *SEC(2006)620 Impact assessment of COM (2006) 232 Soil strategy*

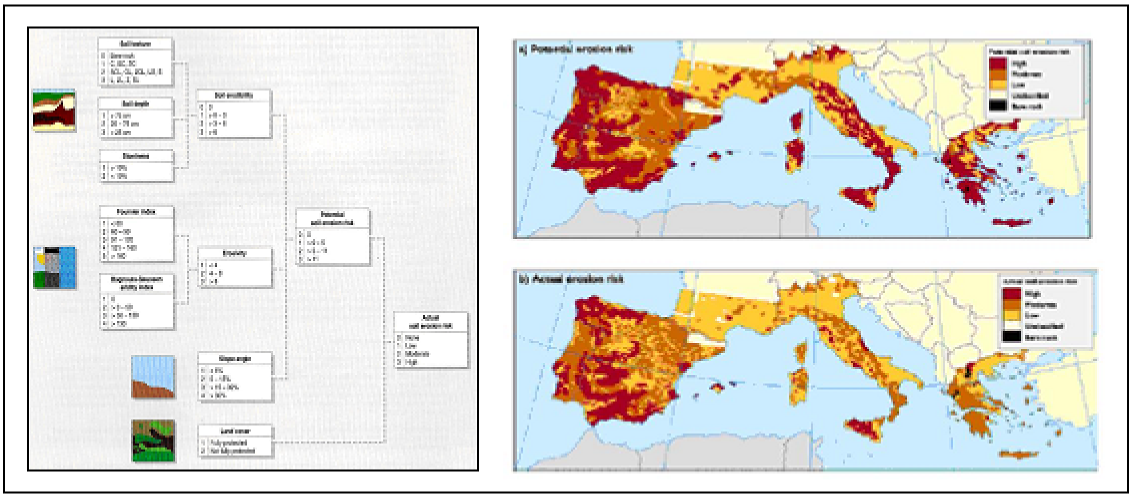

Figure 1. CORINE EROSION model schema and resulting maps (actual and potential soil erosion risk).

Another approach in the delineation of soil erosion was developed by the European Environmental Agency. The map produced has been developed from earlier maps (Favis-Mortlock and Boardman, 1999; de Ploey, 1989) based on local empirical data, and expert knowledge to identify rough zones for which the erosion processes are similar. Hotspots are then highlighted within each zone, and associated with the best estimates of soil erosion rate, from the literature.

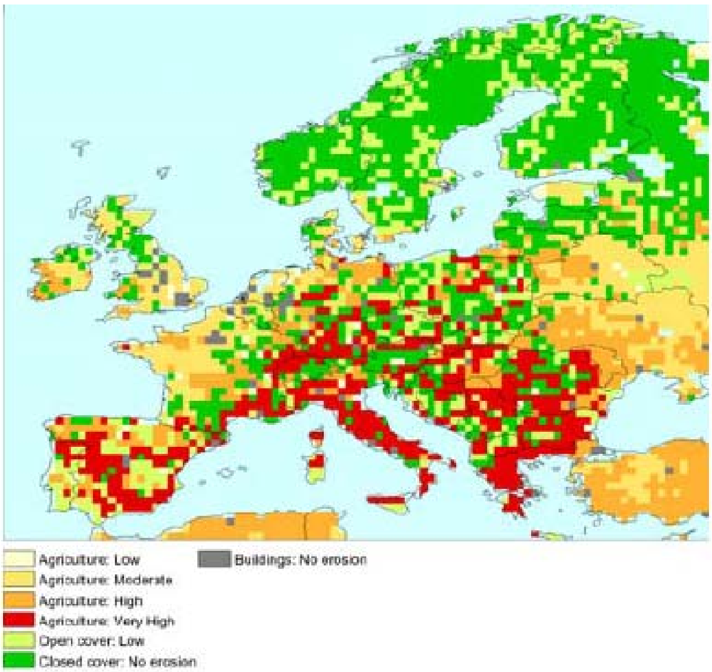

Figure 2. The RIVM water erosion vulnerability for 2050 (RIVM, 1992)

In the 21th century the risk of soil erosion is analyzed and studied through models that are increasingly integrated into a GIS environment (USLE, INRA model, PESERA). The first approach was the Universal Soil Loss Equation USLE (Wischmeier and Smith, 1978). The application of the USLE in Europe by Van der Knijff et al. (2000) is a first attempt to quantify soil erosion by rill and inter-rill erosion, based on a 1km x 1km data set for the whole of Europe.

Following this first evaluation INRA (Institut National de la Recherche Agronomique, France) developed a model based on a hierarchical multi–factorial classification designed to assess average seasonal erosion risk at regional scale. The hypothesis is simple and realistic: soil erosion occurs when the water infiltration rate is lower than the rainfall intensity causing increased run-off and consequent soil erosion. The INRA approach has the advantage of using detail data available at national level but, on the other hand, gives a qualitative estimation (five classes) of soil erosion.

The Pan-European Soil Erosion Risk Assessment (PESERA) represents the evolution of the different models previously above. PESERA is a physically based quantitative model. Unfortunately, as all physically based models, the application of the PESERA model requires good quality data to avoid errors and uncertainties.

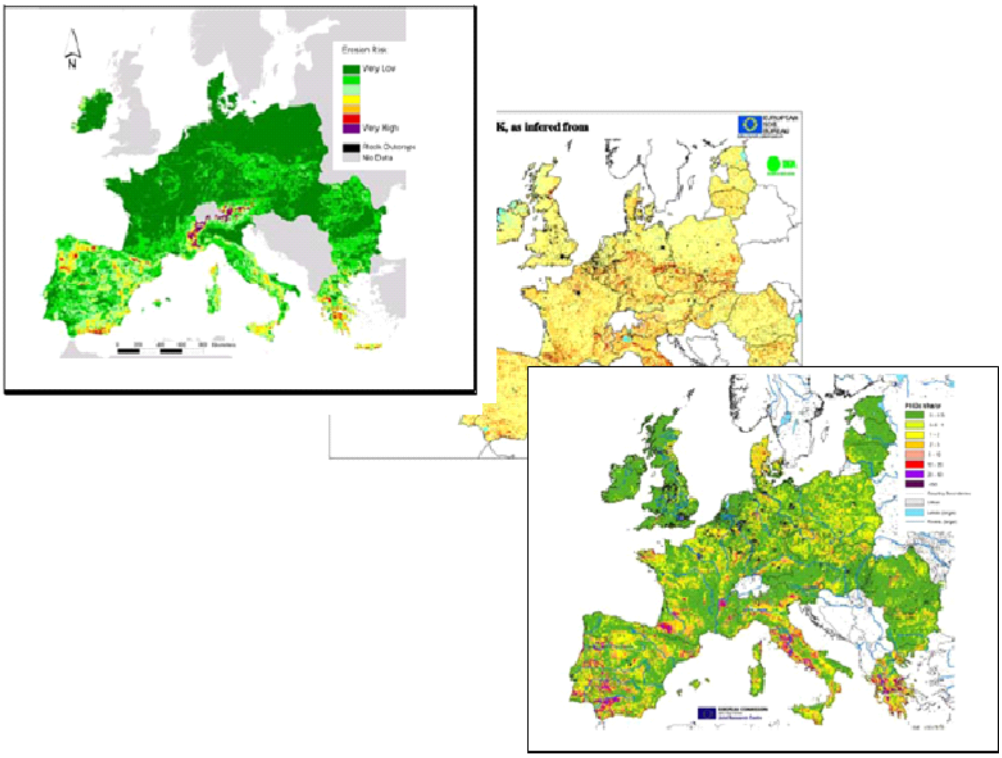

Figure 3. RUSLE, INRA and PESERA Soil erosion risk assessment

The data needs of the PESERA model are mainly based on the data available at European level like MARS climatic data and Soil Geographical Database at a scale of 1:1 million. Many soil related data are derived by pedotransfer rule.

Soil erosion is recognized as one of the most important soil degradation process worldwide. “An estimated 115 million hectares or 12% of Europe’s total land area are subject to water erosion, and 42 million hectares are affected by wind erosion”. [3] At present it is estimated that in the Mediterranean region water erosion could affect the loss of 20/40 ton ha^{-1} of soil after a single cloudburst, and in extreme cases the erosion could be even more than 100 ton ha^{-1}. Moreover the Impact Assessment of the Thematic Strategy underlay that the cost of soil erosion for the EU27 is € 0.7 – 14.0 billion.

[3] *Thematic Strategy for Soil Protection COM (2006) 231*

However, the erosive process can not be regarded as an isolated phenomenon. It is linked with other land degradation phenomena (e.g. compaction induced run-off). Toth et al. (2007) define the common criteria of risk identification throughout Europe according to the following conditions:

- identification of factors/hazards related to threat ('external' factors),
- characterization of receptor ('internal' attributes),
- performance specification, model selection (with data requirements).

Other aspects to be taken into account are on-site and off-site effects of the erosive processes. The following table shows the on- and off-site damage caused by water erosion.

Table 1: On-site and off-site damage due to soil erosion (Giordano, 2002)

Kind of erosion	On-site damages	Off-site damages
Water	• Loss of organic matter • Soil structure degradation • Soil surface compaction • Reduction of water penetration • Supply reduction at water table • Surface erosion • Nutrient removal • Increase of coarse elements • Rill and gully generation • Plant uprooting • Reduction of soil productivity	• Floods • Water pollution • Infrastructure burial • Obstruction of drainage networks • Changes in watercourse shape • Water eutrophication.

2. Soil erosion in a new perspective

All the above described layers related to soil erosion risk provide static and yearly analysis of erosion risk. There are differences between the models used and the output may be qualitative or quantitative. Many factors influence the erosion process; some of them are changing very slowly, others more quickly. A typical example is given by land use. Land use could change very rapidly on the basis of different pressures as policy[4] and price of the agricultural products. Unfortunately it is very difficult, if not impossible, to take into account these changes at a continental scale.

With the PESERA model the risk of erosion of 1990 was compared with that of 2000 using the Corine Land Cover 1990 and 2000 layer.

For policy purposes it is important to define changes in erosion rate with a time span of 10 years but as mentioned before land use change could be rapid.

Another fundamental factor for the soil erosion process is the erosivity. The erosivity is usually defined using climatic data with long time series (10-30 years).

[4] *i.e. change the obligatory set-aside rate from 10% to 0% for autumn 2007 and spring 2008 sowings. The current area under obligatory set-aside amounts to 3.8 million hectares in the EU*

Climate change will produce a change in the distribution and intensity of precipitation. It is essential to take into account this aspect in order to approach soil erosion issues appropriately . Increased off-site damage due to soil erosion is highly related to extremes storm events. Producing a soil erosion risk assessment in a predictive and dynamic way could help to avoid such phenomena.

Figure 4. Estimated change in erosion rate for Europe use CLC 90 and CLC 2000

In order to obtain soil erosion risk forecast assessments it is necessary to have up to date data and forecast data.

One of the possibilities for the land cover factor is to use the NDVI or EVI index derived by the MODIS sensor with a 10 days time series. Unfortunately, especially for northern Europe the NDVI and EVI index are not available during winter time mostly due to cloud cover.

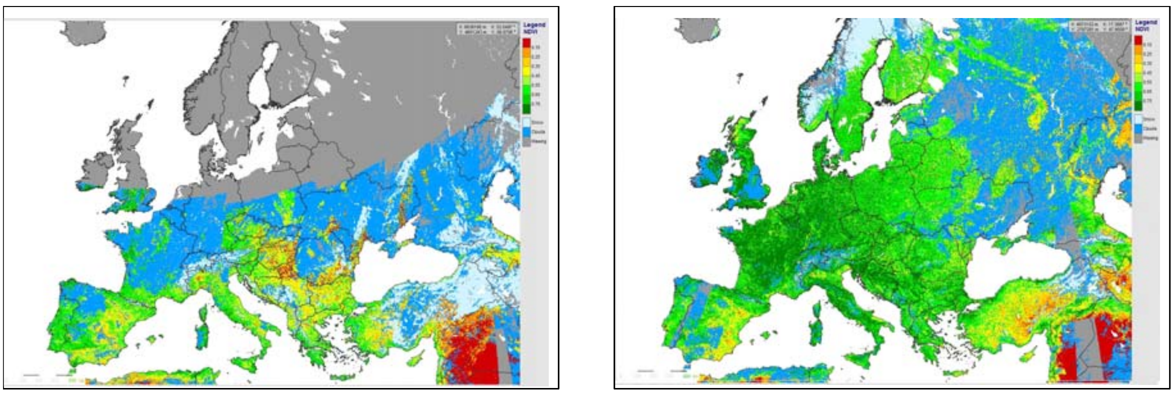

Figure 5. NDVI for Europe derived by MODIS for January and April 2007

Forecas used to determine rainfall erosivity can be supplied with different time scans: hourly, every three hours and every six hours or daily. The data are then provided on different sized grids .

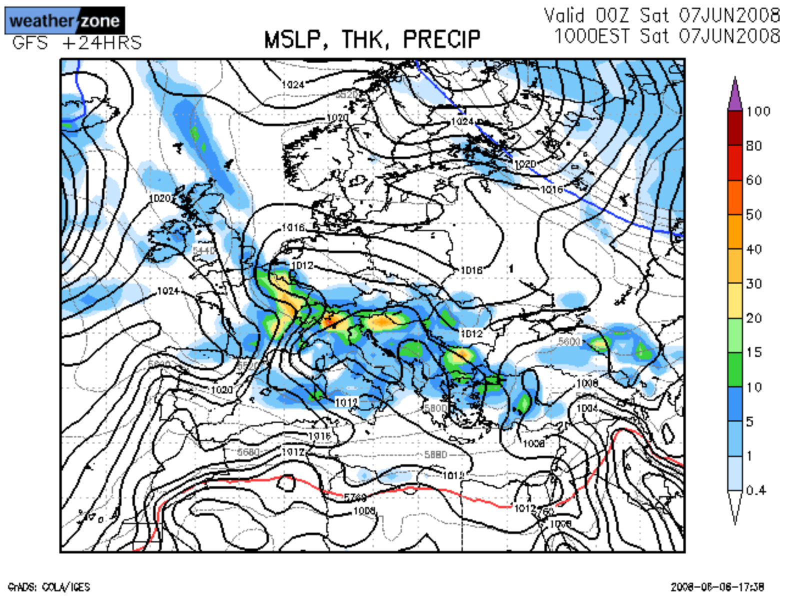

Figure 6. Example of temperature and rainfall forecast map

In literature there are some algorithms available to define the rainfall erosivity using hourly and 6-hourly precipitation data.

Wishmeier and Smith (1978) defined an algorithm for erosivity using 6-hourly data.

$$R = 27.38 * P2.17$$

Where P is the 6-hourly precipitation.

Bazzoffi (2007) defined an algorithm for the daily erosivity taking into account hourly precipitation and the maximum daily temperature.

$$\sqrt{R_g} = -0{,}648472 + 0{,}213979 * \text{mm} +$$
$$+ 0{,}144879 * \text{max\_piog/ora} + 0{,}048597 * T\text{max } °\text{C}$$

Where:

mm is the daily precipitation in millimeters,
max_piog/ora is the maximum hourly rainfall in millimetres,
T max is maximum daily temperature in Celsius degree.

Through the use of these algorithms it is possible to define the daily rainfall's erosivity.

Some steps in this direction have already been done. The Iowa daily erosion project producing daily estimates of rainfall, runoff, and soil erosion for the state of Iowa is particularly interesting. The results of the Iowa daily erosion project are obtained using the WEPP model (Water Erosion Prediction Model). WEPP is a physical based model and requires heavy input data. In order to fulfill all the WEPP requirements some elaboration on rainfall data is necessary to downscale the hourly rainfall to 15 minute periods. The Iowa daily erosion project uses the rainfall data of the day before the elaboration so the model is not able to predict the soil erosion rate but it is estimating yesterday's erosion rate through the WEPP model.

A new model for Europe, the Daily Forecast Soil Erosion Risk for Europe (DaFoSER) will have great implications in many sectors, a.o. civil protection, agriculture and flood risk. In order to reach the proposed results some conditions have to be met:

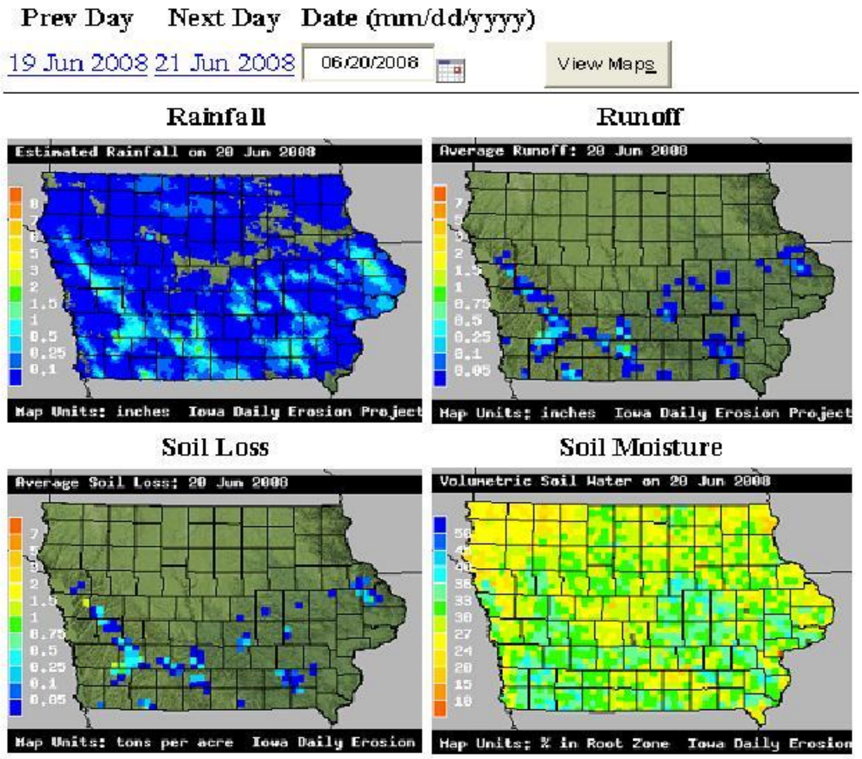

Figure 7. Example of estimated maps of rainfall, runoff, soil moisture and soil loss

- The model should be as simple as possible and adaptable to the different conditions throughout Europe,
- The system requirements should not be too heavy in terms of hardware and software,
- The data processing should be fast and allow forecast data every 3 days . In comparison; the PESERA model requires more than 5 days of processing to obtain the output at European scale,

The necessary input data for the model should be adaptable to a continental level. According to these needs the model will need to be designed in a flexible and adaptable way.

The following figure show the PESERA model schema and the major changes that will be made for the DaFoSER.

3. Conclusion

Currently the feasibility of this new approach for the soil erosion risk for Europe, the Daily Forecast soil Erosion Risk for Europe, is inbeing assessed. Many problems will have to be faced. One of the main issues is the decision on whether the model will become a physical-based or an empirical model. In the mean time a thorough analysis is being carried out on data availability and data needs. Some algorithms relating on the rainfall erosivity are being tested. This test phase is also includes remote sensing and the selection of the most suitable vegetation index for defining the cover factor.

In conclusion, the concept driving DaFoSER is to obtain an instrument able to define the real trend of soil erosion and the understanding of the changes taking place both from a climatic, political and institutional point of view.

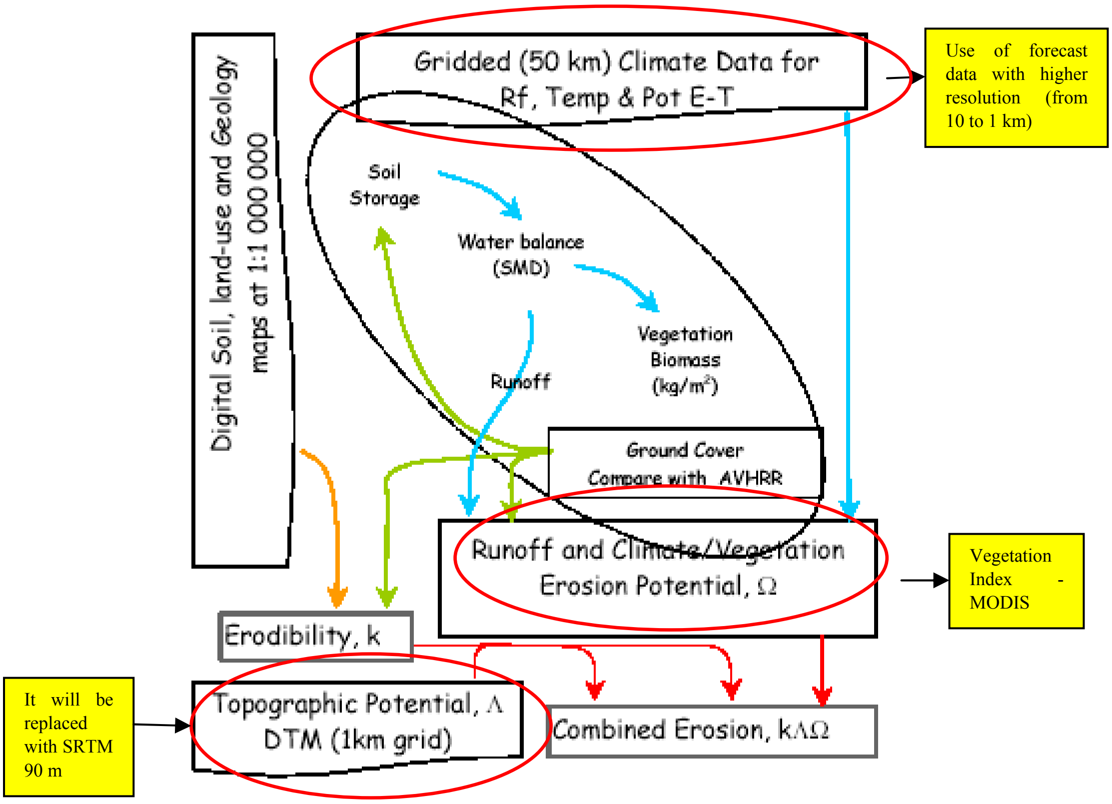

Figure 8. Major changes in DaFoSER compared to PESERA

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Soil erosion risk assessment in the alpine area according to the IPCC scenarios

C. Bosco, E. Rusco, L. Montanarella, S. Oliveri

Summary

The general aim of research was an estimation of actual erosion over the whole alpine space and a spatial analysis of soil erosion trends in different IPCC(Intergovernmental Panel on Climate Change) scenarios.

The Revised Universal Soil Loss Equation (RUSLE) was applied to the whole alpine space. It allowed to produce, with a spatial resolution of 100 m, the map of actual soil erosion and two further maps defining soil erosion rates in IPCC A2 and B2 scenarios. This analysis was carried out by means of the dataset the International Centre for Theoretical Physics (ICTP) of Trieste made us available. It provides daily rainfall values for the years 1960 – 1990 and for the IPCC A2 and B2 scenario 2070 – 2100.

From a comparison between actual erosion and soil losses in A2 and B2 scenarios it comes out that our model does not show relevant raises in erosion rates. However, low variations in soil losses rates are observable. In particular, B2 scenario shows a growth of low entity of soil losses over a significant part of the alpine space. In A2 scenario a clear distinction between northern and southern Alps comes out. Northern part should experience a low reduction of soil erosion, whilst in southern areas a rise of soil losses should take place.

1. A focus on mountain regions

Soil erosion is becoming one of the questions that most deserve the attention of the entire world community. A great number of studies have been carried out aiming at an evaluation of water erosion in the alpine territory. In some areas, in fact, the phenomenon is particularly intense and the erosion rates are high.

Soil erosion is a matter of primary importance in mountain areas. Increasing numbers of tourists, changes in farming/cultivation techniques and climate change are expected to intensify soil erosion in the Alps. The loss of soil from a field, the breakdown of soil structure, the decline in organic matter and nutrient, the reduction of the available soil moisture as well as the reduced capacity of rivers and the enhanced risk of flooding and landslides are processes linked with soil erosion. In all regions with steep relief and at least occasional rainfall, debris flows occur in addition to surface erosion processes. During the last decades the hazard was intensified due to land use change and climate change that all lead to an accumulation of natural disasters.

The general aim of our study was an estimation of actual erosion over the whole alpine space and a spatial analysis of soil erosion trends in different IPCC (Intergovernmental Panel on Climate Change) scenarios.

The analysis of the existing studies on the topic highlights that the main research methodologies have been developed to study erosion in agricultural contexts or hill areas with a mild climate. Therefore, it is difficult to apply these methods in mountain areas, also because of the extreme complexity of the alpine system.

For this reason, some researchers assert that the most common soil erosion models, as USLE/RUSLE or CORINE EROSION, can not be efficiently applied in an alpine environment because they were designed to be used on hilly agricultural areas where sheet and rill erosion processes are prevailing. Furthermore, the above mentioned models are not designed to consider some typical erosion processes of alpine areas as, for example, the debris flows.

An efficient model to analyse the real morpho-sedimental processes, should in theory be able to:

- minimize empirical factors and be based mainly on physically based factors.
- Use strong calculation methods.
- Combine all factors involved in the process.

A step forward has been made in this direction with the introduction of new-generation models, as i.e. WEPP (de Rosa, 2004). However, as regards the research related to erosion and, in this case, alpine areas erosion, the most used model is USLE (in one of its different versions: i.e. USLE, RUSLE). As a matter of fact, it is the only model in which input data can be obtained in different ways (measurement, estimation, interpolation).

Advanced models, as WEPP, have been and still are less used because they are often less flexible to be adapted to situations that have not already been parameterized before. Furthermore, USLE is a model used on differentiated spatial scales.

Another advantage in the use of USLE is related to its flexibility, as it is always possible to set this equation to adapt it to the environment to be analysed.

USLE is, as a matter of fact, a valuable means that has been and still is largely used, nevertheless, a complete development of models like WEPP or EUROSEM will probably cause its decline.

On the basis of the above mentioned considerations, we decided, to reach our aims, to use the RUSLE model. The main reason of this choice is that RUSLE has a more flexible data processing system. A further reason is the acquired experience in the application of RUSLE both on local and continental scale. On the contrary, it is useful to highlight the fact that, as already mentioned, the RUSLE model has been designed mainly for agricultural terrains. Its application in alpine areas could hence lead to a coarse estimation, from a quantitative point of view, of water erosion processes. However, it is necessary to take into account that our main objective is the assessment of the soil erosion in relation to climate change. From this point of view, the analysis of the cartographic printouts should be considered comparative and not absolute.

2. Quantitative analysis of actual erosion on the Alpine Space using RUSLE model

2.1 Input data and factors

RUSLE estimates erosion by means of an empirical equation:

$$A = R \times K \times L \times S \times C \times P$$

Where:

A = (annual) soil loss (t ha^{-1} yr^{-1}).
R = rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}).
K = soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1}).
L = slope length factor (dimensionless).
S = slope factor (dimensionless).
C = cover management factor (dimensionless).

P = human practices aimed at erosion control (dimensionless).

As spatial information regarding human practices aimed at protecting soil from erosion were not available, the P factor was set 1 and, actually, it has not considered.

The procedures used to estimate the different factors are explained in detail in the following paragraphs.

2.1.1 Rainfall-runoff

The RUSLE rainfall erosivity factor (R) indicates the climatic influence on the erosion phenomenon through the mixed effect of rainfall action and superficial runoff. The R factor for any given period is obtained by summing, for each rainstorm, the product of total storm energy (E) and the maximum 30 minutes intensity (I_{30}) (Wischmeier, 1959). Unfortunately these data are rarely available at standard meteorological stations.

The rainfall erosivity factor probably is, among the different components of the soil loss equation, one of the most difficult to derive, above all because rainfall data with adequate temporal resolution are very difficult to obtain over large areas. Rainfall data we could collect are not enough detailed to apply Wischmeier’s procedure to compute R factor over the whole alpine space.

This is the reason because simplified formulas, with lower temporal resolution, were applied.

Many of these formulas use the Fournier’s index (F) modified by Arnoldus (1980), other equations, instead, are exclusively based on the average annual rainfalls (P).

There are limited applications of these formulas at the alpine level and there is no consensus on which are the most appropriate algorithms to determine R factor instead of the EI_{30} in the alpine zone.

We have hence carried out a deep analysis of some of the mostly used formulas (Arnoldus (1980), Arnoldus (1977), Renard and Freimund (1994) - F, Renard and Freimund (1994) – P, Lo et al. (1985), Yu and Rosewelt (1996), Ferrari et al. linear (2005), Ferrari et al. exponential (2005), based on mean annual precipitation or on modified Fournier's Index (Table 1).

Table 1. Commonly applied equations to estimate erosivity

Author	Equation
Arnoldus (1980)	$R = [(4.17 * F) - 152]$
Arnoldus (1977)	$R = [0.302 * (F^{1.93})]$
Renard & Freimund (1994) - F	$R = [0.739 * (F^{.1.847})]$
Renard & Freimund (1994) - P	$R = [0.0483 * (P^{1.61})]$
Lo et al.	$R = [38.46 + (3.48 * P)]$
Yu & Rosewelt (1996)	$R = [3.82 * (F^{1.41})]$
Ferrari et al. (2005) – linear	$R = [(4.0412 * P) - 965.53]$
Ferrari et al. (2005) - exponential	$R = [0.092 * (P^{1.4969})]$

With this analysis we intended to evaluate the applicability of these methods, developed in different climatic zones, on the alpine region.

A statistical analysis was hence carried out to estimate the degree of correlation (Correlation Coefficient [R^2] and Root Mean Square Error [RMSE]) between R factor values computed by means of EI_{30} or using the simplified formulas. The analysis was carried out on rain data with high temporal resolution available for 42 meteorological stations in Veneto region, inside the alpine territory. Data were supplied by ARPAV (Agenzia Regionale per la Prevenzione e Protezione Ambientale del Veneto).

With the aim of computing the correlation between the simplified formulas and Wischmeier's R factor, Pearson (r) correlation coefficient was used.

R^2 is the square of the r correlation coefficient. It can be interpreted as the ratio between y variance imputable to x variance.

RMSE can be computed by mean of the following formula:

$$RMSE = \sqrt{\frac{1}{n} * \sum_{i=1}^{n} ki^2}$$

Where:

n = the number of location subjected to validation.
k_i = the difference between R estimated and R (EI_{30}).
Looking at data distribution (Figure 1), it comes out that all simplified formulas over or under-estimate R factor. Among all the other, with growing over or under-estimations at higher R values, Lo et al. (1985) equation show a systematic over-estimation. The Lo et al. formula shows the highest R^2 and among the lowest RMSE values. Compared to Lo's equation, Arnoldus (1980) formula, that is the wide used equation, shows a lower RMSE value but its R^2 is inferior and its trend inconstant: the higher are R (EI_{30}) values, the higher are the errors. The maximum error caused by Arnoldus is higher than the one using Lo's equation.

We decided hence to apply the Lo et al. (1985) equation to calculate the R factor of the RUSLE.

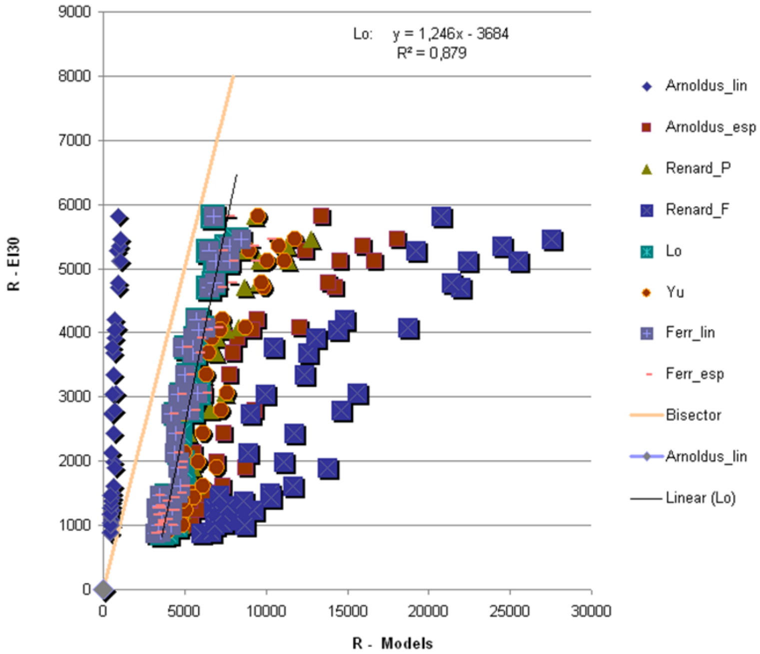

Figure 1. Comparison between R factor values obtained with EI_{30} method and simplified formulas

Ideally, none of the formulas we tested can be considered suitable for a quantitative estimation of erosion on the alpine territory. Unfortunately, the lack of data with adequate resolution got us to apply the best one among them.

The rainfall measurement data we used to determine rainfall erosivity factor on the whole alpine space have been provided by the International Centre for Theoretical Physics (ICTP) of Trieste. These data are the output of a prevision model of the climatic change (RegCM, Regional Climate Model), that provides the daily rainfall values for the years 1960 – 1990 and for the IPCC A2 e B2 (2070 – 2100) scenarios. RegCM is a 3-dimensional, sigma-coordinate, primitive equation regional climate model. Version 3 is the latest release. Different reasons explain the choice to use modelled instead of measured data:

- we decided to use dataset sharing the same origin to make more significant comparisons between actual and future rate of erosion.
- Modelled data have homogeneous spatial distribution and accuracy over the alpine space. They warrant to make analysis with the same level of accuracy over the whole study area.

With the use of GIS interpolation techniques and the application of Lo formula we obtained the spatial distribution of the rainfall erosivity factor in the Alps.

2.1.2 Soil erodibility

The soil erodibility factor K indicates the susceptibility of soils to erosion . It is defined as the unit erosion index for the R factor in relation to a standard fallow parcel (22.13 m length; 9% slope). On this basis, the value of factors such as length, slope, cultivation and anti-erosion actions becomes unitary. K is usually estimated using the normograph and formulae that are published in Wischmeier and Smith (1978). While these equations are suitable for large parts of USA, they are not ideally suited for European conditions. Romkens et al. (1986) performed a regression analysis on a world-wide dataset of all measured K-values, which yielded the following equation (revised in Renard et al.. 1997):

$$K = 0.0034 + 0.0405 * \exp\left[-0.5\left(\frac{\log D_g + 1.659}{0.7101}\right)^2\right]$$

Where D_g is:

$$D_g = \exp\left(\sum f_i * \ln\left(\frac{d_i + d_{i-1}}{2}\right)\right)$$

D_g is the geometric mean weight diameter of the primary soil particles (mm) and, for each particle size class (clay, silt and sand), d_i is the maximum diameter (mm), d_{i-1} is the minimum diameter and f_i is the corresponding mass fraction.

Information on soil surface texture were derived from the 1:1.000.000 Soil Geographical Database of Europe (ESGDB) (Heineke et al., 1998). Texture information in the database is stored at the soil typological unit (STU) level. Each soil mapping unit (SMU) is made up of one or more STUs. Due to the type of data concerning the soil texture in the database, some further data processing has been necessary. The processing led to the creation of a soil erodibility value in relation to the texture class (Van der Knijff et al., 2002). For each SMU, a K-value was estimated for all its underlying STU. Then a weighted average was computed, where the weights are proportional to the area of each STU within a SMU.

2.1.3 Slope and Length

The main innovation of the RUSLE model, in comparison with the original model (USLE), is the LS factor. The factor considers the flows convergence and is the result of the combination of the slope (S) and length (L) factors. Many methods have been proposed to improve the calculation of the topographic factor LS, but just in the last ten years a certain accuracy has been reached thanks to the implementation of GIS systems and of digital elevation model (DEM). The L Factor has been substituted by the Upslope Contributing Area (UCA) (Moore and Burch, 1986; Desmet and Govers, 1996), in order to consider the convergence and divergence of the superficial runoff. The UCA area is where water flows in a given cell of the grid. L and S factors have been determined through GIS procedures carried out using the following relation of Moore and Burch (1986):

$$LS = \left(\frac{A}{22.13}\right)^m * \left(\frac{sen\alpha}{0.0896}\right)^n$$

Where:

A = drainage area of a point belonging to a certain cell of the grid.
α = slope.

As suggested by many researchers, the values m and n are considered respectively as 0.4 and 1.3. For the calculation of the LS factor the DEM SRTM (Shuttle Radar Topography Mission) has been used. The resolution of the DEM is of 90 m.

LS calculation in complex hillslopes is generally problematic for traditional USLE applications, particularly when slope morphology shows great spatial variability (Moore and Burch, 1986; Engel, 1999; Mitasova, 2002). The topographic complexity of the alpine territory, consisting in steep slopes and complex ravine networks, presents significant challenges in estimating the S factor. Therefore, we preferred to modify Moore and Burch’s equation. The S factor has been evaluated using Nearing’s (1997) formula, that provides more reliable results at high slopes (more than 50%) than those provided by RUSLE, which is used in case of lower slopes.

$$LS = \left(\frac{A}{22.13}\right)^{0.4} * S$$

Where:

$$S = -1.5 + \frac{17}{\left(1 + e^{(2.3-6.1 sen\alpha)}\right)}$$

The formula was applied in a GIS environment. The A factor has been substituted with the result of the flow accumulation (Flowacc) multiplied by the pixel dimension (cell size). Flowacc consists of the number of cells bringing runoff water to each pixel in the grid.

2.1.4 Soil cover management

The soil cover factor represents the influence on soil loss of vegetation, terrain cover, agricultural activity, management of agricultural residuals and of soils. The C factor represents the relation between the soil loss in certain agricultural or cover conditions and the erosion that would be obtained from a standard fallow parcel (bare soil). The evaluation of this factor is difficult, because it always depends on changes in terms of environment, cultivations, agricultural activities, residuals management and on the morphology of the plant in the year. The C factor for a certain soil cover typology may have different values. Due to the lack of detailed information and to the difficulties in processing all factors on a large scale, it is difficult to use RUSLE guidelines to estimate the soil cover parameter. Therefore, the average values of literature have been used for this aim (Suri, 2002; Wischmeier and Smith, 1978). The necessary data to establish the C parameter have been provided by the Corine project, a European programme aimed at reproducing maps about soil use, analysing the image of the whole Europe provided by satellite. The calculation of the soil cover factor has been processed using the information layer Corine Land Cover 2000 (CLC 2000) third level. The information layer CLC 2000 is not available for the Switzerland territory. For this area, we decided to use the CLC 1990, in which the Helvetian region is covered. The legends of the Swiss and of the rest of the alpine territory information layers were different. Hence, an intervention aimed at uniforming the data was necessary. To this aim, everything has been traced to the 44 classes of soil use/cover established in the CLC 2000. A C factor value has been assigned to every class, based on literature data.

2.2 Results and discussion

By integrating the different factors of the RUSLE equation, it was possible to generate, on a GIS platform, the Potential and Actual Soil Erosion maps for the whole alpine space. They define, for each cell of analysis, the quantity of soil (t ha^{-1}) annually lost due to erosion processes.

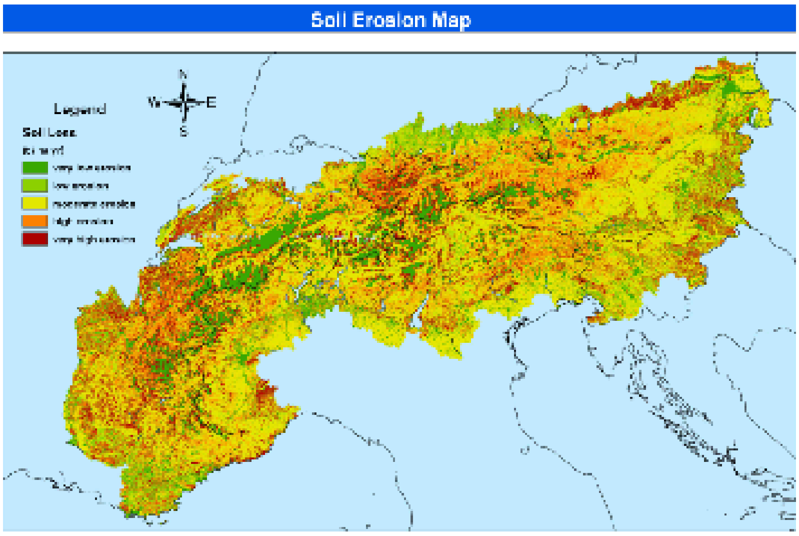

Figure 2. Actual Soil Erosion (t ha^{-1} yr^{-1})

Due to a systematic overestimation of the R factor calculated using the Lo's formula, we preferred to collect the values of soil erosion (t ha^{-1} yr^{-1}) in 5 different qualitative classes, deeming that a qualitative representation of the data obtained was preferable.

The potential soil erosion map points out the soil loss due to the action of physical factors involved in erosion processes. Hence, it does not consider the action of soil cover. The integrated reading of the two maps shows the fundamental role carried out by vegetation in areas potentially exposed to high erosion rates. The mitigating action of soil cover acts reducing kinetic energy drops of water reach the land surface with. It acts on their breaking action and, as a consequence, on translocation of soil particles (splash erosion). Besides, soil cover is a barrier against surface water flowing. This produces a further mitigation of erosive effect (sheet erosion).

By analyzing erosion values obtained with RUSLE application, it is evident that the alpine territory is subject to erosion phenomena. According to the classification we adopted, about 20% of the alpine space shows rather high erosion; nearly 30% shows a middle risk and the remaining 50% a low risk. Nevertheless, due to the extension of the Alpine Space (the way it has been defined by the Convention of the Alps), it is necessary to carry out a more detailed analysis, linked with geo-litho-morphologic and land use/cover parameters. As it has been previously pointed out slopes, slope length, pluviometric regime and soil cover play a crucial role in the erosive process. The study area was hence subdivided in some classes of landscapes, with the altitude acting as discriminating agent. Elevation shows, at least in the Alps, strong correlations with the other factors previously mentioned. The alpine space was therefore subdivided into four elevation zones:

- flat areas (< 300 m above sea level).
- Hill areas (300 – 600 m above sea level).
- Mountain areas (600 – 2000 m above sea level).
- High mountain areas (> 2000 m above sea level).

By analyzing the data relative to the elevation zones it is possible to notice the relative significance of the different factors of the model:

- in the areas below 300 m, more than the 80% of the territory shows low or moderate erosion, but the remaining 20% is characterized by high or very high erosion rates. The observation of the C factor map allows to understand that in these areas the role of cover vegetation is low, because the most of these areas are represented by arable land.
- At higher altitudes (300 – 600 m), the proportion of territory with an erosion rate low or moderate diminishes, whilst nearly 20% of the zone shows an erosion rate very high. This trend is caused by an increase in slopes which produces very high risk levels in areas with poor cover. On the other hand, the presence of wooded areas contributes in keeping high the percentage of territory with low risk level.
- In the mountain zone (600 – 2000 m), the high presence of forests increases, as regards to the lower zones, the percentage of territory with an erosion rate low or moderate and allows a reduction of the areas with very high soil losses.
- In the high mountain zone, erosion presents a very particular trend. More than 40% of these areas are not subject to soil losses. Moreover, more than 30% of the remaining territories are interested by high or very high erosion rates. This is easy to explain with lithologic considerations: at these altitudes, soil is often very thin and bare rocks crops out; but in the areas where soils exist, geo-morphologic characteristics, severe rainfalls and often lacking vegetation cover make them very vulnerable.

After all, without further deepening the item, it is possible to assert that alpine space is, due to its peculiarities, highly vulnerable to erosion risk. But the widespread presence of vegetation cover allows, in a significant part of the territory, to keep it under control and this is the reason because a right management of mountainous region cannot be disregarded.

2.3 Main limits

Erosion assessment in the alpine region, obtained by applying the universal soil loss equation, specifies the quantity of soil yearly moved from a catchment. Every soil particle can undergo different removal and sedimentation cycles. In its way downstream, a huge part of the removed material can sediment due to variations of slope, superficial roughness or land use/cover. RUSLE, which does not consider the sedimentation processes, tends to over-estimate soil loss. Besides, the model is not able to simulate gully erosion, mass movements and riverbed erosion.

The application of RUSLE over the alpine territory, moreover, presented huge difficulties mainly due to data availability problems. Unfortunately, we were not able to collect the whole set of data necessary for a strict application of the model. We have been often forced to make use of simplified equations, as for R and K factors, or use data with sub-optimal geographic scale, like for C, L and S factors. The simplified equation we used for R factor computation, in particular, though preferable to the other available, tends to over-estimate the measured rates of erosivity and makes scarcely meaningful a validation based on measured data.

These and many other uncertainties propagate throughout the model, resulting in an uncertainty in the estimated erosion rate. Despite these deficiencies and shortcomings, the methodology applied has produced valuable information on alpine soil erosion processes and on their distribution. The spatial analysis, in fact, has allowed the identification of areas which are likely to experience significant erosion rates. More detailed input data and more sophisticated erosion models might warrant a better quantitative estimation of soil losses due to water erosion.

It is however worth noticing that, in spite of a proper validation of the results is hardly possible and despite the over-estimation problems, the geographical distribution of soil erosion rates is congruent with the expected results. We draw this conclusion using satellite images and analysis through Google Earth.

3 Quantitative analysis of erosion trends on the Alpine Space using RUSLE model, in different climate scenarios

3.1 Scenarios data

IPCC identified a set of scenarios (Figure 4) based on socio-economic evaluation. They will produce an increase in greenhouse gases concentration and, as a consequence, temperatures could raise from 1.4 to 5.8 °C and rainfall regimes change. Every IPCC scenario comes out from different storylines. Each of them assumes a specific course of future development with diverging final conditions. They cover a while range of key future characteristics such as demographic change, technological and economic development.

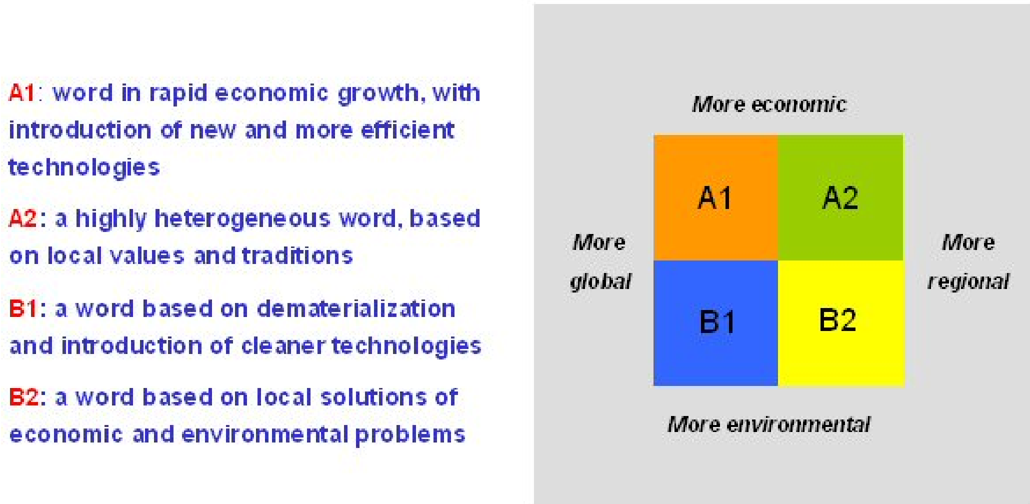

Figure 4. The different IPCC scenarios features

3.2 Input data and factors

Climate change potential in raising soil erosion is evident, but it is difficult to estimate. The aim of our study, in the framework of ClimChAlp project, was giving an estimation of the potential impact of changing climate on soil erosion processes inside the alpine region. The quantitative estimation of soil erosion trends resulting from ongoing climate change was carried out, like for the actual erosion definition, by means of the universal soil loss equation (RUSLE). The factor K, L, S and C coincided with the ones used for actual erosion estimation. Lo (1985) formula was instead used to generate new maps for the R factor. The ICTP provide climatic data, according the output of RegCM model, for A2 and B2 scenarios.

3.3 Results and discussion

The integration of the different factors of universal soil loss equation allowed to achieve two erosion maps based on climatic data referred to A2 and B2 (2070 – 2100) climatic data. These maps have been compared with the map of actual erosion. The analysis allowed the definition of soil erosion trends in relation to different scenarios of climate change (Figure 5 and Figure 6). From the analysis the following observations have been made:

- from a general comparison between actual soil erosion (1960 – 1990) and future soil losses (A2 and B2 scenarios. 2070 – 2100), it is evident that erosion rates remain nearly constant (Figure 7). The spatial extension of the soil erosion classes, in fact, is almost unvaried
- Some evidences arise from a spatial analysis of maps defining, for each grid cell, differences between actual erosion data and A2 - B2 scenarios (Figure 5 and Figure 6). B2 scenario shows a general growth of soil losses over a significant part of the alpine space. The increase is, however, of low entity. From A2 scenario comes out, instead, a strong distinction between northern and southern Alps. Northern part should experience a low reduction of soil erosion, whilst in southern areas a rise of soil losses should take place.

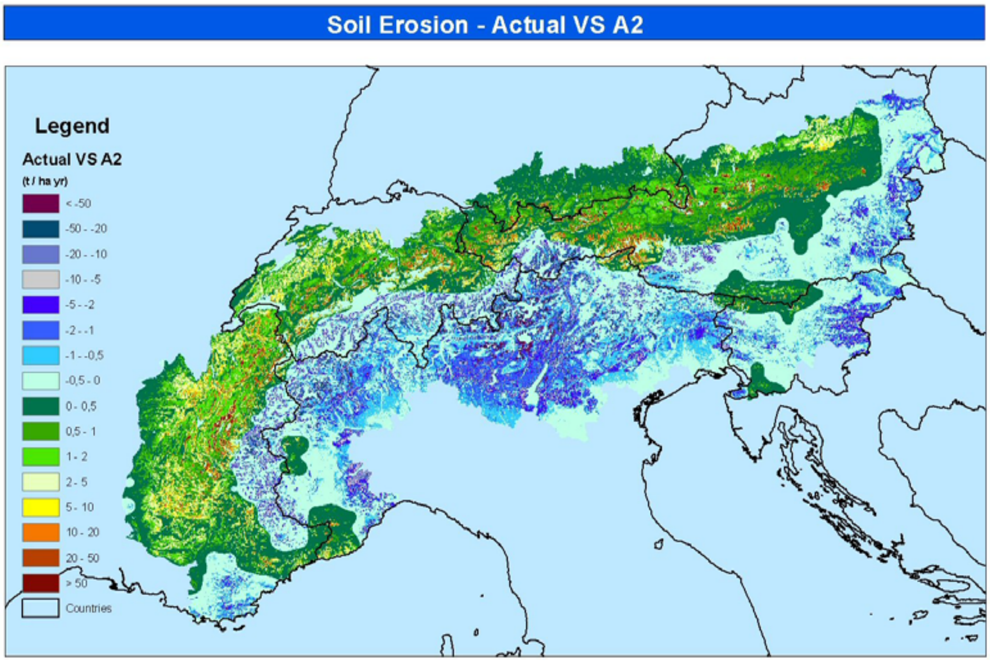

Figure 5. Soil Erosion trend. Actual vs. A2 scenario (t ha^{-1} yr^{-1})

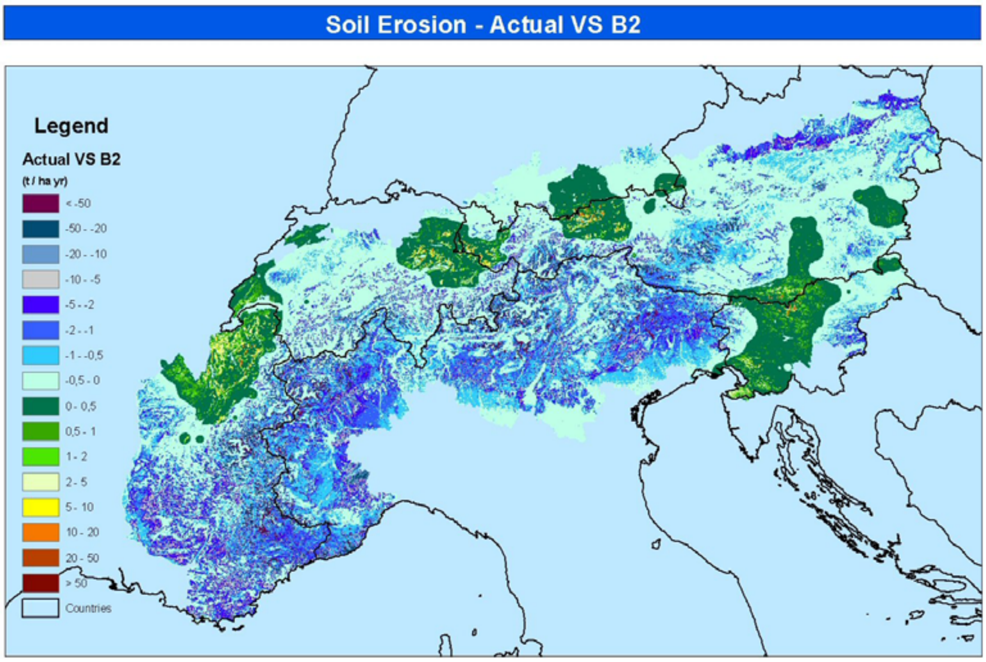

Figure 6. Soil Erosion trend. Actual vs. B2 scenario (t ha^{-1} yr^{-1})

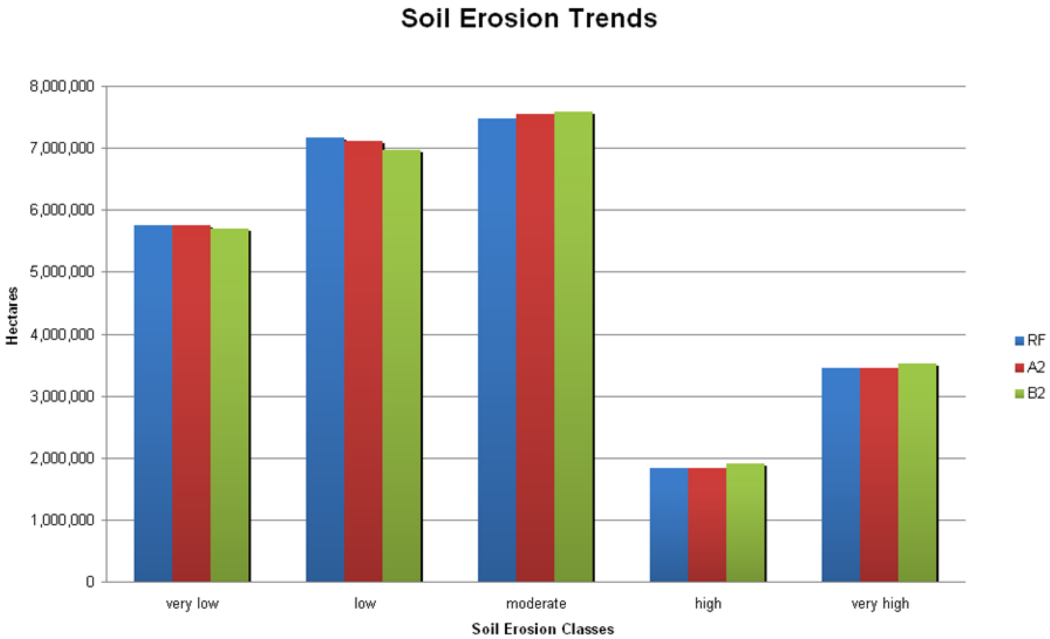

Figure 7. Spatial extension of soil erosion classes in the analysed scenarios.

Ongoing climate change contributes to arise the spatial variability of rainfalls. They should decrease in subtropical areas and increase at high latitudes and in part of the tropical zones. The precise location of boundaries between regions of robust increase and decrease remains uncertain and this is commonly where atmosphere-ocean general circulation model (AOGCM) projections disagree. The Alps are just located in this transition zone. This is the reason because, as a consequence of the expected climate change, a very little variation in soil erosion rates over the alpine space was predictable. RegCM model, which produced rainfall data used in this study, places the transition zone more southward in B2 than in A2 scenario. Due to this difference in the placement of the transition zone, even though A2 scenario foresees heavier climate change than other scenarios, the B2 scenario shows, over the Alps, higher rainfall rates. This is the reason because in B2 scenario a higher number of areas with high erosion are present. In A2 scenario, moreover, prevailing winds come from the south. This explains the sharp demarcation line between northern and southern Alps and the increase of rainfalls on the southern side. B2 scenario is characterized by a low increment in soil erosion rates, even if some isolated areas present an opposite trend, which is difficult to explain. The investigation of these phenomena requires further analysis, going beyond the aims of this study. They are possibly explainable from a modelling point of view and could be due to non linearity problems, easily coming out at these scales. To justify their origin different models should be used, with the aim of a deeper calibration of results. This is the reason because IPCC derived results of its four report on climate change making use of 20 climate models.

As mentioned before, soil erosion trends in the alpine region are mainly attributable to changes in rainfall regimes. A better estimation of soil losses in climate change scenarios could be assured by evaluating future variations of cover management factor.

4. Conclusions

With this study, soil erosion processes over the whole alpine space were estimated, with the specific aim of comparing actual erosion rates (1960 – 1990) with the A2 and B2 (2070 – 2100) IPCC scenarios data.

The alpine region is a highly complex area. The study shows that the territory of the Alps is subject to a high vulnerability with regard to soil erosion. About 20% of the alpine space, in fact, shows a high risk of erosion. In the high mountain zones, in particular, more than 25% of the territory is interested by very high erosion rates. Vegetation cover plays a key role in mitigating the soil loss processes. The vulnerability of the Alps is mainly imputable to the geomorphologic complexity of their territory and to the type of rainfall they are subject to. As regards erosion trends in future climate scenarios (IPCC A2 and B2 data), our methodology points out that over the alpine space raises in soil losses are not expected to be significant. In spite of that, some evidences come out: B2 scenario shows a growth of low entity of soil losses over a significant part of the alpine space. In A2 scenario a clear distinction between northern and southern Alps comes out. Northern part should experience a low reduction of soil erosion, whilst in southern areas a rise of soil losses should take place.

One of the most critical points in soil erosion studies is the estimation of rainfall erosivity. The climate model we used makes available rainfall data with daily resolution. It is not adequate for the strict computation of R factor in the universal soil loss equation, which requires a temporal resolution of 30 minutes.

To get round this problem, simplified algorithms can be applied. By the use of monthly or yearly average rainfall data they allow the computation of R but they do not take into consideration the specific intensity of single rainfall events. As climate change could produce a tendency to tropicalization, with highly intense rainfall events, a complete understanding of climate change impact on erosion trends is difficult to obtain.

As regards future investigations, a downscaling approach will be useful for a better comprehension of water erosion processes in the Alps. It would allow to carry out more and more accurate predictions and estimations at regional or local scales. To this aim, evaluations on possible future variations of timberline, pastures or woods distribution and vegetation cover are critical. It is moreover crucial to take into better consideration the erosivity power of snow melting processes, which can strongly influence soil erosion phenomena, both sheet and rill. Above all, further researches should be carried out with the aim of a better determination of rainfall erosivity factor. It still persists, moreover, the need for a deep calibration and validation of models, by means of field measures and with monitoring activity on soil and their degradation.

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An example of soil threat evaluation: wind erosion assessment using DSM techniques

H.I.Reuter, F.Carre, T.Hengl, L.Montanarella

Summary

The Soil Thematic Strategy - adopted in September 2006 by the EC - is in its political discussion process with the primary goal of soil protection and soil sustainability. Wind erosion is one of the threats for soil outlined in there. DSM with its quantitative prediction of uncertainty has distinctive power for assessing soil relatively to environmental issues (called Digital Soil Assessment). In the paper it is applied for the prediction of wind erosion events using daily long term time series of meteorological data for the last 30 years. A regression kriging (RK) estimation approach used 1200 profile observations together with information on parent material, DEM - and remote sensing parameters to estimate clay, silt and sand content (in %) and its related uncertainty for the area of Czech Republic. Based on these derived parameters, texture class percentages have been derived in four different ways: 1) using the Dominant Soil Surface Texture of the European Soil Database (ESDB); II) using the estimated texture based on the RK, III) as well a Best Case - and IV) Worst Case (minimum clay, maximum sand content) scenario especially tailored for wind erosion. The number of erosive days on bare agricultural soil was computed based on the Wind Force Integral, by using two climate data sets: daily wind speed, precipitation and evaporation data for 1961-1990 and for a climate scenario 2071-2100. Lower and upper limits of clay content and therefore wind erosion aggregate stability show severe differences for each single location, which are not possible to estimate with the ESDB dataset. The consequences of those different numbers are amplified in the number of Erosive days. ESDB and Worst Case-Scenario deliver similar numbers of ED, whereas the RK dataset shows a reduction to one/fourth of the number of EDs. The climate scenarios allow for the forecast that areas in eastern CZ will be more prone to wind erosion events in the future, whereas an overall decrease can be observed, which allow for future predictions of soil quality in this areas.

1. Introduction

The Soil Thematic Strategy has been adopted in September 2006 by the European Commission to improve the protection of European Soils. Wind erosion is one of the threats outlined in there. Erosion of soils by water and wind has been recognized as major threats to soil quality (Goossens 2004; Tóth 2006). Wind erosion especially submits soil particles into the atmosphere, thereby affecting physical and chemical processes. Dust particles in the atmosphere for example affects the radiative forcing, chemical reactions in the atmosphere, as well as biological systems (Kohfeld and Harrison 2001), and therefore might be one source for climate change processes (Miller and Tegen 1998).

The amount of dust emission by wind erosion into the atmosphere from agricultural fields in Europe is assumed to be much smaller than that of natural origin like the Sahara desert. Still, effects can be observed not only on large scale global systems, but also on the European Scale. Emitted soil particles influence air quality (Jaecker-Voirol and Pelt 2000) – compare Clean-Air Protection Act) and have effects on the human respiratory system (Dogan 2002); (Smith 2003)). Wind born soil particles also pollute occurs in rain, water, vegetation and soil (Schulz 1992), Reheis et al. 1995, Sterk et al., 1996, Pelig-Ba et al. 2001). Although some areas gain soil particles due to wind erosion events, even more significant is the loss of fine soil material as well as organic matter from the soil surface in the short term and decrease in soil fertility and water storage capacity in the long term. Goossens (2003) summarized wind erosion effects for on-site effects for soil degradation, abrasion damage and other damaging effects like accumulation of deposits. Further on, off-site effects are divided into short term effects like reduced visibility and deposition of dust anywhere. Long term effects include lung diseases, or contamination/infection via the transported materials.

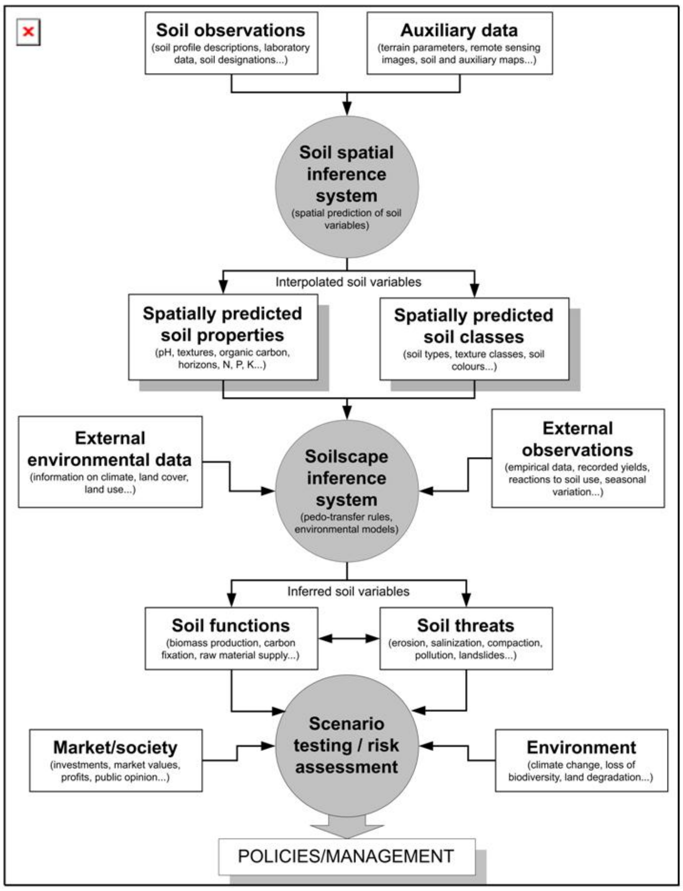

Figure 1. Graphical Representation of the DSM/DSA/DSRA workflow

Digital soil mapping in combination with digital soil assessment has the distinctive power to allow for a complete Digital soil Risk assessment (see for a more in depth discussion Carré et al, 2007; Tóth 2004). Thereby a complete assessment process with its related uncertainties can be performed. In this overview described below we wanted to apply the process as outlined in Carré et al (2007). Starting from soil observations together with several auxiliary datasets, a soil spatial inference system has been developed, which predicts several soil properties. Based on this results, a pedotransfer rule together with land use data is used as a soilscape interference system to allow for evaluating of a soil function – in that case the aggregate stability against wind erosion. The scenario testing has been performed to allow for a risk assessment using a reference and a climate scenario as data coming from the environment. We did not include any market data as well as possible adaption effects from society. The overall goal of this workflow has been to outline areas at risk for a decrease of soil quality using the indicator wind erosion. Thereby, in contrast to previous mapping approaches not only yearly average values on a single country have been used, instead daily long term time series of meteorological data for the last 30 years have been applied in a consistent matter.

2. Methods

A regression kriging (RK) approach used 1200 profile observations together with information of parent material (ESDB V2), DEM - and remote sensing parameters (EVI mosaics obtained from MODIS) to estimate clay, silt and sand content in % and its related uncertainty for the area of Czech Republic (Figure). The continuous predictors have been: DEM, SLOPE, MEANC, TWI, SOLRAD, INSOL, LSFACT, PERC, EVI097, EVI161, EVI193, EVI225, EVI257, EVI289 and the crisp predictors unknown parent material, consolidated clastic sedimentary rocks, sedimentary rocks, igneous rocks, metamorphic rocks, alluvium, Aeolian deposits. For a more in-depth description of some of the data preparation/treatment applied please refer to Rodriguez et al. (2008), the general process of regression kriging is described in detail in Hengl (2007).

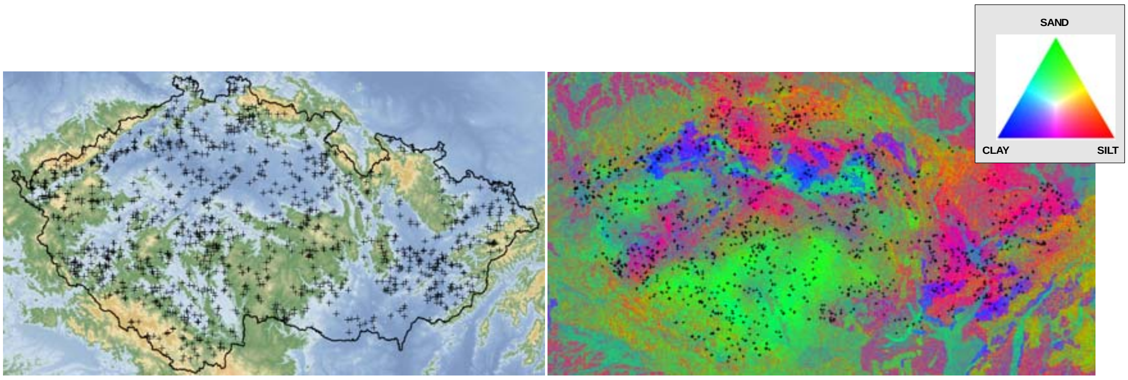

Figure 2. Location of soil profiles used in the RK-approach(left) and predicted soil texture(right).

As a baseline scenario we used the Dominant Soil Surface Texture as provided in the ESDB V2.0 Database (ScI). Further on we derived based on the RK-predicted clay, silt and sand content the following three additional scenarios: the estimated texture based on the RK (ScII), a RK-Best Case (ScIII) - and a Worst Case (minimum clay, maximum sand content) Scenario (ScIV) especially tailored for wind erosion. Depending on the various textured materials reported, different friction velocity thresholds were assigned: 5 m/s for coarse, 7 m/s for medium, 9 m/s for fine textured and 11 m/s for very fine textured soils. Organic soils were assigned a threshold value of 8 m/s.

Two parameters for the evaluation of wind erosion in the context of DSA have been selected. The wind erosion aggregate stability has been derived using a pedo transfer rule (PTF) for the four different texture scenarios using:

$$\text{ASEAGS} = 0.83 + 15.7 \text{ x clay} - 23.8 \text{ x clay}.$$

Second, the Wind Force Integral (WFI-Beinhauer and Kruse, 1994) has been used as a parameter to compare erosivity of the meteorological conditions with respect to the underlying soil surface across different years. The WFI defines the potential transport capacity of the wind at the soil surface as a function of the wind force and the surface moisture providing that the following conditions are met (I) precipitation < 0.3mm in that time step (e.g. no rain event), (II) precipitation (m) < evaporation (m), and (III) the average wind speed (u) in m s^{-1} is above the soil textural threshold (uthr) in m s^{-1}

If all three conditions are met, the WFI is computed as:

$$WFI = \sum_{1}^{n} (u - u_{thr}) \times u^2$$

for the time steps (n) per month/or year. We applied this formula for two climate data sets: daily wind speed, precipitation and evaporation data for 1961-1990 and for a climate scenario 2071-2100, derived from the PRUDENCE high resolution climate prediction. A mask (based on CORINE land use –see Figure 3) was used to differentiate between agricultural and non agricultural areas, which are not subject to wind erosion events. For the agricultural areas we assumed bare field conditions as no information was available for surface cover fraction in a consistent time series.

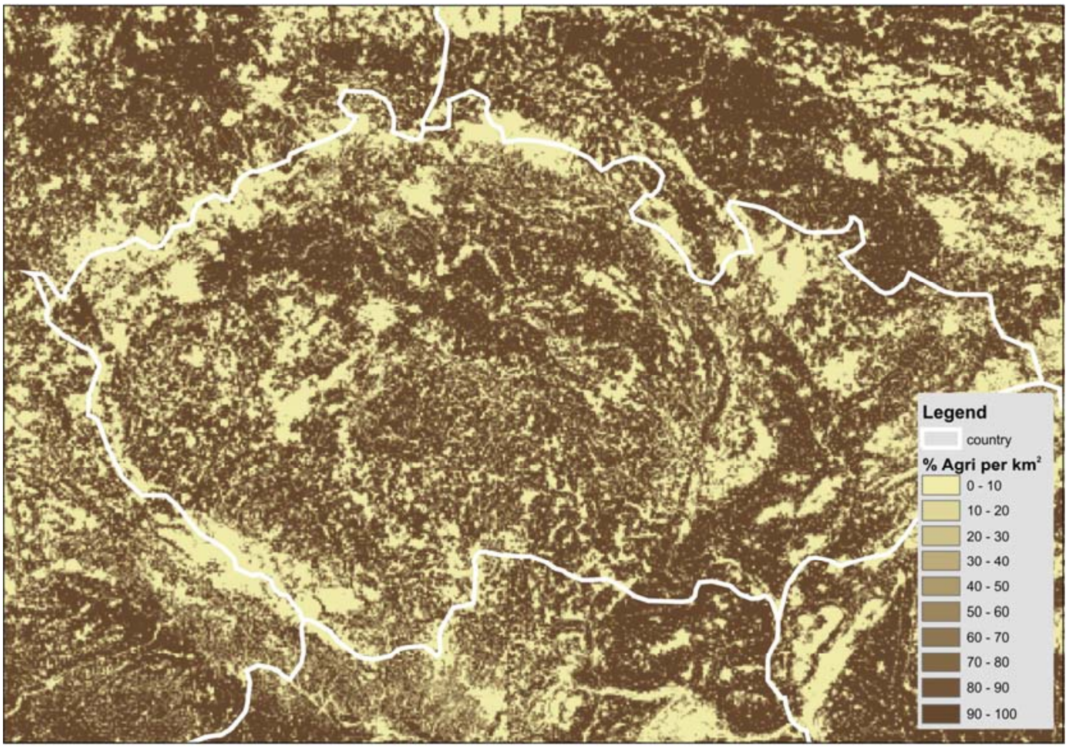

Figure 3. Mask based on Corine2000 describing the percentage of agricultural area in each cell.

3. Results and Discussion:

3.1 Digital Soil Mapping:

The authors observed in the DSM step (e.g. regression kriging) fairly low correlations between their target variables with environmental predictors (0.05 – 0.28). Additionally, a high nugget variation of the residuals has been observed. Possible reasons for that might be for example: insufficient data quality (e.g. sampling strategy, analytical quality, not sufficient accuracy in the coordinates (e.g. 2 digits), chosen pixel size too coarse to accurately predict soil properties, using parameters which do not explain the soil development and soil distribution process in that area, as well as the natural variability.

3.2 Digital Soil Assessment:

The first DSRA application concerning the aggregate stability for wind erosion shows clear differences between Data resulting from the European Soil DB (Mean 2.83 / Standard deviation 0.8) and RK approaches (Mean 3.1, Standard deviation 0.35) (see Figure 4). Lower and upper limits of the clay content show severe differences for each single location, which are not possible to estimate with the ESDB dataset. From the values predicted, it is clear that an increased accuracy of the predictions combined with a decreased uncertainty for each point is needed to allow further decision making.

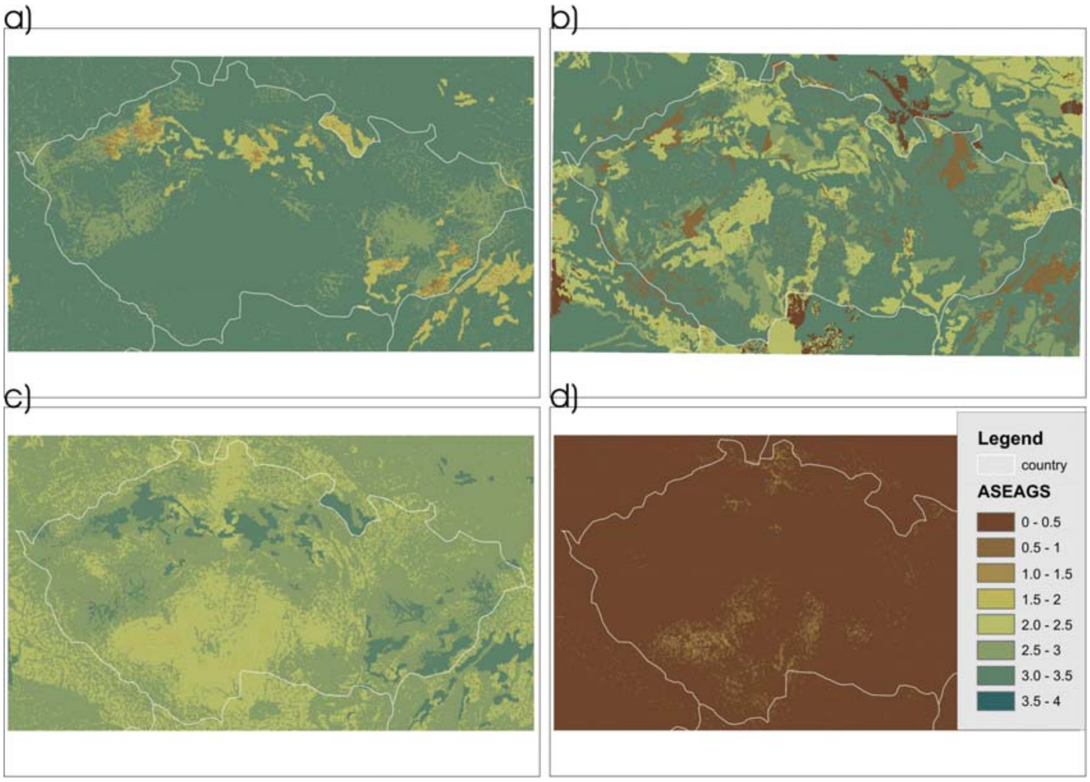

Figure 4. Aggregate Stability derived using a PTF from RK (a), the ESDB database (b), for the 95% lower limit (c) and the 95% upper limit of the RK clay content (d).

3.3 Digital Soil Risk Assessment:

The consequences of those different aggregate stabilities are amplified in the second application in the framework of Digital Soil Risk assessment: the number of Erosive days (ED) for wind erosion on bare soils on agricultural areas. ESDB and Worst Case-Scenario deliver similar numbers of ED, whereas the "average" RK data shows ¼ of the number of EDs.

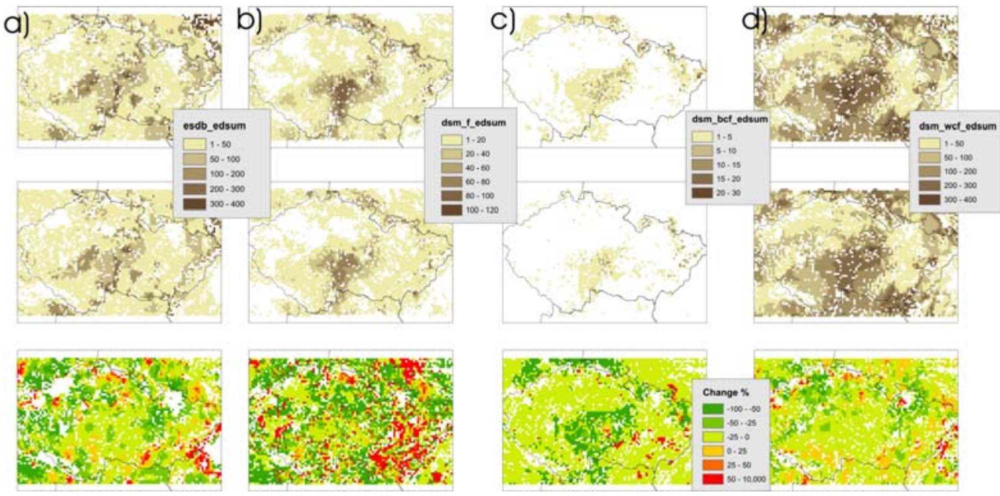

Figure 5. Number of Erosive Days for 30 years, based on texture of the ESDB database (a), from RK (b), for the Best Case Scenario (c) and Worst Case Scenario (d) for a current climate (top row), a climate scenario (middle row) and the change in percentage (bottom).

The climate scenarios allow for the forecast that areas in eastern CZ will be more prone to wind erosion events in the future, whereas an overall decrease can be observed. Still, uncertainty of this prediction is significantly high. A solution to this would be to sample additional points in areas currently under sampled (see Carré et al, 2007 for additional sampling strategies)

4. Conclusion

In this overview we applied DSM for DSA and DSRA to outline wind erosion estimations for part of the Danube basin. Inclusion of DSM techniques in modelling algorithm needs process knowledge as well as data understanding at the soil threats modellers' side. A combination of different datasets (soil, meteorological, land use) in DSA and DSRA allows outlining risk areas, in this case for wind erosion estimations with significantly decreased uncertainties.

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Updated map of salt affected soils in the European Union

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Summary

The aim of this report is to present an updated map of salt affected soils of the European Union. Together with the new map of salt affected soils in the EU a description is given on the methodology and datasets applied to derive the map. The report also contains an overview of the threat salinization/sodification, including a summary of the factors influencing the salt accumulation in soils.

1. Introduction

1.1 Salinization and sodification in Europe

The groups of salt affected soil differ both in physical, chemical, physico-chemical and biological properties as well as in their geographical distribution. The degradation risk in the context of the threat of salinisation/sodification is a measure of the probability and severity of the salinisation/sodification due to natural factors and human activity, that adversely affects one or more soil functions.

Saline/sodic and potentially saline/sodic regions in Europe are among areas having most detailed soil descriptions (profile and analytical databases). The work of Szabolcs (1974) summarized the knowledge on salt affected soils of Europe in a map format, together with a description of classification. Since the 1970's new continental scale data became available and the need for a new soil map evolved. Recently Várallyay et al. (2006) made an overview of the factors of salinization/sodification risk and the representation options on a European dimension. Our introduction of the issue of salinization/sodification is based on this summary.

Várallyay et al (2006) propose a multi-scale assessment of salinization/sodification in Europe that spans from the continental through regional (country) to field levels. The authors recommend delineating "Hot-spot" maps for decision making, and identify the spatial resolution of 1:1.000.000 scale for continental scale operations. On this scale using only 3 main classes can indicate the most significant salt-affected regions. The 3 main classes can be the followings: 1) saline soils 2) sodic soils 3) potential salt affected soils. Within the potentially salt affected soils a special mention may be given to soils that are saline/sodic in the deeper layers.

1.2 Definition and classification of salt affected soils

Saline and sodic soils are soils with high soluble salt content. According to Szabolcs (1974) two main groups of these soils can be distinguished:

1. Soils affected by neutral sodium salts (mainly sodium chloride and sodium sulphate)

2. Soil affected by sodium salts capable of alkaline hydrolysis (mainly $NaHCO_3$, Na_2CO_3 and $NaSiO_3$)

Processes resulting of the formation of salt affected soil can be distinguished as

a) salinization and b) sodification.

a) Salinisation

Salinisation is the process that leads to an excessive increase of water-soluble salts in the soil, in the soil solution.

The accumulated salts include sodium, potassium, magnesium and calcium, chloride, sulphate, carbonate and bicarbonate. A distinction can be made between primary and secondary salinisation processes. Primary salinisation involves accumulation of salts through natural processes due to high salt contents in parent materials or groundwater. Secondary salinisation is caused by human interventions such as inappropriate irrigation practices, e.g. with salt-rich irrigation water and/or insufficient drainage.

b) Sodification

Accumulation of Na+ in the solid and/or liquid phases of the soil as crystallised $NaHCO_3$ or Na_2CO_3 salts (salt „efflorescens”), ions in the highly alkaline soil solution (alkalisation), or exchangeable ion in the soil absorption complex (ESP). Result of the sodification process is the high ratio of exchangeable Na+ within the total exchangeable bases.

From the soil classification point of view, two main groups of these soils can be distinguished: soils belonging to the first group have mainly been named saline and those of the second group, alkali or sodic soils.

Salt-affected soils thus can be classified as:

1. Saline soils: soils in which high salt content dominates the problems.

Saline soils of Europe have developed in the most arid regions. The few exceptions to this rule are caused by salinity of local groundwater or soil forming substrate (Szabolcs 1974).

2. Sodic soils: soils in which high sodium content dominates the problems

As the accumulation of sodium salts capable for alkaline hydrolyzes is based on different biogeochemical processes, soil sodicity occurs under arid, moderate, and humid climate as well (Szabolcs 1974).

From the environmental risk point of view, a third category of soils can be distinguished:

Soils with specific characteristics in certain environmental conditions may be in risk of salinisation

In the World Reference Base (WRB) soil classification system, saline soils mainly occur in the Reference Soil Group of Solonchaks. However, some other Reference Groups may also have a salic horizon (indication - of certain degree - of salinisation) such as Histosols, Vertisols and Fluvisols.

Sodic soils mainly occur in the Reference Soil Group (in the WRB) of Solonetz. However, Solonetz may be associated with Histosols, Gleysols, Chernozems, Kastanozems, Vertisols and Solonchaks.

1.3 Factors influencing salinity of soils

1.3.1 Preconditions of salt accumulation

The preconditions of salt accumulation are as follows:

- Salt source (primary: weathering, volcanic activities; secondary: parent material, surface- and subsurface waters)
- Transporting agents (wind, surface water, subsurface water) lead to accumulation of salts (a, from a large water catchment area to a relatively small accumulation territory; b, from a thick geological deposit to a relatively thin accumulation horizon)
- Driving force to transport
  - relief for surface runoff
  - suction gradient for seep in the unsaturated zone
  - hydraulic gradient for groundwater flow
  - concentration gradient for solute transport
- negative water balance (at least for certain period of the year)
- vertical and horizontal drainage limitations

1.3.2 Main factors influencing soil salinity

The factors influencing salinity of soils can be divided in two groups:

a. natural factors
b. human-induced factors.

a. Natural factors

To summarize the most important natural factors, they are:

- Climate (seasonal variations)
- Soil parent material (acid, alkaline or even saline)
- Land cover and/or vegetation type (bare areas)
- Topography (slope and aspect)
- Soil attributes

The natural factors may be subdivided to 'soil' and 'land' (or 'area') factors. However, it is worth mentioning that the characteristics of soil (i.e. their response to anthropogenic factors) depend on "internal" soil properties and other "external" natural factors of the area as well.

Natural characteristics of the area:

- climate (temperature, rainfall, evaporation, wind characteristics, with their spatial distribution and time variability);
- geology (potential salt sources, sequence and thickness of aquifers and the vertical and horizontal transmissibility of geological layers);
- relief;
- vertical and horizontal drainage conditions;
- hydrology (quality and quantity of surface waters, groundwaters, deep-waters and their fluctuations).

Natural characteristics of the soil:

- texture;
- structure (aggregate status and stability; cracking, shrinkage – swelling characteristics);
- clay mineral composition;
- compaction rate – porosity (preferably differential porosity and pore-size distribution);
- hydrophysical properties (infiltration rate, water storage capacity, water retention, saturated and unsaturated hydraulic conductivity);
- salt content (profile, regime, balances, ion composition).

The above natural (environmental) factors result in salinisation/sodification through the processes of:

- movements of transgression and regressions that in some particular geological conditions bring about an increase of the concentration of salts in groundwater and consequently in soils;
- rise of salt-rich groundwater due to natural factors or human interventions (see below) up to the surface, near to the surface or to the overlying horizons;
- groundwater seepage into areas laying below sea level, microdepression with no or limited drainage;
- floods of fluvial waters coming from areas with geological substrates that release high amounts of salts;
- wind activities that, in coastal areas, bring moderate amounts of salts in soils;

b. Human-induced factors

To summarize the most important human-induced factors, they are:

- Land use and the nature of farming systems
- Land management
- Land degradation

The above human-induced factors may lead to salinisation/sodification through:

- irrigation of waters rich in salts;
- rising water table due to human activities (filtration from unlined canals and reservoirs; uneven distribution of irrigation water; poor irrigation practice, improper drainage);
- use of fertilizers and amendments, especially in situations of intensive agriculture with low permeability and limited possibilities of leaching;
- use for irrigation of wastewaters rich in salts;
- salt-rich wastewater disposal on soils;
- contamination of soils with salt-rich waters and industrial by-products.

2. Materials and methodology to delineate areas of salt affected soils

2.1 Databases

Two major data sources are available to delineate areas threatened by salt accumulation in Europe: the European Soil Database (ESDB 2004) and the map of salt affected soils in Europe compiled by Szabolcs (1974)

2.1.1 The European Soil Database (ESDB) at 1:1,000,000 scale

This database forms the core of the European Soil Information System, developed by the Land Management and Natural Hazards Unit of the Institute for Environment and Sustainability of the JRC. The database is managed in the European Soil Data Centre (ESDAC) by the action 'Soil Data and Information Systems' (Action SOIL).

The database contains a list of Soil Typological Units (STU), characterizing distinct soil types that have been identified and described. The STU are described by attributes (variables) specifying the nature and properties of the soils, for example the texture, the moisture regime, the stoniness, etc. The database also contains information on parent material, that is an important factor in salinization processes. The scale selected for the geographical representation of the ESDB is the 1:1,000,000. At that scale, it is not technically feasible to delineate each STU. Therefore STUs are grouped into Soil Mapping Units (SMU) to form soil associations. The criteria for soil groupings and SMU delineation have taken into account the functioning of pedological systems within the landscape.

The detailed instruction guide (doc. EUR 20422 EN) of the inventory as well as the full data and documentation are available from the EU Soil Portal: http://eusoils.jrc.it/index.html.

2.1.2 The map of Salt Affected Soils in Europe by Szabolcs (1974)

The publication of Szabolcs (1974) has been compiled within the frame of a programme for preparation of the World Map of Salt Affected Soils. The preparation of this map was in pressing importance of the 1970's because the salinity and alkalinity of soils hinder the satisfactory agricultural utilization of lands in many regions. The grouping of salt affected soils according to uniform principles allows their comparison. The experts of the working group preparing the map agreed that the map should be a logical extension of the FAO/UNESCO Soil Map of the World, emphasizing the specific problems of salt affected soils for their better utilization on the global scale. The scale of representation was 1:5.000.000 and was later displayed on the map sheet on 1:1.000.000.

According to the approach of the World Map of Salt Affected Soils two cases were recognized. These were:

a) A class dominated by chlorides and sulphates. This class is to be called: saline

b) A class dominated by exchangeable sodium and/or by sodium bicarbonate and/or by sodium carbonate. This class is to be called: alkali. It is subdivided in:

   a. a sub-class without structural B horizon

   b. a sub-class with structural B horizon.

It was recognized that in the case of Europe, where the dominantly salt-affected soils are alkali soils, the division of class B into two sub-classes were not adequate to indicate important differences as they occur in the continent. Therefore soils in the European map legend subdivide class B into lower, more detailed groupings, as following:

a) A subclass without structural B horizon

b) A subclass with structural B horizon

   1. Solonchak-solonetz and calcareous solonetz
   2. Non-calcareous solonetz with an A horizon < 15 cm
   3. Solodized and/or deeply leachedsolonetz and solod
   4. Solonized and slightly salt affected soils with minor structural formation

While the further subdivision of class B was deemed absolutely necessary by the editors of the map, they agreed that the acid-sulfate soils (thionic fluvisols of the FAO soil classification for Europe) should be omitted from the Map of

European Salt affected Soils. These soils occur in several coastal areas of the continent (for instance in Holland , Finland, Sweden etc.) and differ fundamentally in their properties, genetics and environmental conditions from the common types of European salt affected soils.

2.2 Method of initial delineation of salt affected areas

In the first step of the analysis we needed to select items in the two main databases which have characteristics of salt affected, or potentially salt effected soils. This exercise was the initial delineation of salt affected areas and it was performed for the two databases separately. The procedure for this selection is described below.

2.2.1. Information on soil salinity and alkalinity in the ESDB: the process of determination

The information on salinity and alkalinity that is available directly or through pedotransfer rules in the ESDB is introduced in detail by Baruth, Genovese and Montanarella (2006). The authors give detailed description for the procedure to derive all the salinity or sodicity related information from the database as well, therefore in this report we only provide a brief summary of this procedure.

To estimate the derived attributes SALINITY, ALKALINITY and DRAINAGE, pedotransfer rules (PTR) have been rewritten to accept the standard nomenclature system of the World Reference Base (WRB) of Soil Resources (FAO et al., 1998).

2.2.2 Salinity rule for the ESDB

Review of available information in the ESDB for salinity estimation

In the ESDB, the information concerning salinity is available in the soil name attributes (FAO85-FULL, FAO90-FULL, WRB-FULL). The agricultural limitation attributes (AGLIM1, AGLIM2) also contain information on salt content, however, considering the lower level of reliability confidentiality of these information, the agricultural limitation attributes were excluded from our mapping process.

Attribute FAO85-FULL

The attribute FAO85-FULL describes the soil name of the STU following the FAO-UNESCO 1974 legend, modified for the Soil Map of the European Communities in 1985. Several subdivisions, for which no definition is available, have been introduced by contributors to the ESDB. In the FAO-UNESCO 1974 legend, soil names that give information about salinity are the Solonchaks. In the definition of Solonchaks, the reference to salinity is given by the presence of a high salinity which is defined as soils having, at some time of the year:

• an EC_{se} of more than 15 mmhos/cm at 25°C (1 mmhos/cm = 1 dS/m),

- within 125 cm depth if the topsoil texture class (weighted mean) is coarse, or
- within 90 cm depth if the topsoil texture class (weighted mean) is medium, or
- within 75 cm depth if the topsoil texture class (weighted mean) is fine.

• or an EC_{se} of more than 4 mmhos/cm at 25°C, within 25 cm depth if the pH (H_2O, 1:1) is more than 8.5.

Table 1. Values of the FAO85-FULL attribute that give information about salinity

FAO85-FULL code	Signification
Z	Solonchak
Zg	Gleyic Solonchak
Zgf	Fluvi-Gleyic Solonchak
Zm	Mollic Solonchak
Zo	Orthic Solonchak
Zt	Takyric Solonchak
Gtz	Salo-thionic Gleysol

All soils having a high salinity must be named as Solonchaks except if they have the characteristics of Histosols, Lithosols, Vertisols and Fluvisols when referring to the key for determining soil names. There is no subdivision defined in the FAO-UNESCO 1974 legend considering salinity for these soil groups.

In the ESDB, STUs that have in their name a reference to salinity are Solonchaks and a Salothionic Gleysol (Table 1). For Solonchaks, referring to their definition, they can be considered as having a high salinity. For the Salo-thionic Gleysol, as Gleysols appear after Solonchaks in the key of the FAO-UNESCO 1974 legend, we can consider that this soil does not have a high salinity but a medium one.

Attribute FAO90-FULL

The attribute FAO90-FULL describes the soil name of the STU following the FAO-UNESCO 1990 legend which is a revision of the FAO-UNESCO 1974 legend. In the FAO-UNESCO 1990 legend, soil names that give information about salinity are Solonchaks and salic soils. Solonchaks are soils having salic properties. In the other soil groups, the salinity is used as a criterion for subdivision only for Fluvisols. The Salic Fluvisols are soils presenting salic properties. The salic properties refer to an EC_{se} of more than 15 dS/m at 25°C, at some time of the year, within 30 cm from the soil surface, or more than 4 dS/m within 30 cm from the soil surface if the pH (H_2O, 1:1) exceeds 8.5.

In the ESDB, STUs that have in their name a reference to salinity are Solonchaks only (Table 2). We can consider them having a high salinity.

Table 2. Values of the FAO90-FULL attribute that give information about salinity

FAO90-FULL code	Signification
SC	Solonchak
SCg	Gleyic Solonchak
SCh	Haplic Solonchak
SCn	Sodic Solonchak

Attribute WRB-FULL

The attribute WRB-FULL describes the soil name of the STU following the WRB (FAO-ISRIC-ISSS, 1998). In the WRB, soil names that give information about salinity are Solonchaks, 'salic' soils, or 'petrosalic' soils. In the definition of Solonchaks, the reference to salinity is given by the presence of a salic horizon within a depth of 50 cm. The 'salic' soils are soils other than Solonchaks that have a salic horizon within 100 cm from the soil surface.

The use of several specifiers modifies the definition as follows:

- endosalic: the salic horizon is between 50 to 100 cm from the soil surface.
- episalic: the salic horizon is between 25 to 50 cm from the soil surface.
- hyposalic: the EC_{se} is more than 4 dS/m in at least one sub-horizon within 100 cm from the soil surface.
- hypersalic: the EC_{se} is more than 30 dS/m in at least one sub-horizon within 100 cm from the soil surface.

The salic horizon is a surface or a shallow subsurface horizon which contains a secondary enrichment of readily soluble salts, i.e. salts more soluble than gypsum. The diagnostic criteria for a salic horizon is precisely that this type of horizon must have an EC_{se} of more than 15 dS/m at some time of the year, or an ECse of more than 8 dS/m if the pH (H_2O) of the saturation extract exceeds 8.5 (for alkaline carbonate soils) or is less than 3.5 (for acid sulphate soils). Petrosalic soils are soils having within 100 cm from the soil surface a horizon of 10 cm or thicker which is cemented by salts more soluble than gypsum. There is no accuracy information about the EC_{se} value of this horizon.

In the ESDB, soil names in WRB-FULL that give information about salinity are Solonchaks and salic soils. There are no petrosalic soils. We can consider Solonchaks as having a high salinity. For salic soils, the salinity is also high except for hyposalic soils which have a medium salinity. The difference between the different salic soils will be on the depth of the salic horizon which could be indicateed using the specifiers Endo (from 50 to 100 cm) or Epi (from

25 to 50 cm). Without specifiers, the salic horizon is present from 0 to 50 cm for a Solonchak (by definition), and from 0 to 100 cm for the other possible soil groups.

Table 3. Values of the WRB-FULL attribute that give information about salinity

WRB-FULL code	Signification
SCgl	Gleyic Solonchak
SCha	Haplic Solonchak
SCgy	Gypsic Solonchak
SCso	Sodic Solonchak
SCcc	Calcic Solonchak
SCgyw	Hypogypsic Solonchak
FLsz	Salic Fluvisols
HSsz	Salic Histosols
CLszn	Endosalic Calcisol
VRsz	Salic Vertisols

Adaptation of the PTR with the information available in the ESDB: the salinity rule

The analysis of the available information shows that the information from the soil name can be used to characterise the presence of a horizon having saline properties at a maximum depth of 125 cm. Three classes of salinity are proposed to be estimated in further exercises:

- • low: ECse < 4 dS/m
- • medium: 4 < ECse < 15 dS/m
- • high: ECse > 15 dS/m

Table 4. The rule for salinity.

FAO85-FULL	FAO85-FULL.CL	FAO90-FULL	FAO90-FULL.CL	WRB-FULL	WRB-FULL.CL	SALINITY
*	*	*	*	*	*	low
8	o	*	*	*	*	high
J**	o	*	*	*	*	high
J**	o	*	*	*	*	high
G**	o	*	*	*	*	medium
G**	o	*	*	*	*	medium
Gtz	o	*	*	*	*	medium
Z**	o	*	*	*	*	high
*	*	SC*	o	*	*	high
*	*	FLs	o	*	*	high
*	*	*	*	SC**	o	high
*	*	*	*	**sz	o	high
*	*	*	*	**szn	o	high
*	*	*	*	**szp	o	high
*	*	*	*	**szh	o	high
*	*	*	*	**szw	o	medium
*	*	*	*	**ps	o	high

The salinity rule is described in Table 4. For some soil types, the threshold can be less than 15 dS/m (4 dS/m for a Solonchak (FAO90) having a pH > 8.5, or 8 dS/m for a Solonchak or a salic soil (WRB) having a pH > 8.5 or a pH < 3.5). But, the ECse can be higher than 15 dS/m.

It is considered that these soil types have a high salinity in order to avoid having classes that overlap (these soils have generally other limitations like alkalinity or sulphur toxicity)

2.2.3 Alkalinity rule for the ESDB

Review of available information in the ESDB for alkalinity estimation

In the ESDB, the information concerning alkalinity is available in the soil name attributes (FAO85-FULL, FAO90-FULL, WRB-FULL). The agricultural limitation attributes (AGLIM1, AGLIM2) also contain information on salt content, however, as was in the case of salinity assessment, due to the low level of reliability of these information, the agricultural limitation attributes were excluded from our mapping process.

Attribute FAO85-FULL

In the FAO-UNESCO 1974 legend, soil names that give information about alkalinity are the Solonetz and Solodic Planosols. In the definition of Solonetz, the reference to alkalinity is given by the presence of a natric B horizon. A natric B horizon is an argillic B horizon which has in its upper 40 cm:

- a saturation with exchangeable sodium of more than 15%,
- or more exchangeable magnesium plus sodium than calcium plus exchange acidity (at pH8.2) if the saturation with exchangeable sodium is more than 15% in some subhorizon within 200 cm from the soil surface.

The Solodic Planosols are Planosols having more than 6% sodium in the exchange complex of the slowly permeable horizon. The classification of a soil in the Solonetz group is not only characterized by the high saturation with exchangeable sodium but also by the presence of other characteristics:

- the natric B horizon must have the characteristics of an argillic B horizon and
- must have also a columnar or prismatic structure. There is no criteria 'high alkalinity' as it exists for salinity.

Table 5. Values of the FAO85-FULL attribute that give information about alkalinity

FAO85-FULL code	Signification
S	Solonetz
Sg	Gleyic Solonetz
Sof	Fluvi-Orthic Solonetz
Sm	Mollic Solonetz
So	Orthic Solonetz
Vpn	Sodi-pellic Vertisol

So, all soils having a high alkalinity are not always classified as Solonetz. On the other hand, there is no subdivision defined in the FAO-UNESCO 1974 legend considering alkalinity for other groups than Solonetz, except for Solodic Planosols.

In the ESDB, STUs that have in their name a reference to alkalinity are Solonetz and a Sodipellic Vertisol (Table 5). For Solonetz, referring to their definition, they can be considered as having a high alkalinity. For Sodi-pellic Vertisols, as Vertisols appears before Solonetz in the key of the FAO-UNESCO 1974 legend, this soil could have potentially a high alkalinity. But we have no information in the database that could confirm it. This STU has a dominant land-use of arable land and is drained. It is proposed to estimate its alkalinity to medium.

Attribute FAO90-FULL

In the FAO-UNESCO 1990 legend, soil names that give information about alkalinity are Solonetz and sodic soils. Solonetz are soils having a natric B horizon. A natric B horizon is an argic B horizon which has in its upper 40 cm:

- a saturation with exchangeable sodium of more than 15%,
- or more exchangeable magnesium plus sodium than calcium plus exchange acidity (at pH 8.2) if the saturation with exchangeable sodium is more than 15% in some subhorizon within 200 cm from the soil surface.

In the other soil groups than Solonetz, the alkalinity is used as a criterion for subdivision only for Solonchaks. Sodic Solonchaks are Solonchaks that present sodic properties at least between 20 and 50 cm from the soil surface. The sodic properties refer to saturation in the exchange complex of 15% or more of exchangeable sodium or of 50% or more exchangeable sodium plus magnesium. We thus can consider Solonetz or Sodic Solonchaks as having a high alkalinity.

In the ESDB, STUs that have in their name a reference to alkalinity are Solonetz and Sodic Solonchaks (Table 6). They can be considered as having a high alkalinity.

Table 6. Values of the FAO90-FULL attribute that give information about alkalinity

FAO90-FULL code	Signification
SNm	Mollic Solonetz
SNh	Haplic Solonetz
SCn	Sodic Solonchak

Attribute WRB-FULL

In the WRB, soils having alkali characteristics are Solonetz, ‘natric’ soils, or ‘sodic’ soils. In the definition of Solonetz, the reference to alkalinity is given by the presence of a natric horizon within 100 cm from the soil surface. The ‘natric’ soils are soils other than Solonetz that have a natric horizon within 100 cm from the soil surface. The natric horizon is a dense subsurface horizon with higher clay content than the overlying horizons and that has a high content in exchangeable sodium and/or magnesium. The diagnostic criteria for a natric horizon specify that this type of horizon must have an ESP of more than 15% within the upper 40 cm, or more exchangeable magnesium plus sodium than calcium plus exchange acidity (at pH 8.2) within the same depth if the saturation with exchangeable sodium is more than 15% in some subhorizon within 200 cm of the surface.

‘Sodic’ soils are soils having more than 15% exchangeable sodium or more than 50% exchangeable sodium plus magnesium on the exchange complex within 50 cm from the soil surface. Two specifiers can be used:

- endosodic soils have more than 15% exchangeable sodium or more than 50% exchangeable sodium plus magnesium on the exchange complex between 50 cm to 100 cm from the soil surface,
- hyposodic soils have more than 6% saturation with exchangeable sodium in at least some subhorizons more than 20 cm thickness within 100 cm from the soil surface.

Solonetz, ‘natric’ soils, ‘sodic’ soils and ‘endosodic’ soils are considered having a high alkalinity. For ‘hyposodic’ soils, the alkalinity is medium.

In the ESDB, STU that give information about alkalinity are Solonetz, Sodic Solonchaks, Sodic Gleysols, Sodic Phaeozems, Hyposodic Vertisols, Hyposodic Calcisols or Hyposodic Gypsisols (Table 7).

Table 7. Values of the WRB-FULL attribute that give information about alkalinity

WRB-FULL code	Signification
SNgl	Gleyic Solonetz
SNha	Haplic Solonetz
SNgy	Gypsic Solonetz
SNcc	Calcic Solonetz
SCso	Sodic Solonchak
GLso	Sodic Gleysol
PHso	Sodic Phaeozem
VRsow	Hyposodic Vertisol
GYsow	Hyposodic Gypsisol
CLsow	Hyposodic Calcisol

Adaptation of the PTR with the information available in the ESDB: the alkalinity rule

The analysis of the available information shows that the information from the soil name can be used to characterise the presence of a horizon having sodic properties. The proposed alkalinity rule is described in Table 8. Three classes of alkalinity are proposed for further analysis:

- low: ESP < 6%
- medium: 6 < ESP < 15%
- high: ESP > 15%

Table 8. The rule for alkalinity

FAO85-FULL	FAO85-FULL.CL	FAO90-FULL	FAO90-FULL.CL	WRB-FULL	WRB-FULL.CL	ALKA-LINITY
*	*	*	*	*	*	low
S**	o	*	*	*	*	high
Vpn	o	*	*	*	*	medium
Ws	o	*	*	*	*	medium
*	*	SN*	o	*	*	high
*	*	SCn	o	*	*	high
*	*	*	*	SN**	o	high
*	*	*	*	**na	o	high
*	*	*	*	**so	o	high
*	*	*	*	**sow	o	medium

2.2.4 Classification of salt affected soils in Europe by Szabolcs (1974)

The original aim of the mapping procedure lead by Szabolcs was to derive a geographical coverage of salt affected soils in Europe. The thematic database produced therefore does not require further processing in its semantic component. ! However with regards to the thematic layers in the ESDB and the information derived from these layers a harmonization effort was necessary.

The legend of the Map of Salt Affected Map of Europe consist of the following classes:

- Saline soil, > 50% of the area
- Saline soil, < 50% of the area
- Alkali soils without structural B-horizon > 50% of the area
- Alkali soils without structural B-horizon < 50% of the area
- Alkali soils with structural B-horizon, calcareous, > 50% of the area
- Alkali soils with structural B-horizon, calcareous, < 50% of the area
- Alkali soils with structural B-horizon, non-calcareous, > 50% of the area
- Alkali soils with structural B-horizon, non-calcareous, < 50% of the area
- Potentially salt effected soils

2.3 Updating the map of salt affected soils in Europe

Following the initial delineation of salt affected areas in the two input databases the updating of the information was carried out.

The updating was based on the perception, that the information of the ESDB is derived from recent and detailed soil surveys, therefore more reliable in its content and geographical extent. From this standpoint we were aiming to draw a new map, which contains the ESDB derived salinity/sodicity information, complemented with the information of the Szabolcs map. The latter was especially useful in the assessment of potentially salt affected areas, for which the ESDB did not provide any information.

In this mapping process the establishment of a common legend was the most important component, on which the spatial delineations could be harmonized. Unfortunately in the ESDB there is no direct information on the B horizon of salt affected soils. At this stage the PTR driven definition of structural characteristics of soil units was an option that would further lower the reliability confidentiality of the results, therefore we selected to go for a simplified joined legend. In this legend saline and sodic soils and the share of these soils within the soil mapping unit can be distinguished. With these categories as common platform between the Szabolcs map and the salinity information of the ESDB a clear overview of saline/sodic soils of Europe can be provided.

The delineation of saline/sodic areas in Europe consisted of the following steps:

1. In the first step of the procedure we have prepared digital geodatabases from the map of Szabolcs (1974) in the Lambert Azimuthal Equal Area coordination system ('Digital Szabolcs Map'), the same projection as the ESDB. After the databases were in the same reference system, we were able to match the spatial information and explore the semantic components of the data.

2. Soil mapping units with saline and/or sodic soils were selected from the ESDB and the percentage share of salt affected areas of the polygons have been calculated. (>50 % and < 50% for saline and alkaline soils.) These polygons were displayed on a separate layer.

3. The Digital Szabolcs Map was overlaid by the new salt affected soil layer of the ESDB.

4. On areas where both maps had information on salinity, the information from the ESDB took priority. Areas outside the salinity/sodicity layer of the ESDB had been characterized by the 'Digital Szabolcs Map'.

Following this method we compiled an updated version of the salt affected soils map (Figure 1.)

The legend is designed to include six categories.

1. Saline soils cover more than 50 % of the indicated area

2. Saline soils cover less than 50 % of the indicated area

3. Alkali soils cover more than 50 % of the indicated area

4. Alkali soils cover less than 50 % of the indicated area

5. Potential salt affected areasNo risk of salt accumulation

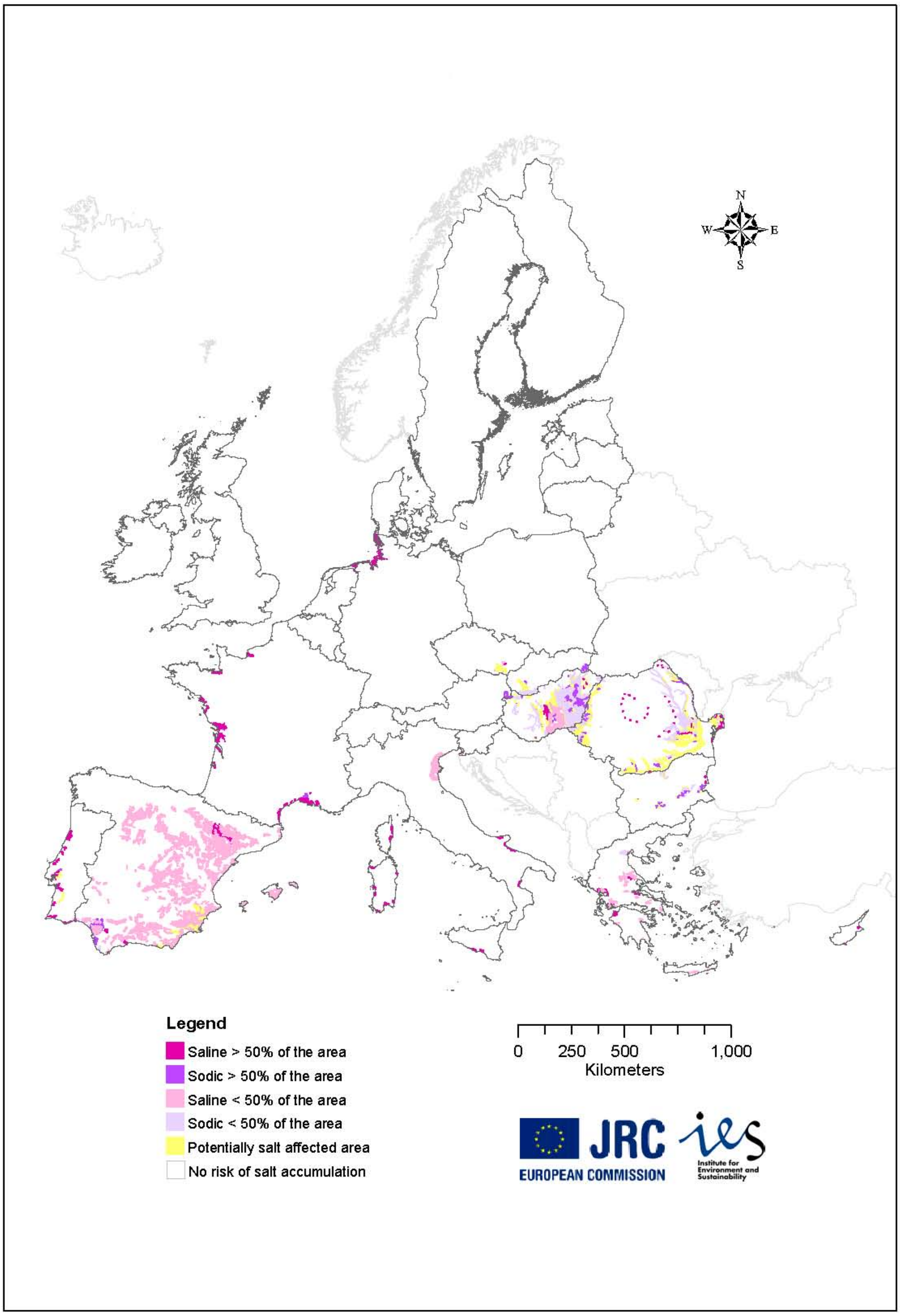

Figure 1. Map of Saline and Sodic Soils in the European Union: Status and Potentials

3. Conclusion and need for further research

Salinisation and sodicification are among the major degradation processes endangering the use potential of European soils. These processes are recognized among the main soil threats identified in the Thematic Strategy for Soil Protection of the European Union (COM2006/231). However, no continental scale assessment for salt affected soils has been carried out in the continent since the compilation of the map “Salt Affected Soils in Europe” by Szabolcs (1974). The most up-to-date information on the salt affected soils with a continental coverage is available in the European Soil Database (ESDB 2004), which stands as the main general purpose soil database in Europe. Although the ESDB provides information on greater taxonomic detail about the types of salt affected soils occurring in the continent, the spatial resolution of the ESDB (1:1M) remains on that of the map of Szabolcs (1974), which was compiled on a scale of 1:1M and published on the scale of 1:5M.

Based on the available information on salt affected soils on the European scale, a new map was generated for the delineation of areas in the European Union that are threatened by salinization or sodification.

Further research is needed to predict the extent of salt affected soils with an increased accuracy in comparison to the continental or global geographical soil databases available at the moment. However, the ancillary information currently available on a continental covarage, namely the (I) SRTM derived digital elevation model for Europe, (II) European Groundwater Database, and (III) European Map of Aridity Index were found to be inadequate to increase the accuracy of the delineations.

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A framework to estimate the distribution of heavy metals in European Soils

L. Rodriguez, H. I. Reuter, T. Hengl

Summary

The paper presents results of interpolating concentrations of eight critical heavy metals (arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc) using the 1588 georeferenced topsoil samples from the Forum of European Geological Surveys Geochemical database. The concentrations in ppm were mapped using regression-kriging and accuracy of predictions evaluated using the leave-one-out cross validation method. A large amount of auxiliary raster maps was used to improve the predictions.The study revealed that this database is suitable for geostatistical analyses: the predictors explained from 23% (Cr) up to 37% (Pb) of variability; the residuals showed spatial autocorrelation. The Principal Component Analysis of the mapped heavy metals revealed that the administrative units (NUTS level3) with highest overall concentrations are: (1) Liege (Arrondissement) (BE), Attiki (GR), Darlington (UK), Coventry (UK), Sunderland (UK), Kozani (GR), Grevena (GR), Hartlepool & Stockton (UK), Huy (BE), Aachen (DE) (As, Cd, Hg and Pb) and (2) central Greece and Liguria region in Italy (Cr, Cu and Ni). The evaluation of the mapping accuracy showed that only the maps of Ni and Pb can be considered satisfactory accurate (prediction accuracy 47% of total variance), marginally satisfactory for As, Cd and Hg (33–35%), while maps of Cr, Cu are unsatisfactory accurate. The critical points to improve the mapping accuracy are: (a) problem of sporadic high values (hot-spots); and (b) relatively coarse resolution of the input maps. Automation of the geostatistical mapping and use of auxiliary spatial layers opens a possibility to develop mapping systems that can automatically update outputs by including new field observations and higher quality auxiliary maps. This study also shows the scientific and political benefits of exchange of geographical information at regional and national levels in the form of additional and coherent environmental information.

1 Introduction

Soil contamination by metals has become a widespread environmental problem in large proportions of industrialized countries. Atmospheric deposition, mining, fossil fuel consumption, irrigation with water, waste incineration, and the use of fertilizers and agrochemicals have been identified as the main human sources of heavy metals to soils (Hutton and de Meeûs, 2001; Hansen et al., 2002). Natural processes as emissions from volcanoes, degassing processes in the Earth's crust, forest fires or the chemical composition of the parent material can be also important sources of pollutants to soils (Løkke et al., 1996; Palumbo et al., 2000).

The European Commission has been preparing a proposal for a framework Directive (European Communities, 2006) that sets out common principles for the protection of soils across the EC. Soil contamination has been one of the recognized degradation threads occurring in European soils. Tóth et al. (2007) cite soil pollution as one important parameter to estimate in order to determine soil quality and soil sustainability. From the aspect of soil pollution, three main questions need to be clarified: (i) which threshold values should be used to classify soil as polluted? (ii) at which locations can high natural background values of heavy metals be expected? and (iii) which methods should be used to explain the spatial distribution of these elements in soils of Europe? Threshold values for soils are difficult to evaluate since the toxicity and bioavailability of heavy metals is not only dependent on the total content in soils but also in other environmental variables. At European level only threshold values related to the application of sewage sludge in agricultural soils have been defined (EU Directive 86/278/EC). The determination of natural background values is controversial because they can release a responsibility of human activities to the overall pollution on soils (Baize and Sterckeman, 2001). Otherwise, it is often difficult to determine the background values that would correspond to a pristine situation since the geochemistry of most of the European ecosystems is greatly influenced by a long history of anthropic activities, and even the concept of a background value is often fuzzily defined (Reimann and de Caritat, 2005; Reimann and Garrett, 2005).

At present, some efforts have been done in mapping the distribution of heavy metals in European soils. Reimann et al. (2003) created maps of heavy metal contents in soils using an Inverse Distance Weighted interpolator on about 740 samples from agriculture fields. The Geochemical Atlas of Eastern Barents region (Salminen et al., 2004) also includes interpolated maps of heavy metals from 1358 sampling sites in the northern part of Europe. Gawlik and Bidoglio (2006) produced maps for Cd, Cr, Cu, Hg, Ni, Pb and Zn in 11 European countries by using empirical relationships with soil parent material and land use. The European Environmental Agency (2006) merged the sampling points from three different soil databases to create a map of the concentration of lead in topsoils across Europe (http://dataservice.eea.europa.eu).

Recently, the Forum of European Geological Surveys (FOREGS) produced the "Geochemical Atlas of Europe" (Salminen et al., 2006) (http://www.gsf.fi/publ/foregsatlas/) with the aim of establishing the geochemical baseline values of 70 elements and compounds in topsoils, subsoils, floodplain sediments, stream sediment and humus. In this work a number of maps of the element contents in soils of Europe were created. The term "geochemical baseline" in this context is not equal to the "background value" since it represents a measurement that does not correspond to a pristine situation. However, these geochemical baselines can provide a perspective on the present status of pollution of the European soils, and serve as a model for future pan-European monitoring projects.

All these works present great differences since they have been created using different databases and different methodological approaches, and this makes that all this available information create some confusion for decision makers and other customers in general.

In this work we propose a framework based in regression kriging to estimate the distribution of heavy metals in European soils. This technique has the advantage of make use of relevant auxiliary information to map the distribution of heavy metal concentrations in soils of Europe as well a means of estimation of the accuracy of the results. In addition, our intention is also to promote the environmental exchange of information at regional and national level being demonstrated the scientific and political benefits, in the form of additional and coherent environmental information, that such exchange would cause.

2 Material and method

2.1 Sample database

The database used in this work has been produced by FOREGS and it is freely available in their web site (http://www.gsf.fi/publ/foregsatlas/).

Figure 1. Location of the sampling points.

We used a total number of 1588 points from topsoils samples although not all values were available at all locations (Figure 1). The original Lat-Long coordinates of the sampling points were transformed to European Terrestrial Reference System.

2.2 Auxiliary GIS layers

The auxilary information used in this study include geology and land cover, MODIS-based vegetation indices, night lights images, earthquake magnitudes, distance to roads and railroads, climatic variables and deposition levels of Cd, Pb and Hg. The map of Geology was obtained from Pawlewicz et al. (2003). This map is a coarse synthesis of the main rock types and their geologic age of formation in Europe (http://pubs.usgs.gov/). The original legend was reduced to 10 classes relevant for mapping of heavy metals: (1) Granites, rhyolites and quartzites; (2) Paleozoic schists, phyllites, gneisses and andesites; (3) Shales and sandstones; (4) Mesozoic Ultramafic, basic phyllites, schists, limestones and evaporates; (5) Jurassic, Triassic and Cretaceous calcareous rocks; (6) Cenozoic serpentinites, gabros and sand deposits; (7) Tertiary basanites and andesites; (8) Neogene and Paleogene calcareous rocks; (9) Quaternary limestones and basaltic rocks; and (10) Other Ultramafic and undefined rocks.

The CORINE Land Cover 2000 map of Europe, generalized to 1 km grid, was used to represent the main classes of land cover. For Switzerland, we used the Corine Land Cover 1990 since no updated information was available. The CLC1990 classes for this country were adjusted to those described in the CLC2000 and both data sets were merged together and aggregated to 1 km resolution. The original 44 classes were simplified to 8 classes: (1) urban infrastructures; (2) agriculture; (3) forest; (4) natural vegetation; (5) beaches; (6) ice bodies, (7) wetlands and (8) water bodies.

Monthly averaged MODIS images of the EVI at 1 km resolution for the period 01/01/2004 to 31/12/2006 were obtained from the MODIS Terra imagery (http://edcimswww.cr.usgs.gov). Seventeen single blocks covering the whole study area were mosaicked and reprojected to the Lambert Azimuthal Equal Area (ETRS89) projection. We performed Principal Component analysis on 19 complete mosaics and used the first 10 principal components of the EVI images.

The 1 km Digital Elevation Model (DEM) was derived from the SRTM30 6 V2 (http://www2.jpl.nasa.gov/srtm/). The SRTM DEM was used to derive a slope map, the Topographic Wetness Index and total incoming solar insolation (Böhner et al., 2006).

The cumulative earthquake magnitude image was calculated by using the global seismology point database (http://earthquake.usgs.gov/eqcenter/). We used the logarithmic measure of the size of the earthquakes and we derived a 1 km density grid by using the kernel smoother and search radius of 10 km.

The lights at night image for the year 2003 (http://www.ngdc.noaa.gov/dmsp/) measures night-time light emanating from the earth's surface at 1 km resolution. The lights at night map contains the lights from cities, towns, and other sites with persistent lighting. This map is a direct estimate of the urbanization level.

The map of distances to roads, airports and utility lines was calculated using the distance operation in ILWIS and the GIS layers from the GISCO database of the European Commission (http://eusoils.jrc.it/gisco_dbm/dbm/).

Mean annual temperature and accumulated precipitation maps were obtained from the very high resolution raster layers created by Hijmans et al. (2005) on a global scale at 1 km grid resolution. The annual potential evapotranspiration (PET) was calculated from monthly temperature data using the method of Thornthwaite (1948). Runoff was calculated as the difference between annual accumulated precipitation and PET.

The estimated annual deposition and emission rates of cadmium, lead and mercury in Europe for year 2004 calculated within the European Monitoring Evaluation Programme (http://webdab.emep.int). The original 50 km grids were down-scaled to 1 km grids using ordinary kriging.

This gives a total of 36 maps-predictors. To minimize multicollinearity, the original predictors were converted to independent principal components (raster maps). Since some predictors show continuous changes and others (land cover, geological units) represent abrupt changes of the values, the final principal component show a hybrid representation of the total environmental conditions (Figure 2).

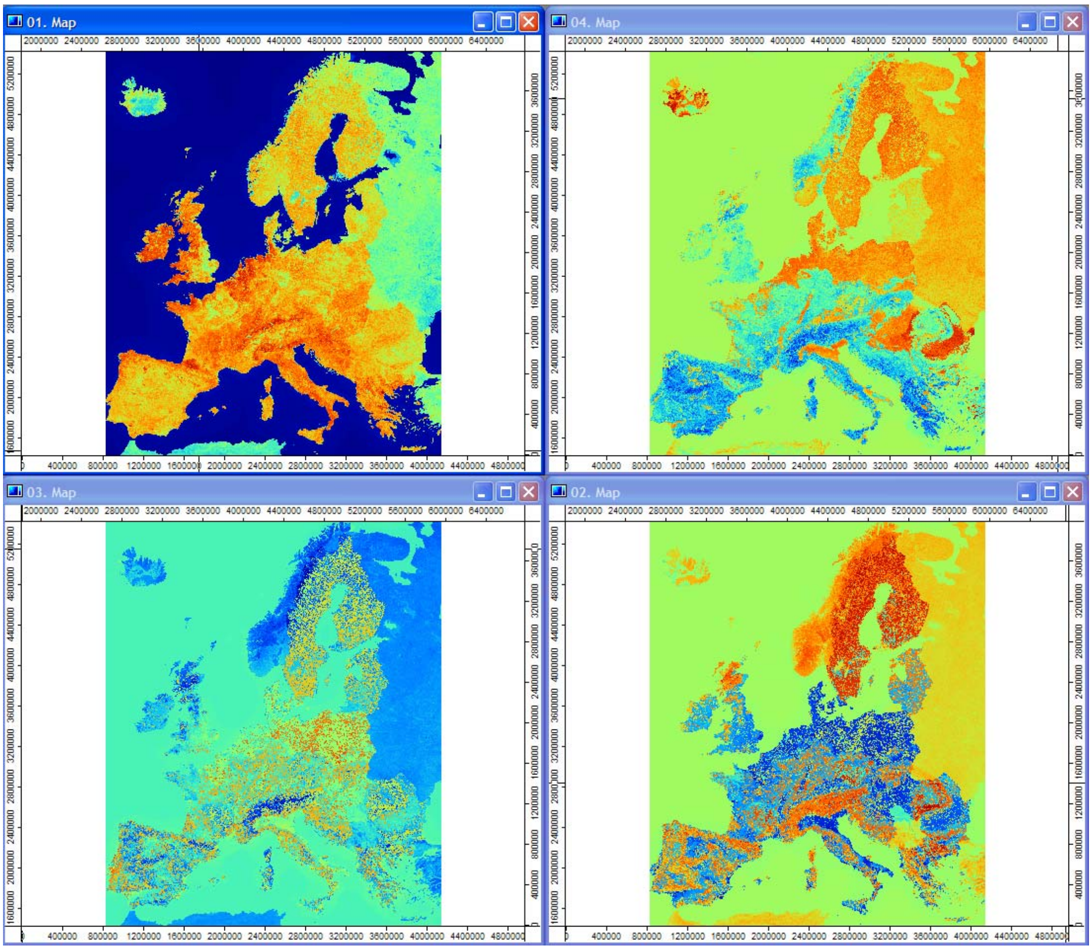

Figure 2. First four components resulting from Principal Component Analysis of the auxiliary variables.

2.3 Geostatistical analysis

The original variables showed skewed distributions. Logit transformations with physical limits set at z_{min}=0 and z_{max}=10^6 were performed to make the data suitable for regression and variogram analyses (Hengl et al., 2004; Papritz et al. 2005). The transformed variables in the point samples were first correlated with the set of environmental predictors (PCA raster maps) by stepwise regression. Then, the residuals of these regression models were analysed for spatial auto-correlation. The final maps were obtained by block regression-kriging to reduce the influence of the local hot-spots on the final predictions. All regression analyses and variogram fitting were performed with the 1 km data, however, the final predictions were generated at 5 km grids. We also derived the overall (mean) normalized interpolation error maps for each element by dividing the interpolation error (universal kriging variance) by the global variance in the point data.

3 Results

3.1 Preliminary analysis

A first exploratory analysis of the samples revealed that some heavy metals are highly correlated. This is the case of Cr and Ni (r=0.85), Cu and Ni (r=0.75), Pb and Zn (r=0.74), Cd and Zn (r=0.69) and Pb and Hg (r=0.69). In summary, the HMCs are well represented using the two main components of PCA. The second component clearly differentiates two groups of elements, As, Cd, Hg and Pb with positive correlation and Cr, Cu and Ni with negative correlation. Similar groupings were found in other studies on heavy metals (Facchinelli et al., 2001; Micó et al., 2006; Zhang, 2006; Luo et al., 2007) and it is often interpreted as indicator of the source of origin of the elements:

anthropogenic for As, Cd, Hg, Pb and Zn, and geogenic for Cr and Ni. The intermediate position of Cu may indicate a joint contribution from both natural and anthropogenic sources.

The correlation analyses between HMCs and the untransformed predictors showed that there are some links between these environmental variables and the distribution of heavy metal content in soils. The results of step-wise regression were all significant at 0.001 probability level. The adjusted R-square ranges from 0.23 for Cr up to 0.37 for modeling of Pb. The most significant predictors were PC04 (related to limestones and elevation), PC02 (agriculture), PC13 (EVI, infrastructures) and PC14 (earthquakes). Much of variation was not explained by these predictors, but the residuals showed spatial autocorrelation.

3.2 Regression-kriging results

Even if in this paper we only present the regression-kriging maps of Ni and Pb (Figure 3 and Figure 4), we can anticipate that high values of Ni and Cr were mainly found in central Greece, northern Italy, central Pyrenees, Slovakia and Croatia.

For the other heavy metals, the higher concentrations are mainly found in Central Europe and it is directly related to human activities. Cd, Cu, Hg, Pb, Zn present a high correlation with Agriculture (r=0.7) and with quaternary limestones (r=0.41), where most of the agricultural areas in Central Europe are located. The use of the fertilizers, manure and agrochemicals are important sources of these elements to European soils. These heavy metals are also highly correlated with the distance to infrastructures and to components 1 and 3 of the EVI images (PCEVI1 and PCEVI3), which depict clearly urban and industrial areas.

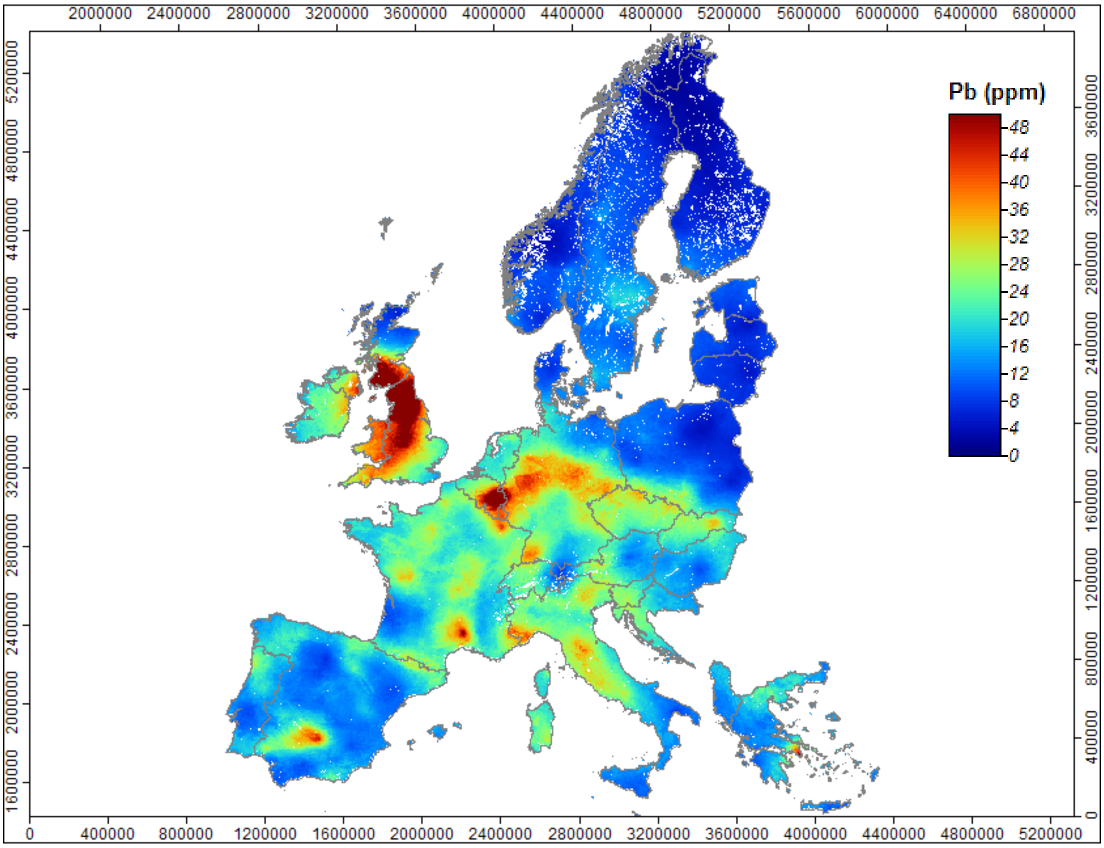

Figure 3. Estimated concentrations of Pb Ni in European soils.

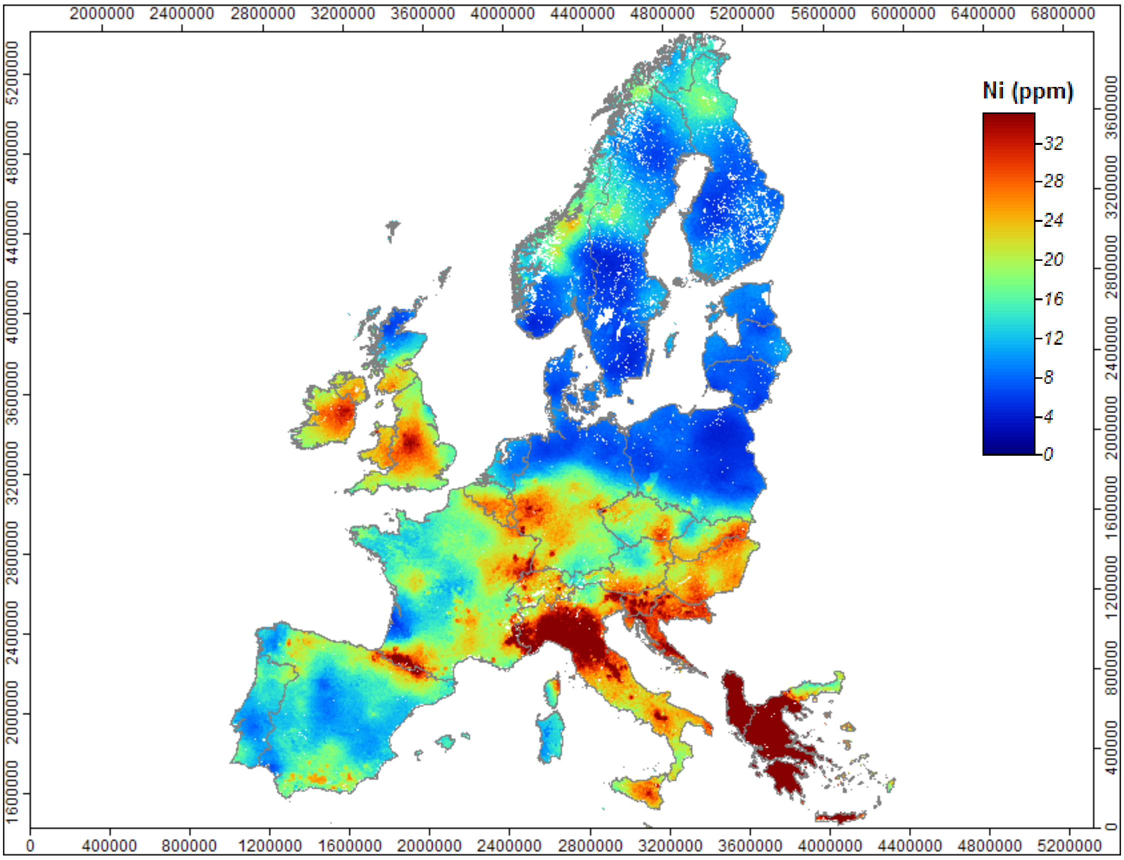

Figure 4. Estimated concentrations of Ni in European soils.

3.3 Validation

The success of the technique was evaluated using the leave-one-out cross validation method. The results of the validation show that the contrast of the maps produced using regression-kriging is in general 20–40% higher than for the ordinary kriging and that the Root Mean Square Prediction Error (RMSPE) is always smaller for regression-kriging, but the relative difference does not exceed 15%. Only the maps of Ni and Pb can be considered satisfactory accurate (prediction accuracy 47% of total variance), while the maps of As, Cd and Hg are only marginally satisfactory (33–35%). Maps of Cr, Cu and Zn are critically inaccurate, i.e. they do not seem to carry much useful information about the distribution of these metals. These results demonstrate that there is quite some difference is how the HMCs vary in space.

4 Conclusions

In the regression kriging maps arise some spatial features that effectively influence the contents of some heavy metals in soils. This is clearly the case of geology or land use, but also relief and density of roads seems to play an important role in the distribution of heavy metals. Such patterns are normally completely ignored in the maps that are produced by other geostatistical techniques such as ordinary kriging.

Even if the validation of the results shows that the regression-kriging is not much more accurate than ordinary kriging and that several maps (Cr, Cu and Zn) have critically low accuracy, our opinion is that auxiliary predictors can contribute to the understanding of the sources of heavy metals to soils (geogenic vs anthropogenic). All this allows an in-depth analysis of the processes that cause the distribution of HMCs, so that also the remediation policies can be selected appropriately.

The main advantages of this mapping framework are that the maps of heavy metals in this paper were produced using a robust and objective geostatistical technique. A technique that can produce the best linear predictions of the values over the whole area of interest, but also an estimate of the mapping error, which can be equally important for decision making and further spatial modeling.

Even if the analytical quality of the FOREGS database is out of doubt, the accuracy of our geostatistical models is highly dependent on the quality of sampling. In the case of the FOREGS database, the sampling density corresponds to cell sizes of 1–5 km, which obviously limits its usage for general scales only. The biggest limitation of the FOREGS dataset seems to be its sampling design that obviously carries limitations: sampling points are somehow clustered and several features (certain land use features and urban soils) have been omitted by the surveyors, which might lead to under or over-estimation over the whole continent.

Finally, we hope that this work will motivate the national agencies in Europe to contribute with additional point data measures but also to share their environmental geographic databases that can be useful to improve future modelling analyses similar to those presented in this exercise. This would impulse the development and implementation of adequate policy control measurements to protect our environment in this specific issue.

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We want to express our gratitude to the Association of the Geological Surveys of The European Union for their courtesy of delivering freely their sample database. We recognize that this work wouldn't be possible without their huge sampling and analytical effort.

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Application of Soil Organic Carbon Status Indicators for policy-decision making in the EU

V.Stolbovoy and L.Montanarella

Summary

Soil Organic Carbon Status Indicators (SOCSI) are a new deterministic approach to examine soil organic carbon (SOC) in order to support policy-decision making with regard to establishing carbon management regimes in cultivated mineral soils of the European Union (EU). The SOCSI comprise a set of characteristics describing amount and qualities of SOC including static parameters such as actual (Act SOC), maximum (Max SOC), minimum (Min SOC) SOC levels, and dynamic parameters which include potential amount and potential rate of SOC change.

According to SOCSI, the cultivated soils in the EU have a considerable amount of potentially degradable SOC varying between 50 and 100 tC ha in southern and northern regions respectively. The potential rate of SOC loss is estimated as High in the northern and central regions and Limited in the southern regions of the EU.

SOCSI indicate that cropland soils have a substantial potential to accumulate SOC varying from 150 tC ha in the northern to 50 tC ha in the southern regions of the EU. However, the potential rate of SOC gain is assessed as High in the southern regions and Limited in the northern regions.

The verification of the SOCSI shows good agreement between predicted and observed rates of SOC gain in the soils of northern Italy.

The uncertainty associated with the SOCSI varies between 20-30% and mainly depends on the quality of available data on SOC contents.

Key words: soil, soil organic carbon, SOC change, carbon sequestration.

1. Introduction

SOC accounts for more than 95% of the total carbon accumulated in pastures and perennial crops and nearly 100% of the total carbon accumulated in cropland ecosystems. SOC contributes to the resilience of agricultural ecosystems, and increases sustainability of rural livelihoods, which minimizes the negative socioeconomic and environment consequences of agricultural practices. In addition, SOC is among the mandatory items to be reported for agricultural land use under the Kyoto Protocol, and it is one with the highest potentials both in terms of enhancement of C sink and reduction of C emission.

SOC is a characteristic that is mostly affected by bioclimatic conditions and land use. However, in recent years both land use and climate have undergone dramatic changes that in turn cause changes in SOC. With regard to the EU, the changes are particularly driven by numerous land use regulations (e.g. Nitrate Directive, Water Framework Directive, Biodiversity, Climate Change, Natura 2000, etc.). In adition, many regions of the EU are experiencing climate evolution, such as temperature rise and changes patterns of precipitation (IPCC, 2007). As a result of combined land use and climate changes in the EU, the loss of SOC is substantial and is estimated at the rate equivalent to 10% of the total fossil fuel emissions at the pan-European scale (Janssens, 2004). A survey of Belgian croplands (210 000 soil samples taken between 1989 and 1999) indicated a mean annual SOC loss of 76 gC m^{-2} (Sleutel et al., 2003).

A large-scale inventory in Austria showed that croplands were losing 24 gCm^{-2} annually (Dersch and Boehm, 1997). SOC losses across England and Wales in 1978-2003 were about 13 million tonnes of carbon annually (Bellamy et al., 2005). Grassland is seen as a net C sink in most European countries. The overall mean C sink is 60 gCm^{-2} annually. However, the uncertainty of this estimate is high (Vleeshouwers and Verhagen, 2002). Most undisturbed organic soil wetlands accumulate organic carbon (OC) at rates ranging between 0 and 80 gCm^{-2} annually, depending on age, climate and the type of wetland ecosystem, e.g., mires, fens, marshes (Tarnocai and Stolbovoy, 2007). The estimate of the OC balance in the Europe’s drained peatlands demonstrates that more OC is lost due to drainage than is sequestered in undisturbed peatlands (ranging from 0 to 47 gm−2 on average annually). In a number of countries this situation is further exacerbated by the extraction of peat and its use in horticulture, agriculture and in the energy sector (ranging from 0 to 36 gm−2 on average annually; Lappalainen, 1996). However, the reported picture on the status of SOC in the EU is mosaic. A lack of systematic SOC monitoring does not allow a reliable overview of the potential amount and possible rates of SOC changes in the region to be estimated.

Managements to sequester SOC in cropland soils are observed in a number of publications in the EU (Liski et al., 2002; Janssens et al., 2004; Smith, 2003; Smith et el., 2000, 2005; Freibauer et al., 2002), in the USA (Lal et al., 1998) and globally (Lal, 2004). These options include reduced and zero tillage, set-aside, perennial and deep rooting crops, more efficient use of organic amendments (animal manure, sewage sludge, cereal straw, compost), improved rotations, irrigation, bioenergy crops, organic farming, and conversion of arable land to grassland or woodland. In addition, SOC sequestration is limited by availability of land, biological resources and land-suitability. These constraints do not take into account soils and particularly ignore current levels of SOC in croplands. The overestimate of the SOC sequestration potential in EU croplands is one of the results of this simplification (Smith et al., 2005).

The latest analysis of the global potential of cropland to sequester SOC (FAO, 2007) indicates locations (referred to as the "soil carbon gap"), where SOC levels are currently low but, where the technical potential for sequestration is medium-to-high depending on soil type, climate soil moisture and land cover conditions. However, this analysis is based on global databases at a coarse scale of resolution and with variable accuracy.

The main objectives of this study are: (1) to introduce a new method SOCSI for, a spatially explicit analysis of SOC levels to support policy-decision making in the EU; (2) to estimate the potential magnitude of SOC changes (both gain and loss); (3) to evaluate the potential rate of SOC changes (gain and loss) and, (4) to overview options for SOCSI-based policy-decisions regarding SOC management across the EU.

2. Methods and Materials

Definitions

SOC is a measure of the total amount of OC in soil, independently of its origin or decomposition. In cultivated (cropland) soil, the SOC is mostly in the form of humus that comprises highly condensate organic compounds (e.g., sugars, starches, proteins, carbohydrates, lignins, waxes, resins and organic acids). Humus is closely associated with minerals and is relatively stable, changing slowly over time. The study considers SOC quality from two perspectives: (1) static, which contains levels (contents) of SOC and, (2) dynamic, which includes the amounts of SOC potentially available for changes and potential rates of such changes. A variety of characteristics describing SOC quality is termed SOC Status Indicators (SOCSI). The principle structure of the SOCSI is shown in Figure 1.

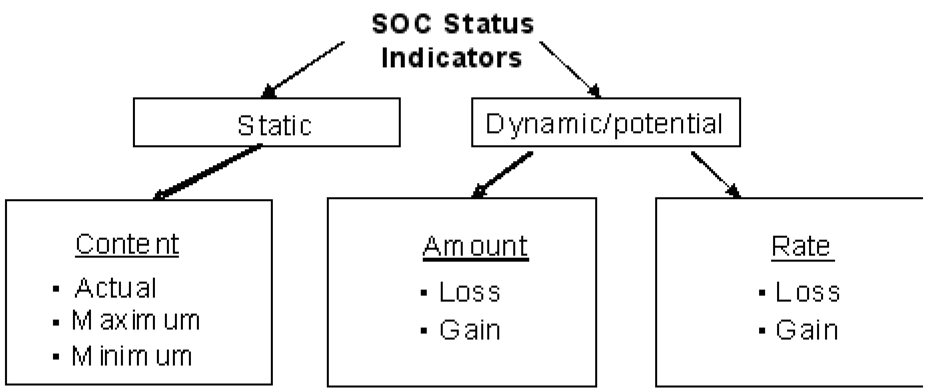

Figure 1. Principle structure of SOC Status Indicators.

Act SOC contents are available SOC values which are the main independent parameters in the SOCSI determination. Data on Act SOC contents can come from measurements (Toth et al., 2006) or maybe generated by models using pedo-transferring rules (Jones et al., 1995), etc.

Most publications report Act SOC in percent (%) of OC. The value in percent shows the relative fraction of OC in a unit mass of soil (e.g. gram of OC in kilogram of soil) related to a specific soil layer (e.g., topsoil (0-30 cm) or total soil (0-100), etc.). However, the value in % is inconvenient for making comparisons because the weight of the soil layer depends on its bulk density (BD) such that the same soil layer is heavier in the soil with the higher BD. The conclusions based on SOC percentage can lead to confusion because soil with a higher OC percentage content may actually contain less OC in weight. In order to account for the effect of BD and, make OC compatible between different soils, the SOC content in percent should be translated into SOC densities (e.g. kgC m^2 or tC ha per certain soil layer). Details on the computation of SOC density can be found in many publications (e.g. see Stolbovoy et al., 2007a). The present study is limited to the 0-30 cm topsoil layer.

The Min SOC and Max SOC threshold values show the range of deviation of Act SOC in a given type of soil (soil typological unit, STU) of cultivated land within a given bioclimatic region.

The dynamic SOCSI include potential amount and potential rate of SOC changes. Both dynamic characteristics depend on the STU and the current level of SOC.

Concepts

Information about SOC levels and dynamics can be found in numerous publications (e.g. Turin, 1965; Orlov, 1990; Stevenson, 1994). Regarding the level of SOC, the study stems from the following basic considerations (Stolbovoy et al., 2007b):

1. The amount and quality of SOC are specific values (characteristics) for a STU (e.g., Podzols, Chernozems, Cambisols have different SOC levels). These characteristics are driven by a combination of the soil-forming factors, such as temperature, precipitation, vegetation type and productivity, parent material, etc. The soil-forming factors regulate the input of OC into soil (mostly vegetation residues) and formation of new humus and, the output of OC from soil including decomposition of vegetation remains and mineralization of existing humus. In a steady state ecosystem (either native or agricultural), the processes of OC input-output are balanced and the level of SOC is at quasi-equilibrium.

2. Land use and land management affect SOC levels and might lead to SOC decline or enrichment (e.g. see Turin, 1965; Goulding et al., 2000).

3. The analysis of deviation of Act SOC content allows us to distinguish Max SOC and Min SOC threshold values that show the potential limits (both amount and rate) of SOC change.

The dynamic nature of SOC can be illustrated by Figure 2.

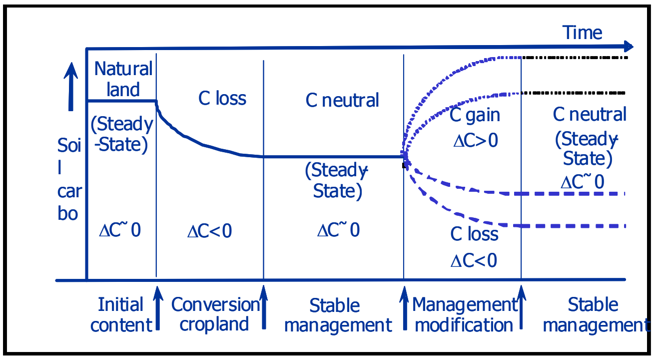

Figure 2. SOC dynamics caused by cultivation and land management changes in agricultural soil (modified from Johnson, 1995). Initial (native) SOC level at the steady state shows OC input-output balanced (ΔC≈0). Conversion into cropland shifts the OC input-output balance towards C loss (ΔC<0). Long-term land management brings the OC input-output processes into a new balance (ΔC≈0) that results in a new steady state of the SOC content. Further modification of land management might lead either to a shift of OC input-output processes towards SOC enrichment (ΔC>0) (dotted line) or towards SOC loss (ΔC<0)(dashed line). The newly established land management leads to a new OC input-output balance (ΔC≈0) and a new steady SOC level.

Method description

SOC levels

The static SOCSI for each STU can be described by Act SOC and critical SOC threshold values such as Max SOC and Min SOC (Fig. 4).

(1) the Act SOC level includes available data on the amount of C (kg m^2 or tC ha) in a given STU. For any STU, the Act SOC content is mainly driven by bioclimatic conditions that control processes of OC input-output and land management which modifies them (e.g., crops, productivity, application of fertilizers, tillage system etc.). Table 1 illustrates that Act SOC levels may vary from 24 to 72 tC ha in different STU in two climate regions in Italy, which is in line with the above considerations on the factors responsible for SOC. In addition the table shows that Act SOC contents in the same STU, but which occur under different climates are different. For example, Dystrict Cambisols in the temperate oceanic climate region contain about 54 tC ha and the same STU occurring in the mediterranean-subcontinental to mediterranean-continental climate region hold nearly 42 tC ha.

Land management is another important factor that controls Act SOC levels in cropland (Fig. 3). As can be seen from the figure, different land management regimes resulted in a 20-30% difference in Act SOC of Cambisols cultivated in the same climate conditions.

(2) Min SOC content is the lowest SOC amount that can be held by a STU. The capacity of soil to hold a minimum level of SOC is mostly controlled by soil texture and clay mineralogy (Stevenson, 1994). Coarse textured soils have low fertility, limited content of water available and are often suffer from droughts. Fine textured (clay) soils are considered marginal for cropping and are not widely used. These soils are difficult for cultivation when dry or wet and not suitable for some crops, e.g. root crops.

The majority of cropland soils are medium textured. Clay minerals in theese soils are smectites (montmorillonite 2:1-layers clays). Due to a relatively low variation in texture and relatively similar composition of clay minerals in cultivated soils of the EU, the deviation of the Min SOC content is rather small (Table 1). The table illustrates that variability of Min SOC levels in different STU is very small (about 20-24 tC ha) and is not dependent on bioclimatic conditions.

The Min SOC level can be associated with biologically “inert” SOC. This fraction is broadly defined as the part of OC which remains in soil over many decades even under bare fallow and without any fertilization. The inert fraction is hardly involved in the transformation of SOC and cannot be influenced by agricultural activities. Inert SOC is generally estimated from the SOC content of “nil plots” in long-term experiments provided the soil has reached quasi-equilibrium (e.g. see Körschens et al., 1998). There are also several models to estimate the value of the Min SOC level (e.g. see Falloon et al., 2000).

Table 1. Deviation of SOC levels (tC ha) in different STU by climate regions in Italy.

Soil	Mediterranean-subcontinental to mediterranean-continental			Temperate-suboceanic		
	Act SOC	Min SOC	Max SOC	Act SOC	Min SOC	Max SOC
Calcic Cambisols	24	24	24	37	24	82
Chromic Cambisols	40	20	83	32	20	51
Dystric Cambisosls	42	20	80	54	20	109
Eutric Cambisols	37	20	66	47	20	85
Vertic Cambisols	36	24	76	41	24	93
Eutric Fluvisols	39	20	62	44	20	86
Gleyic Luvisols	33	20	65	53	20	120
Haplic Luvisols	31	24	61	72	24	144

Under the assumption that variation in the Min SOC contents for a given STU follows normal Gaussian distribution, Min SOC is proposed to be defined by the formula (Act SOC -2δ), where δ is one standard deviation of the Act SOC. Following this assumption, the Min SOC content accounts for 95.45 % of the SOC observations.

(3) Max SOC content is the highest amount of SOC that any STU can hold in given climatic conditions. Generally, Max SOC level depends on the STU and land management. Table 1 illustrates the deviation of Max SOC in different STU (e.g., cultivated Calcic Cambisols in Mediterranean subcontinental to mediterranean-continental climate region contain 24 tC ha and Chromic Cambisols in the same region hold 83 tC ha. In addition the table demonstrates that the same STU occurring in different climates can show substantially different Max SOC contents (e.g., Haplic Luvisols contain 61 tC ha and 144 tC ha in Mediterranean subcontinental to mediterranean-continental region and temperate-suboceanic region respectively).

The Max SOC level in a given STU of cultivated soil can be achieved by a combination of the highest OC input into soil and the land management providing the lowest rate of OC output that, as defined above, includes decay of vegetation remains and humus mineralization. Fig. 4 provides an example of the regulation of Max SOC by different crop rotations and application of different doses of fertilizers.

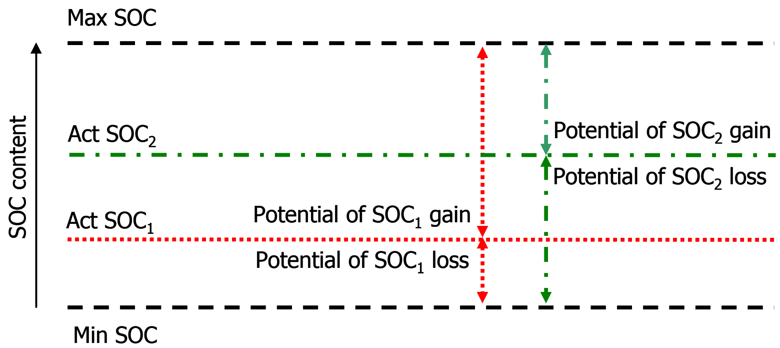

Figure 3. Critical SOC threshold values for two soils belonging to one STU and having a different Act SOC levels. Arrows show different distances to the SOC threshold values that indicate different potentials of SOC gain-loss.

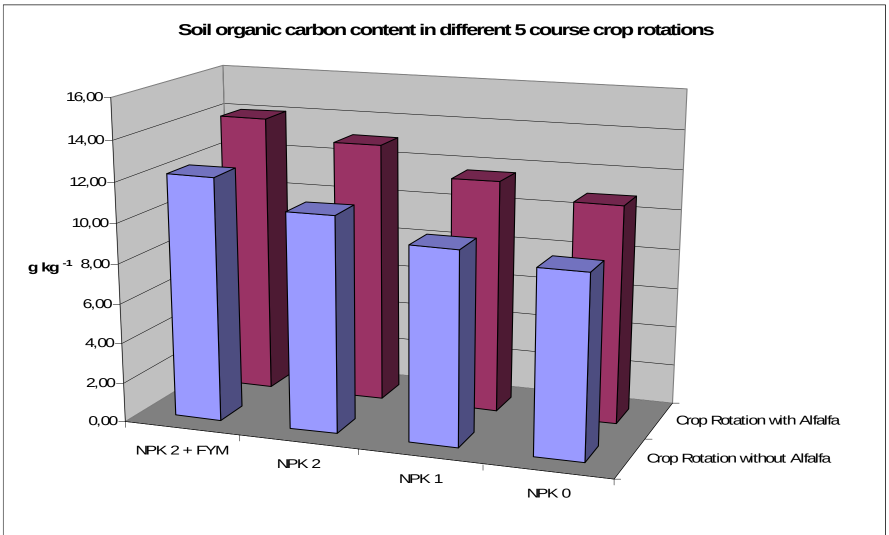

Figure 4. SOC levels in cultivated Cambisols after 20 years of different land management regimes. The experiment illustrates that SOC content is highest where land management includes a rotation with Alfalfa together with double doses of mineral fertilizers (NPK2) and the application of manure (FYM). The SOC level is smallest where the soil is cultivated without Alfalfa and without mineral and organic fertilizers. This field experiment shows that Act SOC level is a function of land management for a given STU. (Source: Toth and personal communications).

Potential rate of SOC changes

Numerous studies demonstrate that the kinetics of SOC change follow an exponential curve in which the rate of SOC gain-loss declines with time approaching a quasi-equilibrium level (Fig. 2 and 5). This suggests that the rate of SOC gain is higher for the soil unsaturated with OC and the rate of SOC loss is higher for OC saturated soil. This general observation is supported by the latest studies, which reported a strong influence of initial SOC content on SOC dynamics (Subramanian et al., 2006). For practical reason, the interval for rate classes of SOC change is set as (Max SOC – Min SOC)/3. The lowest boundary of the rate class is defined by the formula: (Min SOC + (Max SOC – Min SOC)/3). The highest boundary is denoted by the equation (Min SOC + 2(Max SOC – Min SOC)/3).

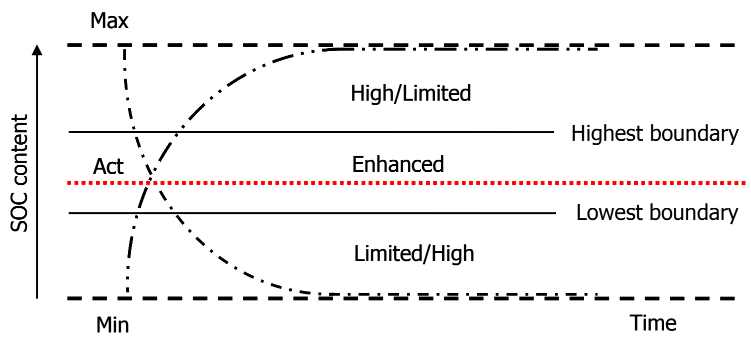

Figure 5. Potential rate classes of SOC change (indicated by horizontal black lines) between Min and Max SOC threshold values (dashed lines). Soils with Act SOC content below the lowest boundary have a Limited rate of SOC loss and a High rate of SOC gain. Soils with Act SOC content between the lowest and the highest boundaries have an Enhanced rate of SOC loss and gain. Soils with Act SOC content above the highest boundary have a High rate of SOC loss and a Limited rate of SOC gain.

Principle equations

The SOCSI concept can be summarized by Table 2.

Table 2. The SOC status indicators and algorithms for their computation.

SOC status indicator	Algorithm
Act SOC	Measured SOC density
Max SOC	Maximum SOC value from all observations
Min SOC	Mean SOC value - 2δ
Potentials to gain SOC	SOC = Max SOC – Act SOC
Potentials to loss SOC	Act SOC - Min SOC
Potential Rate class interval	(Max-Min)/3
Class boundary for Limited potential rate of SOC loss	Act SOC < Min SOC + (Max-Min)/3;
Class boundary for Enhanced potential rate of SOC loss	Act SOC > Min SOC + (Max-Min)/3 and < Min SOC + 2(Max-Min)/3
Class boundary for High potential rate of SOC loss	Act SOC > Min SOC + 2(Max-Min)/3
Class boundary for High potential rate of gain	Act SOC < Min SOC + (Max-Min)/3
Class boundary for Enhanced potential rate of SOC gain	Act SOC > Min SOC + (Max-Min)/3 and < Min SOC + 2(Max-Min)/3
Class boundary for Low potential rate of SOC gain	Act SOC > Min SOC + 2(Max-Min)/3

Input data

The analysis is based on widely available spatially explicit data including soils, land cover, climate and rasterized SOC. All sources are available on the Internet and can be approached through the European Soil Portal (http://eusoils.jrc.it/). The files are indicated below.

Soil data arrived from: \ESDAC\MOSES SGDB4_20m.shp, stu_ptrb.dbf and stu_sgdb.dbf and country.shp

ESDB_v2_CD\soilDB_shapefiles_and_attributes\soilDB_shapefiles_and_attributes

Climatic data come from: \ESDAC\MOSES \soil_regions_v2\climate\eusr_clima

Land cover data originate from: \MOSESDATA\CORINE2000\corine2000v6.lyr

Organic carbon grid is taken from: \ESDAC\MOSES data\octop\octop_insp_directory\octop_insp

Parameterization

The basic parameters to generate the SOCSI were calculated for each STU of cropland by bioclimatic regions. The computation was performed on 1 X 1 km grid cells. All characteristics of the input data are in integer format. Initial raster datasets were assembled in Arc/Info Grid. Microsoft Access is used as the database manager. A link was performed from Access to the resulting GIS database. If the GIS database is recreated, this is automatically updated in the database. All calculations were stored in SQL queries, generating output tables. Forms were used to act as the graphical interface to the database. The resulting SOCSI are shown on-screen form (see below) and can be printed.

3. Results

The main results of the analysis are presented in a set of maps illustrating the distribution of the SOCSI across the EU.

Static SOCSI

Figure 7. Maps of the static SOCSI in the EU.

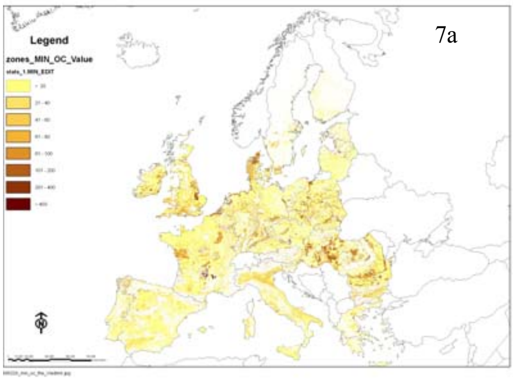

Figure 7a shows Min SOC threshold values that vary from 20-40 tC ha (yellow) to more than 80 tC ha (brown). General distribution of Min SOC threshold values does not follow any geographical pattern. This distribution is in line with irregular allocation of parent materials which define soil texture and clay mineralogy. Characteristics of parent materials are the main controls of spatial distribution of Min SOC threshold values.

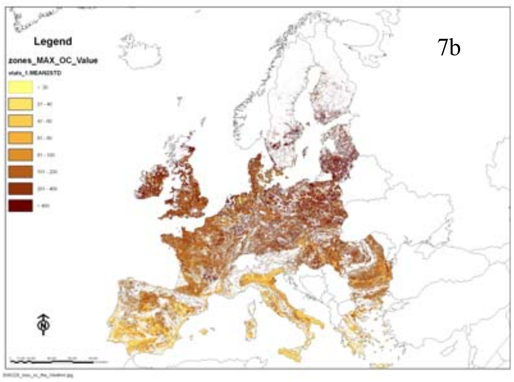

Figure 7b illustrates Max SOC threshold values. The prevailing Max SOC values are more than 100 tC ha (dark brown). These values are observed in the northern two thirds of the EU territory. This distribution coincides with the extent of temperate forest. This bioclimatic zone has a temperature-precipitation ratio favorable for the SOC accumulation. Max SOC values are less than 100 tC ha (light brown). They are observed in the southern part of the EU and coincide with extent of Mediterranean semi-arid bioclimatic zone.

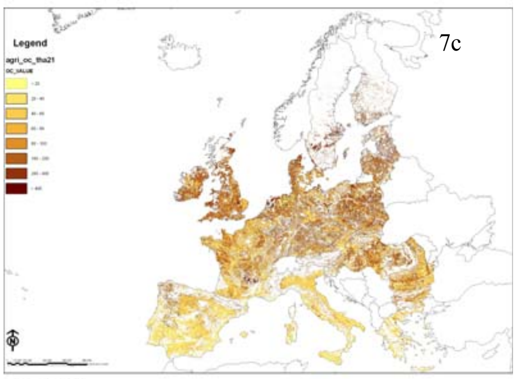

Figure 7c shows Act SOC levels. Distribution of the Act SOC content shows a gradual decrease from north to south. This pattern is in line with the distribution of the main bioclimatic zones. The mosaic pattern in the regional Act SOC contents is explained by a variation in STU and deviations in land management regimes.

Dynamic SOCSI OC loss

Figure 8. Potentials of SOC loss in the EU.

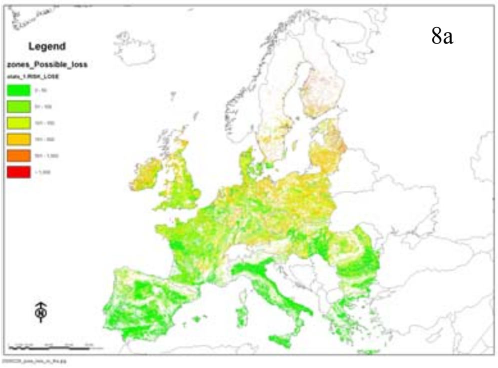

Figure 8a shows that more than 150 tC ha (red) of SOC could potentially be lost in the northern regions. The central part of the EU demonstrates a mosaic distribution of the potentials of SOC loss ranging from less than 50 to more than 150 tC ha (yellow and light green). The southern regions of the EU have low potentials of SOC loss of less than 50 tC ha (green).

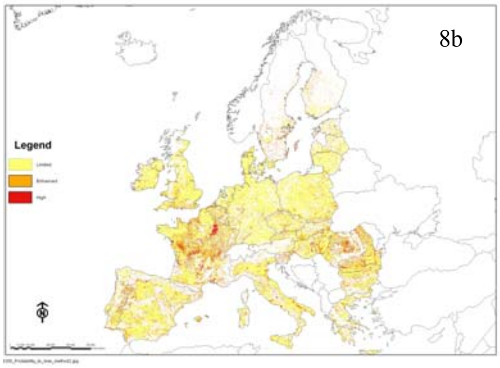

Figure 8b illustrates the zones of potential rate of SOC loss. In general, the distribution of the potential rate zones of the SOC loss is a mosaic. Most of the soils in the northern (Scandinavian) regions have an enhanced potential rate (reddish) of SOC loss. Western and eastern parts of the EU have considerable areas at high potential rate (red) of SOC loss. Southern parts of the EU have a rather mosaic pattern of regions with limited and enhance potential rates of SOC loss.

SOC gain

Figure 9. Potentials of SOC gain in the EU.

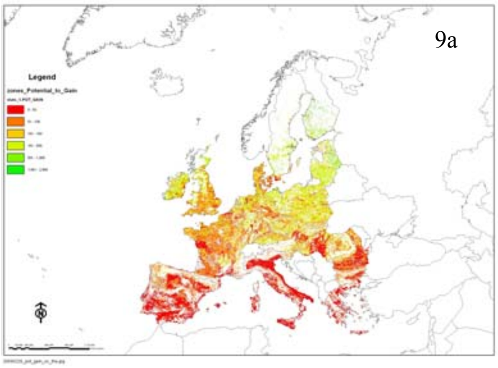

Figure 9a shows that the highest amounts (more than 150 tC ha) of the SOC can be potentially gained in the northern regions of the EU (green). The central regions of the EU demonstrate a mosaic distribution of potentials of SOC gain ranging from less than 50 tC ha to more than 150 tC ha. The southern regions of the EU have potentials of SOC gain less than 50 tC ha.

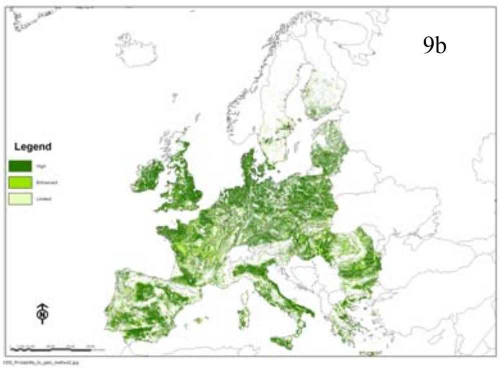

Figure 9b illustrates the zones of the potential rate of SOC gain. Most of the soils in the northern (Scandinavian) regions have a limited potential rate of SOC gain (light green). Rest of the EU has a mosaic pattern of regions with combination of enhance (green) and high (dark green) potential rates of SOC gain.

Verification

Available data allow verification of some SOCSI such as potential rates for SOC gain. For this task we applied two sets of SOC data obtained to monitor the effect of conversion of cropland to forest on SOC. The sites are situated in the Lombardia and Piemonte regions of northern Italy. The plantation in the Lombardia region with slow growing trees (oak, hornbeams) was established in 2003. The site has a developed cover of herbaceous vegetation. The soil is classified as medium-textured calcic Regosol (WRB, 1998). The plantation in the Piemonte region with slow growing oak was established in 1997. The soil was classified as a fine-textured gleyic Luvisol. SOC measurements were performed before and after tree-planting. The first-time observation was employed to elaborate SOCSI indicators. The second-time observation was used to estimate the rate of SOC change. The comparison of the predicted potential rate of SOC gain with measured SOC accumulation rate was used to assess the performance of the SOCSI method. The results of the analysis are presented in Table 1.

The first-time observation of the cultivated soil in Lombardia site shows that Act SOC content was 38.1 tC ha with STD 3.3. tC ha. Min SOC and Max SOC SOC values were defined as 31.5 tC ha and 57 tC ha respectively. Potential SOC gain and SOC loss were 18.9 tC ha and 6.6 tC ha correspondingly. The threshold interval was 8.5 tC ha. The Limited and High class boundaries were established at 40.0 and 48.5 tC ha respectively. The potential rate of SOC gain was estimated as High.

The second-time measurement of SOC level in the soil under the 4-year forest plantation at the Lombardia site observed annual rate of SOC accumulation was estimated at about 5 tC ha.

Similar procedure was applied for the site in the Piemonte region. Due to a lack of data on Max SOC in Piemonte site, a default value from the regional SOC database was selected for the same STU of cropland.

Table 1. SOCSI (tC ha) for cropland converted into forest plantation (data provided by S. Brenna (Lombardia) and M. Piazzi (Piemonte), Italy).

Region	Static				Potential		Threshold interval	Class boundary		Potential rate gain	Observed rate, tC ha yr
	Act	STD	Min	Max	Gain	Loss		Limited	High		
Lombardia	38.1	3.3	31.5	57.0	18.9	6.6	8.5	40.0	48.5	High	5
Piemonte	45.3	10.3	24.7	≈ 90	≈ 45	20.6	≈ 22	≈ 47	≈ 69	High	≈ 3

≈ default value; STD - standard deviation.

The observed rate of SOC accumulation in the Lombardia soil was nearly twice as much as in Piemonte. One reason for this deviation is the difference in the ages between these two plantations. For the Lombardia site the annual rate of SOC gain was taken for the first 4-years period of SOC accumulation after planting which is characterized by the highest SOC gain rate (Fig. 5). For the Piemonte site the annual rate of SOC gain was taken as an average of 10-year period of SOC accumulation.

The observed rates of SOC gain (Table 3) were extremely high for both experimental sites. For comparison, published data on GHG mitigation options (e.g. Smith et al., 2005) report average SOC accumulation rates due to conversion of cropland to woodland for the EU varying between 0.3 (low estimate) to 0.5 (high estimate) tC ha yr. These values are one order less than those in the soils of Lombardy and Piemonte regions. This difference can be explained by dramatic depletion of Act SOC contents in Italy due to long-term cultivation in semi-arid climates. In this situation, the SOC gain rate in Italy is higher than the average for the EU.

The predicted SOCSI show high potential rates for SOC gain which fully coincide with the field observations and laboratory measurements (Table 1). The agreement between estimated and measured rates allowing to conclude that the SOCSI method is operational.

4. Discussion

The study illustrates that agricultural soils in the EU are very different in Act SOC levels and their qualities regarding the potential amounts and rates of the SOC changes. This diversity implies that the measures of SOC management of agricultural soils in the EU should deviate from one region to another. SOCSI can be applied to establish SOC management regimes across the EU.

Generally, the policy of the SOC management can be outlined by the following principle goals: conservation, enhancement and optimization. These goals can be distinguished by three pillars of SOC management that include crop selection and rotation, introduction of technology of land preparation (e.g. tillage), and application of fertilizers to balance organic matter input/output in soil. Detailed examination of the land management options is beyond the scope of this paper. However, it is important to emphasize that SOC level sums practically all natural and humans driven processes in soils. Therefore, explicit measures of SOC managements are closely linked and should be introduced jointly in a system-based manner.

Conservation (maintenance) of SOC level

This goal of SOC management is aimed to keep the Act SOC value stable (Figure 7c), which can be achieved by continuing existing land management practices or by a slight modification of the latter towards the enhancement of OC input and reduction of OC output (e.g. reduced tillage). The implementation of the elements of the SOC enhancement regime is needed to prevent the present-day loss of SOC by agricultural soils in the EU (Janssens, 2004). In the regions where the potential rate of SOC loss is expected to be high (Figure 8), the modification of the crop rotation or residual management can be intensified.

Enhancement of SOC content

This goal may aim at different purposes, such as soil amelioration, shift towards biological agriculture, sequestration of carbon in line with the Kyoto Protocol, etc. To achieve these purposes the soil has to show a high potential amount of SOC gain (Figure 9a) that should likely coincide with the high potential rate of SOC gain (Figure 9b). The most effective means including legumes planting or even conversion into grassland, no-till, etc., can be implemented (Freibauer et al., 2002).

Optimization of SOC content

This purpose is not widely recognized at present. There are a limited number of studies showing that the relationship between SOC and soil productivity is rather complicated and there is an upper threshold of SOC beyond which no further increase in productivity is achieved (e.g. see Toth et al., 2007). Clearly, for the successful implementation of this goal in the future the knowledge of the SOCSI might be of a high significance.

Uncertainty

The uncertainty of the SOCSI outcome can be defined by giving a range of values which are likely to enclose the true value. This may be denoted by the estimate error e.g., value ± uncertainty.

The uncertainty of the SOCSI approach largely depends on the quality of data used to estimate Act SOC levels. Considerable source of the uncertainly which is difficult to assess originates from the use of dominant soils in this study. Very often the soil which occupies a minor area in the polygon and is not accounted might be rich in SOC. The error coming from this exclusion cannot be quantified at present.

The uncertainty of the SOCSI is estimated on the basis of deviation of Act SOC content in the dominant soils. The inaccuracy of initial data was not discussed by the authors (Jones at al., 1995). From assumptions we can suggest that deviation of data on Act SOC content derived from the pedo-transferring rules can be as much as 15-20 %.

The parameters derived from the SOCSI model (potentials amounts and rates of SOC changes) can be estimated by the standard error propagation formula:

$$s(\Sigma) = \sqrt{s_{ActSOC}^2 + s_{Max(Min)SOC}^2}$$, where

$s(\Sigma), s_{ActSOC}, s_{Max(Min)SOC}$ are variances of potential amount and rate of SOC changes, and standard errors for Act SOC, Max SOC and Min SOC respectively.

Applying this formula we estimated the uncertainty of the potentials to gain or loss SOC is within 20-30%. However, taken into account that our calculation stems from general assumptions the result should be received with considerable reservations.

5. Conclusions

1. A new quantitative approach "Soil Organic Carbon Status Indicators" (SOCSI) is developed. The SOCSI comprises a number of SOC characteristics that are combined in two groups: static and dynamic. The static group includes indicators of SOC levels (Act, Max SOC and Min SOC). The dynamic group contains indicators of potentials and rates of SOC changes.

2. Cropland soils in the EU are very different by both the SOC levels and in qualities. Min SOC threshold values vary from 20-40 tC ha to 80 tC ha and do not follow any geographical pattern. This is explained by the mosaic distribution of the parent rocks defining soil texture and clay mineralogy which are the main factors controlling the Min SOC threshold values.

The prevailing Max SOC contents are more than 100 tC ha. These threshold values are observed in the northern two thirds of the EU territory. Distribution of Max SOC contents coincides with the area of temperate forest zone which provides favorable conditions for SOC accumulation. Max SOC contents ofless than 100 tC ha are mainly observed in the southern part of the EU and match with the area of Mediterranean semi-arid bioclimatic zone.

The Act SOC content shows a gradual decrease from north to south. This pattern is in line with the distribution of bioclimatic zones in the EU. The mosaic pattern in the regional Act SOC contents is explained by a variation in STU and a spatial deviation in land management.

3. Cropland soils in the EU show different potentials for SOC loss with the highest amount (more than 150 tC) in the northern regions and the lowest amount (less than 50 tC) in the southern regions. The central regions of the EU demonstrate a mosaic distribution of the potentials of SOC loss which range from less than 50 tC ha to more than 150 tC ha.

The potential rates of SOC loss are High and Enhanced in the northern and central regions and Limited in the southern regions of the EU.

The regions with High and Enhanced potential rates for SOC loss should establish land management regime aimed at the conservation of SOC. The regions with Limited potential rate of SOC loss should be aim for the land management supporting the accumulation of SOC.

4. Cropland soils in the EU have different potentials to gain SOC. The highest amounts (more than 150 tC ha) of SOC can be potentially gained in the northern regions and the lowest amount (less than 50 tC ha) of SOC in the southern regions. The central regions of the EU demonstrate a mosaic distribution of the SOC gain potentials ranging from less than 50 tC ha to more than 150 tC ha.

The cropland soils in the northern regions have Limited potential rate to gain SOC. Most of the central and southern regions have Enhanced and High potential rates of SOC gain.

The regions with a Limited potential rate of SOC gain expect having slow reaction on the enrichment measures. The regions with the High potential rates of SOC gain may be expected to reach more immediate reaction on the SOC accumulation practices.

6. The SOCSI are useful to formulate and diversify policy and decision making regarding SOC management in the EU (e.g., SOC maintenance, enhancement or optimization).

7. An initial verification of the SOCSI in Lombardia and Piemonte regions of Italy shows that the method generates results that agree with the field observations of SOC accumulation rates. However, further testing is needed in different regions of the EU.

5. The uncertainty of the SOCSI varies within 20-30% and depends on the quality of available data on SOC levels. Thus a comprehensive soil characterization is a necessary condition to make policy-decisions on SOC management efficient in the EU.

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Main threats on soil biodiversity:
The case of agricultural activities impacts on soil microarthropods

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Summary

The wide attendance to the Ninth Conference of the Parties of the Convention on Biological Diversity demonstrated that the need to actively protect biodiversity is universally acknowledged. Soil biodiversity represent the largest part of global biodiversity, but its knowledge and the awareness on its importance is scarce. There is still a limited knowledge on the relations between aboveground and belowground biodiversity, between biological diversity and ecosystem functions and especially on the threats on soil biodiversity. This paper provides a short review on soil biodiversity threats and a more comprehensive review on the impacts of agriculture on soil microarthropods.

1. Introduction

Humans have extensively altered the global environment and caused reduction of biodiversity. These changes in biodiversity alter ecosystem processes and change the resilience of ecosystems to environmental change. It is estimated that human activities increased the rates of extinction 100-1000 times (Lawton and May, 1995). In the absence of major change in policy and human behavior our effect on the environment will continue to alter biodiversity.

The recent Conference of the Parties (COP) of the Convention on Biological Diversity[5] demonstrated that the need to actively protect biodiversity is unanimously acknowledged. Biodiversity conservation is essential not only for ethical reasons, but especially for the ecosystem services, that the complex of living organism provide; these ecosystem services are needed for the human being and to keep the planet alive.

Soil represent one of the most important reservoir of biodiversity. The biological diversity in soils is several orders of magnitude higher than that above ground (Heywood, 1995) and is seen as the last frontier for biodiversity on earth (Swift, 1999). Although this importance, study on soil biodiversity are often neglected and the level of knowledge on this subject is scarce. The largest part of soil organisms is still unknown: it is estimated that the numbers of known species of Nematode, Acari and Protozoa, is less than 5% of the total (Decaens *et al.*, 2006).

Indeed the relationships between ecosystem function and biodiversity are particularly evident in soil. Soils, as the control organizing entities in terrestrial ecosystems (Coleman and Withman, 2004), provide a high number of ecosystem services, thanks to the complex communities of organisms living there.

The soil biota plays fundamental roles in soil formation and contribute, directly or indirectly, to many processes such nutrient cycling, waste decomposition, soil structure formation, pollination, etc.

The contribution of soil organism to nutrient cycling in terrestrial ecosystems is well established, and quantified for a number of ecosystems (Swift *et al.*, 1998). Some of these processes, particularly within the N-cycle, are very specific, while other, like the organic matter decomposition, are mediated by a diverse group of bacteria, fungi, protozoa and invertebrate. Pinmentel (1997) estimated in 1,546 billions of dollars, the economic benefits of soil biodiversity. The most obvious service is the waste recycling, while other services are less evident, such as plant pollination; many species of pollinators in fact, have an edaphic phase in their life-cycle.

A necessary starting point, to achieve the objective to preserve soil biodiversity, is to reach an adequate level of knowledge on its extent and its spatial and temporal distribution. If we consider for instance agriculture, studies on soil biodiversity are still scarce in relation to other aspects of agricultural biodiversity (Weigel and Schrader 2007).

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[5] *Ninth Conference of the Parties, held in Bonn (Germany) in May 2008.*

From a general point of view we know that soil biodiversity tends to be greater in forests compared to grasslands and in undisturbed natural lands compared to cultivated fields. However we still have a very limited knowledge on the relationships between aboveground and belowground biodiversity; these two components of terrestrial ecosystems have traditionally been considered in isolation. Only recently there is an increasing recognition of the influence of these components on one other and the fundamental role of the aboveground-belowground feedbacks in controlling ecosystem processes and properties (Wardle *et al.*, 2004).

Also the influence of the abiotic factors on the structure and complexity of soil biota communities is poorly investigated; soil ecology is a young discipline and in many cases is not benefitting from the knowledge of other soil scientist.

The objective of this paper is to briefly describe the potential threats on soil biodiversity, on the base of the DPSIR frame. In many cases we can consider that the main pressures and driving forces acting on biodiversity, are also acting on soil biodiversity; but there are exception and peculiarity that apply only to soil.

2. Pressures and Driving forces

For some extent it is possible to base the evaluation of soil biodiversity pressures indicators, on the global evaluation of biodiversity pressures indicators proposed by Spangerberg (1999, 2007), baring in mind the main differences in the processes affecting above and below ground organisms.

For Europe the main anthropogenic disturbance factors (pressures), have been identified for the three levels of biodiversity: ecosystem, species and gene (EuroStat, 1999; EEA, 2004, 2005).

At the ecosystem level, the main pressures derive from:

- Land use change
- Overexploitation
- Change of climatic and hydrological regimen
- Change of geochemical framework

At species level, the main pressures on soil biodiversity derive from:

- Change of environmental conditions
- Change of geochemical framework
- Competition with invasive species
- Effects of ecotoxics

At gene level, the main pressures derive from:

- Change of environmental conditions
- Effects of ecotoxics
- “Genetic pollution”

Other pressure factors, important for the overall biodiversity, can be considered less important for soil biodiversity. This is the case of habitat fragmentation, and the consequent reduction of biotope size, that theoretically can be also very detrimental for soil biological diversity, but at spatial scales that rarely occurred in practice. In fact there are several scientific evidences on the effects of small scale habitat fragmentation on soil organism, but the dimension of the habitat fragments used in these researches are in the order of few square centimeters, far away from the real world process (Rantalainen *et al.*, 2006; Gonzales and Chaneton, 2002).

It is important to consider that in addition to the above listed pressures, any physical loss of soil, or other soil degradation processes, can lead to loss of biodiversity.

Starting from the analysis carried out by Spangenberg (2007) for the biodiversity in Europe, in Table 1 the main pressures on soil biodiversity, and the related driving forces, are listed.

Table 1. Summary of the main pressures, sorces and driving forces on soil biodiversity.

Pressure	***Source***	***Driving force***
Climate change	The increase in the greenhouse gas emissions in the atmosphere is recognized as the main cause of the climate change: • CO_2 originates when organic materials are oxidized, mainly by burning fossil energy carriers, but also by natural process such as soil and ocean respiration. • N_2O release to the atmosphere originates from agriculture (N over-fertilization), industrial processes and vehicle engines. • CH_4 originates from rice paddies, wetlands, animal breeding and waste site disposals.	Energy consumption Land use intensity Agricultural intensity
Ecosystem/habitat disruption	Land use change and the overexploitation of biodiversity can determine the disruption of ecosystems and habitats. Among the land use change processes the conversion of agricultural land into urbanize areas (soil sealing), and the conversion of natural or seminatural habitats into agricultural land use are the most prominent threat to soil biodiversity.	Land use change Land use intensity
Soil erosion	Soil erosion is a natural process, but is usually exacerbated by the human activities. The overexploitation of pasture or agricultural lands, can promote severe erosion.	Land use intensity Energy consumption (via climate change)
Soil compaction	The use of heavy load machinery in agriculture and the reduction in soil organic carbon content can determine soil compaction.	Agricultural intensity
Chemical pollution	Long range air pollutants Pesticides used in agriculture Persistent organic pollutants Heavy metals Trace elements from industrial processes and vehicle emissions	Agricultural intensity Dissipative use of chemicals
Soil organic matter decline	Decline in soil organic matter is the result of a series of causes, among them: • decoupling of animal husbandry and agricultural activities and consequent reduction of manuring practices • intensification of agricultural practices (frequency and depth of tillage, continuous cropping, narrow crop rotations, etc.) • climate change	Agricultural intensity Energy consumption (via climate change)
Human exploitation	Intensive agriculture Intensive animal husbandry and grazing Forest farming	Land use change Land use intensity Agriculture and animal husbandry intensity
GMO pollution	Accidental, deliberate or residual release of GMOs, with the subsequent establishment of modified organisms or of modified DNA in natural populations.	GMO production, trade and release
Invasive species	Accidental or deliberate introduction of foreign species as a results of globalization (global trade, tourism). The impact of invasive species may be exacerbated by climate change.	Globalization
Habitat phragmentation	The land use change processes and the construction of linear transport infrastructure, generally led to a reduction of natural and seminatural biotope size. This pressure for soil organisms, is not as dramatic as it is for other, above ground, organisms. However it is important to consider that some assets of land use. Even if, for soil organisms, the reduction in size of biotopes is not	Land use change Mobility infrastructures

3. Case study: Impacts of land use change and agriculture on soil microarthropods

3.1 Land use and agricultural practices

Land use is considered to be the main aspect of global change for the next future. In a review on changing biodiversity Chapin III *et al.* (2000) prospect that land use will be the main cause of change in biodiversity for tropical, mediterranean and grassland ecosystems. The fate of soil microarthropods will not differ substantially from this behavior, even if the soil represent a more conservative and resilient environment.

Forests, tropical or temperate, generally represent the biomes with the largest soil biodiversity. Consequently any land use change determining the removal of perennial tree vegetation will produce a reduction of soil biodiversity. In some case forests are followed by pasture or perennial grasslands, while in other case the arable land will take the place of former wooded area. Change in soil biodiversity will therefore be affected by the succession of land use following the forest. Cultivation, for instance, is known to reduce the number and diversity of microarthropod populations from levels observed under natural forest or grassland vegetation.

Reduction of soil biodiversity consequent the urbanization can be also more severe. The urbanization process led to the conversion of indigenous habitat to various forms of anthropogenic land use, fragmentation and isolation of areas of indigenous habitat, and an increase in local human population density. The urbanization process has been identified as one of the leading causes of declines in arthropod diversity and abundance (Davis, 1978; Pyle *et al.*, 1981).

The abundance, biomass and diversity of soil and litter animals are influenced by a wide range of management practices which are used in agriculture. These management practices include variations in tillage, treatment of pasture and crop residues, crop rotation, applications of pesticides, fertilizers, manure, sewage and ameliorants such as clay and lime, drainage and irrigation, vehicle traffic (Baker, 1998). Furthermore differences in agricultural production systems, such as integrated, organic or conventional systems, have demonstrated to affect soil fauna with respect to numbers and composition (Hansen *et al.*, 2001; Cortet *et al.*, 2002).

In the following paragraphs the influence of soil tillage and soil compaction, fertilizer, manure, sludge and pesticides application on soil microarthropods is reviewed. At the end of this chapter some case study concerning the effects of different land use and agricultural practices on soil fauna will be discussed.

3.2 Soil tillage and compaction

Soil tillage operations determine deep modification in soil environment, especially referred to soil architecture (soil structure, porosity, bulk density, water holding capacity), crop residues distribution and organic carbon content. Soil environments directly influence the soil microarthropod community with respect to numbers and composition (Andrén and Lagerlof, 1983) and, according to Farrar and Crossley (1983), their spatial distribution.

Modifications in the habitat of microarthropods overlay their positive effects on soil structure (Bertolani *et al.* 1989; Larink, 1997). Due to the destruction of biological stabilized structure the soil pores created by ploughing are unstable and sensitive to compaction (Ehrnsberger and Butz-Strazny, 1993).

The impact of soil tillage operation on soil organism is highly variable, depending on the tillage system adopted and on soil characteristics. Conventional tillage by ploughing inverts and breaks up the soil, destroys soil structure and buries crop residues (Dittmer and Schrander, 2000) determining the highest impact on soil fauna; the intensity of these impacts are generally correlated to soil tillage depth. Minimum tillage systems can be characterized by a reduced tillage area (i.e. strip tillage) and/or reduced depth (i.e. rotary tiller, harrow and hoe); crop residues are generally incorporated into the soil instead to be buried. The negative impact of these conservation practices on soil fauna is reduced with respect to conventional tillage. Under no-tillage crop production, the soil remains relatively undisturbed and plant litter decomposes at the soil surface, much like in natural soil ecosystems.

The influence on soil organism populations is expected to be most evident when conservation practices such as no-till are implemented on previously conventionally tilled areas because the relocation of crop residues to the surface in no-till systems will affect the soil decomposer communities (Beare *et al.*, 1992). No-till (Hendrix *et al.*,1986) and minimum tillage generally determine an increase in microarthropod numbers. Acari (mites) and Collembola (springtails) have been shown to increase with no-till practices when compared to conventional tillage (Hendrix *et al.*, 1986) because the crop residue cover of no-till provides a readily available food source, moderates extremes of surface soil temperatures, reduces moisture loss, and influences the predominance of certain organisms (House and Stinner, 1987; Perdue and Crossley, 1989; Beare *et al.*, 1992; Doran and Linn, 1994). Conventional tillage (moldboard plowing and disking) cause a reduction of microarthropod numbers as a result of exposure to desiccation, destruction of habitat and disruption of access to food sources (House and Alzugaray, 1989). The influence of these impacts on the abundance of soil organisms will be either moderated or intensified depending on their spatial location; that is, in-row where plants are growing, near the row where residues accumulate or between rows being subjected to possible compaction from mechanized traffic (Fox *et al.*, 1998). This authors also demonstrate that abundance of Prostigmata and Mesostigmata was mainly influenced by soil bulk density and availability of crop residues. According to Filser (1995), Collembola can create large populations under intensive agriculture and are less affected by high management intensity with respect to other soil animals such as earthworms and epigeic predators (Andrén and Lagerlof, 1983; Hendrix *et al.*, 1990; Holland *et al.*, 1994; Wardle, 1995; Sabatini *et al.*, 1997 reported in Filser et. al., 2002). The high management intensity seems to affect mainly epigeal and hemiedaphic Collembola (Çilgi *et al.*, 1993 reported in Filser et. al., 2002), even if Filser et. al. (2002) demonstrated that some epigeal species, such as *Isotoma viridis*, is favored when management intensity is reduced, while the euedaphic *Protaphorura armata* decreased. Other epigeic species however showed an opposite behavior. Soil tillage also influence the sensitivity toward compaction process. Conventional tilled soils have lower load bearing capacity compared to soil subjected to reduced tillage (Ehlers and Claupein, 1994), and consequently are more sensitive toward compaction determined by the increasing wheight of agricultural machinery. Soil microarthropods have different responses to soil compaction. The population size of microarthropods generally decrease with increasing soil compaction, and Collembola seems to be more sensitive to soil compaction than mites (Schrander and Lingau, 1997; Heisler 1993; reported in Dittmer and Schrader, 2000). However soil compaction sensitivity can be very different inside the same taxon; Dittmer and Schrader (2000) evidenced that Collembola species had very different sensitivity in regard to soil compaction: *Mesaphoura krausbaueri* and *Protaphoura campata* abundance decrease with increasing compaction, while *Folsomia fimetaria* and *Sminthurinus aureus* increase with increasing compaction.

3.3 Fertilizer applications

Observations on the impacts of agricultural managements on communities of microarthropods showed that the high input of intensively managed systems tend to promote low diversity while lower input systems conserve diversity (Bardgett and Cook, 1998; Siepel and van de Bund, 1988). It is also evident that high input systems favour bacterial-pathways of decomposition, dominated by labile substrates and opportunistic, bacterial-feeding fauna. In contrast, low input systems favor fungal pathway with a more heterogeneous habitat and resource leading to domination by more persistent fungal feeding fauna (Bardgett and Cook, 1998).

The effects of fertilizers on soil invertebrates are a consequence of their effects on the vegetations and, directly on the organisms. Increase in quantity and quality of food supplied by vegetation is frequently reflected in greater fecundity, faster development and increased production and turnover of invertebrate herbivores (Curry, 1994). The effects of organic and inorganic fertilizers in terms of nutrient enrichment may be comparable, but these two types of fertilizers differ in that organic forms provide additional food material for the decomposer community.

The responses of soil fauna decomposers to mineral fertilizers appear to be variable. Siepel and van de Bund (1988), in an attempt to determine which environmental factors had most effect on the microarthropods community, concluded that N fertilization had a major influence, while factors such as mowing or grazing being of minor importance by comparison. High levels of N fertilization were associated with a decrease in species richness and in numerical abundance of microarthropods (Siepel and van de Bund, 1988). This study also reported changes in microarthropod community structure determined by fertilizers that reduce the proportion of euedaphic collembola groups in favor of hemiedaphic and epigeic collembola. Gunadi *et al.* (2002) reported that there were no significant changes in populations of arthropods in the soil after application of inorganic fertilizer to the tomato and pepper plots. Culik *et al.* (2002) reported that inorganic fertilizer and organic fertilizer did not affect mean density of total collembola and showed little difference in species richness, diversity and evenness between organic and inorganic fertilizer treatments. Seniczak *et al.*, (1998) reported the effects of ammonium-rich air polluted, produced by some nitrogen fertilizers on the mites community. The density of arboreal mites was significantly lower in highly and medium air polluted plots, compared on the control plot, while the density of soil mites was not significantly different.

In a review that compare the soil biological community in fields with conventional and alternative fertilizer strategies (organic or biodynamic) Ryan (1999) concluded that the total soil microbial biomass and the biomass of many specific groups of soil organisms will reflect the level of soil organic matter inputs. Hence, organic or traditional farming practices, that include regular inputs of organic matter in their rotation, determine larger soil communities than conventional farming practices (Ryan, 1999). Also Pfotzer and Schuler (1997) reported that the soil microbial and faunal feeding activity responded to the compost application with higher activity rates than with mineral fertilization. Generally the responses of soil fauna to organic manure will depend on the manure characteristics, and the rates and frequency of application. Herbivore dung, a rich source of energy and nutrients, is exploited initially by a few species of coprophagous dung flies and beetles and, later, by an increasingly complex community comprising many general litter-dwelling species (Curry, 1994). Decomposer populations in soil are often food-limited and benefit from organic amendments, but adverse effects may arise following land-spreading of large quantities of animal manure or related organic wastes in the form of semi liquid slurry. A large number of surface-dwelling arthropods were not affected by slurry in any consistent way. The most abundant collembolan detritivour species, *Sminthurinus aureus* and *Isotomidae spp*., were drastically reduced whereas the phytophagous *Sminthurus viridis* was less affected (Curry, 1980 reported in Curry, 1994).

Miyazawa ***et al.*** (2002) showed that the Collembola population was higher with less tillage, less biocide application and more organic matter input; the mite population was also higher under most conservational treatments and this results suggested that beneficial effects of these practices on the Acari community could be increased by integrating these practices. The mulch application with mineral fertilizer increased the Collembola and Acari populations (Sautter ***et al.***, 1988)

Vermicasts have more favorable physical-chemical properties, increased microbial population, enzyme activities and nutrient mineralization that support plant grow and yeld (Parthasarathi and Ranganathan, 1999). Many studies reported increased microbial activities during passage of food through gut in earthworms (Parthasarathi and Ranganathan, 1999; Burges and Raw, 1967) and higher number of fungi, bacteria and actinomycetes in vermicasts (Burges and Raw, 1967). Parthasarathi and Ranganathan (1999) showed enhanced microbial population and activity in the freshly deposited pressmud vermicasts of *Lampito mauritii* and *Eudrilus eugeniae* in relation to nutrient rich substrate concentrations, multiplication of microbes after passing through the gut, optimal moisture level and large surface area of vermicasts ideally suited for better feeding and multiplication microbes. The presence of earthworms increased the numbers of taxonomic groups of soil arthropods, i.e. isopods, diplopods, spiders, collembolans and ants (Gunadi, 1993). Gunadi ***et al.*** (2002) showed that the conventional compost resulted more mites than vermicompost and there were significant differences in the soil Prostigmata, Mesostigmata and Cryptostigmata populations after application of conventional compost. Vermicomposts produced from cow manure supported more mites (Mesostigmata and Cryptostigmata) and Collembola (Onychiuridae and Isoptomidae) than the other vermicomposts produced from food waste and paper waste (Gunadi ***et al.***, 2002). The mechanisms that increase the numbers of trophic groups of soil arthropods after application of vermicompost probably has a relationship with a high microbial activity in the vermicompost (Gunadi ***et al.***, 2002).

Study related to sewage sludge application on agricultural soils showed an increment of the abundance of Collembola (Lübben, 1989), Carabidae (Larsen ***et al.***, 1986), Oligochaeta (Cuendet and Ducommun, 1990), soil nematodes (Larink ***et al.***, 1990, reported in Bruce ***et al.***, 1999) and Arachnida (Glockmann and Larink, 1989, reported in Bruce ***et al.***, 1999). In some cases, the application of sewage sludge to agricultural land can apport toxic substances that, accumulated in soil, reached potential toxic level for soil fauna (Bruce ***et al.***, 1999). Field studies suggested that metals contained sewage sludge didn't reduce abundance of euedaphic (Lübben, 1989) and epigeic collembola (Bruce ***et al.***, 1997) but may alter their population structure. Bruce ***et al.*** (1999) reported negative effects on collembola community in soil treated with sewage sludge and this effects may be attributed to anaerobe conditions and high ammoniac level. In effects, knowledges related to effects of sewage sludge showed that species more sensitive to toxic substances contained in sewage sludge can disappear, while others, more tolerant, can drastically increase. Acari oribatid (*Rhysotritia ardua ardua*, *Oppia bifurcata*, *Niloppia sticta*, *Striatoppia niliaca* and *Microzetes alces*) increase in soil treated with sludge (Al-Assiuty ***et al.***, 2000) and this effect can be caused by organic matter contained in sludge (Abdel-Hady, 1997). On the contrary, disappearance of *Oppia trifurcata* and *Multioppia bayoumii* may be related to negative effects that sludge has on these species (Al-Assiuty *et al.*, 2000).

3.4 Pesticide application

Pesticides application to the soil can affect the microarthropods communities influencing the individuals performance and modifyng ecological interactions between species. When one or more ecosystem components are impacted by a pesticides, this will affect the microarthropods communities with respect to number and composition. Krogh (1991) evidenced that the application of the insecticide Isofenphos alter both number and composition of different species; in particular Isofenphos determine a negative effect on Collembola, Acarina, Cryptostigmata, Brachychthonoidea, Prostigmata, Mesostigmata e Astigmata. The most evident effect was observed during the first month after the treatment, even if the global effect lasts eighteen month (Krogh, 1991). A soil insecticide chlorpyrifos was most toxic to beetles on the day of application and this level of toxicity declined relatively rapidly with little or no mortality being observed when beetles were introduced three weeks after application (Heimbach *et al.*, 1994).

Pesticides toxicity on soil fauna is determined by different factors, such as pesticides chemical and physical characteristics, species sensitivity and soil type. In fact among soil microarthropod, different taxa showed a variety of responses. Dimethoate application to the soil seems to affect markedly only collembola, while the effect on other edaphic taxa is negligible (Martikainen *et al.*, 1998). A research carried out comparing the effects of five insecticides on soil mesofauna demonstrated that the collembola *Cyphoderus javanus* is more sensitive with respect to the mite *Archegozetes longisetosus* (Pramanik *et al.*, 1998). Within the same taxon, the species-specific responses to a pesticide may favours some species with respect to others. Laboratory tests showed that monocrotophos, methyl-parathion, chlordane e carbaryl cause marked reduction of Cyphoderus individuals while others Collembola species are more resistant (Joy and Chakravorty, 1991). The susceptibility of non-target Coleoptera to pesticides residues, evaluated using an appropriate index (ratio between an exposure function and susceptibility; Wiles and Jepson, 1994) indicated that the staphylinid betlee *Tachyporus hypnorum* would be most affected by deltamethrin residues while the Carabids *Demetrias atricapillus* showed a reduced susceptibility (Wiles and Jepson, 1994).

It is well known that pesticides toxicity on microarthropod communities may be highly variable. A laboratory research, carried out comparing the responses of soil microarthropods to 5 insecticides, allowed to ranks the toxicity of these products: heptaclor>endosulfan>methyl parathion>phosphamidon>dichlorvos (Pramanik *et al.*, 1998); this results were particularly evident for Collemobola and Acari. The insecticide fipronil seems to have a stronger depressive effect on orbatids Acari with respect to carbofuran (Cortet *et al.*, 2002). Other field and laboratory studies showed that the microarthropods total density, especially refered to Acari and Collembola, decreased with aldrin and endosulfan treatement, while dimethoate e phosphamidon determine only a temporary reduction in population density (Joy and Chakravorty, 1991). The soil pesticide concentration is one of the most important factors determining the toxicological response by microarthropods communities. Researches carried out on the Isopod *Porcellionides pruinosus* showed a positive correlation between soil diazinon concentration and the isopods mortality rate; the contemporary presence of diazinon and benomyl determined an increase in isopods mortality rate as a probably results of a synergic effect by the two pesticides on *P. pruinosus* (Vink and van Straalen, 1999). A research for the eavaluation of heptachlor and endosulfan toxicity on soil microarthropods showed the sensitivity of *Cyphoderus javanus* e *Archegozetes longisetosus* to doses 300 times lower with respect to the doses allowed for agricultural applications; furthermore both of these pesticides reduced *C. javanus*, *A. longisetosus*, *Lancetoppia confusaria* e *Scheloribates albialatus* fecondity (Sarkar *et al.*, 2000).

Soil physical and chemical characteristics, such as texture, structure, pH, organic matter content and quality, nature of clay minerals, are important factors determining the toxicity effects of pesticides or other xenobiotics. A study carried out by Joy and Chakravorty (1991) showed reduced toxicity effects, as a function of soil type, according the following order: sand>sandy-loam>clay>organic soil. Often, toxicity of pesticides can related directly to soil organic matter content: acute toxicity of heptachlor, diazinon and parathion on the cricket *Gryllus pennsylvanicus* was in relation to soil organic matter (Harris, 1966 reported in van Gestel and van Straalen, 1994).

However not always pesticides application causes negative impacts on the entire soil microarthropods community. For example for certain type of soil there are evidence that some taxa can obtain a competitive advantage from the application of some specific pesticide. An experiment dealing with benomyl showed that the pesticide application caused an increase of the overall microarthropods community (Krogh, 1991), and especially the Collembola, initially not affected by the treatment, were successively stimulated; the various families of Acari showed different responses: Mesostigmata decreased by 50% immediately after benomyl application, while in Cryptostigmata and Prostigmata the reduction of population took place one week after the treatment. Specie-specific differences in pesticides sensitivity are particularly evident in Collembola: laboratory test to asses benomyl toxicity showed a high sensitivity of *Folsomia candida* compared to *Onychiurus justi porteri*; these differences may be explained by the highest activity of *F.candida* (Tomlin, 1977, citato in Krogh, 1991). Also the resilience of Collembola after pesticide treatments can be specie-specific. After the conversion of conventional to organic farming systems, the colembolan *Entomobrya nicoleti* was not increased while *Isotoma viridis* individuals increased (Frampton, 2000); *Lepidocyrtus spp.* is characterized by longer resilience time than *Sminthurinus elegans* (Frampton, 1997). Field and laboratory experiments on residual toxicity of phosphamidon, dimethoate, methyl parathion and endosulfan showed specie-specific responses: the effects of these pesticides persited more for *Cyphoderus sp*. with respect to *Xenylla sp*. (Joy and Chakravorty, 1991).

3.5 Genetically Modified Organisms (GMO)

The world area planted under GM-crops reached 117 million hectares in 2007 (ISAAA, 2007). There is a great concern on the potential effects of GMO on the environment and on human health. One of the largest uncertainties is the effect of GMO crops on biodiversity and on the fate of modified DNA in the soil.The pesticide resistant GMO Crops represent approximately the 70% of the total, while the insect resistant GMO Crops, and among them the *Bacillus turingensis* (Bt) crops, such as Bt corn and Bt cotton, represent the 20%.

A Bt growing plant will continually produce the Bt protein and release a part of it into the soil. However in several reports (i.e. EPA Report, 2000), there is no mention on the susceptibility of soil-dwelling microbiota to this protein. Most of the experimental activities are carried out using Lepidoptera (i.e. *Helicoverpa virescens*, *Helicoverpa punctigera*), or soil nematodes as test organisms, consequently few information on the effects on soil microarthropods are available. Among the few studies dealing with the evaluation of the effects of commercial GMO crops on soil microarthropods, Pedigo and Bitzer, 2001) reported a lack of any significant deleterious effect of GMO herbicide resistant soybean on the collembolan community in the soil. The scarcity of data on the effect of GMO crops on soil microarthropods, and more in general on soil biodiversity, suggests that further independent studies are advisable.

4. Conclusions

Soil degradation processes, habitat disruption, invasive species and climate change represent the main threats on soil biodiversity. The conservation of soil biodiversity should be integrated with soil protection and, more in general, wth a broad environmental and sustainability strategies. For the European Union this objective could be achieved by broad application of Soil Thematic Strategy and by the effective application of the revised EU Sustainable Development Strategy (EUSDS II). From the review on the potential impacts of agriculture on soil microarthropods, that represents only a limited part of soil biodiversity, it is shown that new threats will affect agricultural soil biodiversity. Especially the use of GMO and the biofuel sector, with the important implications on the agronomic management, are potential threats that are little known. It is therefore evident the need of further investigation on the pressures on soil biodiversity to fully accomplish the objective of its protection.

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Landslide Risk Mapping in Urban Spaces by using ASTER imagery – a comparative case study

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Summary

The aim of this study is to identify landslide probability areas and their threats to elements at risk for two urban areas - Recife (NE-Brazil) and Sarno (S-Italy) - which were affected by serious landslide events during the past two decades. Input Data were derived from all information out from ASTER-satellite imaginary (15-30-60m resolution) to enable a transfer of methodology to other study areas in an easy, fast and cost-effective way.

The Modified Single-Flow-Direction Model (MSF-Model - Huggel et al. 2003) was applied and a suitable constellation of model parameter has been analyzed and discussed by using different reference landslide data (landslide inventory, literature, expertise). Elements at risk were identified by landcover-analysis using the ASTER-VNIR images. The intersection between the landslide probability areas and elements at risk (potential loss) both classified at a high, medium and low scale has allowed the determination of vulnerability within the investigated urban area. The processed results agree with the reference information. Landslide probability and vulnerability in the region of Sarno are much higher and more concentrated than in Recife where low and scattered landslide distribution pattern are dominating. Within the Sarno study area the communes Castel San Giorgio, Bracigliano, Siano, and Sarno and in Recife the politic administrative regions RPA02 (Recife-North), RPA03 (Recife-Northwest) and RPA06 (Recife-South) are representing areas with the highest landslide risk potential.

1 Introduction

Hazard management in settled areas represents one of the most important challenges for urban planners (Smyth and Royle 2000). Arbitrary, non- or mismanaged tendencies of urban settlement in the past and today increase the vulnerability towards human beings and their properties. Landslide hazards become a risk when subjects of protection enter into its spheres of influences. Thus, urban areas with its high density of elements at risk are particularly prone to landslide events with often high economical and environmental damages as soon as fatalities. Apart from this, landslides are part of soil degradation processes and hence representing a threat to soils (Toth 2006). Many studies have applied remote sensing approaches to identify landslide risk zones in an easy-practicable and non time-consuming way (Fourniadis et al. 2006, Kimura and Yamaguchi 2000, Mantovani et al. 1995, Van Westen 2004).

This study tries to apply a similar approach for urban agglomerations which applies only a small number of input data in order to identify landslide probabilities and their risk potential in an easy, fast and cost-efficient way. The results will provide a first approximate delineation of areas of landslide probabilities and vulnerabilities to elements at risk and could be used as a first assessment tool for areas where continuous landslide information is not available yet. Certainly, a first assessment will never reach the quality and quantity of a detailed survey, however in a data and cost limited environment it might prove helpful.

Since the so called Sarno-event 1998 landslide events are recorded and analyzed comprehensively in a national-wide Italian landslide database (APAT 2007). In contrast, the detection of urban landslides in Brazil is mostly confined to fragmentary mapping results. By using the metropolitan city Recife and the rather small urban Sarno region we will discuss, how far the applied methodology is appropriated to determine landslides.

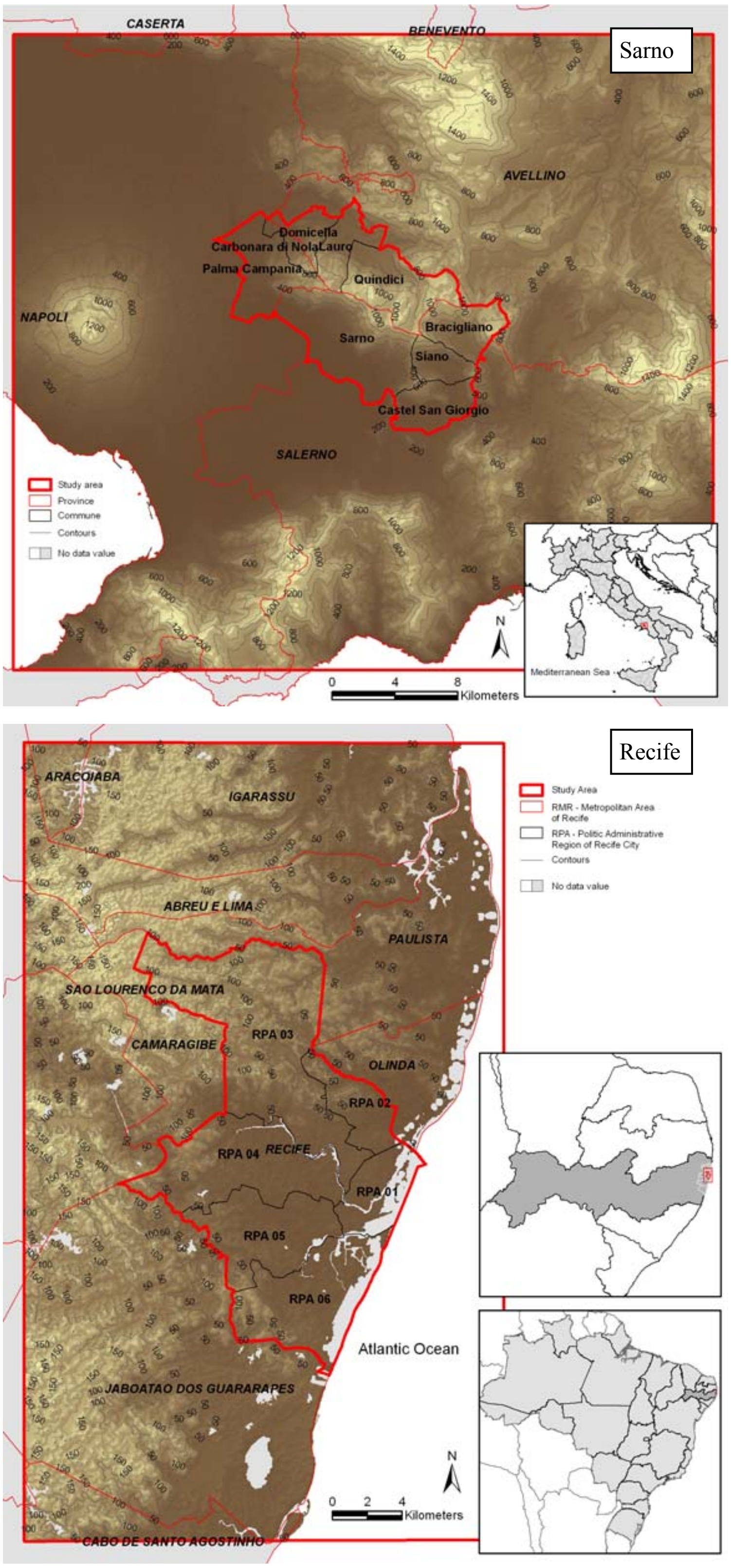

Figure 1. Regional and urban study areas of Sarno (South-Italy) and Recife (Northeast-Brazil)

2. Landslides in urban areas – the examples of the Sarno region and Recife

Metropolitan areas are more vulnerable to disasters because of the concentration of people and activities in defined and limited space, the sheer numbers of people and activities and the proximity to human-made hazards (Anderson 1992). The increase of the incidence of urban hazards events based either on an increasing population already exposed to physical risks or a physical expansion into areas at risk (Smyth and Royle 2000). According to the *Thematic Strategy for Soil Protection of the European Commission* (CEC 2006) main man-induced driving forces for landslides are rupture of topography (e.g. construction works), land use changes (e.g. deforestation), land abandonment and extraction of raw materials. It is widely acknowledged that the modalities of urban landslides (triggering factors, number and intensity of events, losses) are depending on the specific constellation of physical and anthropogenic basic conditions. In this context big cities usually have a greater disaster resilience (Cross 2001) compared to small cities even though former normally have a larger number of elements at risk. In the following the basic urban physical and anthropogenic conditions of the small urban agglomerations in the surrounding of Sarno (South-Italy) and the metropolitan city Recife (Northeast-Brazil) are introduced.

2.1 The region of Sarno

According to historical landslide studies done by Guzzetti (2000), Campania is the second most affected Italian region by landslide disasters. The Italian study area is located in the central-west of Campania in the border triangle of the provinces Naples, Salerno and Avellino, 16 km to the east of the volcano-crest of Vesuvius (Figure 1). The investigated 141.7 km² study area (hereinafter called Sarno area) involves the communes Sarno, Palma Campania, Carbonara di Nola, Domicella, Lauro, Quindici, Bracigliano, Siano and Castel San Giorgio. In that region landslides were already well known in the past but since the event on the 5 and 6 may 1998 they still represent a serious hazard. At that time, Sarno and the adjacent municipalities Quindici, Siano and Bracigliano have been affected by about 150 landslides triggered by long-lasting rainfalls (Brondi & Salvatori 2003). These soil slip-debris/earth flows were characterized from very rapid to rapid speed and with high water content (Cruden and Varnes 1996). Sarno suffered the most devastating mudflow during that landslide incidence which caused 137 fatalities solely in Sarno and 23 fatalities in the other affected communes. Damages to houses and to the production sector in Sarno have amounted to about 26 million Euros (Brondi and Salvatori 2003). Furthermore considerable damages in the infrastructure and visibly changes in the local landscape have been detected (Brondi and Salvatori 2003).

The study area is characterized by the Pizzo d'Alvano Massif with steep slopes which trends from NW to SE and bordered on the SW side by the Campanian Plain Graben. There is a dominance of limestones and calcareous marls intercalations of Cretaceous age (Carbonate Complex). Major regional structures are the anitapenninic trend (NE-SW) and the apenninic trend (NW-SE) (Monti et al. 2007). The elevations vary between 30 m and 1133 m a.s.l. indicating high relief energy with a mean terrain gradient of 34° whereas subvertical limestone cliffs interrupt the morphological continuity of the slopes (Crosta and Dal Negro 2003). The bedrock is influenced by karstic processes which determine both the geomorphological settings and the deep groundwater flow. Pyroclastic deposits (Pumice) on the slopes derive from Vesuvian eruptions and can be observed up to distances of 50 km from the volcano (European Soil Bureau 2001). The depths of pyroclastic layers significantly increase from ridges (1.5 – 2 m) to open slopes (several meters) (Crosta and Dal Negro 2003). The dominance of pyroclastic soils (Andosols) and pumice layers provide landslide enhancing conditions (Cascini et al. 2008). Morphological and stratigraphical interruptions such as road cuts, subvertical cliffs additionally reduce the slope stability (Crosta and Dal Negro 2003). In climatic terms the Campania region is affected by Csa-climate according to Köppen that means warm-moderated conditions with dry seasons in summer and an average rainfall of around 1000 mm. In the study area a high spatial variability of intense rainfall events is typical (Crosta and Dal Negro 2003).

2.2 Recife

Recife is the capital of the state Pernambuco in the north-east of Brazil. The metropolitan region (RMR) extends over an area of 2.798.6 km² and counts about 3 million inhabitants spatially scattered across 14 districts. The urban study are of Recife city (area: 219.5 km²) with its six political administrative regions (RPA1: Centre, RPA2: North, RPA3: Northwest, RPA4: West, RPA5: Southwest, RPA6: South) and 94 municipalities has today around 1.5 million inhabitants and a density of population of about 6800 inhabitants per km² (Figure 1). Compared with other Brazilian megacities such as Rio de Janeiro or Sao Paulo, the population growth of Recife has slowly increased during the last two decades with a rate of about 1 % per year in average (Seliger 2007). At 2004, 38 % of all domiciles in the metropolitan area and 59 % of Recife-city were located in hilly or flood-sensitive areas (Condepe/Fidem 2008). Beginning from the Colonial period until recent times mainly poor classes of population settled in physical vulnerable areas which they occupied legally or illegally. Man-induced driving forces such as transformation of the natural environment without consideration of geological and morphological conditions, initiation of deforestation, cutting of bluff banks, occupation of non-consolidated mounds as soon as modifications of the natural drainage, original slope areas and/or water course have induced the risk of landslides in Recife mainly triggered by high precipitation (Condepe/Fidem 2008). From 1984 till 2003 within the city of Recife 45 debris flows have occurred with in total 112 fatalities (Bandeira 2003). Approximately one-third of the registered landslides caused fatalities. In April 1996, the most destructive debris flow has occurred in the hilly northern part of Recife with 42 fatalities and 2000 homeless persons (Bandeira 2003).

In the metropolitan area of Recife about three-fourth of the territory are characterized by rolling relief. Recife city is mainly influenced by coastal planes with Holocene terraces and alluvial planes that fit smoothly into hilly areas with small amplitudes located in western urban areas. Northward the small hilly areas are replaced by a plain relief with medium amplitudes (Figure 1). The elevations vary between 20 and 140 m a.s.l. Landslide relevant areas are the hilly landforms in the northern and western parts and the transition areas between these two relief units. The geological units in Recife can be divided in the Costal plane (fluvial and fluvial-laguna deposits with sandy-clayey sediments occasional with organic matter), the Barreiras Formation in the north (intermediary sand and clay layers), the Formations Beberibe (homogeneous sandy layers) Gramame e Maria Farinha (clayey sediments) in the northeast and the Formations Cabo (layers with big blocs and gravel with intermediary layers of sand and clay) and Ipojuca (clayey-sandy sediments) in the southwest (Alheiros 1998). In the western part of Recife affiliates the Crystalline Basement with sandy-clayey sediments. Onto the specific sediments the predominant soil types of Podzol (sandy), Vertisol (clayey) and Terra rossa (sandy-clayey) have been evolved. Especially on clayey grounds landslide susceptibility is high (Alheiros et al. 2003). According to W. Köppen Recife has a rainy tropical climate (As') with dry summer (October till February) and rainy season (March till September) and annual precipitation rates occasionally greater than 2000 mm. During the rain-laden period between April and July there are precipitation values greater than 300 mm in average per month and humidness till 90 % (INMET 2006). Such climate conditions abet the chemical weathering of granite rocks as soon as feldspar sediments that increase the clay content and hence the probability of landslides incidences (Alheiros et al. 2003).

3. Methodology

The study is composed of three main working-steps I) preparation of input data based on ASTER satellite imagery, II) determination of landslide probability areas using the Modified Single Flow Direction Model (MSF-Model) developed by Huggel et al. (2003) and III) risk analysis using a simple vulnerability matrix according to Cascini et al. (2005). The analysis of suitable input parameter and the check of model capability for processing landslide probability have been carried out first at a regional study area (Figure 1, big red rectangle). This probability results have either been evaluated with reference data obtained from digital IFFI-inventory for the Sarno case or with information obtained from field trips for the Recife case (Figure 1). In a second step the probability results have been used for a following risk analysis at urban scale. The chosen approach (Figure 2) sees itself as a pre-evaluation carried out theoretically in order to approximately characterize the recent (year of recorded ASTER-image) urban landslide probability and vulnerability.

3.1 Data preparation

ASTER imagery

Two ASTER images of Recife (Scene-ID: AST_L1A.003:2018097607 and Scene-ID: AST_L1A.003:2018097602; Cloud Cover: 0 %; Date: 2003/10/22) and one of the Sarno region (ID: AST_L1A.003:2035431352; Cloud Cover: 0 %; Date: 2006/7/25) have been ordered via the USGS GLOVIS interface. Elevation values were extracted in ENVI4.3 using the AsterDTM2.2 for a pixel resolution of 30 m with a horizontal absolute accuracy better than 50 m and a vertical absolute accuracy of 20 m. Using the generated DEM, the remaining spectral bands of the ASTER imagery was orthorectified. During the DEM generation correlation files were generated, and DEM pixels with correlation values less than 70 (Recife) respectively 20 (Sarno region) have been rendered void. Generally, the correlation in the Sarno region image showed much better values than in the Recife image. The shallow hilly relief of Recife with less contrast makes the derivation of topography parameters more difficult compared to areas with stronger relief characteristics as is the case in the study area of Sarno. Hence, in areas with very shallow and homogeneous relief conditions along the coast of Recife, creation of DTM was limited or not possible.

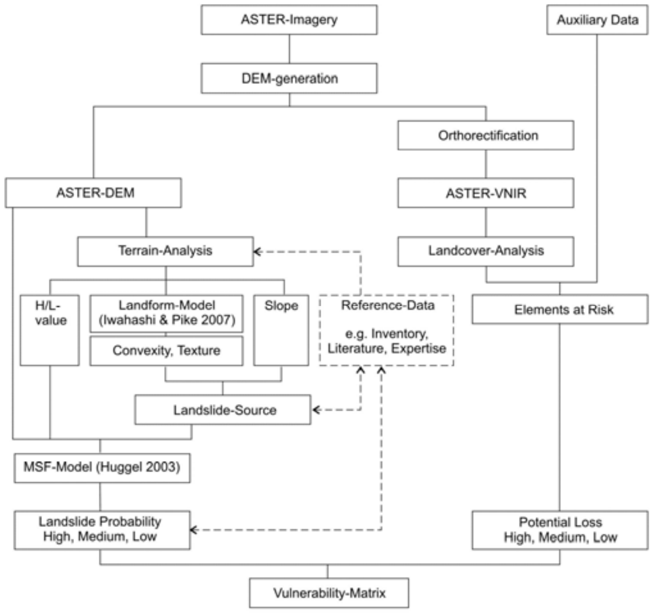

Figure 2. Workflow based on ASTER-imagery.

Another observation is a much higher proportion of clouds and cloud-shadows in the ASTER image of Recife as usual under tropical conditions. Areas covered by clouds, cloud-shadows or waterbodies have been classified using the orthorectified VNIR data in Ecognition 5.0 and were set to void as well. Void areas up 25 cells were filled using an iterative focal mean in Arc/Info GRID. Finally, the DEM was smoothed by applying a low pass iterative filter (three iterations) and sinks were filled using a sink threshold of 10 m. Finally, generated DEM-accuracy was checked with the aid of homogeneity of contours. In case of non-homogenous contour-trends the fill sink and smooth parameters were adapted accordingly. From the prepared DEM all model-relevant parameter were derived by performing an automated terrain analysis (Reuter et al, 2005; Iwahashi and Pike, – see www.geomorphometry.org for scripts).

Land cover information has been classified based on ASTER-VNIR images with a 15 m resolution in Ecognition (Scale Parameter: between 10 - 30 depending on landuse class, Colour: 0.9, Compactness: 0.5, Shape: 0.1, Smoothness: 0.5). This analysis allows a segmentation of image objects and its classification according the nearest – neighbour algorithm. Infrastructure areas could not be extracted from ASTER images due to the fact that spectral signatures were too similar in comparison to agricultural meadow and urban area signatures and classified results were not reliable. Therefore it was decided to use consequently auxiliary data sets for infrastructure. The accuracy of classification in Ecognition is limited by the given image heterogeneity. The more clear and variegated colour scheme merries the differentiation of homogeneity criteria and completeness of classes. Finally all produced data classes have been exported into grids.

Reference landslide information

After the landslide event in Sarno in 1998, Italy launched a national landslide inventory database within the IFFI-Project (Inventory of landslide phenomena in Italy). All information is provided on the project webpage http://www.mais.sinanet.apat.it/cartanetiffi/cartografia.asp. The IFFI database was updated 2006; further updating is continually realized by regions and autonomous provinces. The database includes the landslide inventory in combination with useful vector (urban areas - Corine Land Cover 2000, the roads and railways, the administrative boundaries, the drainage system) and raster layers (digital terrain model, digital orthophoto TerraItaly it2000, Landsat satellite images and IGM topographic map). All recorded landslides contain information on its most important parameters additionally demonstrated by documents, photos or videos (APAT 2007). The identification points of Landslide phenomena's (abbreviated: PIFF) describe landslide information on three levels of increasing detail (Trigila et al. 2007). The slide propagation area (hereinafter called tracks) starting from the PIFF is displayed and characterized according to landslide typology based on the IFFI-Guide of landslide-compilation, appendix one (Amanti 2001). The location of PIFF corresponds to the most elevated point of the landslide crest (Trigila et al. 2007) and can be regarded hence as source points of landslides. Information about PIFF, landslide-tracks and infrastructure has been downloaded in form of pdf from the IFFI-webpage, converted in tiffs and georeferenced in ERDAS, classified and extracted in Ecognition. The PIFF and landslide-track data have each been classified as one layer without doing any differentiations to PIFF-levels or track-typologies.

Unfortunately Recife has not launched a consistent digital landslide database for its urban space yet. Landslide programs like “Vivo o Morro” (Alheiros et al. 2003, Condepe/Fidem 2008) or “Programa Guarda Chuva” (CODECIR 2008) predominantly initiated by local authorities, universities or engineering companies discuss current sensitive landslide areas and its potential risk in a qualitative way but don’t report starting zones (sources) and tracks of former incidents. Therefore experiences of the lead author’s field trip in 2006, image analysis in Google Earth as well as landslide study results within the region Camaragibe westwards of Recife city done by Bandeira (2003) have been taken into account for evaluating processed results in a rather limited way. A distinction in starting and track zones was not possible for the Recife area.

3.2 Landslide probability

The determination of landslide probability according to the MSF-model is preferred because it requires less input data, needs less computation time, is easily available and requires less technical know-how (Gruber et al. 2008). The D8-method forms the basis of the MSF-model by calculating multiple flow directions based on single flow-directions to assign flow from each cell to one of its neighbours in the direction of the steepest descent path (Huggel et al. 2003, Gruber et al. 2008). The inherent uncertainties in our dataset pose a threat to the initial detection of possible urban landslide areas, however could only be overcome by using more advanced techniques like the MFD model (e.g. see Gruber et al 2008), therefore these uncertainties have been accepted as is. More detailed analysis can be realized at a later stage and are desired (e.g. field studies, higher-resolution satellite imagery, complex models, expertise). All working steps for running the MSF-model in Arc/Info are described clearly and precisely in Gruber et al. (2008) stating all necessary ArcInfo-commands for processing and can be found at www.geomorphometry.org.

Application of MSF-Model

The basic input parameters required for MSF-model are a DEM, a source area that defines where the mass movement begins and a stopping condition (H/L-value) of the landslide event. The choice of the source area should be defined as detailed as possible in order to avoid too extensive and unrealistic landslide predictions. The flow direction has been calculated by using the DEM and the D8 algorithm and was converted into grids afterwards. With the aid of the pathdistance command in ArcInfo the flow propagation downslope along the steepest descent path and in the 45° lateral diversion has been calculated. The result (Li) describes the horizontal distance from the starting location of the flow to each cell potentially affected by it. The ratio of Li and the value of the horizontal distance from the source location and the lateral flow divergence (Fri, flow resistance) describe the probability that a certain cell is affected by a debris flow (Pqi).

The definition of a mass flow stopping condition H/L is necessary to constrain the flow propagation and should be therefore defined as detailed as possible against the background of the predominant terrain conditions. H/L is defined by the ratio of the vertical distance of each cell from the maximum elevation of the start location (H) and the appropriate path length (L). The limiting flow-reach condition depends on the selected H/L-value. Normally lower H/L-values should be chosen in areas either with shallow relief characteristics or in areas where larger and more fluid debris flows are expected. Therefore different terrain conditions between Recife (shallow hilly with lower amplitudes) and Sarno (stronger pronounced relief with higher amplitudes) will be reflected in adequate limiting flow reach condition for both study areas. Finally the already modelled Pqi-ratio in consideration of the chosen stopping condition result in a final grid whose cell values represent a qualitative probability of being affected by the simulated mass flow which has a stopping condition equivalent to the defined H/L value (Gruber et al. 2008) All cell values ranging between zero to one (correspond to 0-100%).

Determination of landslide sources

While the DEM can be easily processed from ASTER-images, the definition of a landslide source that approximately agrees with the circumstances in reality is more difficult. It has been tried to define sources by using only three basic input parameters of topography such as slope, local convexity and surface texture by using the DEM of Recife respectively Sarno. The automatic classification of the two local convexity and surface texture was realized by using an unsupervised nested-means algorithm and a three-part geometric signature presented by Iwahashi and Pike (2007) which provides a scale free terrain classification solely based on the inherent DEM properties. The model provides up to 16 classes of terrain forms which can be derived from a matrix that includes the components texture (fine, coarse), convexity (low, high) and slope (steep, gentle). The classes one (steep slopes, fine texture, high convexity) and two (steep slopes, coarse texture, high convexity) were chosen as part of initial landslide areas and were grouped into one terrain-grid. The slope conditions in degree have been derived separately from the DEM; four slope grids with inclinations greater or equal 10° (class 1), 20° (class 2), 30° (class 3) and 40° (class 4) were built. Each slope grid was combined with the former generated terrain grid by what four different source grids both for Sarno and Recife have been produced (e.g. SS1).

For the Sarno study the given PIFF-source of the IFFI-inventory enables a conformity analysis between the self-processed and the reference landslide source. The combined slope and terrain grids were adopted in the way that its intersection area was assigned to the value one (condition fulfilled) and the remaining areas to the value two (condition non-fulfilled). Source areas in the reference grid of Sarno were also assigned to the value one, respectively two for areas with no source information. The conformity analysis between reference source (PIFF-data) and the four self-processed sources was realized by a cross validation in Arc/Info. The result matrix (Table 1) shows four possible combinations of the values 1 (source existent) and 2 (source non-existent) whereas the first number describes the reference source (rs) and the second number the processed source (ps). Hence, the matrix contains four essential statements: ps well predicted (case 1/1, ps hits rs), ps is underestimated (1/2, where rs there is no ps), ps is overestimated (2/1, where ps there is no rs) or no information (2/2, neither ps nor rs). The given percentage values in Table 1 refer to the sum of areas with the cross combinations 1/1, 1/2 and 2/1. A conformity check for the produced Recife sources could not been realized because of the lack of source reference information as already mentioned.

As demonstrated in Table 1 for the Sarno case, the greater the chosen slope for source-processing the greater the underestimation and the less the overestimation compared to the reference source. The low dimension of matching as soon as the high overestimation rates of the processed sources mainly caused by the comparison of punctual data (reference source, PIFF) with spatially data (processed source) because it is hardly possible to produce a self-processed source which matches well with the IFFI point sources. The degree of overestimation could probably be reduced by performing the conformity analysis only with processed medium and high probabilities results. The matching could probably be enhanced by taking into account of more landslide influencing parameters such as landcover, lithology or soils. In general the best processed source zone is represented by high matching values (1/1) as soon as low under- (1/2) and overestimated (2/1) values. In this connection a greater overestimation is more acceptable than a high underestimation. The best matching compromise compared to the reference sources provides the processed source SS3 (input parameter: terrain classes one or two; slope greater or equal 30°).

Determination of probable landslide areas

Using the processed landslide sources, the DEM and stopping condition (H/L-value) landslide affected areas have been predicted by applying the MSF-model. Hence, the calculated areas represent the qualitative probability of being affected by the simulated mass flow (Gruber et al. 2008). In a first step the landslide probability was analyzed and discussed for the regional area and in a second step in detail for the urban space (Figure 1 and 3) In order to find the best prediction-result the prepared four different source grids were processed with six different stopping conditions 0.17, 0.19, 0.21, 0.23, 0.25 and 0.27 within the MSF-model. For Recife the MSF-model was also processed with lower additional stopping conditions 0.11, 0.13 and 0.15 because the number of processed landslide probability areas was under-represented even by applying an H/L-value of 0.17. For the future text each single combination will be named as a combination (e.g. SS30.19) based on the source (e.g. SS3 for Sarno-source 3) and the H/L values (e.g. 0.19 for an H/L value of 0.19)

For the Sarno study the conformity of all 24 predictions possibilities was checked by a comparison to the landslide areas of the IFFI-inventory equally to the procedure applied to source validation. The results in Table 2 show much better matching values (1/1) compared to the matching results of the source analysis, because the success probability between crossing area/area is much higher than between point/area. Therefore the matching values (case 1/1) are higher and the overestimation values (case 2/1) lower compared to the source analysis results. Nevertheless the overestimation of processed probability areas is still high. On the other hand the underestimation of relevant areas is in most times higher than in the source analysis. The modelled landslide probability result SS30.19 represents the best compromise of matching compared to the reference-tracks of IFFI. In other words, the source - defined by intersection of areas with landform classes one or two and areas with slopes greater or equal 30° - and the limiting flow reach condition defined by areas with H/L-values greater than 0.19 provide the best matching. This H/L values perfectly matches with recommendations given by Rickenmann (1999) and Huggel et al. (2003). In this example the best identified landslide source has also provided the best processed probability results where 6.2 % of the processed probability areas hit the reference area, 6.5 % underestimate and 87.3 % overestimate (Table 2).

Table 1. Conformity-analysis between processed sources and reference sources (IFFI).

	Source **SS1**		Source **SS2**		Source **SS3**		Source **SS4**	
	Slope >= 10		Slope >= 20		Slope >= 30		Slope >= 40	
T C 1, 2	16.19	*2.29*	9.97	*2.72*	4.06	*2.85*	0.45	*1.32*
	4.20	*0.59*	10.42	*2.84*	16.32	*11.47*	19.94	*58.29*
	686.21	*97.12*	345.77	*94.43*	121.98	*85.68*	13.81	*40.39*
	1102.79		1443.23		1667.02		1775.18	

Table 2. Conformity-analysis between processed landslide areas and reference areas (IFFI).

	0.17		*0.19*		*0.21*		*0.23*		*0.25*		*0.27*	
S S 1	22.94	*5.59*	20.22	*5.42*	18.27	*5.31*	16.70	*5.24*	15.32	*5.19*	14.00	*5.11*
	46.86	*11.41*	49.58	*13.28*	51.53	*14.98*	53.10	*16.67*	54.49	*18.45*	55.80	*20.38*
	340.86	*83.00*	303.51	*81.30*	274.27	*79.71*	248.81	*78.09*	225.50	*76.36*	204.00	*74.51*
	1398.73		1436.08		1465.32		1490.77		1514.08		1535.59	
S S 2	31.20	*5.81*	27.99	*5.78*	24.44	*5.62*	21.34	*5.47*	18.84	*5.41*	16.60	*5.34*
	38.60	*7.19*	41.81	*8.63*	45.36	*10.42*	48.46	*12.43*	50.96	*14.64*	53.20	*17.12*
	467.36	*87.01*	414,54	*85.59*	365.46	*83.96*	320.15	*82.10*	278.21	*79.94*	240.91	*77.54*
	1272.22		1325.04		1374.12		1419.44		1461.37		1498.67	
S S 3	35.83	*6.02*	33.89	***6.16***	31.76	*6.25*	28.63	*6.11*	25.61	*5.95*	23.24	*5.92*
	33.97	*5.71*	35.91	*6.53*	38.04	*7.49*	41.18	*8.79*	44.20	*10.27*	46.56	*11.86*
	525.32	*88.27*	480.53	***87.32***	438.16	*86.26*	398.82	*85.11*	360.69	*83.79*	322.83	*82.22*
	1214.26		1259.05		1301.43		1340.76		1378.89		1416.75	
S S 4	24.31	*5.74*	23.45	*6.06*	22.28	*6.28*	20.99	*6.48*	19.15	*6.47*	16.75	*6.11*
	45.49	*10.74*	46.35	*11.98*	47.52	*13.39*	48.81	*15.07*	50.65	*17.12*	53.05	*19.36*
	353.97	*83.53*	317.05	*81.96*	285.00	*80.33*	254.01	*78.44*	226.10	*76.41*	204.21	*74.53*
	1385.61		1422.53		1454.59		1485.57		1513.48		1535.37	

Explanations to tables 1 & 2:

	Source **SS1**		Source-grid
	Slope >= 10		Slope-grid
T C 1, 2	16.19	*2.29*	Case 1/1: ps correspond to rs
	4.20	*0.59*	Case 1/2: ps underestimated
	686.21	*97.12*	Case 2/1: ps overestimated
	1102.79		Case 2/2: neither ps nor rs
	[km²]	[%]	

Terrain-grid

	0.17		H/L-value (Stopping condition)
S S 1	22.94	*5.59*	Case 1/1: ps correspond to rs
	46.86	*11.41*	Case 1/2: ps underestimated
	340.86	*83.00*	Case 2/1: ps overestimated
	1398.73		Case 2/2: neither ps nor rs
	[km²]	[%]	

Source-grid

Dark green…best value, grass green…2nd best value, light green…3rd best value

For Recife the results have subjectively been evaluated for a small pilot area in the northwest of Recife which was subject of a field study by the author in 2006 and by analysing results of a landslide study done by Bandeira (2003) in the Camaragibe region (Figure 3). The attempt to create an own reference landslide dataset by digitizing visible landslide areas in Google Earth failed because of difficult or non-sufficient perceptibility of former or recent landslide zones. The best landslide probability results are represented by the case RS10.13. The best result both for Sarno (SS30.19) and Recife (RS10.13) have been selected for a detailed urban landslide analysis which discuss the proportions and distributions of areas prone to high (probability greater or equal 0.66), medium (probability between 0.33 and 0.66) or low landslide probabilities (probability below than 0.33) (Table 3 and Figure 3).

3.3 Landslide vulnerability and Landslide risk

According to Cascini et al. (2005), a risk increase in urban agglomeration is related to an increase of demographic pressure and territory mismanagement. Depending on the protection value of elements at risk, risk can be described as acceptable, tolerable or unacceptable whereas the highest attention at all is given at risks to lives. In order to quantify the existent risk within the investigated urban spaces, a vulnerability analysis has been carried out by using a pragmatic approach. A vulnerability matrix contrasts the landslide probability levels (high, medium, low) with the levels of potential loss of elements at risk (high, medium, low) (Cascini et al. 2005). The following elements at risk have been classified and assigned to one of the three potential loss classes (in brackets): Built up areas with high housing density (high), Built up areas with medium or low housing density (medium), infrastructure (medium), meadows and agriculture (low), forest (low). All mentioned data with the exception of infrastructure have been derived from the ASTER-VNIR images in Ecognition. Information about roads and railways have been gained either from the IFFI background information (Sarno) or from municipal data of the Prefecture of Recife provided in 2006 and, hence, represent the only auxiliary information which has not been gathered from ASTER-imagery. Non-classifiable areas (e.g. clouds) have been defined as no data values. Overlaying and intersecting the layers of landslide probability and element at risk in GIS allowed the delineation of areas with high, medium or low landslide vulnerability (Figure 4). Thus all remaining areas are described either by no vulnerability or no data values.

4. Results

4.1 Landslide probability assessment

More than a quarter of the total investigated regional study area around Sarno with a size of 1809.4 km² is prone to landslides with high (10.5 % of study area), medium (11 %) or low probabilities (7 %). The main landslide areas range from the north to the south and from southwest to east along the mountain edges of the Pizzo d'Alvano Massif. The Vesuvius marks a further spot with mainly high landslide probabilities along its slopes, which can be confirmed based on the IFFI landslide areas. Focused on the urban area with a total size of 141.7 km² areas with medium landslide probabilities dominates (16.2 % of the urban study area) followed by areas with high probability (13.8 %) and low probability (9.3 %). About 76 % of all identified probable landslide areas are represented by areas with medium or high probabilities. Particularly for the urban analysis the reference landslide tracks overlay the processed high and medium landslide areas pretty well (Figure 3). The converse situation can be observed in the north-east of the regional study area. Here the model has assigned only a small number of landslide probabilities even though many reference tracks are existent. This is owed to the fact that within this zone starting sources have not been processed because of the relative homogeneous and gentle local terrain conditions. Consequently in areas without delineated source areas, the MSF-model is not able to process areas of landslide probabilities. Only 5.1 % of the regional study area (size: 1280 km²) of Recife is jeopardized by landslides (Table 3). Areas with medium probabilities marginally dominates (2 % of the study area) followed by areas with low probabilities (1.6 %) and high probabilities (1.5 %). Potential landslide areas are ranging from the southwest to the northeast starting from the transition zone between coastal plane and the hilly areas. The scattered distribution of the processed probability areas is caused by the undulated terrain conditions with small scale differentiations. Noticeable line-like landslide probabilities areas can be observed in zones where rivers cut in mountainous regions. These zones are characterized by comparatively high relief energy. Within the city of Recife only 4.2 % of the urban area is prone to landslides. In this regard the urban region RPA03 is the main landslide affected area in Recife. 1.6 % of the urban area is characterized by medium landslide probabilities followed by areas with high probabilities (1.4 %) respectively low probabilities (1.3 %). Modelled high probability areas in northern, north-eastern and south-western urban parts are corresponding well with the high sensitive geological formations Barreiras, Gramame e Maria Farinha as well as Ipojuco.

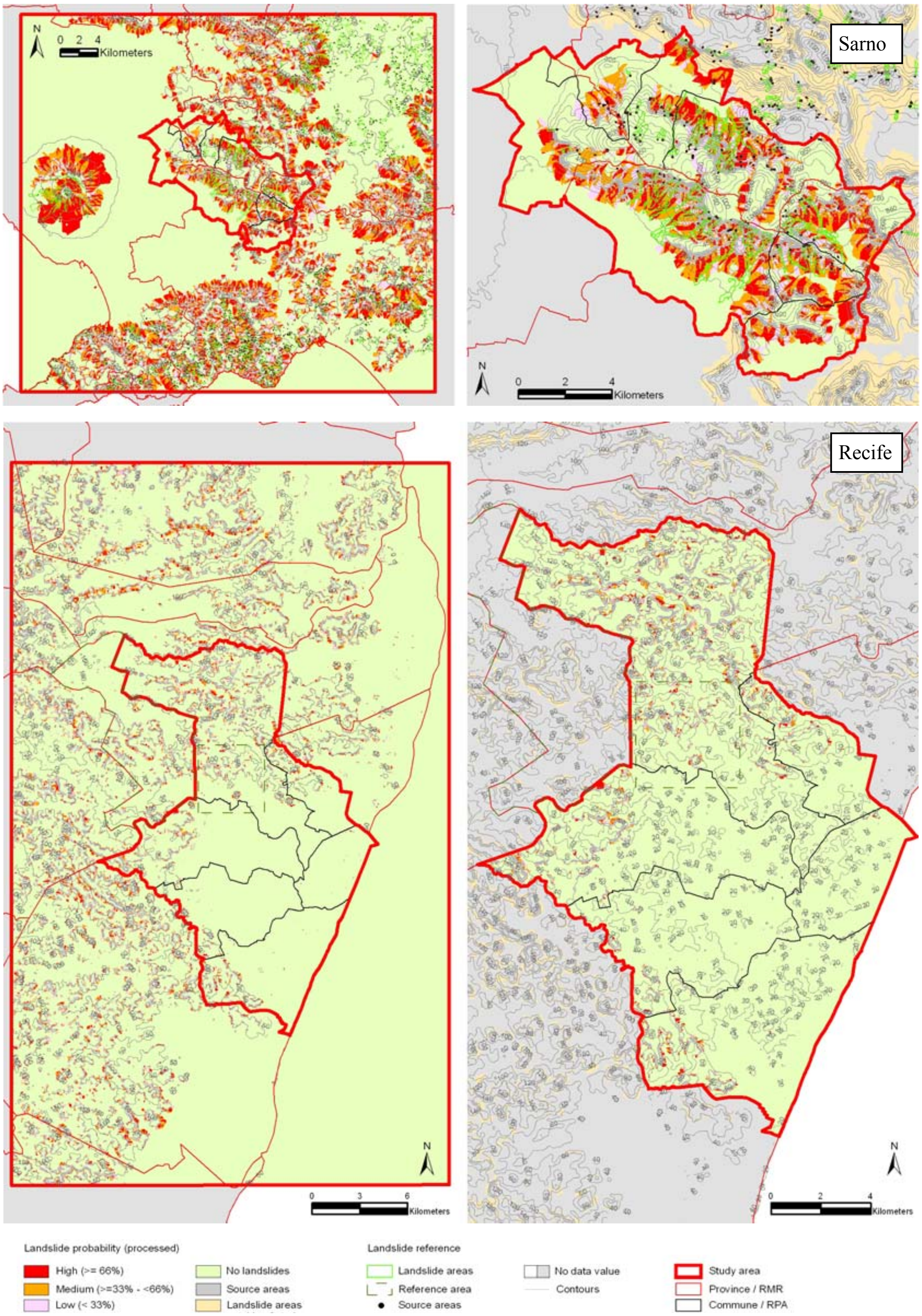

Figure 3. Landslide probability at regional and urban scale

Table 3. Landslide probability analysis at regional and urban scale.

	High Probability			Medium Probability			Low Probability			Total			
	Slide-Area [km²]	A [%]	B [%]	Slide-Area [km²]	A [%]	B [%]	Slide-Area [km²]	A [%]	B [%]	Slide-Area [km²]	Study-Area [km²]	A [%]	B [%]
SARNO – SS30.19 (suitable combination of model parameter)													
1	189.62	*10.48*	*36.86*	198.40	*10.97*	*38.57*	126.40	*6.99*	*24.57*	514.42	1809.38	*28.43*	*100*
2	19.57	*13.81*	*35.09*	23.02	*16.24*	*41.28*	13.17	*9.30*	*23.62*	55.76	141.71	*39.35*	*100*
RECIFE – RS10.13 (suitable combination of model parameter)													
1	19.51	*1.52*	*29.63*	25.31	*1.98*	*38.45*	21.01	*1.64*	*31.91*	65.83	1280.01	*5.14*	*100*
2	2.98	*1.36*	*32.43*	3.41	*1.56*	*37.14*	2.80	*1.27*	*30.44*	9.19	219.51	*4.19*	*100*

Explanations: 1…regional study area, 2…urban study area, A…percentage values refer to corresponding study area, B…percentage values refer to sum of all landslide probability areas

According to the landslide probability analysis the study area of Sarno is much more prone to landslide than the study area of Recife. Almost 40 % (where 30 % are areas with high or medium probability) of areas jeopardized by landslides contrast with only about 4 % (where 3 % are areas with high or medium probability) in the urban space of Recife. Nevertheless the wide scattered distribution of hazard zones beyond the coastal plane in Recife increases the risk to elements at risk in a certain way because it makes a prediction of danger difficult. For the example of Sarno processed landslide areas are more concentrated, predictable and somehow easier to manage.

4.2 Landslide vulnerability assessment

The main areas of the Sarno region (80.9 %) are characterized by low potential loss due to the predominance of areas covered by forest, meadows respectively used by agriculture (Table 4). Elements at risk with high and medium loss potential with surface ratios of about 19 % (high: 9.6 %, medium: 9.4 %) are mainly located on slope bottoms and ranging clockwisely from the east to the noth parallel to the boundary of the study area (Figure 4). Almost 40 % of the study area is vulnerable to landslide (Table 5). Most areas are characterized by low vulnerabilities (22.6 % of the study area) followed by areas with medium vulnerabilities (13.3 %). Only 3.4 % of the whole study area, mainly concentrated in the communes of Bracigliano, Siano, Castel San Giorgio and Sarno, is exposed to high vulnerabilities. This area distribution is in line with results already observed by Cascini et al. (2005, 2008) and Brondi and Salvatori (2003).

Within the city of Recife areas with high and low loss potential cover almost the same surface ratios each with about 41 %, the remaining areas are characterized by elements at risk with medium loss potential (12.8 %) (Table 4). While elements at risk with high potential loss are predominantly concentrated on the coastal plane, the forests, meadows and agriculture areas (low potential loss) are mainly located in the hilly northern and western urban parts which have been assigned as areas with high landslide probabilities. On this account it is expected that Recife is exposed to only a small number of vulnerable areas that has been proven by the processed vulnerability values. Therefore only 0.7 % of the urban study area is characterized by a high, 1.2 % by a medium and 2.2 % by a low vulnerability (Table 5). In total 4.2 % of the urban areas are exposed to varying vulnerability. Main threatened urban spaces indicated by relatively large numbers of high vulnerabilities are located in the northern parts RPA02, southern parts of RPA03 and western parts of RPA06 (Figure 4).

Table 4. Potential loss classes of elements at risk at urban scale.

	Elements at risk - Potential loss									
	High		Medium		Low		No data		Total	
	[km²]	*[%]*	[km²]	*[%]*	[km²]	*[%]*	[km²]	*[%]*	[km²]	*[%]*
Sarno	13.54	*9.55*	13.31	*9.39*	114.58	*80.86*	0.28	*0.20*	141.71	*100*
Recife	90.74	*41.34*	28.05	*12.78*	90.45	*41.20*	10.28	*4.68*	219.51	*100*

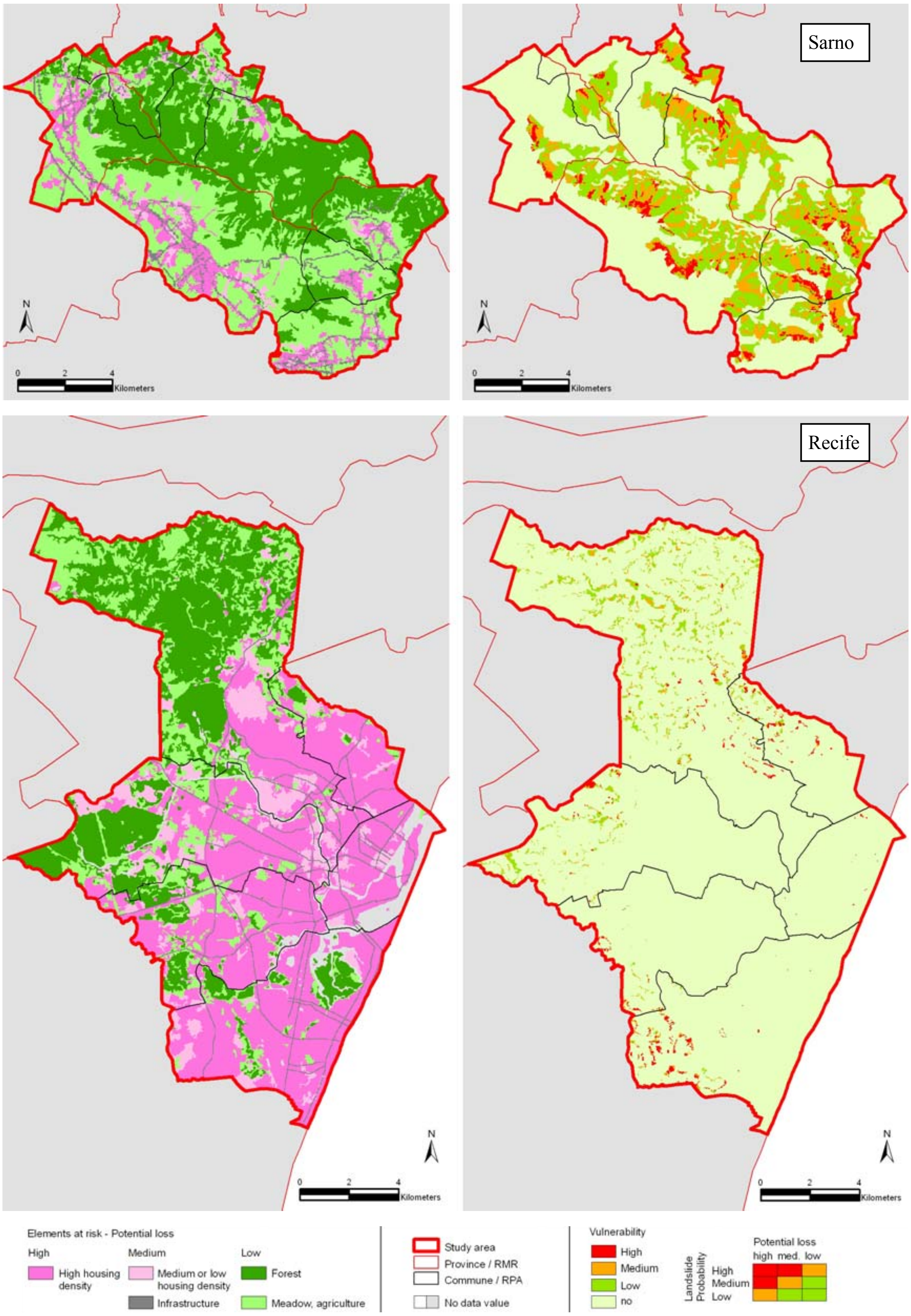

Figure 4. Elements at risk, potential loss and landslide vulnerability at urban scale.

Table 5. Vulnerability analysis at urban scale

			Sarno		Recife	
			[km²]	*[%]*	[km²]	*[%]*
Vulnerability	High	Hh	1.49	*1.05*	0.61	*0.28*
		Hm	1.59	*1.12*	0.37	*0.17*
		Mh	1.67	*1.18*	0.52	*0.24*
		Sum	**4.75**	***3.35***	**1.50**	***0.69***
	Medium	Lh	0.56	*0.40*	0.35	*0.16*
		Mm	1.89	*1.33*	0.39	*0.18*
		Hl	16.45	*11.61*	1.98	*0.90*
		Sum	**18.90**	***13.34***	**2.72**	***1.24***
	Low	Lm	0.75	*0.53*	0.29	*0.13*
		Ll	11.85	*8.36*	2.13	*0.97*
		Ml	19.42	*13.70*	2.48	*1.13*
		Sum	**32.02**	***22.59***	**4.90**	***2.23***
Total			**55.67**	***39.28***	**9.12**	***4.16***

In terms of vulnerability the investigated urban area of the Sarno case study is much more threatening (39.3 % of all areas) compared to the situation in the city of Recife (only 4.2 %). Even if more than the half of the urban area of Recife is characterized by elements at risk with high or medium potential loss (Recife: 54.1 %, Sarno: 18.9 %), its vulnerable areas reach dimensions ten times lower compared to those of the Sarno region. In general a high vulnerability to landslides can be identified in those areas where high protection goods invade into landslide sensitive areas. In Recife high and medium vulnerable areas (1.9 % of the study area) are mainly located in the hilly northeast and southwest and probably caused by informal occupation of poor levels of population. For the Sarno region it is assumed that high and medium vulnerabilities (16.7 % of the study area) are mainly provoked by urban mismanagement.

5. Conclusion

Landslides in urban areas represent a serious threat to human beings and their properties. Easy-applicable, cost-efficiency, fast-realisable approaches applicable to urban spaces are increasingly needed in order to identify areas of probabilities and vulnerabilities. For the urban areas of Sarno-region and Recife, both affected by landslides within the last two decades, a low cost, least input data approach has been presented successfully in order to quantify the current landslide probability and vulnerability. Former was determined both at a regional and an urban scale by using the MSF-model using a DEM, source areas and stopping conditions. All required parameters have been derived from ASTER-images which, hence, represent the only input data source of the model.

For the Sarno case study a detailed conformity analysis between processed result and reference information has been carried out which revealed a satisfying matching respectively underestimation and a tolerable overestimation of the documented landslide situation. Better conformities could be probably obtained with a more detailed description of landslide source areas by using more input parameter such as geology, soils or precipitation data.

The results show that landslide probability and vulnerability in Sarno are much higher and more concentrated than in Recife where low and scattered distribution pattern are dominating. Main threatened areas in the Sarno study area are the communes Castel San Giorgio, Bracigliano, Siano, and Sarno. Latter three were affected by the disastrous landslide event in May 1998. In Recife northern parts of RPA02, southern parts of RPA03 and western parts of RPA06 are the most threaten urban areas. These areas represent a serious risk for urban lives and properties when the typical local triggering factors such as strong rainfall and sensitive underground conditions (sandy-clayey sediments, Vertisols and Podzols in Recife; limestone, calcareous marls and Andosols in Sarno) are fulfilled. Additionally much higher relief energies in Sarno probably cause landslides events which are much faster and more destructive than those ones in Recife where smaller scale variations of relief are typical.

The results prove the applicability of the chosen approach to urban spaces. In further surveys the methodology could be refined (e.g. map geology from the ASTER imagery). Additional datasets as more detailed satellite imagery, improved DEMs (e.g. LiDAR), auxiliary input parameter (soil or climate data) can be applied in order to test and discuss the degree of result improvement especially under urban aspects. However, one of the advantages of the applied approach is the relatively low cost least data input.

Generally, all generated model results are an approximation of reality and have to be evaluated with field reference data and field knowledge to adapt the model input parameter (landslide source, H/L-value) if necessary. By writing a model script, combining all the steps described throughout the paper, the approach could be run semi-automatically only with an ASTER image, a small number of auxiliary data (in that example: infrastructure) and a reliable digital reference data base.

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Implications of soil threats on agricultural areas in Europe

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Summary

Agricultural areas are affected by a variety of soil degradation processes. Contamination as a result of agricultural practices influences the capability of the soil to provide nutrients for plant growth and its buffering and filtering capacity. Accumulation above saturation point will lead to the pollution of ground and surface waters. Soil organic carbon is mostly affected by land use and climate. Temperature and precipitation influence carbon mineralisation or accumulation. Recent changes in land use and climate have resulted in SOC loss, possibly limiting the soil's ability to provide nutrients for stable plant production and reducing the soil's water retention capacity. Erosion is a natural process but is intensified and accelerated by human activities. Soil loss reduces the soil's fertility and contaminates the aquatic ecosystem. The climate's erosivity and the soil's erodibility determine the extent of erosion. The soil's ability to resist erosive meteorological conditions depends on soil texture, organic matter content, vegetation cover, roughness, and the field size. The salinity of soils is influenced by climate, soil parent material, land cover and/or vegetation type, topography and soil attributes. Secondary salinisation is caused by human interventions such as inappropriate irrigation practices. High levels of salinity in soils provoke the withering of plants. Soil structure is destroyed by an excess of sodium on the exchange complex, due to the lack of oxygen the soil is not capable of sustaining neither plant growth nor animal life. Compaction can create significant damage to soil functions as infiltration rate, redistribution of water and nutrients, root development, direction of root growth as well as economical damage by decreasing crop yields and soil productivity. Cultivation becomes more difficult and can cause soils to stop being economically viable. Agricultural practices addressing soil degradation processes are manifold. In a diverse landscape like the European, there is not one cure for all. Here no-tillage, minimum tillage, soil cover, cover crops, crop residues management, crop rotation, use of mineral fertilisers, ridge tillage, agroforestry, conservation buffers, contour farming, intercropping, subsoiling and bench terraces are discussed.

1. Effect of soil degradation on agriculture

Fertile agricultural ecosystems and healthy soils form the basis of sustained economic growth. If large parts of European soil are in future no longer fit for optimal agricultural production as food and biomass, caused by contamination, loss of organic matter, erosion, salinisation, sodification or compaction, then soil quality (Tóth et al., 2007) and thus important economic activities will no longer be sustained (EEB, 2006).

1.1 Contamination

The major source of contamination in Europe is inadequate waste management which has lead to a large number of contaminated sites. Contamination also enters the soil system from atmospheric deposition and sedimentation from surface waters (Van Camp L. et al, 2004). The focus here will be on the contamination that can be influenced by changing practices, namely contamination that occurs as a result of agricultural practices where the land user modifies ecological processes in the soil through additions of nutrients, exogenous organic matter and pesticides to increase productivity or to protect the current state of the land.

Despite the enforcement of environmental protection policies and legislation, actual consumption of pesticides in the EU has not decreased between 1992 and 2003. As a result of misuses of pesticides, including overuse, the percentage of food and feed samples containing pesticide residues that exceed maximum regulatory limits, has not decreased over the last ten years. 5% of food and feed samples still contains unwanted residues of pesticides above the maximum regulatory limits. This percentage has not declined between 1996 and 2003. According to the European water suppliers' organisation, pesticide contamination of raw water is very severe in lowland rivers. A high proportion of this water is contaminated beyond the 0.1 µg/L threshold value and has to undergo pesticide removal treatment before it can be distributed as drinking water[6].

Since the mid Eighties, a progressive reduction in fertiliser consumption has been recorded and this trend has continued in the period 2000-2003. At EU 15 level, the reduction registered in the period 2000-2003 compared to the previous period 1996-1999 was 6% for nitrogen and 15% for phosphate fertilizers respectively, with the downward trend also continuing in 2004 and 2005. An estimated overall decline of 5% in manure nitrogen burden is derived from declining cattle, sheep and poultry numbers between the periods 2000-2003 and 1996-1999 but the pig numbers are stable (European Commission, 2007).

The effect of contamination on agriculture can be severe. Excess phosphorous and nitrogen from fertilisers, manure or acid depositions containing these elements influence the capability of the soil to provide nutrients for plant growth and its buffering and filtering capacity. Reduced plant growth is clearly linked to lower yields. Accumulation above saturation point will lead to nutrient leaching polluting groundwater, waterways and coastal systems causing eutrophication. (Van Camp L. et al, 2004). Several legislations address parts of the contamination problem (table 1).

Table 1. Legislations addressing contamination problem.

Legislation	Content
Water Framework Directive, 2001	A framework for assessing, monitoring and managing the ecological and chemical status of all surface and groundwater.
Nitrate Directive, 1991[7]	Legislation concerning the protection of waters against pollution caused by nitrates from agricultural sources. Focus on the detection, designation of vulnerable zones, monitoring and code of good agricultural practice.
Thematic Strategy on the sustainable use of pesticides (proposed)[8]	The purpose of this Strategy on pesticides is to address the actual *use* phase of the pesticides life-cycle (e.g. the temporary storage of pesticides at farm level, the management/calibration of application equipment, the protection of operators, the preparation of the spraying solution and the application itself) since it is a key element for determining the overall risks that pesticides entail.

On a practical level, the following practices can be applied to remediate effects of contamination (McGill, 2002; Mitchell, 2001):

- Liming: This is a technique to reduce soil acidity. Limestone (calcite), primarily calcium carbonate ($CaCO_3$), is discharged to neutralize acid waters and soils.
- Soil washing: Soil is physically removed from the contaminated parcel, followed by treatment at a plant on or off-site.
- Soil vapor extraction: Soil contaminants are extracted through a system of wells and pipes, to be created. Usually an effective technique but very expensive.
- Excavation: Soil is physically excavated, moved to a landfill and replaced with new one with heavy machinery and at a relatively high cost. If necessary, the new soil layer can be isolated from deeper non-excavated layers by applying geotextiles (synthetic blanket-like materials) that work as impermeable barrier to contaminants. Geotextiles are relatively low-cost themselves, but subject to tear.
- Microbial/fungal -remediation: Selected microbes/fungi are used to transform contaminants into a less toxic form. Microbes can be very effective in the treatment of hydrocarbons, polycyclic aromatic hydrocarbons (PAH)'s, pesticides, and polychlorinated biphenyls (PCB)'s. Cost is generally relatively low, and timeframe is short. However, increased toxicity from certain metals can appear.
- Phyto remediation: Selected plants are used to extract contaminants or to degrade them in the soil. Cost is low but timeframe can last up to several years and contaminated plants must be disposed of.

1.2 Loss of organic carbon

The organic carbon content of a soil includes all carbon containing constituents as undecomposed organic vegetation residues, soil fauna and humus. It is not homogenous but made up of heterogeneous mixtures of both simple and complex substances containing carbon (C). Soil organic carbon (SOC) can be divided into different pools based on composition and decomposition rate.

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[6] *http://ec.europa.eu/environment/ppps/strategy.htm*
[7] *http://ec.europa.eu/environment/water/water-nitrates/directiv.html*
[8] *http://ec.europa.eu/environment/ppps/strategy.htm*

Firstly, the labile pool includes easily decomposable organic materials which stay in the soil for fairly short periods, from a few days to months. They are important as food and energy sources for soil organisms, as source of plant nutrients (nitrogen and phosphorus) and for promoting stability of large soil aggregates. This pool also includes microorganisms, many of which are involved in the actual decomposition and recycling processes. Secondly, the slow pool includes the well decomposed and stabilised organic materials, often referred to as humus. They stay in the soil for many years and are important for stabilising soil structure (micro aggregates), improving water holding capacity and retaining plant nutrients, e.g. cations. Thirdly, the inert pool includes biologically very resistant organic materials which are thousands of years old. Chemically they are similar to charcoal and because of their charge properties and porous nature they can retain cations and improve soil physical properties.

For a healthy soil, all three pools of organic carbon are present and are needed to serve different functions of the ecosystem. Therefore SOC is a good indicator of soil health. Not only total SOC but the proportion of the different pools is also indicative of the "health" status of the soil. Organic carbon inputs can restore and increase soil health.

Many degraded soils lack labile and slow pools of organic carbon. To restore soil health, organic C material needs to be added that can replenish both the labile pool (to increase food source for soil microbes and provide available nutrients for plants) and slow pool (to improve soil structure and soil physical properties) (Chan Yin, 2008).

Soil organic carbon (SOC) accounts for more than 95% of the total carbon accumulated in pastures and perennial crops and nearly 100% of the total carbon accumulated in arable land. SOC enhances the resilience of agricultural ecosystems. SOC can be a source of greenhouse gases through the formation of CO2, CH4 and N2O. It can also be a sink through C sequestration under an organic form (Van Camp et al., 2004).

The amount of carbon in any soil is a function of the soil forming factors including climate, relief, organisms, parent material and time. Soil type and properties (e. g. soil texture) contribute to the explanation of the initial carbon content. Sandy soils are normally low in organic matter (OM); in contrast many soils rich in clay (e.g. Luvisols) or amorphous products (e.g. Andosols) can accumulate OM in a stable form (humus).

SOC is a dynamic characteristic that is mostly affected by land use and climate. Climate (mainly temperature and precipitation) has a critical influence on carbon mineralisation or accumulation. This influence explains the existence of a north/south climatic gradient with high carbon content in the cold northern part of Europe and in mountainous areas and lower concentrations in the hot semi-arid southern part (Mediterranean). Another influencing factor is the soil hydrology: organic rich soils (e.g. peats) are normally formed in anaerobic and wet conditions which favour accumulation of undecomposed vegetation residues. Considering the whole of Europe, 22 million ha of soils have more than 6% organic carbon; in contrast 74 % of the soils, in the southern part of Europe, have less than 2% organic C (Van Camp et al., 2004).

Under permanent agricultural practice, the SOC content is in equilibrium with climate, temperature and precipitation, due to the nature of the crop residues and OM management. Recent trends in land use and climate change resulted in SOC loss at a rate equivalent to 10% of the total fossil fuel emissions at pan-European scale (Janssens et al., 2004). A survey of Belgian croplands (210 000 soil samples taken between 1989 and 1999) indicated a mean annual SOC loss of 76 gC m^{-2} (Sleutel et al., 2003). A large-scale inventory in Austria displayed that croplands were losing 24 gCm^{-2} annually (Dersch and Boehm, 1997). Carbon losses from soils across England and Wales in 1978-2003 were about 13 million tonnes of carbon annually (Bellamy et al., 2005). Grassland is seen as a net C sink in most European countries. The overall mean C sink is 60 gCm^{-2} annually. However, the uncertainty of this estimate is high (Vleeshouwers and Verhagen, 2002).

SOC enhancement in agriculture can be achieved through reduced soil disturbance and a decrease in the SOC mineralization rate (e.g. reduced or zero tillage, set-aside land, growth of perennial crops, etc.) or through an increased input of organic materials into the soil (e.g. application of manure, crop residues, fertilization, etc.) or both (Lal, 2004). Cover crops can reduce decomposition by reducing the soil temperature. Because of the dependencies of fauna functional groups in the foodweb, the use of pesticides, herbicides or other management practices that disrupt the food webs, will affect organic matter cycling and soil organic carbon sequestration (Johnson, 1995).

A loss in OC content can limit the soil's ability to provide nutrients for sustainable plant production. Soil organic matter provides the physical environment for roots to penetrate through the soil and for excess water to drain freely from the soil. It can hold up to 20 times its weight in water, contributing to the water retention capacity of soils. A healthy level of organic carbon in a soil contributes to a good agricultural productivity.

Carbon sequestration is seen as an option to mitigate climate change and combat organic carbon loss from agricultural soils. Both organic and conventional agricultural systems can sequester carbon: organic systems four times more than conventional systems: 4000 kg CO2/ha versus 1000 kg CO2/ha. On the other hand, farming often still contributes to greenhouse gas (GHG) emissions by causing organic matter loss and through high inputs of fertilisers (EEB, 2006). Melero et al. (2006) found that organic fertilisation increased total organic carbon (TOC) content more than conventional fertilization and positively affected soil organic matter content, thus improving soil quality.

1.3 Erosion

Erosion is a natural process but is intensified and accelerated by human activities, such as deforestation for agricultural purposes, changes in hydrological conditions, overgrazing and inappropriate cultivation techniques and/or cropping practices. The impact of raindrops on soil causes the breakdown of aggregates into smaller parts, which are re-deposited between aggregates on and close to the surface, forming 'soil crusts'. Soil crusts seal the surface, limit infiltration and increase runoff. Bare soils are more vulnerable since the raindrops are not intercepted by vegetation and are subject to the full impact of the raindrops. Due to the loss of topsoil, the soil becomes less fertile and the aquatic ecosystem contaminated. In both cases off-site impacts are as damaging as on-site effects.

Table 2. On and off-site damage from water erosion

On-site damages	**Off-site damages**
• Loss of organic matter • Soil structure degradation • Soil surface compaction • Reduction of water penetration • Supply reduction at water table • Surface erosion • Nutrient removal • Increase of coarse elements • Rill and gully generation • Plant uprooting • Reduction of soil productivity	• Floods • Water pollution • Infrastructures burial • Obstruction of drainage networks • Changes in watercourses shape • Water eutrophication

"An estimated 115 million hectares or 12% of Europe's total land area are subject to water erosion, and 42 million hectares are affected by wind erosion" (European Commission, 2006). With a very slow rate of soil formation, any soil loss of more than 1 t ha^{-1} yr^{-1} can be considered as irreversible within a time span of 50-100 years (Van-Camp et al., 2004).

Wind erosion occurs when two conditions coincide: favourable meteorological conditions (e.g. high wind velocity) and favourable ground conditions (loose particles on a susceptible surface and lack of surface protection by crops or plant residues). Wind erosion itself is a natural phenomenon, but might be accelerated by human influence, especially cultivation or overgrazing. The climate's erosivity and the soil's erodibility determine the extent of wind erosion. The climate's erosivity depends on the interactions between intensity, frequency and duration of wind velocity and wind direction, amount and distribution of precipitation, humidity, radiation, snow depth and evaporation (Funk and Reuter, 2006). The soil's ability to resist erosive meteorological conditions mainly depends on soil texture and organic matter content, which influences the water holding capacity and the ability of the soil to produce aggregates or crusts (Chepil, 1955). Other influencing parameters are vegetation cover, roughness, field size (Funk and Reuter, 2006).

Wind erosion decreases soil fertility by removing the finest and nutrient rich top parts of the soil (van Lynden, 1995). It impacts on air quality, which in turn influences the global climate due to changes in atmospheric radiation balance by aerosols (Shao, 2000; EEA, 2003; Goossens, 2003). It damages crops up to the point where arable land has to be taken out of production (Schroeder and Kort, 1989; Jönsson, 1992; Veen et al., 1997) and pollutes adjacent areas when the wind speed reduces and soil particles are deposited. Wind erosion removes soil organic matter, resulting in a decrease of the aggregate stability of the soil, which impacts negatively on the denitrification potential (removal of nitrogen by microorganisms) for biomass productivity, and even decreases SOM storage in terms of climate change mitigation.

When the humus or topsoil is eroded away by water or wind, the fertility of a soil diminishes and lower yields are returned. Soil erosion can lead to the blocking of dams through sediments deposition downstream. This results in a loss of water holding capacity downstream causing floods.

Erosion mitigation and prevention can be obtained through conservation tillage, using an increased percentage of soil cover as well as through the use of reduced or no tillage. Drainage systems that avoid excess water on the soil, limiting and diverting water run-off (and therefore soil erosion) or avoiding water stagnation on flat surfaces (that cause anaerobic soil conditions) can help prevent water erosion. Superficial drains can be further structured with grass cover, small dams, rocks or irregular routes to avoid water erosion. Field patterns and size, accompanied by hedgerows, buffers, drainages, also influence soil erosion.

Measures to prevent wind erosion are manifold, they intend to increase the resistance of the soil surface and reduce the surface wind speed. Creating a rougher surface stimulates results in a slowdown of the wind velocity (Stull, 1988). A long term and expensive measure is the arrangements of hedges on field level perpendicular to the prevailing wind direction which can give a protection of up to 25 times their height (Nägeli, 1943). By applying no tillage or reduced tillage, larger soil clods are created; more area is sheltered due to the shielding of the leeward side of clods or furrows against wind action and particle impact (Potter and Zobeck, 1988). Soil cover is the best measure against wind erosion. Soil cover greater than 10 % reduces wind erosion rapidly, and complete prevention occurs when the soil cover is over 40 % (Morgan and Finney, 1987; Funk, 1995; Sterk, 2000). Other measures appropriate at field level are water table management, using drainage, irrigation systems and crop management according to good farming practices.

1.4 Salinisation, Sodification

Salinisation is the process that leads to an excessive increase of water-soluble salts in the soil. Primary salinisation involves salt accumulation through natural processes. The main natural factors influencing the salinity of soils are climate, soil parent material, land cover and/or vegetation type, topography and soil attributes.

Sodification is the process by which the exchangeable sodium (Na) content of the soil is increased. Na^+ accumulates in the solid and/or liquid phases of the soil as crystallised $NaHCO_3$ or Na_2CO_3 salts (salt "effloresces"), as ions in the highly alkaline soil solution (alkalisation), or as exchangeable ions in the soil absorption complex (ESP).

Secondary salinisation is caused by human interventions such as inappropriate irrigation practices, e.g. with salt-rich irrigation water and/or insufficient drainage.

The most influential human-induced practices leading to salinisation are the irrigation of waters rich in salts; rising water table due to activities as filtration from unlined canals and reservoirs; uneven distribution of irrigation water; poor irrigation practice, improper drainage; use of fertilisers and amendments, especially in situations of intensive agriculture with low permeability and limited possibilities of leaching; use of salt-rich wastewaters for irrigation; salt-rich wastewater disposal on soils and contamination of soils with salt-rich waters and industrial by-products.

High levels of salinity in soils provoke the withering of plants both due to the increase of osmotic pressure and the toxic effects of salts. In most cases, when alkalinity processes take place, the high pH level does not permit for plant life. Excess of sodium, on the exchange complex, results in the destruction of the soil structure that, due to the lack of oxygen, is not capable to sustain neither plant growth nor animal life. Alkaline soils are easily eroded by water and wind. Salinisation increases the impermeability of deep soil layers, eliminating the possibility to use the land for cultivation (Van Camp L. et al, 2004).

Soil salinisation affects an estimated 1 to 3 million hectares in the enlarged EU, mainly in the Mediterranean countries. It is regarded as a major cause of desertification and therefore is a serious form of soil degradation. With recent increases in temperature and decreases in precipitation, characteristic of the climate in recent years, the problem of salinisation in Europe is getting worse (Van-Camp et al., 2004).

Prevention management practices could include quality control of irrigation water (water coming from salt-rich areas) and stabilisation of the groundwater table.

1.5 Compaction

Soil compaction is the rearrangement of soil aggregates and/or particles due to a reduction or even disappearance of voids and pores between aggregates and particles, causing the soil to become denser.

Soil properties largely determine the natural susceptibility of a soil to compaction. Man induced soil compaction (or secondary compaction) is exclusively caused by soil use and soil management. Natural susceptibility to compaction depends on soil texture; it ranges from sandy (least susceptible) – loamy sandy – sand loamy – loamy –clay loamy – loam clayey – clayey soils – to clays (most susceptible to natural compaction) (Woods et al., 1944). This is due to the weight of the soil, the stability of soil structure and the soil's water regime. Clay soils are the heaviest, and the upper parts of the soil compacts the lower parts. They have a high water holding capacity and it is difficult to maintain the proper soil moisture (around field capacity) for their cultivation. Usually they are too wet and thus fragile for the use of heavy machinery.

Soil compaction can induce or accelerate other soil degradation processes – e.g. erosion or landslides. Compaction reduces the infiltration rate, which increases runoff in sloping areas. In flat areas compaction can cause water logging, resulting in the destruction of aggregates causing crust formation. In sloping areas the presence of a compacted layer with low permeability can cause the upper part of the soil, once saturated with water and thus heavier, to slide, subsequently resulting in mass movements and landslides.

Compaction can create significant damage to soil functions (mainly infiltration rate, redistribution of water and nutrients), root development, direction of root growth as well as economical damage by decreasing crop yields and soil productivity (DeJong *et.al.*, 2001). Compaction reduces rooting depth and plant growth resulting in a lower harvest. Cultivation becomes more difficult demanding more energy. This can cause soils to stop being economically viable.

Soils with high and very high natural susceptibility to compaction need to be cultivated appropriately with regards to their soil moisture conditions (field capacity) and crop rotation patterns that require few external entries on the field. Root crops are generally regarded unsuitable for such types of soil. Reduction of pressure through use of appropriate tires and permanent traffic lanes can also minimise compaction.

2. Agricultural practices addressing soil conservation

In this section agricultural practices will be described along with their effect of the soil threats.

Cultivation methods usually pursue several objectives: soil preparation for seedbed; weed control; plant protection against insects and fungi; plant nutrition; water supply. Soil preparation techniques can be classified according to the depth of their actions and to their mechanical effects on soil strata: inversion or non-inversion, mix or fragmentation.

The combination of these three mechanical effects defines the level of soil disturbance of a certain technique. The table below summarises the main techniques applied in agriculture for soil preparation and their impact on soil strata as well as their respective level of disturbance.

Table 3. Classification of soil preparation techniques

Techniques	Depth of soil preparation (cm)	Inversion	Mix	Fragmentation	Level of disturbance
Subsoil tillage (or sub-soiling)	40 - 80	No	No	Yes	Low
Soil decompaction (or decompression)	15 - 40	No	No	Yes	Low -medium
Ploughing	15 - 40	Yes	Yes	Yes	Very high
Deep soil preparation	15 - 30	No	Yes	Yes	High
Reduced tillage	5 - 15	No	Yes	Yes	Medium - Low
No tillage (or direct sowing)	5	No	No	No	None

Soil disturbance has an influence on the physical properties of soil. When the soil strata are not inversed, *soil porosity* increases due to earthworm activity. These macro pores have a vertical profile, are very resistant to pressure loading and allow a higher rainfall infiltration rate (Tebrügge and Düring, 1999). *Improved aggregate stability* means a better resistance to the impact of raindrops and surface sealing.

Non inversion tillage prevents accelerated mineralization of organic matter by soil micro-organisms. By maintaining soil aggregates, the contact surface between organic matter and soil organisms is reduced and so is the availability of nitrogen (Guerif, 1994). By minimising the mineralisation, **reduced or no tillage** *increases the organic carbon rate* in the top soil layer, therefore also *increasing aggregate stability*. 75% of the organic carbon from the crop can be found in the uppermost 5 cm (Labreuche J. *et al*., 2007). Between the surface and the deeper layers, a decreasing gradient of organic carbon occurs. This gradient depends on the duration of the use of reduced tillage or no tillage and on the type of practice: the less the soil is disturbed, the more the gradient is apparent.

The effect of **tillage practices** (conventional or not) on soil *compaction* depends mainly on the soil humidity during tractor passages (the number of passages), the initial condition of the soil and crop rotation. Soil permeability, soil porosity and topsoil compaction influence run-off and erosion.

The effects of adopting **simplified tillage techniques** on pesticide and fertiliser leaching are highly variable (Tebrügge and Düring, 1999; Holland, 2004; Labreuche *et al.*, 2007; Köller *et al.*., 2005). Especially under **no tillage**, the *increase of soil macropores* (and the better vertical orientation and connectivity of these biopores) facilitates a more rapid movement of water and of the pesticides within. *Macropores* created by earthworms on the contrary are covered with organic matter and retain agrochemicals, preventing the vertical transport. Adsorption and breakdown of pesticides are higher under no tillage or reduced tillage due to **soil cover.** The increase in *soil organic matter* and *biological activity* in the soil's surface has a positive effect on *surface water quality* by reducing the lateral losses of pesticides by *reducing water runoff* compared to conventional tillage (Labreuche *et al.*, 2007; Köller *et al.*, 2005, Tebrügge and Düring, 1999).

Soil biodiversity is influenced by **reduced tillage, no-tillage methods,** *soil fertility, soil structure, soil organic matter* (amount and distribution), **time of cultivation, use of pesticides, crop residues management, crop rotation, use of mineral fertiliser**, etc..(Holland J.M., 2004). **Reduced tillage or no-tillage** methods increase the amount and change the composition and distribution of *biological activity* compared to conventional tillage (Javůrek M. *et al.*., 2006, Holland J.M. 2004). The impact of the **simplified tillage** system on *earthworm populations* feeding on organic matter is very positive (Tebrügge, in Labreuche, 2007). **Organic farming** tends to increase *species abundance* and *richness* across a wide-range of taxa (Hole et al., 2005). *Microbial biomass*, which acts itself as a reservoir of plant nutrients such as N and P was found significantly higher under **organic** than conventional management (Truu et al., 2008).

Crop rotation is a planned system of growing different kinds of crops, in the same sequence, on the same land and over three or more years. Crop rotations help to better use natural resources and therefore improve or maintain *soil fertility, reduce erosion* and the *build-up of pests, spread the workload, reduce risks of weather damage and the reliance on agricultural chemicals*, and generally increase net profits.

Cover crops (including catch crops which are cover crops planted to prevent nutrient leaching) and **soil cover (**permanent or temporary living soil cover, crop residues left on field, mulching) diminish the raindrop impact and significantly reduce *soil loss and run-off*. This leads to a *better soil structure* due to better *aggregate stability* and *soil organic matter content*. Crop residues left by no tillage increase soil humidity and decrease soil temperature. Moreover, the increase of *soil organic matter* and *biological activity* can increase the amount of pesticides fixed (adsorption) or their breakdown in the top soil layer, and reduce pesticide transfers by run-off (Labreuche *et al.*, 2007; Köller *et al.*, 2005).

Ridge tillage is the system of cultivating crops on pre-formed ridges alternated with furrows protected by crop residues. **Ridge tillage** has potential to increase nitrogen use efficiency by plants (Schlinker et al., 2007; Henriksen et al., 2006) because, given the conformation of ridges, precipitation slips away on the ridge's side, thereby *reducing leaching* of inorganic N, usually applied on the top. More over, the ridge tillage has also an important effect on *soil moisture (increase), erosion (reduction)* radiation absorption and *decomposition of organic matter*. *Soil microbial biomass* has been found to be two to three times higher in fields with ridges, which may affect N-turnover (Henriksen et al., 2006).

Many authors suggest that **agroforestry** has positive effects on *soil fertility* maintenance, *erosion* control (Torquebiau, 2000), *water holding capacity, carbon sequestration* and *nitrate leaching* in intensively managed agricultural landscapes, through the potential of the tree roots to *recover nitrogen* from below the crop rooting zone (Reisner et al., 2007). However, effects of agroforestry on the environment are highly variable, depending on biophysical conditions, management, choice of crops and tree species (Palma et al., 2007). Torquebiau (2000) also reports that **agroforestry** contributes to a vast series of additional services (e.g. *microclimate improvement, biodiversity enhancement, watershed protection*) and multiple products (e.g. food, wood, fodder, mulch, fibres, medicines), but could present some constraints as well, mainly concerning potential competition between trees and crops for water, light and nutrients.

Conservation buffers, or **buffer zones**, are areas or strips of land maintained in permanent vegetation. Many different types are used (Filter strips, field border, windbreakers, grassed waterways, riparian buffers, etc) and they aim at preventing *soil erosion from wind and water, reducing leaching of nutrients and drift of pesticides* from arable fields into water bodies, roads or other areas, *enhancing biodiversity* and diversifying productions. Most studies report that **buffer strips** next to arable lands can significantly *reduce the volume of suspended solids, nitrates and phosphates transported by agricultural runoff to water bodies* but their effectiveness remains linked to the mechanisms by which these pollutants are transported (Muscutt et al., 1993).

Contour farming involves that field activities such as ploughing, furrowing and planting are carried out along contours. Contour farming aims to create detention storage within the soil surface horizon and *slow down the rate of runoff*, thus giving the *water* the time to *infiltrate* into the soil. The effectiveness of contour farming for water and soil conservation depends on the design of the systems, but also on soil, climate, slope aspect and land use of the individual fields.

Intercropping is defined as the growth of two or more crops in proximity in the same field during a growing season. The simultaneous growing of different crops (that have sufficiently different niches) on the same field could bring about a more efficient exploitation of the field's resources (Hauggaard-Nielsen et al., 2001). In intercropping, the competition or complementarity between plants enhances the overall stability of the system, including a significant resilience against pests, diseases and weeds.

Subsoiling is the operation to form cracks in soils with deep hardpans for *improving infiltration rates and root penetration*. The implement reaches down below the ploughing depth to break up a compacted layer that is beyond the reach of normal tillage equipment.

Field studies by Pagliai et al. (2004) showed that the use of ripper subsoiling and minimum tillage, on a loam soil, *increased macro and microporosity* and its homogeneity in the profile compared to conventional tillage systems. Moreover, Piovanelli et al. (2004) concluded that this tillage combination on a loamy-sandy soil can be regarded as excellent conservation system as it permits a better *sequestration of carbon*, reducing CO2 emission into the atmosphere. The effects of subsoiling are influenced by many parameters as combination of practices, type of crop and soil, climate and local conditions (precipitation just before and after subsoiling operation), period of soil cultivation and soil water status in growing season (Henriksen et al., 2007).

Bench terraces consist of a series of level or nearly level platforms built along the contour lines at suitable intervals and generally sustained by stone walls. Terraces are created as a measure to stop or reduce the degrading effect of soil erosion by *intercepting surface runoff*, facilitating its infiltration, evaporation or channelling it at a controlled velocity *to avoid soil erosion* (Dorren and Rey, 2004). The efficacy of terraces against surface runoff and soil erosion depends on local conditions and dimensions, form and stability of the terraces, assuming proper maintenance. Efficiency greatly benefits from the application of additional conservation practices such as contour ploughing, strip cropping and permanent soil cover.

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Multi-Scale European Soil Information System (MEUSIS): novel ways to derive soil indicators through Upscaling.

P. Panagos, M. Van Liedekerke

Summary

During the last years the need for a coherent approach to soil protection has come on the political agenda in Europe and was therefore introduced as one of the thematic strategies to be developed within the Community's 6th Environment Action Programme. At present, the most complete source of soil information at European scale is the European Soil Database (ESDB) at scale 1:1,000,000. This database has been developed jointly with partners in participating countries resulting in the only harmonized coverage of digital soil information for Europe and it is available in the European Soil Portal (http://eusoils.jrc.ec.europa.eu/).

*The **INSPIRE** (Infrastructure for Spatial Information in Europe) directive has embarked on a common framework for spatial data in the European Union. One of the INSPIRE ideas is to conduct reporting and analysis of environmental information on the basis of **a** harmonized and hierarchical system of grids with a common point of origin and a standardized location and size of grid cells. In other words, the INSPIRE grid system proposal could lay the basis for a **Multi-scale European Soil Information System (MEUSIS)**, a system whereby soil data produced at a certain scale can easily be integrated or compared with soil data produced at another scale, provided that the rules for representation of the data are equal at all scales. "Upscaling" is the term used to describe the process of reducing a set of values in an area down to a single value representing such area as a whole in order to allow comparison and integration to other data sets.*

1. Introduction

The Multi-Scale Soil Information System(MEUSIS) can be a suitable framework for building a nested system of soil data that could facilitate interoperability through a common coordinate reference system, a unique grid coding database, a set of detailed and standardized metadata and an open exchangeable format. In the context of INSPIRE Directive, MEUSIS may be implemented as a system facilitating the update of existing soil information and accelerating the harmonization of various soil information systems. Moreover, MEUSIS should provide data for the assessment of soil conditions at different levels of details and provide a structure so that coherent and complementary data, available at a nested set of geographical scales, can fit together.

In order to achieve this ambitious objective, Joint Research Centre (JRC) proposes a common standard with specific rules for the collection of harmonized soil information. The common standards include some specifications as the common grid system, co-ordinate transformation, Labeling, Spatial Resolutions, Data Exchange Format and Metadata information. The data providers most likely will need to process their original soil data (held in traditional vector-based soil databases) in order to fit the proposed grid system.

In environmental data like the soil one, it is common to generalize accurate data obtained at the field to smaller scales using either the pedotransfer rules or knowledge of experts or even some statistical solutions which combine single values of spatially distributed data. The most common statistical process for generalization is averaging the values within the study area.

Upscaling – Aggregation

Spatial data and scales in environmental research cover a wide range. Since the research community wants to develop a common understanding of the impact of management changes on larger scales, there is often a difficulty because of the lack of effectively transfer data or information among scales. The way that those data exchanges or model results across space scales can be effectively accomplished is critical to help everyone understanding the methodology on how they can move information across scales.

In cartography, "Upscaling" or "aggregation" are the terms used to describe the process of reducing a set of components or values in an area down to a single value representing such an area as a whole. Aggregation implies simplification, the degree of variation in the considered area is reduced and thus there is an obvious loss of information. The aggregation method should be carefully chosen in order to display the maximum spatial variation of the data in a comprehensive manner according to the scale or representation. There are many methods for Upscaling and the optimal one would be the one which ensures that the new value of the whole area is the most adequate according to the objectives of the Upscaling. Soil variables can be represented either as quantitative variables (numeric parameters) or as qualitative (Classes).

For the quantitative variables like the Organic Carbon Content, the most appropriate method for Upscaling is called "zoning" in which the Mean or the Mode of the single values of an area is used to define a larger area. In soil domain, the use of Mean or Mode statistical indicator is recommended for Upscaling, instead in other cases (where critical limits or values is an important factor assess environmental data: floods, natural hazards) the percentiles, minimum or maximum can be more appropriate statistical indicators.

For the qualitative variables such as Dominant World Reference Base (WRB) Soil Types, the problem is related to the semantic component of the soil types. The recommended way to perform an aggregation is firstly to reduce semantically the soil units (variability of the soil types) and then to explore the most frequent (Dominant) value. Obviously this is more complicated than the quantitative aggregation and specific upscaling rules must be defined beforehand.

Often the assumption in Upscaling is that the aggregation of values to a larger scale is linear in nature or some statistical methods will make approximations to natural variability. This process results in loss of detail, and more critically, may produce erroneous relationships when the process is non-linear or when the natural variability is very high. Our objective is to minimize those errors coming from the non-linear relationships and to find out the cases where there is such a high natural variability.

2. Slovakia Multi-scale Soil Information System

The present paper uses the results of the case study implemented in Slovakia in 2006 and the resulting Slovakia Soil Database. The objective of the present paper is to present the feasibility and applicability of the proposed methodology which is described in the next section (Upscaling Methodology).

The European Commission, Joint Research Centre has collaborated with the **Soil Science and Conservation Research Institute** (Slovakia) in order to test the MEUSIS principles in some Slovakian Regions. Because of their specificity in terms of soil geography and main threats to soils, the MEUSIS Pilot Areas in Slovakia has contributed in the analysis of the feasibility of such approaches.

In MEUSIS, all geographical information (Attributes and Geometry components) are represented by the grid of regular spatial elements (pixels). The representation of various spatial resolution detail is following the INSPIRE recommendations and three **spatial resolution levels** of geographical information have been defined for MEUSIS:

- 10 km^2 resolution grid (10km x 10km)
- 5 km^2 resolution grid (5km x 5km)
- 1 km^2 resolution grid (1km x 1km)

The above mentioned grids follow the Inspire Recommendations and are proposed for the development of Spatial Data Infrastructures in various resolutions. In order to investigate the comparability of datasets in various resolutions, data have been collected in 3 different spatial resolutions (Large scale: 10 km^2 , Medium scale: 5km^2 , Small scale: 1km^2). Having built the Slovakia Soil Database in this way, a system of spatially nested hierarchical grids has been developed.

In The Figure 1, the 3 spatial resolutions are illustrated representing the 3 levels of data collecting in the Slovakian Country:

- Level 1 (L1): The soil cover is described according to 10km x 10km pixels (regular cells to be interpreted as elementary mapping units with associated attributes); the pilot area is approximately 49,026 square km (The whole Slovakia surface). The resulting shape file 10km.shp (10km x 10km) contains 509 cells covering the all surface of Slovakia.
- Level 2 (L2): The soil cover is described according to 5km x 5km pixels; the pilot area is approximately 10,489 square km (Districts of Trnava and Nitra) and is included in the Level 1 pilot area. The resulting shape file 5km.shp (5km x 5 km) contains 475 cells covering the whole surface of the districts of Trnava and Nitra.

- Level 3 (L3): The soil cover is described according to 1km x 1km pixels; the pilot area is approximately 4,409 square km (District of Trnava). This pilot area is included in the Level 2 pilot area. The resulting shape file 1km.shp contains 4,409 cells covering the whole surface of the districts of Trnava.

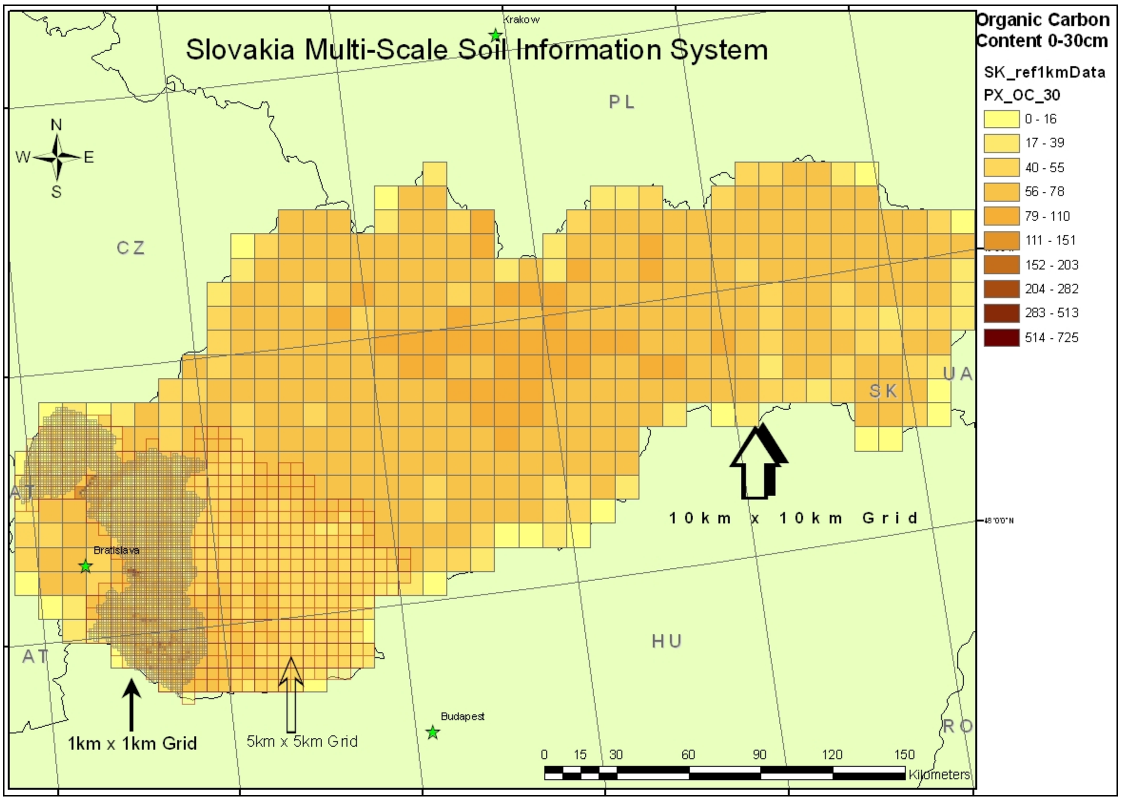

Figure 1. Slovakia MEUSIS: Multi-Scale Soil Information System.

Data Sources

In order to fill in the database in the 3 resolutions, the data provider (Soil Science and Conservation Research Institute in Slovakia) has used the following information sources:

- Digital Soil Map of Slovakia at scale 1:400,000 (PM400)
- Digital georeferenced database of soil ecological units at scale 1:5,000 (PEU-DB)
- Set of digital regional maps of geo-factors of landscapes – soils at scale 1:50,000 (GFZP)
- Georeferenced Database of Agricultural Soils in Slovakia (GDPPS)

Since the data are requested in 3 different scale, the data provider has used information from the above mentioned data input sources in order to fill in the data tables. In figure 2, the allocation scheme per Level (L1, L2, L3) is presented.

Some basic allocation rules have been defined in order to complete the data input according to the most accurate pixel scale. So, each pixel should be filled with the most detailed information source, but in case of lack of information the next more general information source provides the data value. For example, in case of L2 (5km^2 grid cell, if the GFZP database doesn't cover 100% the pixel, then the data are taken from the more general information source which is the PM400.

The detailed description of the detailed GIS operations is not among the objectives of this paper but the reader may find more information in the "Multiscale European Soil Information System – Pilot Project in Slovakia (Skalsky, R. et al., 2006)".

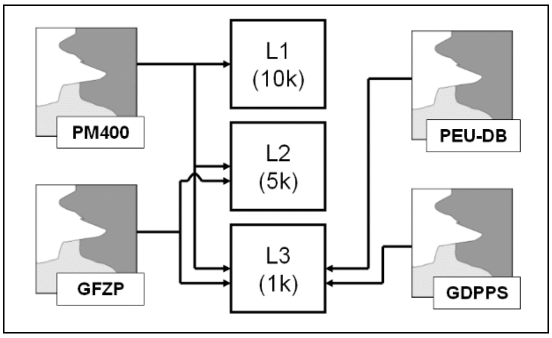

Figure 2. Input Data allocation Scheme

3. Upscaling Methodology

In order to describe the Upscaling Methodology we will use as an input the Organic Carbon Content 0-30cm field (quantitative variable) of the Slovakia Soil Database. We are using the ArcMap ESRI Software in order to perform some specific functions in this methodology. We will take as an example the upscaling of 5km^2 to the 10km^2.

- Rasterize the Shape file. The shape file 5km.shp is rasterised and as a result a corresponding Raster file is produced (*Output: c_5km_polygon*[9]).
- Aggregate the raster data (*Input: c_5km_polygon, Output: Ag_5kmTo10km*). Use the Aggregate function in order to generate a reduced resolution version (10km) of a raster. Some useful tips in order to proceed in this operation:
  - Aggregation Technique establishes how the value of each output cell will be determined. In our case, the Mean is used as the most appropriate technique. It is interesting to execute the same operation with the Median.
  - Factor by which to multiply the cell size of the input raster in order to obtain the desired resolution. Since the Desired resolution is 10 and the input resolution is 5, then the Cell Factor is 2 as it should satisfy the following equation:

    Output Cell Size = Input Cell Size x Cell Factor
  - The Extent of the output area should be the same as the Extent of Original Data that we will compare later. In this case the Extent is the same as the original 10km.shp. Attention should be paid to this fact as each output cell should fit with the original cell of the same size in order to allow further process and comparison.
  - The Result of the aggregation is a float number. A transformation of the Float number to Integer is requested and the “Int” operation has to be executed (*Input: Ag_5kmTo10km, Output: Int_Ag5kmTo10*).
  - The Output Raster Data (Cell size of 10km) have to be converted to vector data in order to allow us to make the comparison (and other statistical operations) with the original 10km.shp data (*Input: Int_Ag5kmTo10, Output: Polyg_int_ag_1to10*).

- Execute the UNION operation in order to join the 2 datasets (*Input: Polyg_int_ag_1to10, 10km.shp - Output: Union_1to10Km*).
- Export the cells that have values coming from both sources (in order to allow comparison) to a data table (*Output: 5to10.dbf*)

4. Upscaling Results from 5km^2 towards the 10km^2

Using the above described Upscaling Methodology, the 475 cells of 5 km^2 yield 134 cells of 10 km^2. As a rule described in the INSPIRE principles, 4 cells of 5 km^2 are requested in order to produce a cell of 10 km^2. In this case we have some additional cells as due to extend of the upscaled data, the input 5 km^2 were less than 4 (cases near the borders, upper left or down right corners).

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[9] *The use of filenames is indicative and may be used as a basis for common terminology*

In Figure 3, the reader may observe the upscaled data as a result of the above described process. The upscaled data of 10 km^2 scale are located in the centre of the cell and have a blue font over the source data of 5 km^2 scale in light grey font. The use of transparency allows the reader to justify the correspondence between source and resulting data. In the majority of the cases the upscaled data is the Mean of 4 input data values of 5 km^2 (except if there are less than 4 cells which perform the Mean operation for the upscale).

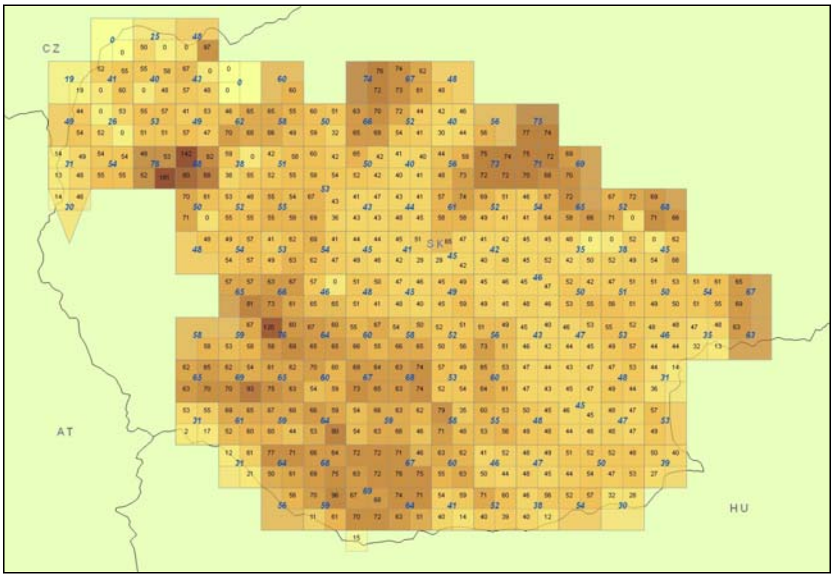

Figure 3. Upscaled results from 5km towards 10km

After executing the Union operation, 131 cells of 10 km^2 fit both in the original and in the upscaled data. In detail the original 10 km^2 data and the upscaled data may be viewed and compared in the Figure 4.

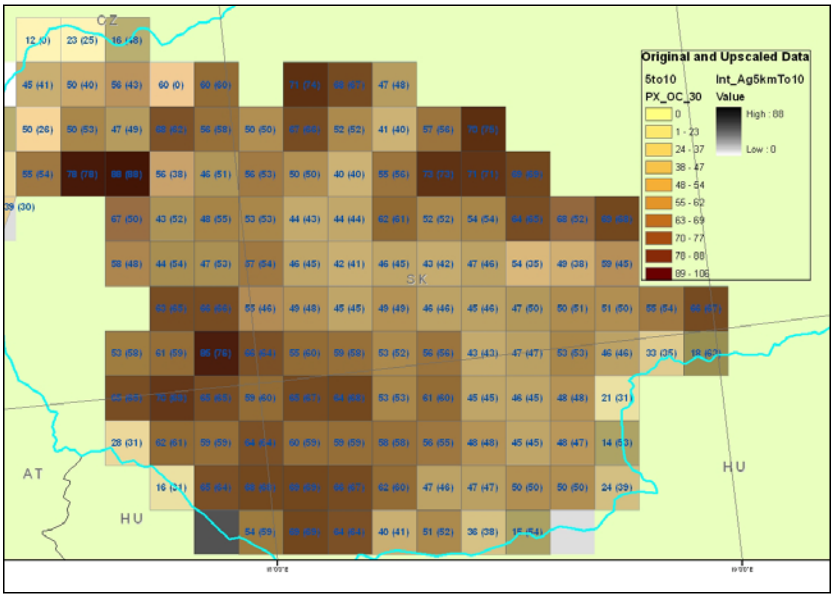

Figure 4. Comparison of Original and Upscaled Data (5km^2 towards 10km^2)

In each of the 131 cells, you may view both the original value and the upscaled one in parenthesis. Also the graduated brown colour ramp is used to give a visual representation of the Organic Carbon distribution (according to the original data) whenever the graduated grey background is used for the representation of the upscaled data.

In the Figure 5, the reader may compare the correspondence between the original 10 km^2 values (X Axis) and the upscaled data (Y Axis). It is obvious that there is a linear relationship between the 2 datasets as there is a major concentration of data values near a line. The scatter diagram helps to identify the close values between the original and the upscaled datasets with some exceptions of extreme differences.

Comparing those 2 resolutions (10 km^2 Original data, 10 km^2 upscaled data from 5 km^2), it is obvious that there is a strong relationship between them. A more detailed Statistical Analysis will follow in order to take into account all the statistical descriptors and make clearer the conclusions of this comparison. The Statistical Analysis is used to access the adequacy of the methodology employed for the aggregation and to find how this can be improved.

In this case the proposed approach is not just a theoretical one, but can reach significant results using real data. In the past, there were many theoretical references to an ideal Multi-Scale Soil Information System which can be a system of hierarchical grids. It is the first time we proceed with testing of MEUSIS Methodology using both GIS operations and Statistical Analysis (Descriptors, Scatter Diagram).

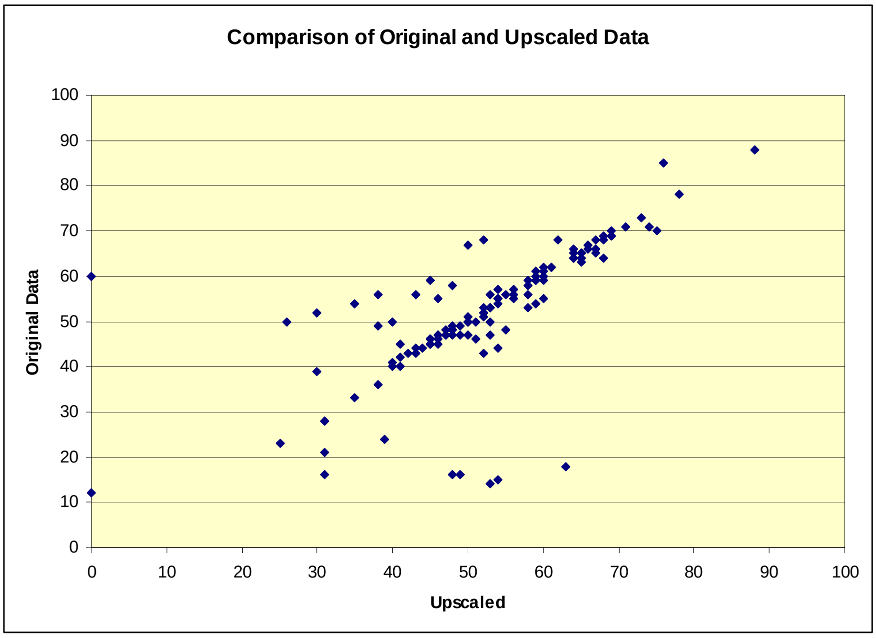

Figure 5. Scatter Diagram of the Original and Upscaled data

In Table 1, some descriptive statistics are shown in order to allow the better understanding of the 2 data series. Commenting the results of the statistical analysis, we notice the following remarks:

- The Mean of both data are very close. This result may be explained in the following ways:
  - The data sources for both the 10 km^2 values and the 5 km^2 values are the same.
  - Or the use of above mentioned Upscaling method is producing satisfactory results as the upscaled data are close to the original data.
- The Median which represents the value for which half of the data are below this value is very close for both data distributions. Also both Median Values are close to the Mean Values.
- The Mode value which represents the most frequent value in data series is also quite close for both data distributions.
- Using collectively the 3 above mentioned measures of tendency (Mean, Median, and Mode), we conclude that there are no extreme values that can affect the data distribution. Having in mind all those statistical

indicators we can certify that there is a small-medium variability regarding the Organic Carbon Content in our case.

- The Standard Deviation measures the dispersion of the data around the Mean. Since we have noticed that the distribution of the upscaled data tends to be normal, we may use the Standard Normal Distribution in order to estimate with 95% probability the Range in which a random upscaled data value X will be included according to the Equation 1:

$$P(-1.96\sigma \leq \mathrm{X} - \mu \leq 1.96\sigma) = 0.95 \qquad (1)$$

Where μ, σ the Mean and the Standard Deviation. Resolving the Equation 1 for the Upscaled data distribution, we determine the range of possible values in Upscaling between 26.10 and 78.10 with a probability of 95%.

- The Coefficient of Kurtosis determines the Kurtosis of the data distributions. In both cases the Coefficient is less than 3 which means that the both distributions are platykurtic (flat shape) for both data arrays. In the Upscaled data the Coefficient of Kurtosis is close to 3 which means that the distribution tends to be normal (Mesokurtic).
- The Coefficient of Skewness in both cases is Negative which means that the left tail of the distribution curve is longer (flatter).
- The Coefficient of Correlation can determine how strong the relationship between the 2 data distributions. The value 0.693 determines a quite strong relationship between the 2 data distributions (It is also obvious from the Scatter Diagram in Figure 5).

Table 1. Descriptive Statistics of the Upscaling Process from 5km^2 towards 10km^2

Statistic	**Upscaled Data**	**Original Data**
Sum	6825	6819
Count (Cells)	131	131
Mean	52.10	52.05
Maximum	88	88
Minimum	0	12
Median	52.00	53.00
Standard Deviation	13.27	14.29
Mode	53.00	50.00
Coefficient of Kurtosis	2.60	1.05
Coefficient of Skewness	-0.76	-0.72
Coefficient of Correlation	0.693	

In the above mentioned statistical analysis, we could continue by performing a Regression Analysis but it is not the main objective of this paper to reach such a conclusion.

5. Upscaling Results from 1km^2 towards the 10km^2

Using the Upscaling Methodology, we will demonstrate the updating from 1km^2 towards the 10km^2. In order to update one cell of 10km^2 it is requested 100 cells of 1km^2. In the Slovakia Soil Database, there are available 4,409 cells which are upscaled to 61 cells of 10km^2 (There are cases where less than 100 cells may be upscaled to 1 cell of the upper scales). The Methodology applied is the one which has been described above and the results are shown in Figure 6.

After executing the Union operation (Joining 2 datasets), the 61 common cells (Original 10km^2 and Upscaled ones) are compared in figure 6 where the reader may view both data values. The blue outline reflects the borders of the area where data in scale of 1 km^2 have been provided.

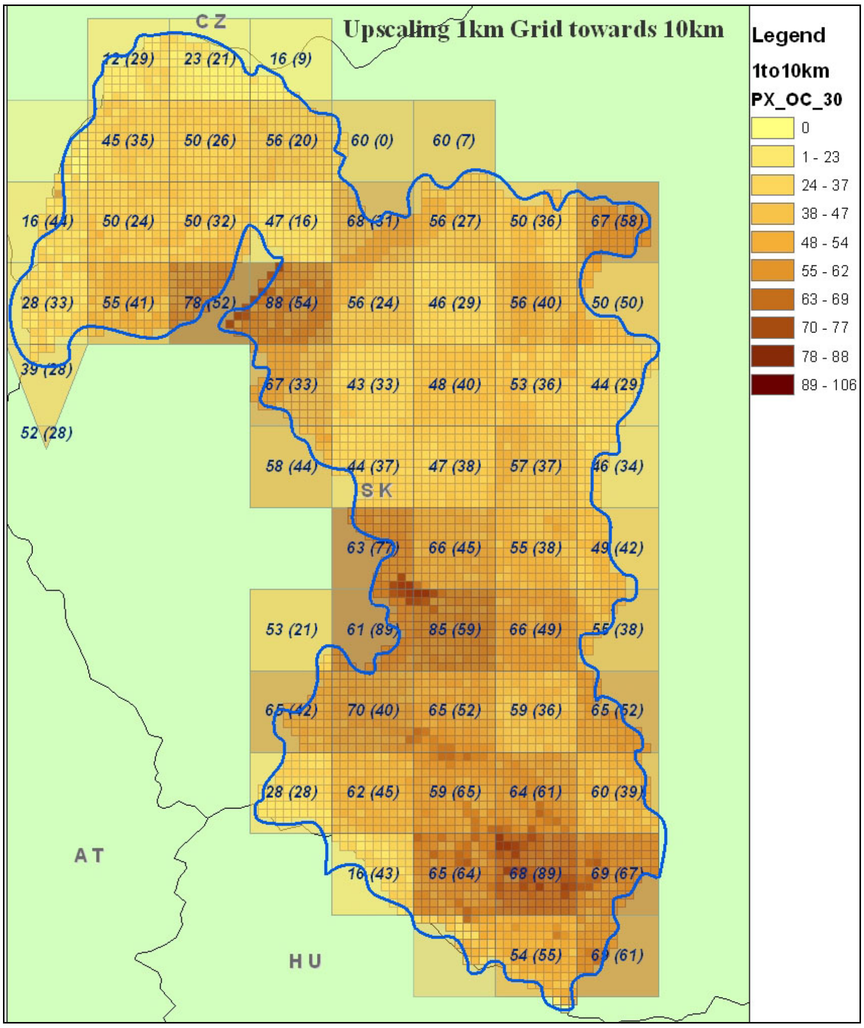

Figure 6. Comparison of Original and Upscaled Data (1km^2 towards 10km^2)

Proceeding with a Statistical Analysis, some statistical descriptors are shown in the following table 2 on which we notice the following remarks:

- There is a slightly significant difference between the 2 Means (40.20 and 53.64) which may be explained as the upscaled data tend to have lower values than the original ones.
- Regarding the Median and the Mode, there is even a larger difference between the 2 datasets due to the trend of the upscaled data to have lower values.
- The Coefficient of Skewness is positive in the upscaled data (Right tails is longer) and negative for the original data (Left tail is longer).
- Regarding the upscaled data, we can estimate with probability 95% that a random upscaled value may be in the range between 6.02 and 74.38 according to the Equation 1.
- The Coefficient of Correlation has a value of 0.449 which express a medium relationship (neither strong, nor weak) between the 2 data distributions.
- The Differences noticed in the Upscaling process from 1km^2 towards 10km^2 doesn't mean that the upscaling methodology can be rejected as it is more possible to have a better result in a certain area of 10km^2 by upscaling 100 data values than having one random value in this large area. It can be considered that the average of 100 samples is more accurate than the random sample in a 10km^2 area.

Table 2. Descriptive Statistics of the Upscaling Process from $1km^2$ towards $10km^2$

Statistic	Upscaled Data	Original Data
Sum	2452	3272
Count (Cells)	61	61
Mean	40.20	53.64
Maximum	89.00	88.00
Minimum	0.00	12.00
Median	38.00	56.00
Standard Deviation	17.44	15.70
Mode	28.00	50.00
Coefficient of Kurtosis	1.03	1.10
Coefficient of Skewness	0.58	-0.81
Coefficient of Correlation	0.449	

6. Upscaling Results from $1km^2$ towards the $5km^2$

In this case, the hierarchical grid system requests 25 cells of $1km^2$ in order to update 1 cell of $5km^2$. Executing the same Upscaling process, we will reach some conclusions which are in between the 2 above mentioned Upscaling exercises. In order to update one cell of 5 km^2 it is requested 25 cells of $1km^2$. In the Slovakia Soil Database, there are available 4,409 cells which are upscaled to 208 cells of $5km^2$.

7. Conclusions

- MEUSIS methodology can be proposed in many cases where the data owner does not allow the distribution/publication of detailed data but is willing to distribute upscaled data (in higher scale). MEUSIS can be considered a valuable methodology which contributes to the degradation (without losing the real values) of very detailed data and may allow the scientific community to access valuable information without having any copyright problems.
- MEUSIS is an effective shared methodology whenever a large soil survey may take place with many data providers and a project co-ordinator who has to implement the final consolidated data. Each partner receives a number of cell ID's for which he had to report the soil data according to the instructions and exchange format described in an Data Exchange Protocol.
- Upscaling has a serious drawback in case that the source data in the lower scale have a high spatial variability. This has been shown in the case Upscaling exercise from $1km^2$ towards the $10km^2$. In a latter stage, we should investigate the use of other statistical descriptors (Median or Mode) in the Upscaling process instead of the Mean in order to view which one can fit better.
- Since National or Regional Soil Information are organized in a complex Databases (Variety in Data Availability, representation differences, various scales, different attribute precision, Inconsistency), the MEUSIS Methodology may be proposed as a pipeline in order to upscale soil information from this detailed level towards European level. MEUSIS may overcome all the above mentioned limitations and problems.
- MEUSIS Methodology can be used also for other Environmental Indicators which have small or medium spatial variability.

Acknowledgements

The data for the application of MEUSIS methodology were provided by the Soil Science and Conservation Institute in Slovakia.

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Inspire Directive 2007/2/EC of the European Communities: http://inspire.jrc.it/directive/l_10820070425en00010014.pdf

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Abstract

This report is summarizing the results of recent research activities on the fields of soil degradation, soil quality and soil information systems performed in the Joint Research Center, in collaboration with partner institutions. An overview is given about the main soil threats (erosion, compaction, salinisation, landslides, decline of soil organic matter, biodiversity decline and contamination) and a soil quality concept with relevance to the Thematic Strategy for Soil Protection.

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