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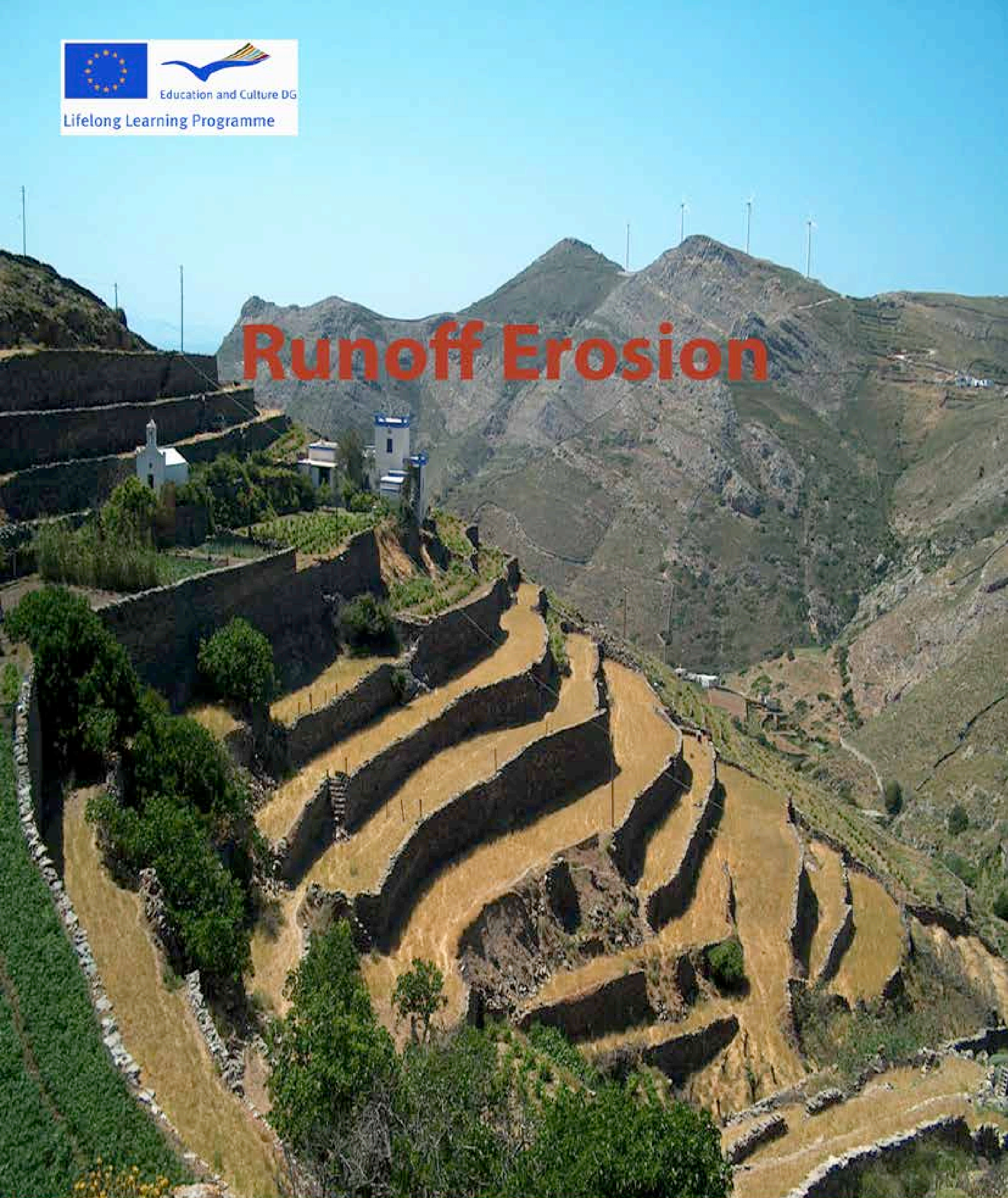
Runoff Erosion



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ACKNOWLEDGEMENTS

The editors would like to thank Evina Liosatou and Eleana Karkani for their support during the preparation of this handbook.

Table of Contents

PART I – THEORY OF RUNOFF EROSION

CHAPTER 1

RUNOFF EROSION – THE MECHANISMS

By

Niki Evelpidou

1. WATER - EROSION

Erosion is classified in different categories depending on the factor that initiates it. Among the most important factors, which cause the detachment of the soil, are the wind, the water, gravity and the land uses.

The superficial runoff due to rainfall and irrigation constitute the most crucial factors of erosion processes. The types of water-erosion are classified into subcategories based on their location inside the watershed.

The study of water-erosion usually takes place within the boundaries of the drainage basin. The most important types (Fig. 1.1) of water-erosion are taken place in rills, interills, ephemeral streams, permanent engraved streams and the river beds. Rill and interill erosion appear at slopes, where runoff has the form of terrestrial flow. The ephemeral stream erosion is found on the small depressions of the relief, where runoff occurs with small concentration flow, due to the topography. The depth of ephemeral erosion is generally small, resulting to the easy alluvium of the channels. The permanent, engraved, streams appear in the same topographical locations as the ephemerals; however, they cannot be easily filled with erosion material, since they are quite deep. The main types of rill erosion are the headward and oblique recession of the wall sides (embankment erosion); each can produce significant quantities of sediment.

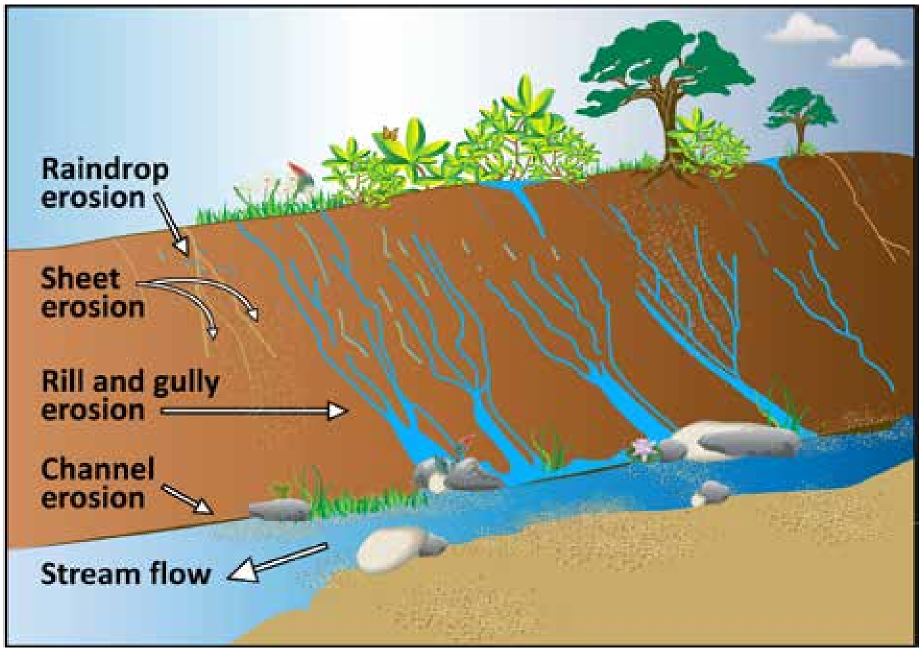

Fig. 1.1: Types of water induced erosion.

1.1 Geographical distribution

In order to better understand erosion caused by surface runoff, it should be examined within the boundaries of a drainage basin. Drainage basins vary in size, from an acre up to some thousands of acres. The smallest and simplest drainage basin is composed of areas with surface flow, which are located near an individual branch of the drainage network (Fig. 1.2). A large drainage basin consists of a series of smaller basins, which include smaller class branches of the drainage network (Fig. 1.2).

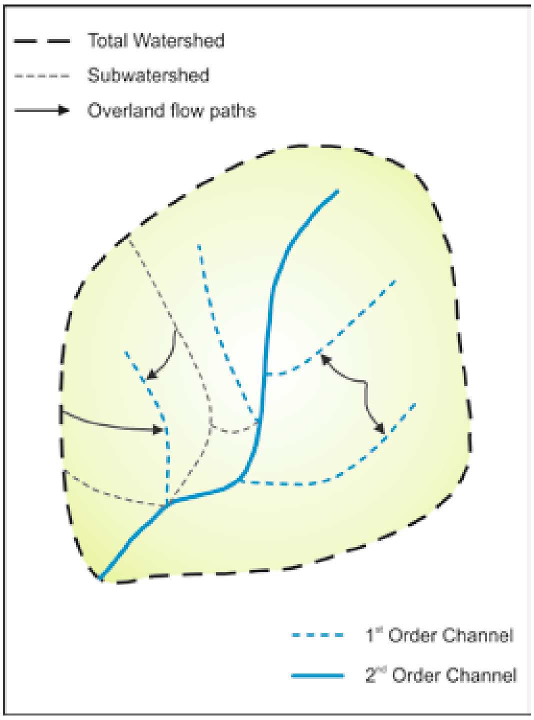

Fig. 1.2: Schematic representation of the hierarchy of the hydrographic basins.

Two important elements in erosion studies are the flow and direction of the water. Water can be regarded as flowing in two types of channels: i) the free channel type ii) the pipe type (Fig. 1.3). An open channel has a free water surface exposed in atmospheric pressure, and corresponds, in nature, to a branch of a drainage network, whereas the pipe type occurs under hydraulic pressure (Chow, 1959). The first type of flow is encountered in rills, streams and basins. Flow under hydraulic pressure occurs in saturated soil, through the porosity of the consisting material.

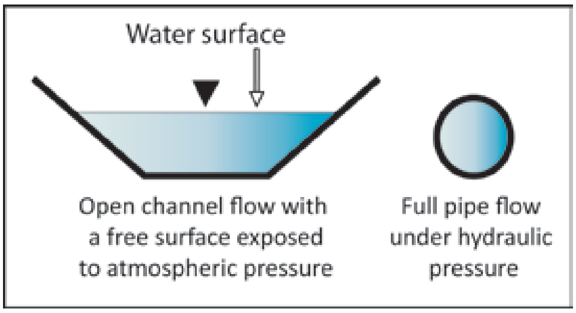

Fig. 1.3: Open channel flow and pipe flow.

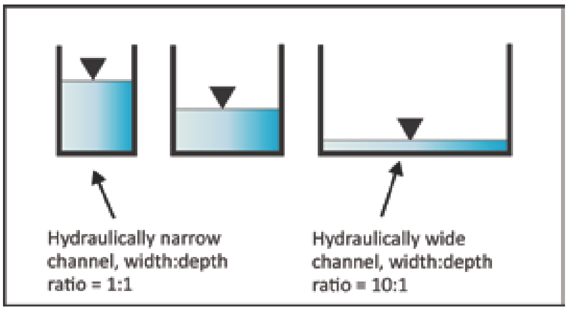

Fig. 1.4: Width – depth ratios of the channels.

The shape of the cross section of

the drainage network branches' varies significantly (Fig. 1.4) from deep engraved channels, such as permanent streams, to wide, shallow channels, such as ephemeral streams.

A simple measurement of a channel's shape is the ratio of width - depth. For a narrow channel this ratio is small (e.g. 1:1), whereas for a wide channel it exceeds 10:1; for the unformed surface flow, this analogy tends to infinity. The branches of a drainage network develop as a natural part of the landscape's evolution. The drainage area is delimited from the areas between a branch and its watershed. In the drainage area, flow does not occur through a specific branch, but water flows freely until it is concentrated in the branch of the drainage network (Fig. 1.5).

Fig. 1.5: Areas of surface runoff and concentrated flow. Photo by C. Centeri.

1.2 Types of superficial erosion

1.2.1 Rill and interill erosion

The first phase of erosion is superficial erosion, occurring with relatively small rates and constitutes a uniform displacement of soil. Rill erosion begins, as erosion becomes more intense. Deep engraved channels are developed as rill erosion evolves into stream erosion.

According to empirical estimations, erosion in a stream initiates when the superficial erosion approaches approximately 6-7 tn per acre per year. Because the erosion of the areas between the streams is uniform, it is practically considered as superficial erosion. This erosion rate can reach 20 tn per acre per year, at the mountain crests, which separated by streams (Meyer & Harmon, 1985). In reality, sedimentation may appear in streams, when the inclination along them is relatively small, as well as, when the interrill erosion is intense.

Rills and interrills constitute the areas of terrestrial flow. According to Emmett (1970), Foster (1971) and Meyer et al. (1975) water accumulates in very small branches of the hydrographic network, which are called rills, even when erosion does not occur within them. In case erosion due to runoff occurs, it will be observed in these areas. Furthermore, erosion can also be observed in the interrill areas. Surface runoff in these areas, also known as Hortonian overland flow, develops when the intensity of the rainfall exceeds the infiltration capacity of the soil (Horton, 1933). The total erosion due to surface runoff is defined by rill and interrill erosion.

The location and shape of a rill are controlled by the microtopography of the slope, and not the macrotopography of the broader area. The microtopography of the slopes is often determined by the crop characteristics. The initial form of small streams, which are developed through the cultivation of soil, may evolve into a network of streams and small basins. Streams are very small channels that can be eliminated by the processes of an area with natural vegetation and, subsequently tend to form in new places. The surface of a mechanically graduated slope may appear uniform; however, intensive erosion due to surface runoff often leads to the formation of streams (Fig. 1.6).

Fig. 1.6: Rill and interill erosion.

Surface runoff appears also in natural, non-cultivated slopes. The characteristics of flow are often defined by the stems and roots of the plants and by the local depositions, which can create an anomalous surface so that runoff is gathered in small channels between the obstacles. Vegetation can be so dense that runoff is almost continuous along the slope and, finally, obtain the characteristics of a constant surface runoff. Generally, the runoff occurring in streams is of great importance due to the significant erosion it causes.

Fig. 1.7: Splash erosion: the impact of the raindrops. Photo by Centeri.

Surface runoff and the transferred sediment move sideways in the interills towards their adjacent areas. Interills often present low length inclinations, less than one meter. In these areas, the extraction of material is usually caused by the impact of the raindrops (Fig. 1.7), whilst in the rills, it results from surface runoff.

Erosion and deposition areas can be recognized at the slopes, with sedimentation taking place mostly in the concave side of the slope (Fig. 1.8). Erosion, which is defined as soil loss, mainly appears at the upstream of the slope, while deposition is observed at the downstream. The quantity of sediment accumulating upstream is less than the soil loss due to erosion downstream. The total length of the uniform and convex slopes forms an area prone to erosion. Furthermore, slopes with small curvature do not present significant deposition. Other factors can also result in deposition, apart from the shape of the slope, such as the dense vegetation or other obstacles that dramatically decelerate erosion.

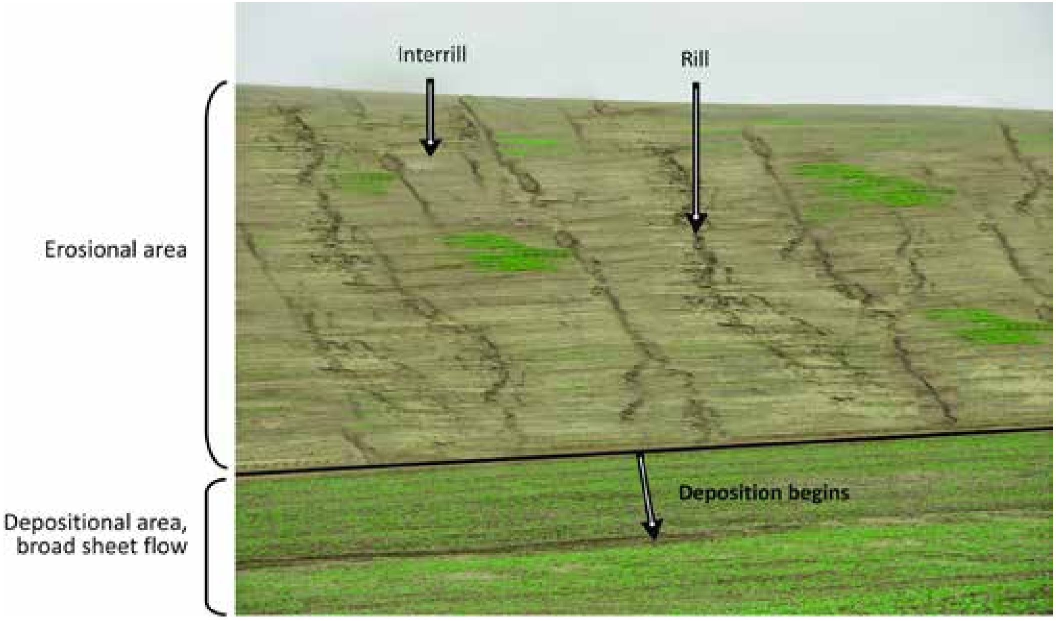

Fig. 1.8: Erosional and depositional areas on a hill slope.

1.2.2 Areas of concetrated flow

An area of concentrated flow is distinguished from a stream, by its morphological location and the role of macrotopography in the definition of the position of the channel of the concentrated flow. The location of the channel of concentrated flow is exclusively determined by the macrotopography of the landscape. In case the channel is aggregated, the channel of concentrated flow reshapes to its previous location. Despite the fact that the location of the concentrated flow is controlled by macrotopography, artificial channels can be constructed at the slopes in order to reduce slope's inclination and rill erosion.

1.2.3 Ephemeral stream erosion

Ephemeral streams are usually observed in areas, where the natural balance is disturbed by agricultural activities. The quantity of sediment produced, in this case, is equal to the amount of sediment deriving from rill and interrill erosion in a specific area (Foster, 1985; Thomas et al., 1986).

In agricultural areas, ephemeral streams develop due to common cultivation practices and, they usually evolve due to material transport from the terrestrial areas to adjacent ephemeral streams. Ephemeral streams are forced by surface macrotopography to regain their previous location after the aggregation, due to agricultural activities. Their name -ephemeral -is also owed to this periodic sediment re-enrichment and reshape from erosion. Over the years, these channels are gradually merged, while, in some cases, they remain engraved with vertical lateral walls (Fig. 1.9).

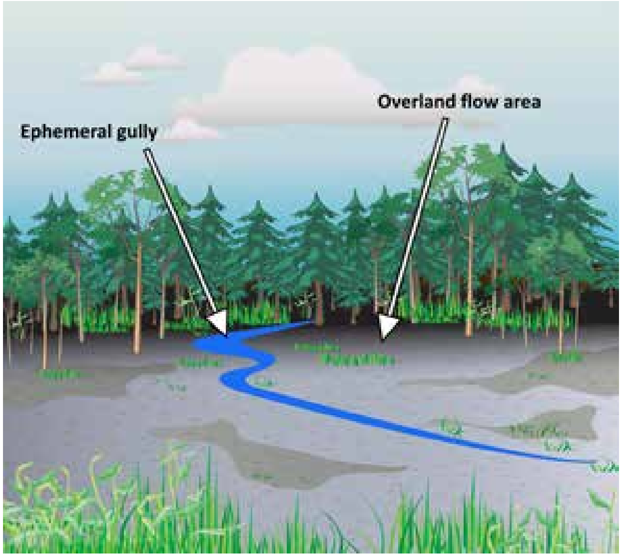

Fig. 1.9: Blending of ephemeral gully areas with overland flow areas.

Cultivations result to the development of a superficial zone in the area of the ephemeral stream which is more vulnerable to erosion than the non-cultivated soil. Flow leads to deep erosion of the ephemeral stream and gradually in lateral erosion, resulting to a wide, shallow channel of great width-depth ratio.

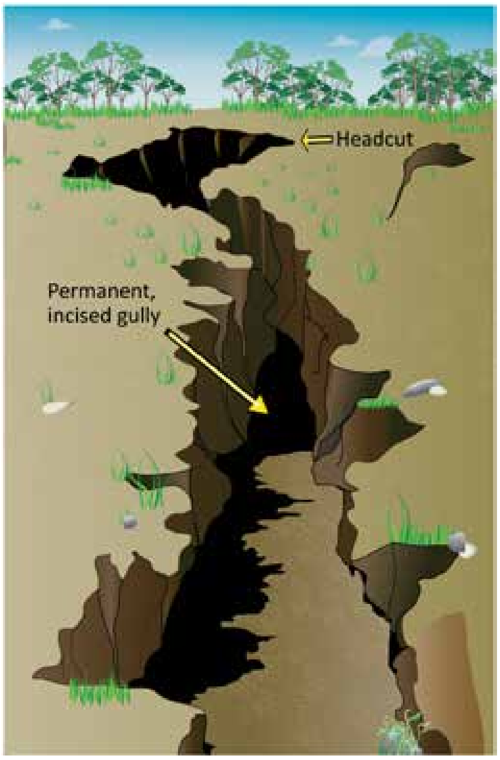

Fig. 1.10: Permanent incised gully.

1.2.4 Permanent, incised gully erosion

Permanent, incised gullies are observed in both natural and disturbed soils (Fig. 1.10). In agricultural soils, these gullies are defined as very deep channels (Foster, 1985). Changes in land use are commonly responsible for the development of gullies. Permanent, incised gullies are usually juvenile and develop in a short period of time. Depending on the flow occurring within them, these permanent gullies are, often, engraved, broad and deep channels (Heede, 1975; Piest et al., 1975). Headward erosion can also lead to the formation of a permanent engraved gully.

When the lateral walls consist of resistant rocks, they are low and vertical, as opposed to when they consist of loose material. Erosion in permanent, incised gullies is periodical and varies in an annual base (Piest et al., 1975). For example, the vegetation that may develop on the materials ending up in the gully, can protect them from detachment. The weight of the material that has accumulated by the undercutting of the lateral walls stabilizes their base, preventing further collapse into the gully. However, an incident of high runoff can cause intense flow and, as a result, move the detached soil material and destabilize the lateral walls from the gully.

1.2.5 River-bed erosion

The river-beds of the drainage network constitute an integral part of the relief. Although their development is independent of human interventions, activities in the upstream and within the channels can greatly affect river-bed erosion. The characteristics of the river-bed, including the degree and shape of the spiral route, facilitate the flow and regulate the sediment load that deposits within (Schumm, 1977). Consequently, changes in land use also transform the river-bed, in addition to altering runoff and sediment deposition. River-beds are constantly changing; however the alteration of steady beds can be imperceptibly slow (Trimble, 1977).

The abrupt changes in land uses, such as the intensive agriculture which significantly increases the upstream runoff, can destabilize the bed and lead to erosion phenomena. This erosion can widen the bed and lead to the regression of the knick-points in very high rates, producing great sediments loads. The most active erosion locations in the river-bed are commonly found in the outer part of the meanders' inflexions, where the channel's embankment retreats several meters during heavy storms. River-bed erosion can be controlled by reducing runoff rates through flow adjustments, constructing expanded intersections, installing control structures in the channel, expanding protective embankments and deflecting the flow away from the bed walls (Shields et al., 1995).

Stream channels in non-disturbed forest soils are typically permanent, however logging and road constructions can drastically disturb them (Dunne, 1998; Elliot, 1999; Grace, 2000; Warrington et al., 1980). Such disturbances can increase surface runoff as well as the sediment load that results in the river-bed. Irrigation ditches that are built along the roads often transfer the increased runoff and the sediment, originating from roadway erosion and other disturbed areas, directly to the river-bed.

The instability of the river-bed can be additionally caused by dramatic changes in the sediment load, which approaches the streams from the upstream areas. The agricultural growth in the 19^{o} c. and the beginning of the 20 c. led extensive areas to erosion (Trimble, 1977). The high levels of rill and interrill erosion, in recently deforestated and cultivated soils, produced and distributed a great amount of sediment in the stream valleys, which were eventually aggregated.

1.2.6 Erosion processes in the watershed

The amount of sediment distributed in the basin's exit, which results from erosion processes occurring in its interior, is called product of erosion in a drainage basin. Sediment production is a measure of the mean erosion occurring within a watershed. It is the total amount of sediment produced from any kind of source: terrestrial flow, ephemeral streams, permanent engraved streams and the areas of channels-streams.

1.2.7 Erosion due to the snow melting

Erosion from terrestrial flow can result from the melting of snow however it depends on a particular combination of conditions. Runoff due to the melting of snow may appear after frost and can be significant (McCool et al., 1995; 1997; Van Klaveren & McCool, 1998). Rainfall in a soil after frost is a common phenomenon. The soil is particularly vulnerable to erosion and even the slightest amounts of surface runoff, from low, but steady intensity rainfall, can cause high rates of rill erosion. Ground sensitivity in this type of erosion is most evident in cultivated soils rather than in the natural non-disturbed ones.

1.2.8 Erosion via porosity

The water usually flows within the soil, right beneath its superficial layer. The soil can consist of macro-pores, other small interstices and channels, which have been created by plant roots, insects, animals, etc. The formation of 'pipes' through these pores, can cause heavy flow and result in erosion phenomena. The initial diameter of the pipes is, usually, of the order of some mm but can be broaden to even one meter.

When a pipe is close to the ground surface, its upper part might collapse and form an open groove or a permanent, engraved flow (Zachar, 1982).

Similar flow can occur, when the superficial layer has a coarse texture and is located above a dense soil zone that will deter the downward flow of the water. The water flows towards downstream, along the crest of the dense soil zone, and escapes through the lower part of the slope, making this area more vulnerable to erosion compared to the neighboring ones (Huang & Laften, 1996). The increasing humidity of the soil in the lower topographical sections and the concentration of the ground's humidity in the cavities is a common phenomenon. The increasing ground humidity, and sometimes the water filtration, makes soil vulnerable to erosion and results to the acceleration of rill and ephemeral stream erosion (Huang & Laften, 1996).

1.2.9 Erosion due to irrigation

Irrigational water reaches the soil through elevated sprayers, surface runoff, or underground vaporizers. When the methods of use of irrigation water are applied appropriately with vaporizers or underground systems, erosion is small. However, when the superficial method for the management of irrigation water is applied, the induced erosion is significant. Through this type of irrigation, the surface water is inserted from upstream of the grooves and begins to flow towards downstream. This irrigational system can cause significant erosion, especially to the upper parts of the grooves, where the rate can be up to 4 times larger than the mean erosion rate of the particular area (Trout, 1996).

2. MAIN FACTORS THAT CONTROL SOIL EROSION

The environmental conditions that define the type and rate of erosion occurring in a specific area depend, mainly, on five factors:

- climate,
- soil,
- morphology,
- land uses
- weathered cap and,
- the soil management

Apart from water erosion, the aforementioned factors control aeolian erosion as well, but in a different way. In this book, the focus is on erosion that results from surface runoff.

Water-erosion is the dominant geomorphic process for the larger part of the earth’s surface. Sedimentation and water-erosion include the processes of detachment, crawling, transportation and deposition of the soil particles. The main forces that characterize these processes are:

- the shear stresses, which are activated by the impact of raindrops
- surface runoff

Water-erosion is a result of these forces and relates to the soil resistance against detachment processes. With the onset of water-erosion, the soil particles behave as sediment. Sediment deposition is the amount of eroded material, which is distributed in a particular area and may originate either from the eroded sections of a slope (soil loss), or from a drainage basin (sediment production). The capacity of sediment transfer through surface runoff is also related to the shear stresses applied on the soil, the size and the density of the sediment particles. Sedimentation takes place when the available quantity of sediment is larger than the transfer capacity, and results to sediment accumulation on the ground surface. Consequently, erosion rates, sediment production and depositional rates are determined by the environmental conditions.

The erosive action of precipitation, which depends on climatic conditions, controls the forces that are applied to the soil and trigger water-erosion. The vulnerability of soil to erosion is determined by the soil properties. The forces applied to soil are affected by the morphology, the vegetation, the formation of the surface, the presence of biogenic material and land uses, and define the degree of soil vulnerability to erosion.

Each of the aforementioned factors are analyzed and correlated in the following paragraphs.

2.1 Climate

The climate refers to the weather and the variability of weather conditions through time. Erosion can be affected both directly and indirectly by the climate. Precipitation is the most important climatic variable that affects runoff-erosion. The erosive ability of rainfall can be described through different variables. These variables include the amount of rain, kinetics, impetus and intensity of every rainfall event (Foster et al., 1982; Wischmeier, 1959). Simple observations show that the intensity and amount of rainfall are the two main precipitation variables, which define the erosive ability of a storm. The intensity of rainfall refers to erosion per unit of precipitation, which leads to an overall estimation of the erosive ability of a storm, when multiplied with the amount of rainfall.

This simple relationship is obvious for erosion, due to the impact of the raindrops, and can be also applied to erosion due to surface runoff. Surface runoff erosion is related to the rate and amount of runoff. The quantity of runoff is associated with the quantity of rainfall minus the amount of infiltration, and the maximum rate of runoff is related to the maximum intensity of precipitation minus the infiltration rate.

Although the erosive ability of rainfall is sufficiently described by the quantity and intensity of rainfall, a further examination regarding the size of the raindrops is considered necessary. The forces that are exerted in the soil by the impact of the raindrops depend on their size. A very small drop, which impacts the ground with low velocity, exerts a very small force to it and causes a small degree of erosion, despite the quantity and intensity of rainfall. This result is described by a variable, the kinetic force of a raindrop impacting the ground, which equals half of the drop’s mass times the velocity of the impact squared.

The size and impact velocity of a raindrop constitute two closely linked terms/concepts. A small raindrop has a small impact velocity and consequently small impact energy. A storm is composed of thousands of raindrops. As a result, the total kinetic energy of the storm is the sum of the kinetic energies of the individual drops.

$$\sum_{1}^{n} E = \frac{mv^2}{2}$$

The final velocity of the raindrops when they impact the ground is directly linked to the drop’s diameter.

Because erosion constitutes a complex phenomenon, it requires experimental research in order to define many of the relationships describing it, such as the erosive ability of precipitation. Experimental research shows that a reliable index of the erosive ability of precipitation is the result of the total kinetic energy of a storm multiplied to the maximum intensity of 30 minutes (Wischmeier, 1959). The mean total kinetic energy of all storms, occurring during one year, is an index that can be calculated by precipitation data (Wischmeier & Smith, 1978). Annual precipitation height and, hence its annual erosive ability, vary considerably from year to year. Moreover, the erosive capability varies during the same year.

Another significant parameter is the temporary erosive ability, since it interacts with changes in vegetation and soil conditions and therefore, has an important impact on the annual erosion. The greatest erosion appears when the period of maximum erosive ability corresponds to the period that soil is most exposed to surface runoff and raindrop impact. The erosive ability is significantly decreased, when these two periods do not correspond to the period of maximum sensitivity.

The form of precipitation, which is primarily defined by temperature, significantly affects the erosive ability of rain. Snowfall is not erosive whereas high intensity rainfall can be highly erosive. Furthermore, soil conditions are affected by temperature and the interplay between soil conditions and the form of precipitation can result to very high or very low erosion rates, during different periods of the year. Snowfall, as well as rainfall, in a frost ground will not cause any erosion, but the runoff caused by the melting of snow and the rainfall in defrost soils can cause very high erosion rates.

Although the amount and intensity of rainfall are significant erosive factors, the amount and rate of runoff should also be taken into account. Additionally, the infiltration, which is the result of the accordance of the earlier soil humidity and soil sealing, should be calculated and taken into consideration. Surface runoff occurs when the precipitation rate exceeds infiltration rate. Consequently, comprehending the hydrology of the study area is vital for the integrated study of runoff-erosion.

Vegetation acts protectively against the forces of the raindrop impact and surface runoff. Plant roots, organic material and the products of decomposition increase the soil's resistance to erosion. The climatic variables of precipitation, temperature, evapotranspiration and, in some cases, soil humidity, affect the biomass and decomposition within the soil's profile. These climatic variables affect the growth and decay of vegetation which, in their turn, affect erosion significantly. The quantity of vegetation is in direct correlation to the quantity of precipitation. Vegetation, or its decomposition products, e.g. falling leaves and roots, increase the organic material on the surface of the soil and in the interior of the soil cap, and therefore reduce the soil's susceptibility to erosion. The amount of organic material coming from vegetation depends on both the plant biomass, which was initially produced, and the remaining quantity after other uses. The remaining organic material on the surface of the soil is lost as time passes due to decomposition processes. Decomposition appears quicker in areas characterized by high values of temperature and precipitation.

2.2 Soil

In general, the term soil refers to the loose material that covers the earth's surface and can be distinguished from solid rock (Govens & Poesen, 1986). The soil performs many functions, feeding and supporting the plants growing on it, and constitutes the base on which roads and buildings are constructed. Furthermore, it produces sediments, due to erosion, and constitutes the means for water circulation.

Soil evolves dynamically through time. Initially, the bedrock is weathered into unstable mineral material that operates as the parent material in soil genesis. The sediment originates either from aggradational processes through water, or from the wind. Finally, the development of soil reflects the combined effects of climate, topography, organic material and time.

Certain soils are much more vulnerable to erosion than others (Agassi et al., 1996; Bryan, 1977; Foster et al., 1985; Foth & Turk, 1972). Apart from the soil properties, other factors also affect the soil's vulnerability to erosion, such as the presence of organic material.

In order to measure the soil's vulnerability to erosion, a constant term of reference was established, in which all external affects are excluded, apart from the influence of soil properties. An example is the case of a limited area of land, where soil is kept fallow (without vegetation) and its upper and lower parts are periodically cultivated. The sediment from the limited experimental area is collected and illustrated graphically as a function of vulnerability to erosion, where the x-axis shows erosivity and the y-axis shows soil loss. The inclination of the line corresponds to an empirical measurement of the vulnerability to erosion.

The most important property in many processes, and especially in erosion, is the composition of the soil. Soils, which are rich in clay, present low rates of vulnerability since they are resistant to detachment. Soils, rich in sand, present low rates as well, at least where vulnerability index is based on the total kinetic energy (practically, the amount of precipitation) and the intensity of rain. Due to reduced runoff, such soils produce less sediment, because of the high infiltration rates. Even when soil particles are easily detached from sandy soils, only a small surface runoff is observed for this detachment and transportation. Soils with medium-sized composition are the most vulnerable to erosion. In these cases, soil particles are distracted and the eroded sediment can be easily transferred. The permeability of soil profile is related to its vulnerability to erosion, since it affects surface runoff. The presence of, e.g. limited layers of clay, near the ground surface can significantly increase vulnerability, especially in sandy soils that are easily detached. An increase of the surface runoff on these soils can lead to an important increase of erosion. Another factor that influences vulnerability is the soil structure, which reflects the sorting of the organic and inorganic materials (Grissenger, 1966; Moldenhaeur & Wischmeier, 1960; Wischmeier et al., 1971).

Soil's vulnerability to erosion varies during the year. Soils are most vulnerable during the months in which the energy, the amount of precipitation and soil humidity are highest. The quantity of surface runoff erosion is increased by high soil humidity. Low temperature increases the humidity of soil, since evaporation is low. As a result, soil vulnerability tends to be higher during the last months of winter and the first months of spring, rather than in the end of summer. Additionally, vulnerability is reduced during the last months of summer because the higher temperatures increase the biological activity in the ground and produce organic compounds, which function as composing factors of the soil.

Defrost soil is particularly vulnerable to erosion due to its saturation that leads to greater surface runoff (Van Klaveren & McCool, 1998).

The compaction of soil and the diffusion of the fine particles that are generated from the impact of raindrops can lead to the development of a thick layer on the soil surface (Bradford & Huang, 1992; Bradford et al., 1986; Summer & Stewart, 1992). This layer is called crust when it is dry and sealing when it is humid; it limits the infiltration rate, which, in turn increases runoff and erosion.

Soils can vary depending on their vulnerability to erosion due to the raindrop impact and to surface runoff. An example where soil is much more vulnerable to runoff erosion rather than from raindrop impact is the case of runoff in defrosted soils. A soil, rich in clay, can be particularly resistant to runoff erosion compared to soils rich in mud. Erosion caused by the raindrop impact is uniformly distributed on the impact surface, while runoff erosion increases with distance along the route of runoff (Meyer et al., 1975). The affect of morphology in erosion processes is larger when the soil is more vulnerable to runoff erosion, rather than to erosion from raindrop impact.

Erosion differs in a landscape since the soil properties vary as well. For instance, the behavior of soil on the slopes' top is defined, to a large extent, by the parent material (Renard et al., 1997). On the contrary, the lower parts of the slopes reflect the properties of a developed soil in a depositional environment, which is quite different from the developing soil of the slopes top. The fluctuations of humidity conditions have a significant impact in soil development and its consecutive properties. The lower parts of the relief and the cavities of the ground tend to be more humid than other areas, which results to different soil properties due to anomalies in the developing environment.

2.3 Morphology

The morphology refers to the geometry of earth's surface. The length of the slope, its inclination and its shape in section and in plan-view are the most important geometrical variables. Slopes with stable inclination along their length are called uniform; in these cases, erosion increases along the slopes. The most common forms of slopes are presented in figure 1.10.

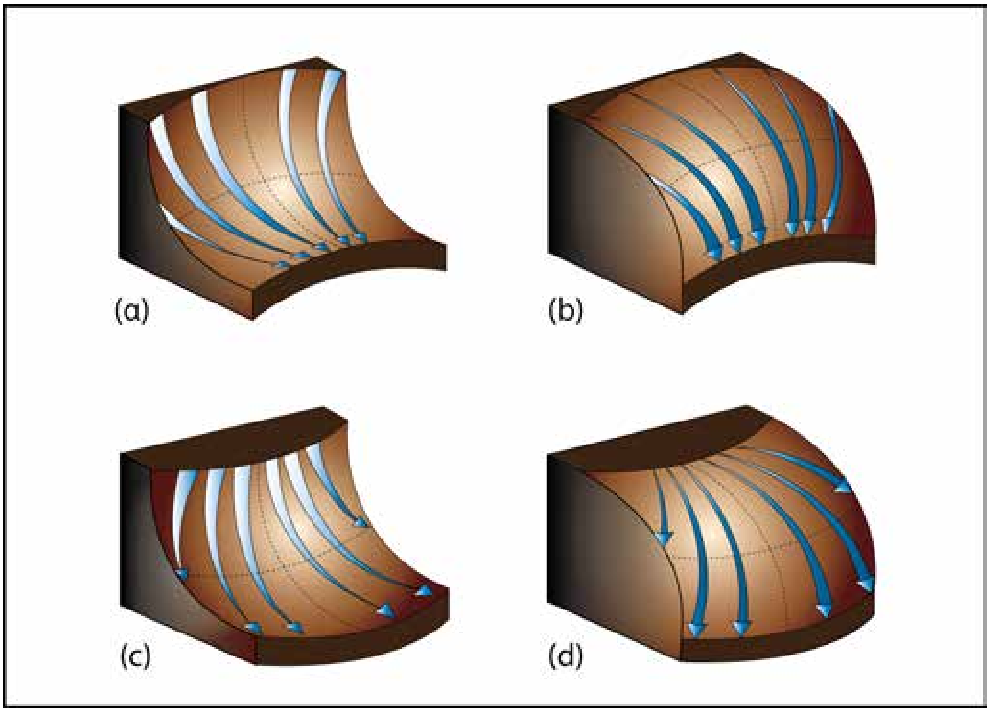

Fig. 1.10: Various forms of slopes (a) concave-concave form, (b) convex-concave form, (c) concave-convex form, (d) convex-convex form and the effect of morphology on surface runoff.

The sediment gathered at the end of a uniform slope differs as a function of the slope's length and it also depends on the amount of erosion caused by raindrops impact in relation to erosion caused by runoff. Moreover, erosion depends on the inclination of the uniform slope (McCool et al., 1987). Erosion increases linearly with increasing inclination. Nonetheless, in the case of runoff erosion, the inclination has greater affect than on erosion caused by raindrop impact (Foster, 1982).

Therefore, erosion on a part of the slope is a function of distance from the starting point of surface runoff and the inclination in that location (Foster et al., 1977). In a location towards downstream, the rate of erosion will be high. For a given location, erosion is proportional to the inclination. This principal can also be implemented in non-uniform slopes, which are most common in nature. A non uniform slope has varying inclination along its length. Other characteristics may differ as well, but only the deviation of the inclination is taken into account during the morphological study of the slopes. In a convex slope, the inclination increases constantly along its length. Erosion on a part of the slope is related to both the inclination and length. The maximum erosion rate is much larger in a convex slope than in a uniform, and the accumulating sediment at the end of the convex slope is also more than in the uniform.

A concave slope is the opposite of a convex. Although the inclination in a concave slope is large, runoff is minimum (Fig. 1.11). The maximum erosion rate is slightly smaller in a concave slope than in a uniform. The same applies for the sediment production, since the steep inclination of a concave slope does not get small enough to lead to deposition.

Fig. 1.11: The inclination in a concave slope is large, but the runoff is minimum. The topography of a slope, which defines runoff, is in direct relation to its transfer capability. As a result, deposition takes place at the lowest part of concave slope and erosion at its upper part.

When the available amount of sediment is larger than the transporting ability, sedimentation takes place on a slope. The available sediment for transfer correlates with the amount of sediment produced, in the upper part of the slope. The topography of a slope, which defines runoff, is in direct relation to its transfer capability. As a result, when the steep inclination of a slope is flat enough at its lower part, it can present significant amount of deposition; this reduces significantly the sediment transfer at the end of the slope. A complex slope with a convex upper part and a concave lower, behaves as a combination of convex and concave slopes.

Figure 1.12 illustrates a typical topography, which consists of three basic parts.

· The slopes, where runoff occurs parallel along the slope (D). The discharge rate, across the width of the hill remains the same, along route of the flow.

· The second important part is at the edges or the tops of the slope (F). In these areas, flow deviates so much that outflow, per unit of width, is reduced downstream and results to a reduction of the erosion rate.

· The third important part is the cavities that cause the convergence of runoff (G). In these areas, the outflow, per unit of width, increases towards downstream with a corresponding increase in erosion. The areas of concentrated flow are formed in cavities where runoff converges.

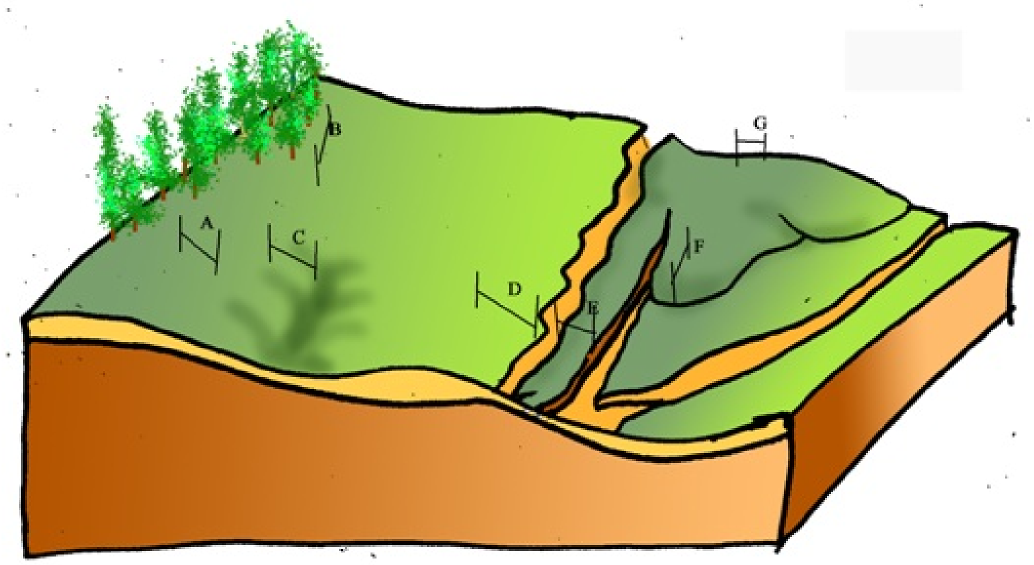

Fig. 1.12: Typical topography consisting of three basic parts.

Topography has a direct impact on erosion. Soil humidity tends to be larger at the bases of slopes and in cavities, rather than in the upper parts (Weltz et al., 1998). Vegetation, biomass and the presence of organic material on the surface and the interior of the soil tend to be greater in areas where humidity is high. This contributes to the spatial fluctuation of erosion, the formation of runoff and, last but not least, to the affect of the slope's inclination from the impact of raindrops (Foster, 1982).

2.4 Land uses

The land uses parameter has a special affect on erosion processes. The term "land uses" refers not only to the general use of the land, but to the applied management as well. For instance, one land use can characterize an undisturbed area, in which all human activities are forbidden. Forest areas are another land use, where disturbance is generally restricted.

Vegetation can be much greater in areas of controlled grazing as opposed to areas where overgrazing takes place. Vegetation species are primarily defined by climate and soil; however the amount of vegetation is controlled by human interventions, such as the management of cultivated areas and pasture lands. Vegetation combined with soil disturbance, which affects erosion, is defined by management.

Generally, land use widely refers to the vegetation cover, the applied cap and the level of vegetation and soil management, along with the support practices that are implemented in order to control erosion.

Land uses have an impact not only on the forces that are applied on the soil but also on the soil’s resistance towards them. Three elements of land use are more important for runoff erosion:

- the vegetation and its role
- the surface cap, e.g. deriving from plant remains, and
- the mechanical disturbance of the soil (Renard et al., 1997; Weltz et al., 1998; Wischmeier, 1975).

2.5 Weathered Cap

2.5.1 Tree foliage

The foliage of the treetops constitutes the overground part of the plants that prevents the raindrops, which does not contact the ground to influence surface runoff. Thick foliages ‘block’ a significant percentage of rainfall to reach the surface of the ground. A quantity of rainfall evaporates before it reaches the ground. This obstruction of the rainwater can be important with thick foliages, of large extent per unit of soil, and it results to a significant decrease of the surface runoff.

Part of the rainwater, which was ‘blocked’ by the foliages of the treetops, flows along the surface of plants, approaching their stems and moving towards the ground. The erosion by the impact of these raindrops is zero. Although this water becomes part of surface runoff, the erosivity of its runoff is reduced because of the travel time from the obstruction point to the surface.

Part of the ‘blocked’ rainwater forms water drops which fall down to the ground. If the bottom of the canopy is close enough to the soil’s surface, the erosivity of this water is much smaller than the one from raindrops that were not stopped. However, if the bottom of the canopy is high enough, the height of the falling drops is enough to approach the erosivity of the non-stopped rainfall (Chapman, 1948).

The percentage of soil that is covered by foliages and their density defines how much foliages affect erosion. Between plants there may be an open area, where raindrops can directly impact the ground, without foliages limiting their erodibility. Erosivity from foliages can be reduced by plants, whose canopy bottom is close to the ground (US Department of Agriculture, Forest Service, 1987).

The foliages of the treetops are alternated during the year, through natural processes. After maturity, plants lose their canopy as leaves lean and fall down. The maximum control of erosion can be accomplished by the selection and management of the plants, in order to have maximum canopy of foliage during the periods of maximum erosivity.

The soil properties, the humidity and other microclimatic characteristics affect differences between areas resulting to a spatial variation. Different plant species can appear at the same place, such as the thick grass cap under a bush. The canopy foliage of grass has a greater affect in the reduction of erosion than that of bushes.

2.5.2 Characteristics of vegetation

The trunk of a tree and the area of its total diameter are equally significant factors in the processes of erosion. Erosion can be eliminated in the part of the ground which is covered by the spread area of the plant. Thick branches decelerate runoff and reduce erosivity. Plant stems can actually accelerate erosion around them. In general, branches and their spread area decrease erosion.

After the harvest, the entire biomass of an annual plant’s root, such as wheat, remains in the soil (Reeder et al., 2001). This material constitutes the decomposing biomass which will reduce vulnerability and significantly increase infiltration. Water can flow sideways of the living roots and the rotten dead roots that lie in the rills. Through this process, the infiltration is increased in the rills, whereas runoff is decreased and erosion is limited.

The aboveground biomass is integrated into the soil through cultivation or other mechanical disturbances, which mix the plant materials within the soil. Biomass can be added on the ground by fertilizers and other organic materials. Worms and other organisms living into the soil incorporate the plant material, in the upper 50 mm of the soil (West, 1990). Plant remains are more effective when they are near the surface rather than when buried deeper. Flora and fauna activities improve the structure of the soil and reduce erosion.

Erosion is reduced as the amount of biomass increases from plant production (Renard et al., 1997). The production of underground biomass differs spatially, like the aboveground biomass. The quantity of the remaining organic material depends on the decomposition rate and that, in its turn, depends on the temperature and humidity of the soil, which are inextricably connected to air temperature and precipitation.

In some occasions, lines of vegetation are developed on a slope or in the edge of a stream so as to retain the sediment. These lines, which are also known as filter strips, significantly decelerate runoff, and result to deposition and reduction of sediment transfer in the area. Filter strips that are placed at a hill's slope are known as buffer strips, and they result to the decrease of erosion while preserving the sediment on the slope. The alternative cultivation of strips includes planting some lines with thick vegetation in alteration with others with intensive cultivation. Strips of thick vegetation and other crops alter so that any kind of vegetation can grow on every strip. The width of the strips in the alteration system is, usually, the same. Cultivation in strips reduces the amount of sediment that moves to the downstream of the slopes. The effectiveness of the vegetation strips depends on the density of plants and on the ability of the strip to decelerate runoff.

2.5.3 Soil cap

The surface cap is in direct contact with the soil, protecting it from the impact of raindrops and decelerating runoff. The soil cap can form from plant remains that accumulate from falling leaves, from harvest remains, protective layers of hay, artificial material added on the surface and living parts of the plants that contact the ground. Research has shown that the affect of the soil cap on erosion is directly related to the percentage of the covered surface. The impact of the soil cap varies according to the climate, the topography and soil conditions (Box, 1981; Meyer et al., 1972).

The protective cap of hay, which is applied in construction areas, is less effective in the reduction of erosion than harvest remains in a rural area. The harvest remains of the previous year are very important in the preservation of the cultivation systems that are used widely for controlling erosion in cultivated areas (Simanton et al., 1984).

Cryptogams are collections of mossy, algae and lichen, which develop across the soil profile. The resulting organic crust of soil can reduce infiltration rates by preventing flow in the large pores of the soil and therefore increase runoff and soil erosion. In other cases, cryptogams can improve porosity and infiltration by increasing soil aggregation and asperity (West, 1990). Furthermore, cryptogams protect soil from both types of erosion, runoff and raindrop impact.

3. MECHANICAL DISTURBANCE

Apart from soil management, soils properties are also crucial in soil's vulnerability to erosion. The presence of organic material, as already mentioned, affects vulnerability.

Furthermore, mechanical disturbances affect not only the vulnerability of the soil but also the erosivity of the raindrop impact and surface runoff. Mechanical disturbance decelerates runoff and reduces the vulnerability to erosion, by creating superficial hardness (ruggedness). Moreover, the hardness of the surface forms cavities where water stagnates, resulting to the reduction in raindrop erosivity and to local storage of the transported sediment.

In nursery conditions, the soil is approximately twice as vulnerable to erosion after cultivation, than after an extensive period of a few years, with no mechanical soil disturbances. On the contrary, the superficial plant material is often buried and mixed with the underground plant remains by the mechanical soil disturbance, in order to significantly reduce erosion.

Soil can be artificially developed, in order to redefine the runoff from a downhill route to another around the slope; a practice known as contouring. This redefinition of flow prevents the erosivity of runoff.

Terraces are often formed in slopes, in order to reduce their length and consequently control erosion caused by runoff. Terraces can, also, be developed as part of a process known as "chain" in natural leas, by removing trees and bushes. A long chain or wire is stretched between two vehicles that move parallel through the ground surface and uproot the highest vegetation of the ground. The remains from the broken plants accumulate regionally and act as terraces to control and direct runoff around the slope.

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CHAPTER 2

LARGE SCALE APPROACHES OF RUNOFF EROSION

By

Stephane Cordier

1. INTRODUCTION

The aim of this chapter is to place runoff erosion into broader spatial and temporal scales. The change in spatial scale is necessary since the runoff erosion processes form part of global erosion, with a transfer of sediments from the upstream parts (of a slope, of a catchment), to the downstream parts, e.g. from source to sink. The sediment transport first occurs on the slopes due to runoff erosion combining the action of water and gravity. The sediments are subsequently able to be transported as fluvial deposits by streams and rivers. They may potentially be transported to sea (Vis et al., 2009), or deposited in the floodplains (or alluvial terraces). Following this, the rivers have to be considered as reliable recorders of runoff erosion, allowing sediment budget to be reconstructed, in particular for small to meso-scale catchments. They hence provide additional informations about slope evolution. This is all the most important as research focus in many areas on fluvial evolution rather than on slope dynamics. However, it is essential to underline that the sediment transfer from slope to river (as well as the transfer along the river) is neither instant nor continuous, and that many relays may take place between the initial erosion and the final deposition. Following this, the change in spatial scale requires the time dimension to be taken into consideration. This chronological approach is naturally not intended to replace the morphological (landforms and processes), pedological (post-sedimentary pedogenesis) and stratigraphical (relative chronology) approaches, but to enrich a multi-proxy reconstruction of runoff induced slope evolution. Absolute chronology is actually fundamental in three ways:

1) Firstly, it makes it possible to assess erosion rhythms, which are a major topic for current quantitative geomorphology. Such rhythms may not only be estimated at a multi-year timescale, but also at significantly longer timescale ranging from the historical periods to the Late Quaternary. Such a perspective is fundamental as the current context emphasizes the recent soil degradation (including runoff erosion as well as salinization, desertification etc) in relation with human influence. This interpretation cannot be denied in a context of increasing world population and induced pressure on soil (especially cultivable lands). However, this evidence should not obscure the fact that runoff erosion plays for millennia a significant role in slope

evolution. This is typically illustrated by the famous words of Plato in the Critias (fourth century BC):

"And so, like it is on the small islands, our recent land has become looking like bones of a sick body if compared with the former times: all the fat and soft land has come down and the bold skeleton remained only. In those past times our land was intact, with high hills, and the so called now stony plains had an abundant lush land, the mountains were covered by broad forests. (…) Among our mountains there are such which now feed bees only, but still roofs made of trees cut not very long time ago are preserved there (…), and the land was giving unbelievably rich pasture to the flocks. The water which Zeus was sending was a fruitful one, not like now when it is fading away with no use, flowing to the field out of the bold land". Even if this text does not mention relations between land-use and runoff erosion, it is a major testimony that, already twenty-four centuries ago, Greeks were aware of the catastrophic impact of soil erosion (including in particular runoff processes).

2) Secondly, the realisation of numerical ages is very important to unravel the complexity of slope dynamics, in relation with the common relays and temporarily storages on the slope prior to the final deposition at the bottom of the slope or in the floodplain. This complexity is well illustrated by the fact that eroded sediments may finally be deposited above younger sediments due to long storages between first erosion and final deposition (Porat et al., 2007; Fuchs et Lang, 2009).

3) Thirdly, the chronological framework allows a better understanding of the origin of colluviation processes in terms of forcing. In case of runoff erosion, the internal (tectonics) forcing may be considered as negligible except in some specific cases (Larue, 2005). The runoff erosion should actually be considered as a response to either an external (climate) forcing, or to an anthropogenic forcing.

This distinction is an important matter for geoarcheological research (reconstruction of long-term interactions between man and landscapes). It is also a necessary step for any strategy for reducing the effects of runoff erosion. This is the case in Mediterranean areas, which are especially exposed to runoff erosion due to the characteristics of climate (heavy rainfalls during autumns), to an ancient anthropogenic pressure, and to the presence of steep slopes. Prime examples are also provided in other areas, for example in Africa during European colonization

(Rossi, 1998): when the Belgian colonisers arrived in Rwanda in the early 20th century, they noticed the presence of steep slopes which were cultivated. Due to the occurrence of heavy rainfall related to the tropical climate, a runoff erosion hazard has been decreed, leading to the realisation of huge « coloniser works »: 462 000 km of anti-erosion ditches were built until the independence of Rwanda in 1962. These works were found to be fully useless, as the traditional farming practices were sufficient to reduce runoff erosion, enabling an equilibrium between the men and the landscapes. Furthermore the realization of these ridges coupled with the introduction of colonizer practices had negative consequences as they led to increased hypodermic landslides (Rossi, 1984). Similarly, the Peuls living in Fouta-Djalon area (Guinea) have been found guilty for the deforestation of the bowé (plateaus covered by a ferruginous crust, typical for the Sahelian margins), as they carried out slash and burn farming for breeding. Subsequent morphological research demonstrated that the ferruginous crust largely predated the Peuls arrival in Guinea, and that no forest would have ever been able to grow in these bowé: the Peuls only adapted their pastoral system to the environmental conditions (Rossi, 1998).

Similar problems exist at longer timescale, as illustrated by the lavaka landscapes in Madagascar (Neboit, 1991): these lavakas (“holes” in Malagasy) corresponds with wide gullies (up to several hundred meters deep and wide) developed in the weathered siliceous rocks. They are typically associated with savannah. The origin of these lavaka remains for several decades a complex matter, as research balance between natural forcing (dryer periods during the Holocene enabling the development of savannah, morphogenetical system related to aggressive tropical climatic conditions) and anthropogenic forcing (land clearing allowing the development of gullies).

Following this, the obtention of a chronology for soil erosion is of great importance to unravel the slope response to climate and anthropogenic forcings during the last millenia. For the last decades, this chronology can be provided by maps, aerial photographs, satellite images, or by the testimony of local populations (societal memory). Useful information can also be derived from high resolution analyses of the soil and slope deposits (morphological and pedological memory). The dendrochronology finally allows a precise chronology to be obtained in particular for landslides, by studying either the outermost tree-ring of a buried dead tree, or the deformations of the tree-rings for a still living tree (in case of slow rotational or tilting landslides). For longer term (a few centuries to a few millennia), the archaeological artifacts may be very useful, bearing in mind that their chronological resolution may sometimes be low. A better resolution can however be obtained using numerical dating (geochronological approach). Despite several methods can theoretically be applied to slope deposits related to runoff erosion (Lang et al., 1999), two methods are commonly used, both belonging to the so-called radiometric dating: the radiocarbon method and the Optically Stimulated Luminescence (OSL). The present chapter is divided into three sections: the first one is devoted to a presentation of these main methods (physical principles, application in the case of slope deposits, advantages and limits). Case study will then be exposed to assess the role of these methods to identify the forcing and for the reconstruction of sediment budget including not only the slopes but also the fluvial systems.

2. RADIOCARBON AND OPTICALLY STIMULATED LUMINESCENCE DATING APPLIED TO SLOPE DEPOSITS

2.1 Physical principles

2.1.1 Radiocarbon dating

Radiocarbon is one of the first dating methods which became applied widely in geomorphological research. Developed in the 1950's, the method is well known and routinely used, so only the basic principles will be here described. This method is based on the presence in any living material (plant, animal etc), of an unstable carbon isotope (^{14}C) in addition to the stable isotopes (^{12}C and ^{13}C). Due to the exchange related to the carbon natural cycle, the carbon is absorbed by plants and animals (through photosynthesis and food chain). As a consequence the proportion of ^{14}C in living organisms is similar to that present in the atmosphere. This proportion is low (±one atom of ^{14}C for a billion of carbon atoms). This proportion starts to decay at the death of the organism, with a half-life of 5730±40 years. The decay can be measured using the classical counting methods (gas counting or liquid scintillation counting), which allow the radioactivity of a given sample to be estimated using detectors. However, these methods require a sufficient bulk for the measurements to be precise enough. This explain the development of the AMS (acceleration mass spectrometry) as an alternative method: the latter actually presents the advantage that it allows the number of non-disintegrated radioactive atoms (even a very few) to be directly counted. The comparison between the residual ^{14}C in the sample and the original content allow an age to be derived, with 1950 as reference year (this choice being allocated to the fact that the atmospheric carbon contents were later influenced by nuclear bomb tests). The obtained age is thus provided in years BP (before present = before 1950). However, later research demonstrated that the level of atmospheric radiocarbon was not constant. Following this, a calibration procedure was developed, based on sample for which an independent age control is available (dating by another method, allowing the obtained age to be validated). The corrected age can thus be given in years BC (Before Christ) or AD (Anno Domino, After Christ).

Due to the existence of an international standard for the original radiocarbon level, the carbon dating presents the major advantage that all the obtained ages are directly comparable, no matter which laboratory performed the measurements. In addition, the radiocarbon method exhibits a low error range (only a few %). The errors directly relate to the measurements and include the instrumental error (noise of the instruments, measuring time) and, for the corrected ages, the uncertainty derived from the calibration procedure. The latter can be more or less important, depending on the considered period.

2.1.2 The optically stimulated luminescence dating osl

The Optically Stimulated Luminescence OSL developed more recently. Since the 1990's, it replaced progressively the thermoluminescence method TL (the latter being applied since the 1970's) for the dating of a wide range of sediments (aeolian, fluvial, but also slope-deposits). In contrast, the thermoluminescence remains applied to volcanic material and archaeological artifacts (potteries etc).

Luminescence methods (TL and OSL) are based on the estimation of the impact of radiations on the crystalline structure of minerals while they are shielded from light (Aitken, 1985; Wintle, 1997; Aitken, 1998; Duller, 2004; Vandenberghe, 2004). The main studied minerals are quartz and K-rich feldspar, which can be found in almost all sedimentary environments. The radiations (α, β, γ) come from radionucleides which are present in the mineral and its natural environment, mainly uranium, thorium (and their decay products), potassium, and for a small proportion from the cosmic particles (Aitken, 1985). They lead to the emission of electrons which are subsequently trapped in crystalline lattice defects. Some of the traps are considered ‘unstable’ (“shallow traps”), which means that an electron inside will not remain trapped for the whole duration of burial. On the contrary, defects situated deeper inside the lattice have a higher thermal lifetime. These “deep traps” (stable traps associated with high energy levels) can adequately be used for dating. The total amount of trapped electrons within a crystal is proportional to the total energy absorbed and retained by the crystal (or dose), hence the time it was exposed to radiation (Fig. 2.1). As soon as the mineral is exposed to sunlight, for example during its transport, trapped electrons absorb the photon energy (from the Sun), and are released from the traps. The rate of release depends on four main parametres: i) the kind of mineral: the eviction occurs faster for quartz than for feldspars (reduction of the luminescence signal by a factor of 100 in ~20 s for the former, but a few minutes for the latter; Godfrey-Smith *et al*., 1988; Mercier, 2008); ii) the intensity of the light flux; iii) the wavelength of the stimulating light: quartz are especially sensitive to short wavelengths from the visible spectrum (*e.g.*, UV, blue, green), *i.e.*, these are more efficient to stimulate and release the electrons. Feldspars have the specificity of being sensitive both to short and to near-infrared or infrared wavelength (800-950 nm; Bøtter-Jensen et al., 1994); iv) the sensitivity of the trap to light. Released electrons can recombine with another kind of crystalline defects (“holes” reflecting electrons vacancies).

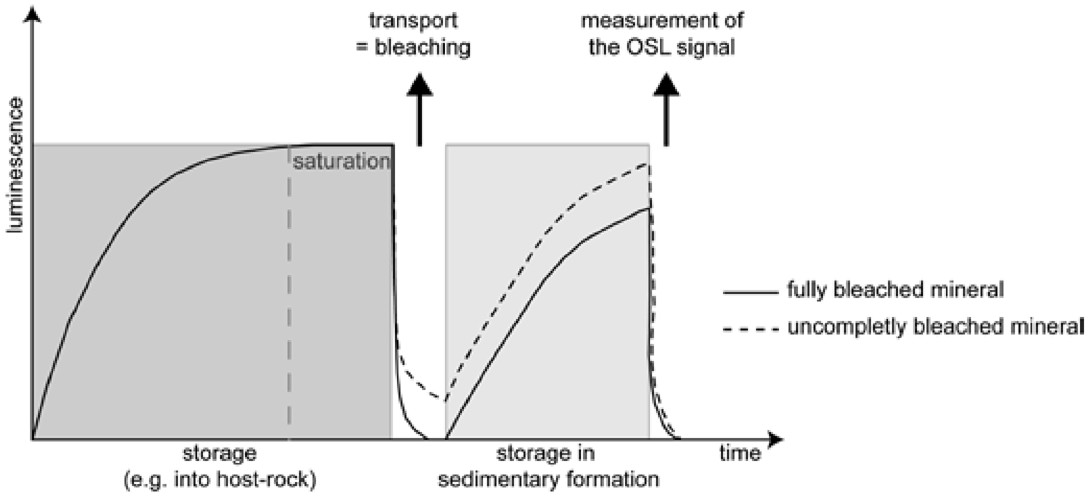

Fig. 2.1: Acquisition and release of the luminescence signal

This recombination generates the emission of light (the luminescence signal) which can be measured in laboratory through heating (for TL) or through light stimulation for OSL (Huntley et al., 1985). The intensity of the signal is proportional to the amount of released electrons. The wavelength of the signal is allocated to the nature of the mineral: the OSL from quartz is typically measured in ultra-violet (340-370 nm wavelength), while quartz also emit in blue (460-500 nm wavelength) and in orange-red (600-650 nm wavelength; Huntley et al., 1991). These latter wavelengths are however less used to avoid interactions with the stimulating light. Feldspars mainly emit in 330-550 nm wavelength (Huntley et al., 1991; Stokes, 1999) and are typically measured in the 400-550 nm wavelength (blue light).

Based upon these physical principles it is possible to recognise the importance of the luminescence dating for sediments and in particular for slope deposits: 1) prior to sediment transport (*e.g.*, when the mineral is stored within host-rock), natural radiations generate the trapping of electrons and the build-up of a latent luminescence signal; 2) when a grain is produced by mechanical erosion and transported, it is exposed to sunlight. This leads to the release of trapped electrons ("bleaching") and the emission of the luminescence signal. 3) after its transport the grain is "definitively" buried under a sedimentary cover. It is exposed again to radiations and accumulates trapped electrons (Fig 3.1). Much later the grain is sampled in the field and stimulated in the laboratory using either visible light for quartz (OSL *stricto sensu*, typically performed with blue or green LEDs), or infrared light for feldspars (Infrared Stimulated Luminescence IRSL). The analysis of the OSL signal (Fig 2.2) makes it possible to estimate the time elapsed since burial, *i.e.*, the age of the sediment. This age is derived using the following equation: Age (in a) = P (in grays = Gy = J/kg) / D_r (in Gy/year). P is the palaeodose (the total amount of dose absorbed by the mineral throughout its burial). It is estimated in laboratory by the determination of the equivalent dose D_e, *i.e.*, the artificial dose which is necessary to induce a luminescence signal similar to the natural one. D_r is the dose rate (the rate at which the sediment is exposed to natural radiations).

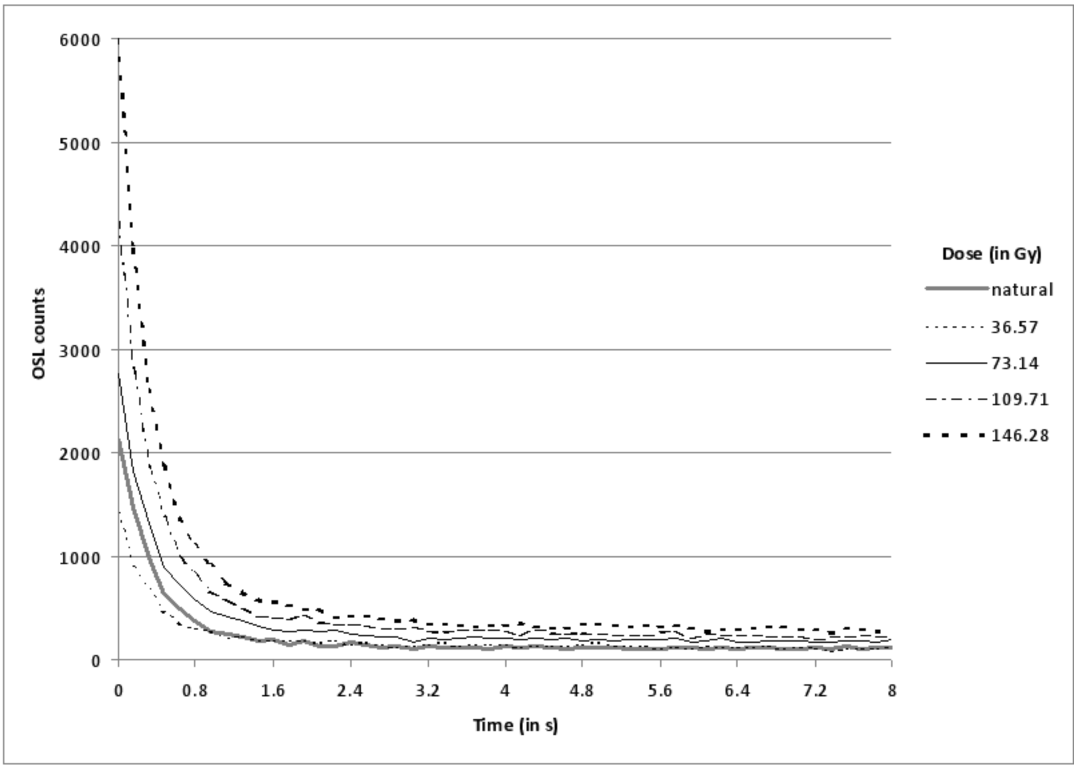

Fig. 2.2: Example of OSL decay curves for natural and regenerative doses.

Even if both methods are based on environmental radioactivity (radiometric datings), OSL and radiocarbon methods are very different in terms of functioning. It is consequently important to underline, in a comparative way, the advantages and limitations of each method for the dating of colluvial deposits associated to runoff erosion.

2.2 Potential of radiocarbon and osl for dating slope deposits

2.2.1 Direct versus indirect dating of slope processes

A main difference between both methods concerns the dated material: while radiocarbon is typically applied to organic matter (pieces of wood, charcoals, pollens, peat etc), OSL is applied to minerals (mainly quartz and feldspars) constituting the sediment. Following this, OSL is more commonly applied to sandy to silty sediments (e.g. loessic material, granitic sands etc) rather than to sediments originated from a chalky or marly mother rock. In contrast, the radiocarbon method can be applied to any kind of sediment containing organic matter: it also appears more applicable, even if datable organic remains are not able to be found on all the slopes (Fuchs et al., 2004).

However, OSL presents a considerable advantage for the dating of slope deposits associated with runoff erosion, since this method allows the sedimentation phase to be directly dated. In contrast, the radiocarbon method does not allow the age of the sedimentation to be directly estimated (Howard et al., 2009), as it only provide age estimate for organic matter included in the sediment body of interest: a time lag (with unknown duration) may actually exist between the death of the organic material and its incorporation in the sediment, leading to age overestimation. This time lag is not significant when the transport occurs as a single event, or when the colluviation phases are separated by long stable periods enabling the development of an organic-rich soil. However, it is important to keep in mind that radiocarbon does not date directly the slope erosion events.

Caution is also needed when dating (either by radiocarbon or by OSL) a sediment layer situated below the slope deposits of interest (e.g. a peat layer overlain by colluvia): the derived age will correspond with a maximum age for the studied event. Interpretation is even more complicated when the radiocarbon age concerns a layer located above the slope deposits: in that case, the obtained age can only provide an order of magnitude for the timing of slope erosion (the death of the organic material can actually be either younger or older than the runoff erosion event).

2.2.2 Other source of age under- or over-estimation

Errors in age determination are also possible for both methods, regardless of the procedures or of the abovementioned caution related to the significance of the age estimates. In the case of radiocarbon, the greatest source of error is due to modern carbon contamination, especially through rootlet intrusion. In the case of OSL (and especially when dating slope deposits), a main problem is the potential incomplete bleaching: to be precise and accurate, OSL dating actually requires a transport event long enough to ensure a complete bleaching of the grains (zeroing of the dosimeter). Such a complete and homogeneous bleaching is commonly observed in Aeolian or fluvial sediments. In contrast, colluviation involves short transport (a few metres to a few hundreds of metres), which can be mass transport. The latter is obviously more favourable to the preservation of organic matter rather than a complete bleaching of all grains. Following this, the OSL dating method cannot be considered as a standard to date routinely slope deposits (especially in case of sliding or solifluctions). However, it is important to underline that colluvial sediments have been successfully dated in many cases.

Several reasons explain this success: first, the reset of the OSL signal is fast, as only a few seconds of light exposures are necessary to obtain fully bleached grains. Such a bleaching is all the most possible that the grain can be exposed 1) before the transport, 2) during the transport (especially in cases of multi-events transport), and 3) immediately after deposition, before the grains are buried by younger colluvia. The latter is particularly important to enhance the possibility of bleaching, especially with the role of bioturbation and mechanical processes. A second reason relates to the recent methodological improvements that characterize the OSL method and in particular the SAR protocol (see below): such improvements include a reduction of the aliquot size (enabling a better recognition of heterogeneous bleaching, e.g. incomplete bleaching for some grains), as well as the statistical processing of the equivalent dose distribution (see below). This led to the obtaining of more reliable age estimates.

2.2.3 Age ranges and accuracy

Another main difference between OSL and radiocarbon concern the age range covered. Radiocarbon ages can be accurately provided up to 35 ka BP (with the counting methods) or even 50 ka BP (using the AMS method). This includes the end of the Upper Pleistocene and the Holocene, which corresponds with the period of development of agriculture, and consequently of anthropogenic forcing on slopes. In contrast, radiocarbon does not allow very young ages (a few decades) or old (beyond the Middle Pleniglacial) ages to be obtained. OSL characterizes by a wider potential, with age ranging from present time (sediments deposited a few years ago) to ±100-150 ka (OSL dating of quartz) and even a few hundreds of thousands years (Infra-Red Stimulated Luminescence IRSL of feldspars; Juschus *et al.*, 2007). OSL method may hence be applied to runoff induced slope deposits deposited during the whole last glacial-interglacial cycle.

Both methods exhibit a relatively high precision (typical error range of ±5% for radiocarbon, 5 to 10% for OSL). It is also important to underline the fact that an international standard is available for radiocarbon, enabling all the ages obtained to be directly compared, whatever the laboratory which performed the measurements. In the case of OSL, and despite the generalization of the SAR protocol as a standard protocol, the procedures are more variable (stimulation time, measurements temperatures etc). However this limit is not really significant, as independent age control (performed where possible) confirm the reliability of OSL age estimates.

2.2.4 The importance of independent age control

On the basis of this comparison, the OSL dating method appears as the most useful method to provide a chronostratigraphical framework to runoff erosion processes (Table 2.1): it actually makes it possible to date directly and accurately the sediment deposition, for a time period ranging from a few years to the last glacial-interglacial cycle.

Method	Radiocarbon/AMS	Optically Stimulated Luminescence
Dated material	Organic matter included in the sediment	Quartz and feldspars (sand/silt) from the sediment itself
Dating of slope processes	Indirect (possible hiatus)	Direct
Age range	Up to 40 ka	Up to 100-150 ka (quartz) Several hundreds of ka (feldspars)
Typical error range (%)	5	5-10
Source for age underestimation	Contamination	Saturation (quartz) Fading (feldspars)
Source for age overestimation	Contamination	Incomplete bleaching
Age interpretation	Existence of an international standard	Influence of the procedures
Independent age control based reliability	Excellent	Excellent

Table 2.1: Comparison of the OSL and radiocarbon dating

However, the method can only be applied to sediments containing sand- or silt-sized quartz or feldspars, which is not systematic especially in limestones areas. The sediments must also have experienced sufficient transport for all the grains to be completely bleached. Following this, the possibility to use this method largely depends on the catchment characteristics, in particular the morphometry (which influences the potential length of transport), the sediment grain size, and the soil structure (influences the type of runoff erosion and hence the degree of bleaching).

The radiocarbon seems to be useful in a wider range of study areas (the presence of organic matter being the main condition) and can also provide accurate age estimates. However, its use for runoff dynamics reconstruction may be limited by the fact that radiocarbon does not allow the transport event to be directly dated (Table 2.1).

As for any research dealing with the past geomorphological evolution, the chronostratigraphical framework must, as soon as it is possible, be obtained using a combination of both methods (in addition where possible with archaeology). This is the only way to get an independent age control which allows the chronology to be validated. It is also important to obtain a high number of age estimates in the study area: this allows both potential aberrations to be detected, and the complexity of slope evolution to be reconstructed.

It is finally important to underline the necessity to avoid linear correlations between the obtained age estimates and a given climatic or anthropogenic event: despite it may be tempting to allocate a runoff erosion period with a more humid climatic period when the chronology suggests this, caution must be taken to this allocation: only a large-scale study, highlighting the occurrence of runoff erosion on various kind of slopes, can lead to conclude to a climate forcing.

This explains that any dating procedure should be preceded by an important field work (morphological, pedological and stratigraphical analyses). The latter does not only aims to define the sampling strategy, but also to raise the main questions datings should be able to answer.

2.3 Field and laboratory procedures

2.3.1 Field work

Field work is an essential step to reconstruct past slope evolution in relation with runoff erosion. It first provides the opportunity to analyse the morphological and stratigraphical framework. This makes it possible to select the best sampling places (at the scales of the slopes and of the trenchs or cores). The most favourable sites typically correspond with slopes exhibiting concave profiles (reflecting upstream erosion and downstream accumulation), and without significant amenities (buildings, roads, railways etc). The sampling can be undertaken from a pit, or using cores. In both cases, a preliminary stratigraphical recognition (precise location of OSL and radiocarbon samples) is essential to select the best sampling locations and to ensure a good interpretation of the obtained age estimates.

For radiocarbon dating, the material should be sampled away from bioturbated layers (e.g. roots) to avoid carbon contamination. For OSL dating, samples should never be exposed to daylight, as this which would lead to the resetting of the OSL signal (Fig 2.3). Two grain-sizes of sediment are commonly utilised for OSL dating: the “coarse grain analysis” uses fine to medium sand (90-250 μm); the “fine grain analysis” uses fine silt (4-11 μm). In case of pit sampling, once the sample has been removed the remaining hole can be used for an *in situ* determination of the dose rate. The dose rate may also be determined in the laboratory, for example using a gamma spectrometer (Fig 2.4; Aitken, 1998; Hossain, 2003).

Fig. 2.3: Example of sampling position for OSL dating

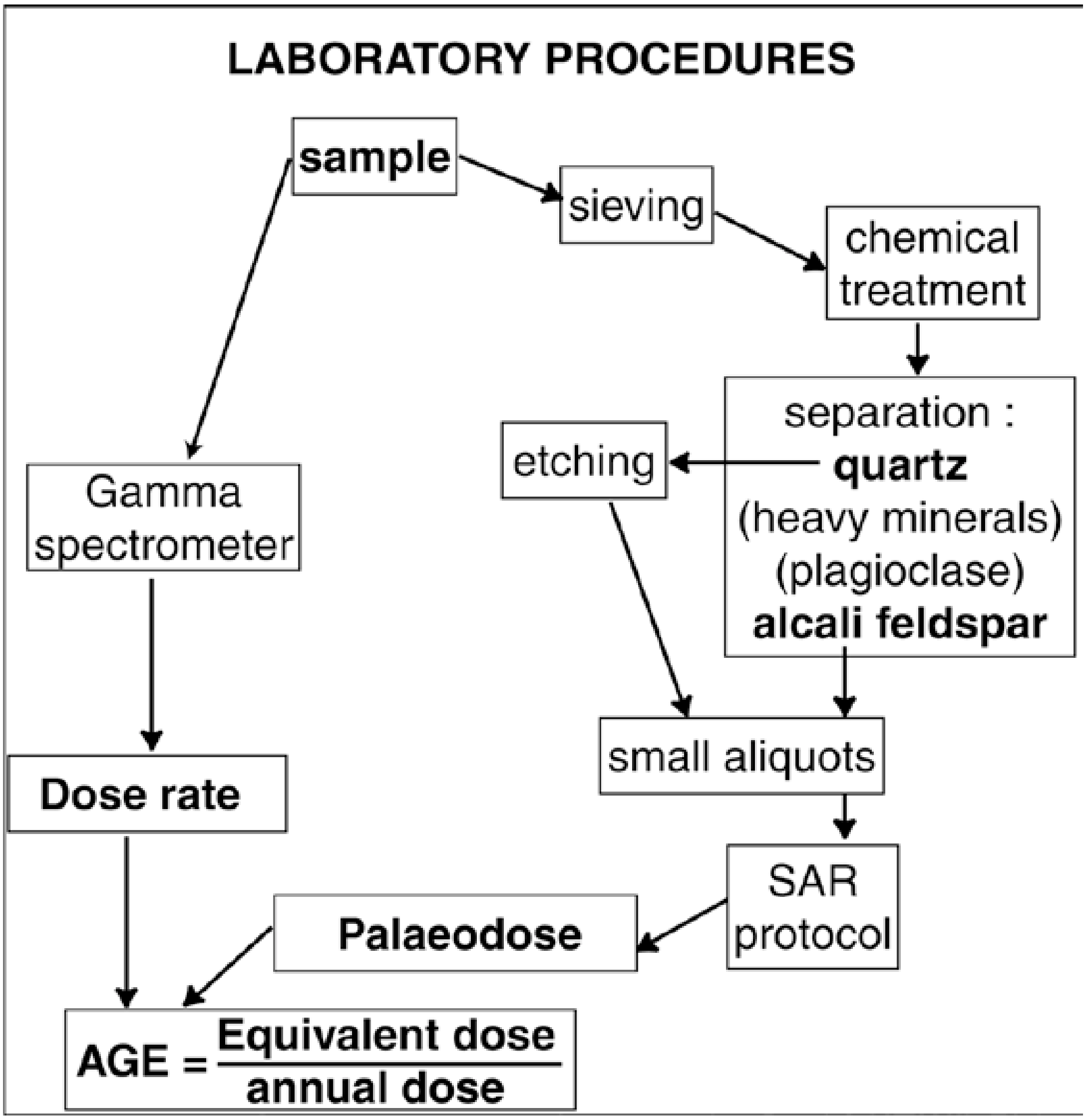

Fig. 2.4: Laboratory procedures performed for OSL dating

2.3.2 Laboratory procedures for osl dating

-Preparation for equivalent dose determination

Sample preparations for the luminescence measurements must be performed in the lab under subdued red or orange light, to allow the manipulation of sediments without affecting the latent luminescence. The preparation typically includes a sieving (to select the most representative grain size), chemical treatment (to dissolve or remove carbonates, aggregates and organic materials), heavy liquid separation (to isolate the mineral under study) and an etching (HF treatment) of quartz grains to dissolve residual feldspar and to etch the outer layer of the grains (Fig. 2.4). The grains are then mounted and sticked on 10 mm diametre discs in order to obtain subsamples (termed “aliquots”). The number of grains is highly variable: “large” and “medium” aliquots contain several thousands of sand-sized grains and several hundreds of sand-sized grains, respectively. For slope deposits, it is recommended to use small aliquots (several tens of sand-sized grains) or single grains, in order to be able to detect potential insufficient bleaching: medium or large aliquots may lead to an averaging of the palaeodose estimate in case of partial bleaching, since most of the aliquots will be likely to include both well bleached and poorly bleached grains.

-The SAR procedure

Several techniques were developed over the last two decades to measure the equivalent dose from the OSL and IRSL signals (Vandenberghe, 2004). The main procedure which revolutionised the optical dating is the “Single Aliquot Regenerative” (SAR) protocol, which is now a standard protocol used in luminescence dating (Wintle and Murray, 2006). The SAR protocol includes the measurement of several OSL signals for a single aliquot (Table 2.2).

1. Preheat 2. Blue-LED stimulation : **measurement of the natural signal Ln** 3. Give test-dose d. 4. Cut-heat 5. Blue-LED stimulation : **measurement of the signal induced by the test-dose Tn** 6. Give regenerative dose x1 7. Preheat 8. Blue-LED stimulation : **measurement of the regenerative signal Lx1** 9. Give test-dose d. 10. Cut-heat 11.Blue-LED stimulation : **measurement of the signal induced by the test-dose Tx1** 12. Give various regenerative doses xn and repeat step 7 to 11 *Checking of recuperation and recycling:* Give a regenerative dose = 0 s. and repeat step 7 to 11 Give a repeated regenerative dose and repeat step 7 to 11 *Checking of feldspar contamination:* Give the same regenerative dose Preheat IR diodes stimulation at 125°C for 100 s. Repeat step 8 to 11.

Table 2.2: the SAR protocol applied to coarse grain quartz.

The signals correspond first to the natural signal, then to signals obtained after artificial irradiation of the aliquot at determined doses (termed “regenerative doses”).

Basically the comparison of the natural signal with the artificial luminescence signals should make it possible to interpolate the palaeodose directly. The procedure is however more complicated, as artificial irradiation leads to the trapping of electrons in unstable traps of the mineral crystal lattice. Since the measurement of the luminescence signal occurs immediately after irradiation, these electrons are very likely to be released during the optical stimulation, leading to an overestimate of the signal. The unstable traps must consequently be emptied before the measurement. This is achieved by a preliminary heating ("preheat") of the aliquot (Table 2.2). Laboratory treatments may actually induce sensitivity changes in the grains, which means that the signal obtained after a subsequent stimulation may be affected by the preheat and irradiation conditions. To monitor this the SAR protocol includes the administration of a constant "test-dose", followed by a short heating ("cutheat") to deplete electrons in shallow traps, but without inducing a sensitivity change in the grains (Mercier, 2008), prior to the measurement of the induced signal ("test signal"). The SAR protocol can be summarised as follows (Table 2.2): i) preheat and measurement of the natural luminescence signal (L_n); ii) administration of a test-dose, heating, and measurement of the induced signal (T_n); iii) several cycles corresponding with irradiations at various regenerative doses, leading to the measurement of several artificial luminescence signals (L_x). Each measurement is followed by the administration of the test dose and measurement of the test signal (T_x). The equivalent dose is then calculated from the normalised signals (L_x/T_x), by interpolating the L_n/T_n ratio on the growth curve formed with the L_x/T_x ratios (Figure 2.5).

The SAR protocol typically includes complementary tests to validate the results. These tests basically correspond with: i) a repeat of the very first regenerative dose which was given after the L_n and T_n measurement. The ratio between the normalised signal and that obtained after the first cycle (or "recycling ratio") has to be closed to unity (with a common tolerance of ±10%; Murray and Wintle, 2000). This shows that sensitivity changes were corrected using the test dose. A recycling ratio significantly different to unity means that for a similar dose the two signals are not the same: the aliquot is consequently discarded for the equivalent dose determination; ii) a luminescence measurement for a regenerative dose equal to zero. If the normalised signal is theoretically equal to zero, a weak signal is often induced by the transfer of electrons during the preheat process. This recuperated signal must be small compared to the natural one (with a tolerance of 5%; Murray and Wintle, 2000; Roberts, 2008), otherwise it means that this thermal transfer is too high and the aliquot has to be rejected; iii) a measurement of feldspar contamination, in case of quartz-OSL measurements. This test is important because feldspar are not only stimulated by infrared light, but also by the blue or green light used for quartz. Hence, the presence of feldspar contaminates the luminescence one wish to record from quartz. iv) a measurement of anomalous fading for feldspar. This test may be performed using a SAR protocol including variable delays between irradiation and measurement of the signal to estimate the fading to be estimated. Accurate ages are then obtained by inserting this fading in a correction model.

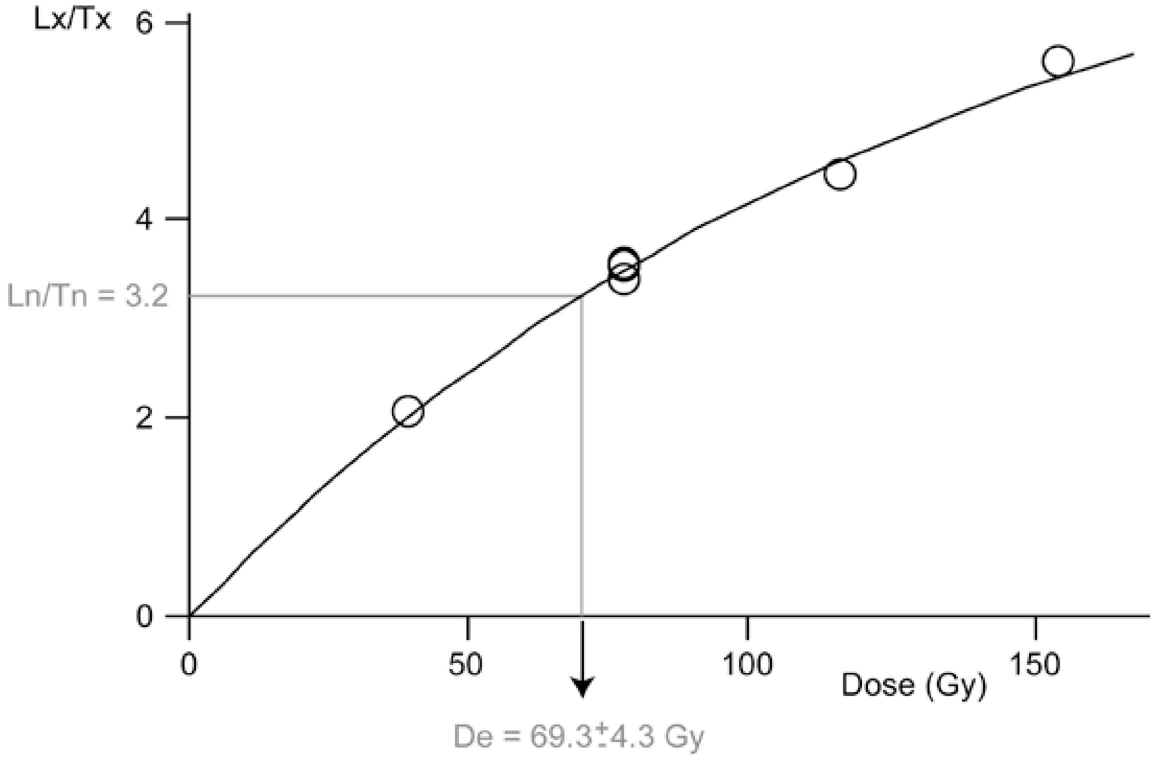

Fig. 2.5: determination of the equivalent dose using the SAR protocol.

-Determination of the equivalent dose (D_e) using statistical models

The SAR protocol creates as many equivalent doses as aliquots, with the exception of those which had to be discarded after the abovementioned tests. In case of heterogeneous bleaching (juxtaposition of well-bleached and unsufficiently bleached aliquots), the D_e distribution for coarse grains shows a scattering (with fine grains the number of grains will anyway be too high to avoid averaging): some aliquots can present a very high equivalent dose, which greatly overestimate the age of the last transport event. This explains why the mean or the weighted mean are not appropriate in estimating the accurate equivalent dose. Several alternatives using statistical models have been proposed to obtain a reliable De value, as confirmed by independent age control (Lauer et al., Geochronometria). The main models are the "Minimum Age Model" (Galbraith and Laslett, 1993), which makes it possible to identify the most bleached aliquots; the "Finite Mixture Model" (Galbraith and Green, 1990; Galbraith, 2005) when the observed distribution of equivalent doses present discrete populations; the use of the lowest 5% value as De estimate (Olley et al, 1998); the leading edge technique (Lepper and McKeever, 2002). As for the sampling strategy the choice of the model is allocated to the kind of sediments and comparison with independent age control is useful, as shown by H. Rodnight *et al*. (2006) for fluvial sediments. The relevance of these models increases with the number of aliquots. The number of 50 aliquots is sometimes considered as a minimal value to ensure a reliable equivalent dose determination (Rodnight, 2008), but it is important to keep in mind that the number of aliquots to be measured depends on the sample and increases with the scattering.

3. DATING OF SLOPE DEPOSITS: FORCING, SEDIMENTATION RATES, SEDIMENT BUDGETS

The aim of this section is to propose a review of recent research which used OSL dating (often combined with radiocarbon) to reconstruct the timing of past runoff erosion. The exposed case-studies typically focus on the last thousands years or tens of thousands years (e.g. from the Late Pleistocene to the Holocene), enabling past slope dynamics to be reconstructed and the forcings to be assessed.

3.1 From late pleistocene climate forcing

Several long-term geochronological studies of runoff erosion induced deposits have been performed in Eastern and Southern Africa, in a tropical climate. In South Africa, M.L. Clarke et al. (2003) analysed the colluvial infill of clay depressions eroded as badlands and overlain by Permian sandstones hills (area of Voordrag, Eastern South Africa). The slope deposits correspond to a thick sequence (up to 18 m) and include several palaeosoils levels. This demonstrates that morphogenic periods (leading to slope deposition) alternate with pedogenic periods. Intensive geochronological approach was developed, including nine IRSL dating of feldspars from slope deposits and eleven radiocarbon datings of the pedogenetized horizons. Numerical datings yielded age estimates spanning the last glacial-interglacial cycle, the oldest and youngest colluvia being deposited at the end of the MIS 5 (between 99±9 and 95±9 ka), and during the Lateglacial, respectively. Furthermore, the radiocarbon dating demonstrates that two main period for pedogenesis took place, at ca 45-40 ka and 23-17 ka, respectively. The radiocarbon ages are more or less consistent with the IRSL datings (independent age control). The comparison with the palaeoclimate records indicates that the soil formation took place under humid conditions (rainfall exceeding 600mm/yr), contemporaneously with rock weathering. In contrast the colluvial deposition is allocated to a decrease of the vegetation cover related to slightly drier conditions (less than 600 mm/yr) coupled with intense rainfall events enabling the transport of sediments. The Voordrag site hence represents a major place for unravelling the relations between climate change and runoff erosion in tropical conditions.

Late-Pleistocene slope deposits have also been studied in Tanzania, in the Irangi Hills area (Eriksson et al., 2000). This low mountain area is composed of altered Precambrian schists and gneiss. Slope are strongly dissected by large gullies (up to 20m) forming a badland landscape. Sheet and rill erosion are also very active, even if morphological evidences suggest that most of the sheet erosion occured before the formation of the badlands. Optical dating of sandy colluvia induced by sheet erosion demonstrated that their deposition took place during the Lateglacial (14500 to 11400 years ago). Like at Voordrag, runoff erosion is allocated to the development of less humid climate conditions, therefore reducing the vegetation cover on slope while episodic heavy rainfall are still possible. This arid period preceded a more humid climate period, recognized at a local scale (pedogenesis on top of the colluvia) and at a regional scale (palynological evidences). This more humid period led to the development of vegetation on the slope, reducing the effect of runoff erosion.

These case-studies are important since they point out that runoff erosion may predate human occupation of slope for agricultural purposes, highlighting the climate forcing on slope evolution. Similar results have been obtained in temperate areas, for example in Greece (Fuchs et al., 2004): OSL dating of a colluvial complex in the Phlious basin (see below) yielded Pleniglacial to Lateglacial ages (between 22±2 and 11±1 ka) for the oldest slope deposits. Following this, it is clear that a background of long-term climate induced runoff erosion can be recognized in both tropical and temperate areas. However, the situation becomes more complex during the last millennia, with the development of agriculture leading to an increased human influence on slope evolution.

3.2 To an increasing Holocene anthropogenic influence

Several reconstructions covering the Holocene period have been performed, in Africa and Europe especially. In Africa, research conducted by M.G. Eriksson et al. (2000 in the Irangi Hills demonstrated that the development of gullies in the Lateglacial colluvia (see above) took place during the last millenium. Despite no evidence (such as archaeology) was available to unravel the historical land-use evolution, the climate stability which characterizes the region during this period led the authors to conclude that the recent increase of runoff erosion was induced by anthropogenic forcing.

More research was carried out in Europe, both in Mediterranean areas and in Northwestern Europe (especially in Germany). Concerning the Mediterranean basin, a reference case-study has been recently proposed for the Phlious basin (NE Peloponnese, Greece; Fuchs et al., 2004; Fuchs and Wagner, 2005 ; Fuchs, 2007). This 25 km² basin corresponds with a depression filled by Neogene rocks (marls overlain by conglomerates) and Quaternary sediments. The marls are today exposed to active anthropogenic erosion with various processes especially including rill erosion and gullying. Reconstructing soil erosion history in this area thus corresponds with reconstructing the history of human occupation, as periods with no significant land use allow the development of vegetation, reducing runoff erosion.

OSL samples have been taken at the basis of a 250 m long slope in the northern part of the basin. Sedimentological observations (especially based on soil colour) suggest that several phases of slope deposition occurred. 28 samples from three profiles were dated using an adapted SAR protocol. The latter was applied to small quartz aliquots (less than 100 grains) in order to detect incomplete bleaching. Despite no radiocarbon-based independent age control was available (lack of suitable material), the dating results first reveal internal consistency of the dataset (stratigraphic order). This provides evidence not only for the accuracy of the results but also for the fact that the periods of colluviation allocated to runoff erosion were distinct in time. While the basis of the sediments can be allocated to the Upper Pleistocene (Upper Pleniglacial to Lateglacial, see above), most of the slope deposition occurred during the Holocene. They also allow the calculation of sedimentation rates (which can be basically defined as the inverse of the deposition time). Despite the precision of OSL dating (7% in mean) does not systematically allow a clear correlation with the archaeological evidences, the age estimates indicate that a significant period of colluvial formation (sedimentation rates >0.5mm/year) took place during the Neolithic (7^{th} millennium BC). This results were confirmed by combined OSL and radiocarbon dating performed in the fluvial sediments of the Asopos River, which indicate a high sediment input in the fluvial system draining the basin at the beginning of the 7th millenium BC (Fuchs and Wagner, 2005). Other main periods for runoff erosion on slopes are the second part of the Bronze Age (second millennium BC), the Roman period, and the last centuries, with sedimentation rates exceeding 1mm/year. In contrast, colluviation was less significant during the Early Bronze Age (5^{th} to second millennium BC) and the Early Iron Age (beginning of the first millennium BC). Comparison with human occupation clearly demonstrates that significant colluviation coincides with the main period of farming, suggesting that the anthropogenic forcing is more important than the climatic forcing in the study area, the latter being able to be considered as constant during the Holocene. The main exception concerns the Early Bronze Age which exhibits low colluviation rates despite significant occupation. This result are allocated by the authors either to a reduction of farming activity or to the development of amenities (such as terracing) in order to reduce soil erosion.

In contrast with this case-study which shows a more or less linear response of slope to human forcing, several studies focusing on colluvial sediments from Southern Germany show that more complex interactions could occur. In 1999, A.Lang and S.Hönscheidt analysed slope deposits associated with solifluction processes on loess and Triassic marls in the area of Stuttgart. They combined the use of radiocarbon and OSL dating with the study of archaeological remains. The OSL age estimates are consistent with the stratigraphy and range between the Younger Dryas and the Middle

Age. In contrast, the age estimates yielded by radiocarbon dating and archeology do not show any stratigraphical consistency. This apparent contradiction is explained by the fact that OSL provide age estimates for the last depositional event, while ages derived from organic matter and artifacts correspond with the first input of the material in the colluvial deposits. In other words, this indicates that the colluvial transport was not related to a single event, but to several events. This assumption was confirmed by the recognition of both anthropogenic ditches and natural sinks allocated to periglacial processes. This made it possible to provide a model reconstructing the main sediments transport events since the Early Neolithic. In particular, the model suggests that until the Iron times most of the deposition occurred in the upper part of the slopes. In contrast, colluviation in the lower slopes mainly took place during the last 3 millenia, with sedimentation rates ranging between 0.6 and 1.2mm/year.

Another example of a complex landscape response is given by the study of M. Fuchs et al., focusing on the 97 km² Aufsess River catchment (Bavaria). The soils consist of weathered Triassic rock mixed with sands and loess. 9 OSL and 2 radiocarbon datings were performed on colluvial sediments. The OSL datings were performed using the SAR protocol applied to small aliquots to ensure the detection of any incomplete bleaching. The age estimates are consistent between OSL and radiocarbon dating, and range between the Middle Holocene (5 ka BP) and the modern period. On the basis of this choronstratigraphical framework, the sedimentation rates show a first increasing at the end of the Neolithic period (0 to 0.2 mm/year). Regional archaeological and morphological evidences (Leopold et al., 2007) suggest this increase of slope deposition is likely to be related to the development of farming. Another strong increase of sedimentation rate for colluvial deposits is recorded during the last 500 years (sedimentation rates ranging between 1 and 2 mm/year).

The results obtained in the Mediterranean basin as well as in Germany can be extrapolated to many areas: the Neolithic Revolution and, a few thousand years later, the Roman colonisation have actually been recognized as two major periods of land clearing leading to slope instability in several European regions. These two periods have for example been recognized in the Luxembourgian Moselle valley (Naton et al., 2009): in the area North of the famous village of Schengen, the lower terrasse (+3m) sediments are overlain by 50 to 80 cm of colluvia originated from the marly bedrock (Fig 3.7). The slope deposits are divided in a basal dark unit, and a top light-colour unit. Many archeological evidences have been found in the basal deposits, ranging from the neolithic period (linear pottery culture village of the Early Neolithic) to the Iron age (storehouses, grain silos) and the Roman times (villa). The deposition of such dark colluvia is consequently related to human induced soil erosion spanning the last millennia BC. In contrast, the upper light-colour deposits overlain the archaeological structures : following this we assume these upper deposits to be allocated to land clearing performed by the Roman, in particular for wine growing: the presence of vineyard during Roman times in this area (as it is still the case nowadays) is actually confirmed by ornamentation on a monument built during the second century AD.

These case studies hence demonstrate the potential of geochronological approach to reconstruct the timing and importance of anthropogenic forcing in slope evolution. One should regreat that research focusing on historical slope evolution obviously remains sparse, especially in comparison with those focusing on fluvial systems. However, the latter are able to provide indirect reconstruction of slope dynamics: runoff erosion actually leads to a sediment transfer not only on the slopes, but also towards the streams and rivers which can hence be considered as useful witnesses of runoff erosion.

3.3 From the slopes to the fluvial systems

In several areas the research took both the slopes and the floodplain into consideration, allowing the sediment transfer in the catchment to be reconstructed with a high resolution.

This is especially the case in the Aufsess River catchment (Bavaria), the analyse of slope deposits was coupled with morphological investigations in the floodplain, and 64 OSL and 12 radiocarbon dating of fluvial sediments (and intercalated organic matter). The comparison with the results for colluvions demonstrate that the fluvial response was delayed and started only 3 ka ago, i.e. ±2 ka after the first record of slope deposition. As for the abovementioned study of Lang and Hönscheidt (1999), this time lag is assumed to reflect temporary storage on the slopes. Similarly a strong increase of sedimentation rate for colluvial deposits is recorded during the last 500 years (sedimentation rates ranging between 1 and 2mm/year), while strong fluvial deposition took place in the fluvial system during the Middle age. This fluvial deposition is assumed to result for intense human-induced (forest clearance) soil erosion on the slopes. The lack of colluvial record for this event is explained by a direct transfer of the sediments towards the floodplain, without storage on the slopes. These cases study hence demonstrate how useful it is to take into consideration not only the slope deposits but also the derived fluvial sediments, to get a full understanding of the erosional processes in a given catchment. In a similar way, the results obtained for the Luxembourgian Moselle (see above) have been confirmed by the study of an oxbow-lake filling in the Roman city of Trier (SW Germany), located ca 40 km downstream of Schengen: the study shows that the fluvial sedimentation rates strongly increased during the Roman times, from 0.14 to 2.6 mm/year (Zolitschka and Löhr, 1999).

In contrast with these cases, many research only used the fluvial deposits to reconstruct the past slope evolution. A first example is provided by the historical terrace of the Alfeios which buries the antic city of Olympie (Greece; Dufaure, 1976; Dufaure and Fouache, 1988 cf Neboit): this lower terrace was dated to the 8th century AD on the basis of archaeological and historical data. The formation of this terrace obviously reflects strong runoff erosion processes in the Alfeios catchment. This erosive crisis is allocated to a major upheaval related to the Slav invasions, leading to the abandonment of the city of Olympie and to the development of a pastoral system which generates a strong pressure on slopes without protecting amenities. Another example concerning the Northwestern Europe is provided by the study of the small valleys bottom filling in the Sarthe catchment (NW France; Larue, 2002): the valleys are incised in plateaus formed in Cretaceous sands, marls and limestones. Despite the slopes are only slightly inclined (3 to 10%), they are affected by intense runoff erosion which includes both rill erosion and gullying. The floodplain sediments show an alternation of slope deposits (sands and marls, mainly originating from the bottom of the slopes) and peat layers. This made it possible to demonstrate that several periods of runoff erosion (transfer of sediments to the valley floor) took place. These periods were separated by calm periods associated with the formation of peat. The radiocarbon dating of these peat yielded age allocated to the Early Bronze Age, to the end of the Iron Age, and to the Late Mediaeval to Modern periods. In contrast the main phases of runoff erosion (leading to sedimentation rates of a few mm/year) are related to the Neolithic, Iron Age, Roman to Early Mediaeval and post-modern periods. The combination of this chronology with the sediment grain size and pollen composition makes it possible to demonstrate that the Neolithic colluviation was related to a climate deterioration. Assuming that the human pressure remains more or less constant during the last millenia, the following erosional periods are, in contrast, allocated to a combination between the climate degradations (e.g. the Little Ice Age) and human clearing ("climato-anthropic dynamics"). A similar reconstruction can be proposed a a larger scale, as demonstrated by the research of G.J. Vis et al. (2010) in the Tagus valley (Spain and Portugal). The authors focused on the floodplain sediment. They combined a sedimentological approach (grain size, magnetic susceptibility) and radiocarbon dating of organic matter to show that fluvial sedimentation rates increased at the end of the Iron age (2500-2300 years BP) and mainly during the mediaeval period. Due to the lack of evidence for any significant climate change at that time, the authors concludes that the change in fluvial sedimentation are directly related to human induced runoff erosion in the Tagus catchment. These results can be compared with those previously obtained for the Ebro River, in Northern Spain (Van Zuidam, 1975, in Neboit, 1999): in this area, the presence of archaeological remains in the lower terrace sediments indicates that a major period of sedimentation took place between the Iron Age and the Roman times. In contrast, no major sedimentation was recorded for the last millennia, and the Early Mediaeval artifacts have been preserved close to the modern surface.

The potential obtention of high resolution chronological framework for slope and floodplain deposits is also of great importance to allow sediment budgets to be estimated, for a given timescale (Brown et al., 2010). The reconstructions typically focus on the Holocene and include not only the colluvial sediments, but also the fluvial deposits included in a given catchment (e.g. Brown et al., 2009; Fuchs et al., 2010). The assessment of alluvial sediment volume can be easily achieved in both small first order catchments (less than 10km^2) and larger catchments, by calculating the width of the floodplain and the sediment thickness (measured using cores) at regular places along the river (Brown et al., 2009). The relations between slope and fluvial dynamics are however more difficult to be assessed, especially in large basins, because of the presence of several sedimentary units showing different functioning. Furthermore, the quantification of the sediments eroded from the slopes is complicated by the fact that the distinction between the in-situ horizons and the colluvial deposits is not systematically obvious. A possibility to achieve it is to get a typical soil profile which is representative for the catchment or subcatchment (assuming this area to have more or less homogeneous characteristics). This reference profile should be coupled with high-resolution field studies (opening of trenches or coring, topographic measurements) and modelling to allow soil truncation as well as accumulation layers be recognized and quantified using GIS. This procedure was applied for the Holocene evolution of the Aufsess catchment (Fuchs et al., 2010). The quantification indicates that 38 millions of m^3 of sediments were eroded from the slopes in the whole Aufsess catchment. 58% of these sediments were stored as colluviums, while 42% reached the alluvial floodplain (hillslope sediment delivery ratio). The volume of Holocene sediments stored in the floodplain represents 9% of the eroded sediments. This led the authors to conclude that the proportion of eroded sediments which were carried out of the catchment (sediment delivery ratio SDR; Asselman, 2003) is of one third (42-9=33%).

3.4 Anthropogenic versus climate forcing?

As demonstrated above, fluvial systems can be very useful to assess the intensity of runoff erosion. However, it is essential to keep in mind that the fluvial record is not directly linked to processes presents on slopes: first, a time lag may exist between the runoff erosion and the sedimentation in the floodplain. Secondly, fluvial systems may also be influenced by internal forcing (tectonic events) or by minor climate change, which in contrast may have no influence on runoff processes. More generally, the understanding of slope response to climate or anthropogenic forcing during the last millenia remains a complex matter, for two main reasons:

-on the one hand, runoff erosion is often the result of a combination between human influence and climate, and the distinction between respective influence is difficult.

-this is all the more the case as anthropogenic forcing is difficult to assess: human may actually either increase natural erosion (e.g. through land clearing) or reduce it through amenities. Furthermore, an increased runoff erosion in a given area can no be constantly related to an increased human settlement (development of farming): the example of the Alfeios River in Greece (see above) demonstrates that similar evolution can be allocated to the abandonment of the area leading to a deterioration of amenities built to stabilize the slopes.

The recognition of controlling factor may hence be difficult, as illustrated by the research of B.Devillers (2003) on the Gialias River catchment (Central Cyprus): even if some correlations may be proposed (soil formation associated with a reduction of anthropogenic pressure at ca 800 BC and 900 AD, fluvial aggradation contemporaneous with significant human presence after 1000 AD), no evidence allows to confirm this hypotheses, and the relations between (fluvial) dynamics, anthropisation and climate remains unclear. Another example concerning the Mediterranean basin can be found in the Pouilles region (SE Italy): in the area of Metaponte, the coastal streams are deeply incised in their floodplain, which can be considered as a terrace. The thickness of the sediment typically ranges between 10 and 20m. The sediments (mainly sands, silts and clays) include organic matter and many archaeological remains. The derived chronological framework demonstrates that important fluvial sedimentation took place during the second part of the Holocene, and in particular during third millennium BC and during the Antiquity (Greek and Roman times). Complementary research in the neighbouring Sicilia indicate that a a regional incision took place during the Bronze Age (second millennium BC). The regional chronology obviously suggests a relation between the fluvial sedimentation and the human occupation (anthropogenic forcing). However, such a correlation does not allow the Bronze Age incision to be explained: no evidence actually indicates that this period corresponds with a reduction of human pressure on the slopes.

This demonstrates that the slope response to either climate or human forcing is much more complex than expected. This may first be explained by the fact that (especially when focusing on the beginning of human occupation) there is a high uncertainty concerning the real land use and its secular evolutions: geochronology and archaeology actually only provide information of local and chronologically contrained significance, and much more data is required to obtained regional high resolution reconstruction. Research conducted in the French Alps (Leveau, 2007) actually point out the necessity to avoid any generalisation, e.g. by systematically associating the Roman period with an increased runoff erosion.

Secondly, the slope evolution since the development of farming can in most cases obviously not be explained only by climate or by human forcings. For example, the recent evolution of the southern Italian rivers should neither be allocated only to climate (as was the case in the pionner research of C.Vita-Finzi, 1969) nor only to human (as suggested by the difficulty to explain the Bronze Age regional incision). It is very likely that this evolution (eg the role of runoff erosion) results from a combination of both forcings. With the development of farming, human societies actually were able to replace the climate in changing the état surface on slopes: this state is actually mainly controlled by the land-use, rather than by the climate. Following this, the effects of climate can be increased, as heavy rainfall may take place on unprotected soils which might have been covered by vegetation without human intervention.

4. CONCLUSION

The present chapter aimed to demonstrate that despite their limits, radiocarbon and Optically Stimulated Luminescence OSL were useful tool for recontructing colluviation periods which took place during the Holocene and historical periods, in association with runoff erosion. The chronological framework derived from the application of these methods (coupled where possible together and with archaeological evidences) allow a better understanding of the relation between sediment formation and human or climate forcing. The increasing of human influence during the last millennia was in particular well recognized in various areas (Africa, Mediterranean basin, Western Europe). However, it is important to avoid any determinism: even if there is a clear increasing of human impact since the Neolithic (and especially since the Roman times), slope response is often complex and should be considered as resulting from a combination of forcings. Following this, the numerical ages obtained with OSL, radiocarbon or archaeology should not be used to allocate the slope dynamics to a given forcing. They only aim to contribute to answer the following question: “in which part does the human land-use modifies (and in particular amplify) the climate effect of climate on the slope evolution?”.

The research carried out by M. Fuchs demonstrates that a better understanding can be provided by taking into consideration both the colluvial and the fluvial dynamics (radiocarbon and OSL being also useful for the latter). Even if fluvial systems may be influenced by other forcings (eg tectonics), a study considering both slopes and floodplain allow sediment budget to be estimated. Such budgets may subsequently be used for the modelling of sediment transfer associated to runoff erosion.

CHAPTER 3

MEASURING PRESENT RUNOFF EROSION

By

Tomás de Figueiredo

1. INTRODUCTION

Soil erosion by water or, with a narrower focus, runoff erosion, address natural phenomena that integrate and combine in a sometimes rather complex way several physical processes, involving energy and mass transfer from the atmosphere to surface ground and along the actual gradients over ground surface (Chapter 1). This complexity, in turn, is reflected in the assessment complexity and on that of the interpretation of assessment results. In order to mitigate such difficulties and constraints, it is important to clearly identify the object under assessment. However, this is not a simple matter and some examples may be called to illustrate it.

As found by Rodríguez-Blanco et al. (2010), after measuring for 3 years stream discharge and suspended sediment in a NW Spain small catchment, inferences from measurements could be done on spatial distribution of erosion within the catchment; however, results obtained by these authors do not provide information on rainfall event rates, nor in long term rates. On the contrary, results obtained by Tomás (1997) do not allow comparable conclusions as processes observed do not include linear erosion and no inference can be derived for the catchment scale. Nevertheless, Tomás (1997) treated a 30 years record of soil loss from erosion plots under a traditional wheat-fallow crop rotation in S Portugal, providing very reliable long term soil loss rates in agricultural fields, either event or annually based. From the work reported by Vandekerckhove et al. (1998), Vandekerckhove et al. (2000) and Figueiredo (2009), no average soil loss can be computed for cereal fields in Bragança area, NE Portugal, because they actually measured gully volumes following extensive incision after one heavy rainfall period; single extreme event data; inference on gullying risk and associated soil loss rates are, however, possible.

Time and spatial scales very much affect erosional processes occurrence and rates. In fact, erosion is a spatially distributed phenomenon, involving sediment transfer along hill-slopes and from slope plans to the linear structures of the natural drainage network. As well, erosion is a time discontinuous, episodic, phenomenon, following the lack in time continuity of the erosive agent, the rainfall. Moreover, erosion phenomena combine continuous and threshold type of mechanisms, meaning that erosional responses may vary from nil to very high magnitude. This range is determined by changes in erosion processes contribution to total loss, from splash everywhere in a field or catchment in virtually all rainfalls (negligible to low loss), to interrill erosion in some events with Hortonian overland flow generation (low to severe loss), to gully incision in very few heavy and prolonged rainfall episodes, with topographically concentrated overland flow running over saturated soil (severe to very severe loss).

Assessing runoff erosion encompasses these wide ranges of processes time and space occurrence and continuity. Therefore, no such thing as a normalized erosion measurement methodology exists, and a wide set of methods historically developed, according to research needs and progresses in specific technology and instrumentation. The principles to be followed when assessing erosion are:

- to set the basic time and space measurement unit according to information requirements and practical feasibility;
- to accept the black box approach for those units and ensure that it is experimentally respected (known inputs and outputs through system boundaries);
- to withdraw any wish to extrapolate measurement results to other time spans and spatial scales.

Modern geology approaches deduced denudation rates from sedimentological studies, although classical geomorphology used on-site soil properties to assess erosion status and trends (e. g. Ahnert, 1998). Erosion measurements effectively started when the phenomenon was extensively perceived as a soil degradation problem (Dust Bowl, in the early thirties of the XXth century, Hudson, 1981). Sediment washed from runoff plots and suspended sediment load in rivers were the two main assessment methodologies applied. The use of soil properties, either observed in the field or determined indoor, to derive soil degradation conditions due to runoff erosion widespread when surveys in remote areas were extensively performed, as it was the case of conservation projects in developing countries (Hudson, 1981; Morgan, 2005). Rainfall simulators developed as more information was needed to deeply understand erosion processes but also to allow vast trial replication, far more expensive and virtually not feasible if done with runoff plots. Tracing methods are still the less commonly applied methods for erosion rates measurement due to instrumental sophistication. GIS-based mapping and erosion models, independent or coupled, are increasingly used tools for mapping, predicting or simulating future scenarios of erosion conditions and the corresponding rates. However, these are out of the range of this text, as they will be treated separately and in detail in Chapter 5.

General classification of methods to assess runoff erosion requires definition of the consistently applicable criteria, coupled with the practical goal of reaching easy application. The proposed categories are designed to meet those requirements. For each section the layout is: comprehensive list of methods or types of methods and for each of these, scope, description, discussion with application conditions, pros and cons.

2. FIELD SURVEYS

Field surveys are currently performed to obtain a direct insight on actual erosion processes occurring and their spatial distribution within the survey area, to derive estimates of past or actual rates and trends of soil loss, to assess and eventually classify land degradation status (Hudson, 1981; Morgfan, 2005). Field surveys may be also performed under specific conditions, as those following an extreme and evidently damaging rainfall erosive event, providing therefore a timed picture (Vandekerckhove et al., 1998; Figueiredo, 2009). Furthermore, they may be performed in sample areas as part of a cartographic approach to wide regions where erosion or land degradation has to be assessed, this way, contributing to draw the regional picture of the problem or status (LADA, 2009).

Field surveys are basically an organized way to collect the necessary information for the survey purpose, which, normally, cannot be obtained with similar detail and quality by other means, as existing map or remote sensing techniques. As so, it is important to clearly define the specific objectives of the survey for the area selected for observations, which may be more geomorphic process-oriented or degradation assessment-oriented. Also, as costly, time and labour consuming operation, surveys should be carefully prepared, assembling prior to field work, all information needed to draw an anticipated picture of the area and limit the information that might be redundantly collected there. Scheduling of the survey is a matter of operation management.

A field survey comprises visual observation of erosion features and signs in the field as, for instance, runoff flow paths, crusts, surface rock fragments, rills and gullies. Observations are appropriately recorded together with notes on impressions or perceptions built on-site, that may provide interpretations helpful to data treatment. Some involve extensive data collection, others are much simpler; some add field measurements of soil properties or erosion features, others simple record their occurrence. Data collection has to be geo-referenced either locating sites in a sufficiently detailed map or by means of a GPS device; forms might be prepared in hardcopies or in a portable electronic device as a PDA, tablet or laptop (Eicher, 2005; Fleskens and Strosnijder, 2007; LADA, 2009; Zanden, 2011).

Examples of recording sheets are found, for instance in Zanden (2011), and in LADA (2009). Hurni (1990) published a form specifically focused on agricultural land. Other less oriented forms are provided in Goudie et al.cols. (1990).

Results obtained with this approach are limited in quantitative terms, although in cases information collected might provided the basis for quantifications on soil loss rates or amounts. On the contrary, these methods are very valuable to identify active processes and their spatial distribution, as well as damage determined by those processes during erosive events, on-site or off-site. They are also helpful for locally calibrating models or indirect quantitative assessments via soil properties and ground features.

3. FIELD MEASUREMENTS

Research methods to quantify erosion rates in the field, under actual conditions, evolved through time towards a short number of types, specific to processes measured, which may be setup with more or less instrumental sophistication. Besides, according to measured object, they can be grouped as: (i) methods materially measuring soil export from a known area by runoff erosion (splash collectors, sediment traps, runoff plots); (ii) methods measuring topographic changes associated to runoff erosion (erosion pins, benchmark-based micro-relief surveys, rill and gully volumes); (iii) methods assessing particle transfer fluxes by runoff erosion (tracers of several kinds). Indicated methods are described in the following sub-sections.

3.1. Splash measuring devices

Splash is measured with cups or boards (Fig. 3.1). Splash cups may be of two types: source-cups (soil containers) and sink-cups (splash collectors) (Morgan, 2005). In the first case, a container (metal, PVC or glass cylinder), filled with the testing soil, is exposed to rains and the mass loss due to splash removal of soil particles is measured after each rainfall or a period of precipitation. Early studies of the erosive characteristics of rainfalls, by Ellison cited by Hudson (1981), applied this method with sand as test material.

Fig. 3.1 : Splash cup, 5.5 cm diameter installed at Lamas de Podence, Macedo de Cavaleiros (left), splash boards installed in a outdoor micro-plot experiment with simulated rock fragments, installed in Bragança (right).

The second type is more commonly used and consists of a PVC cylinder, inserted into the soil with its top rim at a similar slope gradient as that of the surrounding ground, that should outcrop a few millimeters to avoid incoming runoff washed particles (Poesen, 1986,; Morgan, 2005; Fonseca, 2005) (Figure 3.1). Inside the cup, a filter paper traps incoming particles splashed from the surrounding area. Drainage of precipitation water has to be ensured, and can be obtained with a hole deeper than the cup itself. After each rainfall or group of rainfalls, the filter paper cone is replaced by a clean one and the mass of trapped particles is measured in lab.

Splash boards, are rectangular plates (metal, plastic), placed vertically besides the area under assessment, that receives particles splashed and collects them in a container at its foot (Morgan, 2005) (Fig. 3.1). Mass of these particles is measured in lab. Splash boards have problems in windy areas and their height cannot be shortened because trapping efficiency declines (Figueiredo and Poesen, 1998).

The mass loss from soil containers or the mass of particles trapped in cups or by board requires corrections before being taken as splash loss. Correction procedures to account for the size of cups are provided by Poesen and Torri (1988). As well, Figueiredo et al. (2004) adds correction procedures to account for the presence of rock fragments in soil surface. Furthermore, splash measurements are sometimes taken as a surrogate of soil detachability; however, in sloping areas, there is an actual net splash transport downhill which can be assessed with a model developed by Poesen (1986).

3.2. Sediment traps

Sediment traps collect soil particles washed by runoff along hill-slopes (Dunne, 1977). The most common model is named Gerlach trough and consists on a buried metallic box with a gutter upslope, to drive in washed material, and a exit pipe to drain the box of excess runoff water (Morgan, 2005) (Fig. 4.2). Gerlach trough is covered to avoid collecting precipitation and normally has an internal grid to separate large (organic) washed material from sediment (Coelho et al., 1990). This is deposited in the box bed, or in a removable container, which is pull-out ad cleaned in every sediment collection, therefore facilitating this operation. Sediment trapped is oven-dried and weighted and referred to runoff contributing area to express results in soil loss mass per unit area.

The system is commonly installed down-slope unbounded areas, where the contributing drainage area is fairly defined by natural ground topography (Coelho et al., 1990). However, it can be applied for bounded areas, therefore becoming a special case of erosion plots. Nevertheless, they are designed to collect runoff but simply washed soil particles, unless additional or alternative design elements allow runoff storage or sampling, as it is the case of bottles to where runoff water and suspended sediment are conveyed through a pipe, after reaching a certain level inside the box (Fig. 3.2).

Fig. 3.2: Gerlach trough: sketch (left) and an example installed in the field (right).

A series of stakes inserted into soil normal to surface, along contour lines, holding a plastic sheet adequately fixed into soil, can trap washed particles by upstream runoff (Dunne, 1977). As water is diverted laterally or surpasses the trap, sediment accumulates upslope and. Erosion rate estimates may be based on measurement of the deposited oven-dry mass removed in each collection operation, or on deposition volume changes, assessed via changes in relative height of deposition surface during the observation period.

3.3. Runoff plots

Runoff or erosion plots are the most commonly applied method to measure runoff and soil loss from areas affected by interril and, eventually, also rill erosion (Hudson, 1981; Roose, 1994; Lal; 1994; Morgan, 2005). They are bounded areas with size, shape and boundaries adequate to purpose and means available to perform measurements, which correspond to runoff water and washed soil collected in devices installed down-slope, normally preceded by a gutter on their front edge. Such devices vary in instrumental or structural sophistication depending also on means and purpose of measurements, and have to account on plot size and characteristics, that determine potential runoff and sediment yields (Figs. 3.3 to 3.6).

Plot boundaries may be earth ridges, wood or metal plates inserted in soil (both repaired every time leaks are detected), or concrete walls, in the case of well designed and funded long term monitoring programmes (Fleskens and Stroosnider, 2007; Fonseca, 2005, Figueiredo et al., 2011; Figueiredo and Ferreira, 1993, Tomás, 1997, respectively) (Fig. 3.3 to 3.6). Plots may vary in shape but are normally rectangular, although microplots installed in semi-arid matorral by Bochet et al. (1998) had a shape fitting shrub plant canopy ground projection. Plot size varies sharply, indicative surface area and slope length being: (i) micro-plots, smaller than 5m2 (less than 4-5m long); (ii) meso-plots, from 10 to 500m2 (5 to 50m long); (iii) macro-plots, larger than 500m2 (longer than 30m). Plot size has to be adjusted according to the local land use and management practices being monitored for assessing their erosional response. For example, when mechanized operations are part of the crop management system being tested, plots must be large enough to allow in a tractor for carrying out such operations, otherwise they have to be manually performed. Micro-plots can be installed in an heterogeneous land cover area, if they are intended to assess the independent erosional response of land cover patches; however, large plots have to be installed in the same area if assessment is focused on the global response is that results from the connectivity of such patches.

Fig. 3.3: Erosion micro-plots (1 m wide): in a young forest plantation (3m long, Lamas de Podence, Macedo de Cavaleiros), in a burnt shrubby area (4m long, Aveleda, Bragança).

It is necessary to stress that measured soil loss in each case should be the outcome of actually active erosion processes occurring in plot areas, even though, in some cases this is not a simple design problem, because of temporal changes in vegetation cover and ground condition, and because of the episodic pattern of precipitation and the threshold pattern of runoff and erosion processes. In very large plots, most of the recorded soil loss events do not outcome from the entire plot area, whereas small plots cannot represent the most significant erosion events, issued from a runoff concentration virtually impossible to occur in such small size.

Runoff and sediment collection devices can be as simple as tanks connected to the upstream gutter, installed at plot front edge, by means of a conveyor, in which total washed material (soil and water) yielded in an erosive event is stored. Gutter may be concrete, PVC or metal half-pipe or a metal plate onto the ground. Conveyors, shorter or longer, may be part of the gutters (concrete) or independently installed (half-pipe, rigid or flexible pipe). Both elements are selected according to purposes and conditions of the monitoring system installed, but, in any case, they have to ensure fast conveyance and no loss of material from plot to tank (Fig. 3.5 to 3.5).

The system may apply from small plastic tanks or bottles (5 or 10l, for example), to large plastic or metal tanks (50, 100 or 200l, for example), or even trenches open in the soil at plot front edge, covered with a plastic sheet, that also collect rainfall (Fig. 3.4 and 3.5). The collection procedure normally comprises: (i) water volume measurement (or estimate from a calibration height-volume curve for each container); (ii) suspended sediment sampling, e.g. with a beaker, after thorough stirring of tank contents, and sample oven-dry mass determination; (iii) if present, removal of bed load after draining most runoff water, oven-dry mass determination of total bed load or, when too large, wet-weighing of total bed load and sampling it for for water content. Total soil loss in each event is the sum of suspended sediment oven-dry mass (the product of runoff volume by sediment concentration) with bed load oven-dry mass (if present), referred to plot area. It should be noted that, sediment resting on gutter, actually exported from plot but not able to reach the container, is accounted for as bed load. Runoff may be also converted to equivalent height (volume collected divided by plot area, expressed in mm).

In large plots, where large runoff volumes can be expected, the problem of storage volume of tanks is overcome by setting small tanks in series (not more than 3 in total), connected with pipes and runoff divisors. In this case, the first tank collects water and the soil washed from plots, but the remainders receive only a part (e.g. 1/11) of the incoming clear runoff water, the rest being spilled out (Tomás, 1997; Figueiredo, 2001) (Fig. 3.4).

Fig. 3.4: Erosion meso-plots in an olive grove, with earthen boundaries and open runoff and sediment collecting pond (Valbom dos Figos, Mirandela).

Fig. 3.5: Erosion meso-plots in a vineyard (Quinta de Santa Bárbara experimental station, Pinhão, Dourio region) (photo by Jean Poesen).

Recording systems allow a detailed insight on runoff water and soil loss evolution during an erosive event and this is especially useful when studying erosion processes or modeling. Recording devices for measuring runoff are connected to a data-logger, from which the data can be directly or remotely retrieved or accessed. They may be: (i) inspired in limnigraphs (river stage recorders), in which a sensor (e.g. a pressure transducer) measures runoff water level conveyed through a stable section in the flow circuit (channel segment or a spillway, metal, plastic or concrete); (ii) inspired in rain gauges, in which a runoff convey pipe drips on a tipping-bucket device, or a multiple bucket tilted metal wheel, that rotates when a bucket is filling in with runoff (Morgan, 2005) (Fig. 3.6).

Fig. 3.6: Runoff tipping-bucket recorder (Haute Normandie, France).

Runoff recorders may be coupled with mentioned above systems, so as tanks collect total sediment exported during the erosive event, while runoff evolution is is recorded in detail. However, sediment concentration evolution may also be monitored installing sampling bottles filled in during the erosive event water diversion scheme, collected afterwards and oven-dried for sediment mass determination. More sophisticated systems use turbidity sensors to record sediment concentration evolution during runoff event. Both allow coupled drawing and analysis of sedigrphs and hydrographs for each event recorded.

3.4. Ground level monitoring

Measuring height differences relative to reference surfaces or points, and repeat them through time, provides estimates of soil loss amounts and rates, associated with erosion processes taking place in a given observation area. Methods for measuring runoff erosion considered under this heading include erosion pins, benchmarks for micro-relief surveys and root exposure surveys (Dunne, 1977).

An erosion pin comprises a metal rings resting over the soil surface, fixed there by a nail. Distance from nail top to ring is repeatedly measured through time. Pedestals may form as the ring protects the soil underneath from raindrop impact, and they have to be removed prior to measurement. Nails are taken as local benchmarks and changes in ring position are attributed to soil loss due to runoff erosion. A large number of pins is required for a fair representation of surface evolution, because wash and deposition occur in the same area of observation as sediment is transferred along the hill-slope, and because preferable flow paths and ridges have clearly different pattern of surface evolution.

Benchmarks of any kind, installed, as concrete pillars, or naturally set in place, as embedded rocks, serve as references for micro-elevation surveys f neighbouring areas affected by runoff erosion. Repeated surveys help assessing wash rates from eroded volumes estimate. Tree root exposure is an indicator of land degradation by severe runoff erosion. Assessment of eroded volumes requires micro-elevation surveys as in the above indicated cases, however with increased practical difficulties. A particular case of benchmark based measurements is the erosion bridge developed by Shakesby et al. (2002). The device is perforated metal bridges kept leveled during measurements, as its 2 supporting fixed length legs step over 2 benchmarks. The surface soil profile between the 2 legs (or rods) is obtained measuring vertical distances to ground surface taken from bridge with c. 50 metal sticks sliding through the bridge holes down to the soil surface. Time changes in average elevation are estimates of erosion rates of soil between benchmarks.

Removal of soil particles by runoff erosion in interrill areas also lowers ground surface, actually corresponding to the geomorphological concept of denudation rate. Due to ground surface roughness it is hardly possible to describe it as sheet removal, but such approach has been adopted for long. However, it is a useful approach to assess local erosion rates, provided extrapolations are performed very carefully. In fact, the heterogeneity of surface ground features makes them simply point assessments that may not represent larger land tracts. If land lowering rates are converted in equivalent soil loss rates (mass per unit area and time), then a critical conversion factor has to be estimated – soil bulk density – that is normally taken as equal to the actual one, locally measured, assessed or assumed. Anyhow, errors in measurement or estimate have much larger consequences for results in the height differences than in bulk density.

3.5 Gully erosion assessment

Linear erosion features primarily result from the incision of ground surface, and further scouring of the soil profile, by concentrated overland flow, to which might be added effects of other processes as splash, sheet flow and mass movements at micro to meso-scale (Govers and Poesen, 1988). They are normally classified according to size and stability, rills referring to short-living structures lower than 900 cm2 cross-section area (1 ft2) (Morgan, 2005). Large stable incisions are called gullies, in spite that these are labeled ephemeral when they meet size requirements but are fresh incisions that may be erased by regular tillage operations as it is the case of rills.

The basic principle in gully erosion assessment is to estimate gully volume and refer it the estimated catchment area draining to the gully system (Vandekerckhove et al., 1998; Vandekerckhove et al. 2000). This is based on the assumptions that actual gully volume is simply due to linear erosion) meaning that sedimentation and other processes occurring in gully walls are not considered), and that topographically defined catchment area contributed with erosive overland flow to gully incision and development. For most cases, these assumptions are practically acceptable, taking into account either the constraints associated with assessing the contribution of process other than concentrated overland flow to gully actual configuration, and with actual runoff contributing area determination, or the accuracy of estimation procedures applied to outcome gully volume.

Approaches to estimate gully volume may be based in: (i) direct measurements (ruler based or geodetical); (ii) remote sensing techniques (low and high altitude aerial photos) (Vandekerckhove et al. 1998, Figueiredo, 2009; Dunne, 1977; Vandaele et al., 1997, respectively).

In the first case, at selected points along the gully measures are taken to estimate the respective cross-section area, which are integrated over the length of the gully segment they represent to output gully segment volume, the sum being total gully volume. Number of sample sections, measurements performed to estimate cross-section area and integration procedures, depend on gully size on one hand, and required accuracy of estimates on the other hand. In fact, very large gullies (several meters deep and several hundreds of meters long) require geodetically performed measurements. Smaller gullies may be approached with direct measurements with a ruler for gully cross-sections and and a tape for gully segment length (Fig. 3.7). Area of complex cross-sections may be accurately assessed with a needle profilemeter but it is a rather time consuming technique. A much simpler approach consists in assuming a certain regular cross-section shape (triangular, rectangular, trapezoidal, and parabolic) and performing the measurements required determining its area (normally top width and average depth). For a more accurate integration, sections should be selected so as to define gully segments with regularly changing cross-section area, therefore avoiding abrupt changes in size and shape within the segment length.

Remote sensing techniques for gully erosion assessment are applied when the scale of assessment, the extent of the area under assessment, and the expected or required detail of assessment results. Accordingly, the range in resolution of the aerial photos used in this approach is quite large, depending on purposes and practical conditions to perform the assessment. These include available time, consistency and quality of the available information (photos), quality of ground references for photo-interpretation. Ries & Marzoff (2003) used photos taken from a blimp to study gullies in Spain, whereas Vandaele et al. (1997) made use of aerial photos in Alentejo, Portugal.

Gully erosion rates can be calculated if a temporal reference exists. Normally this means repeated observations through time, in a monitoring scheme. However, in fully installed permanent structures as large gullies dating techniques may be used to obtain a temporal reference for calculating rates. This is the case of dendro-chronology of plant roots exposed in gully walls. It should be noted that rates calculated are averages that do not, and could never be, representing the actual changes in process rates over time, as erosion sensu lato is an episodical phenomenon.

Fig. 3.7: Measuring a rilled forest road (Lamas de Podence).

3.6. Tracers

Methods presented here-above are based in the measurement of an amount of soil loss (either the actually exported or the one that was exported from the incised areas where it is not anymore). Rates are then computed referring those amounts to the source area and to time interval where and when the erosive agent acted. However, a different approach may be taken to assess runoff erosion rates and this is to assess directly the soil flux downhill monitoring the concentration of tracers along the slope, through time.

Tracers are supposed to have ability for displacement similar to that of soil particles being eroded. Iron sawdust can be placed in contour bands onto the ground at regular distances down-slope and displacement of such particles monitored measuring electromagnetic activity along the slope, at time intervals, so as to have a picture of the redistribution of the tracer and relate it to erosive agent. The lack of a reliable relationship between soil and tracer displacement (due to different particle density, for instance) is an important limitation of this method, a problem that is overcome when tillage erosion is assessed (Borselli and Torri, 2001).

Cesium 137 (137 Cs) is an artificial radioactive isotope that is used as tracer for erosion and sedimentation studies (Quinton, 1994). It was absorbed by soil particles worldwide after fallout of atmospheric radioactive species resulting from nuclear test during the late 1950's. As soil particles are redistributed in the landscape due to erosion and sedimentation so the activity of this radio-nuclide is decreased or increased when compared to reference stable sites. Furthermore, a comparison is also necessary with another reference which is that of time evolution of 137 Cs levels in soils according to date of measurement in reference site, as radioactive decay occurs since the fallout. The technique requires soil sampling and a gamma spectrometer for measuring radioactive decay in the soil samples. A sampling scheme adequately designed allows mapping redistribution of soil particles within a land tract, and this information is normally used to calibrated erosion and sedimentation models that replicate such redistribution. Berylium 7 is also being tested as tracer in erosion studies (Marestoni et al., 2009). These authors tested the method in experimental plots and found out that, as penetration of 7Be in soils hardly reaches 3 cm soil depth, it is a reliable approach to runoff erosion assessments. Besides, tracing 7 Be in these experimental plots very well matched soil loss measurement following common procedures. Unlike the 137 Cs, the 7 Be is a naturally occurring isotope of cosmogenic origin, with worldwide deposition on soils, yet very much dependent on rainfall.

A special case of tracers use in erosion studies is that of rock fragments. Unlike the above described, in this case, the tracer is supposed to have no displacement while the fines are eroded. Assessment of the relative concentration of rock fragments is then compared to a reference stable point with similar soil allows the computation of erosion rates, as far as a time reference is also possible to be assumed.

Globally, field measurements are the core references for any other erosion assessment. They are meant to provide the best approach to the actual occurrence of erosional processes. Due to the complexity of “real world” conditions, crossed with currently found instrumental constraints, care has to be given to the design and implementation of field measurements. Otherwise, results lead to misinterpretations and compromise the robustness any reference has to possess.

4. EXPERIMENTAL SIMULATIONS

4.1 About simulations

Due to the very high time and space variability of erosion factors, experimental simulation is quite often the approach adopted to study and assess erosion (Hudson, 1091; Morgan, 2007). This approach by-passes some of the difficulties associated with installing and monitoring field experiments and with performing field surveys. Simulations allow triggering rainfall erosion events and controlling their precipitation characteristics and ground or soil conditions, according to convenience. These advantages shorten time required for obtaining research results.

However, the main advantage of simulations is the possibility they provide to control factors and processes, bounding systems to be studied with control of their boundaries. Such conditions are not possible to obtain in the field, considering the large set of interactions between factors and the chaining of processes that are commonly found outdoors, which are the main constraint to scientifically grounded interpretation of assessment results.

Simulations face the problem of reliability in representing the real world, even though it is a fragmented real world. This is a key issue when comparing simulation with field results in erosion experiments. In fact, natural rainfall is hardly replicated by simulators, due to technical limitations, and soil samples used in simulations hardly replicate natural soils, due to disturbance or scale. Scale is often a limitation in simulation experiments, as size of the experimental setup is upper limited by actual instrumental conditions, meaning that those experiments are normally small scale.

All advantages and constraints considered, experimental simulations were and are, undoubtly, a very valuable approach for the advancement of knowledge of runoff erosion processes.

4.2 Rainfall simulators: general

Addressing simulation in erosion experimental research commonly means focusing in rainfall simulation and simulators. As the agent, without which no process occurs, emulating a rain shower has been since long a core research concern. Steps towards this goal were grounded in the deep insight on natural rainfall characteristics that are required to set a reference to be replicated. Drawing the full picture of precipitation characteristics and of how they relate either to easy measurable rainfall parameters, or to synoptic conditions or climatic features is still a hard task. However, since Ellison and with later definite contributions by Laws & Parsons, Ghadiri & Payne, Sfalanga & Torri (cited by Hudson, 1981), natural rainfalls were experimental studied and described with the existing technologies and results obtained are the basis for actual rainfall erosivity estimates. A large step forward in more recent years, not yet comprehensively accomplished, was provided by disdrometers, an equipment used for the refined measurement of rainfall characteristics (Tomás, 1997; António, 2010).

To summarize, it is possible to set a reference for natural precipitation characteristics, namely rain drop size distribution and its variation with rainfall intensity, for certain geographical regions, normally accepted as reliable even for other regions. The relationship between rainfall kinetic energy per unit rainfall height and rainfall intensity expresses that variation in useful form, since kinetic energy of raindrops provides the work necessary to breakdown exposed soil aggregates and start erosion processes. On the other hand, such relationship changes geographically, the data sets for deriving it being compiled in the USA, South Central Africa and with a much shorter record length in Italy /Wischmeier and Smith, 1978; Hudson, 1981; Raglione et al., 1980).

The simulated rainfall should match natural rainfalls with certain frequency and duration, or, inversely, selected for the specific experiment. This means that a frequency analysis of rainfalls for the area to be emulated in experiments should exist. The most practical result of the mentioned analysis is the Intensity-Duration-Frequency (IDF) curves for the selected weather station. IDF curves provide a tool for setting or knowing the frequency (expressed in terms of return period) of the precipitation being simulated with a certain duration and intensity. Although a relevant approach, this should be complemented with natural rainfall kinetic energy frequency analysis, which is a much more difficult task because it requires performing the frequency analysis itself with basic rainfall data records whilst IDF curves may be published or available and can be used without a prior analysis (Tomás, 1997; Figueiredo, 2001).

Rainfall simulation characteristics are duration, intensity (normally kept constant during the experiment), kinetic energy of the dropping water at ground or other reference level (e. g., canopy) and spatial uniformity within the target area. Additionally, if technical constraints do not ensure a steady water flow in the simulator, temporal changes should be known as well. As duration is arbitrarily selected (preferably with a sound justification given the natural and experimental conditions), all other characteristics need to be measured, assumed or neglected, according to accuracy required for the experiment.

Full calibrating runs with the simulator should include sampling the shower with cups placed in the target area at the required level (normally ground level). Cups may be of various materials (plastic, metal, glass), sizes (accommodating target area size, number of cups and sampling intensity accepted), and shapes (simple cups or funnel-topped where funnels are the actual interception device). The longer the run the large the amount of water intercepted by cups, the smaller the relative experimental error. Volume or mass of water captured in the cups averaged and adequately converted to mm h-1 allows assessment of simulated rainfall intensity. Uniformity of water distribution in target area is accessed via statistical dispersion of water amounts collected in cups (Tomás, 1997; Bompastor et al., 2009).

Assessment of simulated rainfall kinetic energy requires additional measurements, namely fall height, initial drop velocity and water drop size distribution. The latter is assumed zero for drippers but has to be assed in the case of sprinklers, the two simulator types described later. To do so, discharge from the simulator nozzles is measured conveying sprinkled water to a bucket and measuring time to fill it or to reach a defined volume. The nozzle diameter is normally given by the commercial provider, otherwise it is measured. Velocity is computed with discharge and cross-sectional area of flow in the nozzles (Bompastor et al., 2009). If the simulator is multi-nozzle, uniformity should be assessed too, with data dispersion analysis. Height is measured from dripper or nozzle to ground surface or other upper reference level and it is normally fixed according to experimental convenience. Water drop size distribution can be assessed in several ways according to instrumentation available. The most reliable device is the disdrometer, which has a sensor plate hit by falling water drops and the detected signal is converted in drop size distribution, allowing the computations of kinetic enegy and intensity (Tomás, 1997). Acoustic sensors, stroboscopic photos, high time resolution image capture are other means of reaching the same goal, with or without measurement of the falling velocity (Tomás, 1997; Morgan, 2005). The most traditional and low-technology input methods is the flour pellet method, that consists in exposing a pan filled with a thick uniform layer of flour to the water shower at the required level, for a short time span so as to allow pellet formation and limit flour soaking. Pellets formed represent drop hitting flour but a calibration procedure is required to adequately assess water drop size from pellet size (Hudson, 1981). A set of sieves screens pellets by size and a drop size distribution curve can be drawn from the mass pellets trapped in each sieve and the size limits defined by each pair of sieves (Bompastor et al., 2009) (Fig. 3.8). The larger the set of sieve the more accurate is the curve. The curve allows derivation of D50 of the simulated rain shower, meaning the median diameter of water drops, that halves the total precipitation (the mass of water drops) (Hudson, 1981). Kinetic energy can be computed with this data.

If the height of simulation is to be change, the previous measurements have to be repeated for several heights (Bompastor et al., 2009). Operational parameters of the simulator that affect or better control simulated rainfall conditions, should be tested during calibration procedures, and be the actual simulation parameters during experimental runs. They include for instance water pressure at outlet in the case of sprinklers (Fig. 3.8). Simulations should only start when steady state at the defined simulation conditions is reached, however normally a short time after starting operating the simulator. To limit water loss, a closed circuit water flow should be possible. Water saving is a crucial issue for simulations, especially under outdoor conditions in remote areas, where water availability is seriously limited.

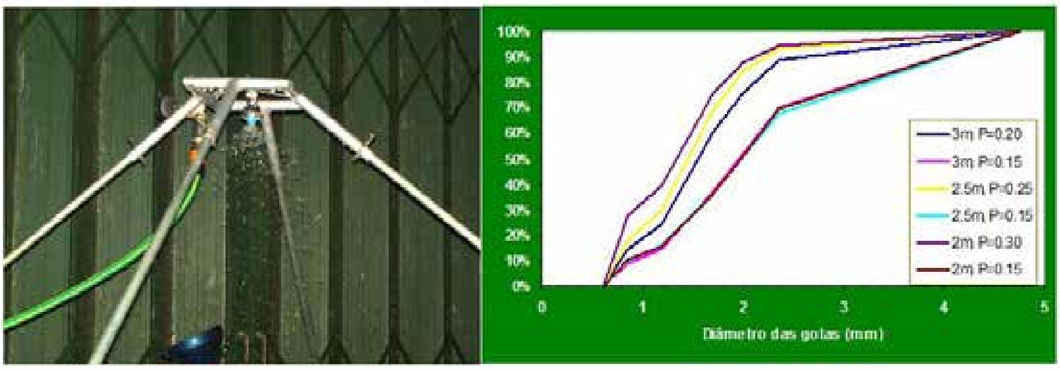

Fig. 3.8: A portable single nozzle sprinkler-type rainfall simulator: calibration indoor and drop size distribution curves according to height and pressure (Bragança).

4.3 Rainfall simulators: types

Simulators can be classified by several criteria (Hudson, 1981; Morgan, 2005). They may be portable or fixed structures. The latter work indoor, meaning in a hangar or laboratory and may heavier or lighter structures according to purposes and material conditions to install them. They generally allow larger target areas, more accurate control of simulation conditions, eventually with extended capabilities that may comprise simultaneous rainfall, runoff and subsurface flow simulation and control, wind and moving storms effects, water quality regulation (Zheng et al., 2004; de Lima and Singh, 2003; Shainberg et al., 1991, respectively). Conversely, portable simulators are normally designed for field conditions, working over smaller target areas, generally with light structure, straight-forward operation, a short set of operational parameters or simple fixed ones, more concerned with water saving design and operation (Fig. 3.9).

Besides, the main criterion for splitting simulators in two main types – drippers and sprinklers – is the drop forming process that sharply differs in the two cases and has important consequences for the characteristics of the simulated rainfall, as well as to operational conditions (Hudson, 1981; De Ploey, 1983; Morgan, 2005).

Fig. 3.9: A protable single nozzle sprinkler-type rainfall simulator at work in the field: experimental setup (top), measuring shrub height (bottom left), run on bare micro-plot (bottom right) (Aveleda, Bragnaça).

In drippers, drops form under low pressure (few centimeters equivalent water height), flowing out from a tank at kept at constant water head through narrow tubes (syringe needles, glass or plastic quasi capillary pipes, larger pipes with an axial wire or string fixed centrally). Water drops initial velocity is taken as zero and dropping frequency (mean total discharge from the tank) increases with water head, whereas drop size in determined by the drip device characteristics, namely the narrow tubes internal diameter. In drippers, water drops are very uniform and so the simulated rain shower represents only the D50 of the natural rainfall being simulated and not the range of sizes observed in nature. Furthermore, drops fall from each dripping device onto the ground always in the same position, meaning that some points are severely impacted while others are not at all impacted. To overcome this limitation, drippers should be moved during rainfall simulation runs, in a determined or random pattern, or the trajectory of the falling drops have to be disturbed, for instance by means of fans, or even the drops pass an intercepting mesh after leaving the dripping devices and are reworked to produce the actual simulated rain shower with a totally new drop size distribution (Alexandre, 1989; Bryand and De Ploey, 1983, respectively).

In sprinklers, drops are the result of rapid water flow under hydraulic pressure passing a nozzle where it is under atmospheric pressure. The spray produced includes a large range of drop sizes, therefore approaching natural rains drop size distribution. Nozzle characteristics, namely internal diameter and spray angle determine maximum discharge and the size of target area (also affected by simulation height). However, in sprinklers the set of operational parameters, and their interactions, affecting simulated rainfall characteristics is quite large. For example, besides nozzle characteristics, discharge is positively affected by pressure, meaning that intensity increases as pressure increases, but this induces a finer spray, meaning a decrease in the D50 of water drops. As so, sprinklers normally yield a low D50 of the water drops when compared with that of natural rainfalls with similar intensity, or, stated differently, when compared with natural rainfalls, rains simulated by sprinklers with a similar D50 of water drops have a much higher rainfall intensity than the natural ones (Tomás, 1997; Morgan, 2005; Bompastor et al. 2009). To overcome this limitation, typical of sprinklers, some models incorporate a rotating metal disk, with and an open window that intermittently allows the free jet flow, while during part of each rotation cycle the et is intercepted. This way, the intensity is lowered but the water drop characteristics are not changed (Hudson, 1981; Tomás, 1997). Multiple-nozzle sprinklers allow larger target areas but the problem of uniformity of drop distribution within the target area, normal in sprinklers, persists or it is even enhanced due spray cone interception /(Hudson, 1981; Morgan, 2005).

In both types, kinetic energy of simulated rainfalls depends on water drops falling height, even though in sprinklers flow pressure in the hydraulic circuit promotes higher drop velocity than in the case of drippers. Only under laboratory fixed structures falling heights can allow approaching water drop terminal velocity, as so approaching natural rainfall conditions. Therefore, in most cases, simulated rainfalls have a lower kinetic energy than natural rainfalls for the same duration and intensity. In the most common cases both types of simulators perform similarly to this respect (De Ploey, 1983).

Outdoor simulations impose special concern about power supply to work pumps and ensure steady pressure in the hydraulic conditions (meaning the need of a fuel motorized power generator), water availability (water tanks) have to be transported to the field), feasible simulator height and appropriate positioning (to ensure a vertical water jet) and wind (the spray cone has to be protected in windy areas by means of a plastic curtain around the simulation area) (Bompastor et al., 2009).

5. MEASUREMENT OF RUNOFF EROSION RELATED SOIL PROPERTIES

5.1 What are runoff erosion related soil processes?

The hydrological response of a land tract that results in soil and water losses by runoff erosion is dependent of many soil and ground features and some of them are decisive for the rate and magnitude of those losses. Assessing the parameters that represent such features might be, therefore, a step towards an indirect assessment of susceptibility to erosion or to help understanding and explaining (empirically or by means of models) runoff erosion direct assessments, performed with the methods and approaches developed above (e. g. Morgan et al., 1998).

The focus of this section is on soil properties and processes that affect ground hydrological response to an erosive rainfall. It is to be noted that, vegetation characteristics are the prior and in most cases the significant factor controlling that response. However, it is assumed that the most critical condition is that of scarcely covered or bare soil, due to the very effective protection role of vegetation. As so, attention is given only to soil and surface ground and not to vegetation, adding that, with or without vegetation cover, erosional response is also determined by those soil and ground features.

Selected properties and processes are those related to water intake by soil, to water flow at surface, to soil resistance to soil particles breakdown and transport by erosion processes, to water redistribution within soil profile.

5.2 Infiltration and soil permeability

During a rainfall, soil intakes water reaching the surface, in a process named infiltration, and proceeds, even after rain ceased, as flow in a porous medium through the soil body, in a process named redistribution. The bulk water flow velocity in soil pores is the key factor controlling both processes and it is named soil permeability or soil hydraulic conductivity (Hillel, 1998).

Hydraulic conductivity is normally measured in saturated soil samples, disturbed or undisturbed, in laboratory permeameters (Fig. 3.10). In these, water flow discharge passing through the samples is measured and the hydraulic conductivity computed according to Darcy law:

$$v = k\ I = k\ DH / L$$

where v is the flow velocity, k is the hydraulic conductivity, I is the hydraulic gradient, DH is the hydraulic head loss (meaning the difference in height between the inlet and outlet of water surface at atmospheric pressure), L is the length through soil (meaning the soil sample length).

Discharge is computed measuring outcoming water volume (V) in a certain time interval (Dt) and velocity is computed with discharge (Q) passing the cross sectional area of the soil sample (A):

$$Q = v\ A$$

Hydraulic conductivity is computed combining the above sated, form given sample size (A and L), measuring in each runoff DH and V and Dt.

Samples are normally cylindrical cores, filling in rings or tubes (metal, glass, plexiglass, plastic), when working with disturbed soil, the filling in procedure being precisely defined to avoid differences in compaction level along the soil column and between samples. Metal rings or cylinders, sharp edged in one side, are normally used in undisturbed samples, as they are the field sampling container, directly placed in the permeameter after transport to the lab.

Samples are previously and adequately water saturated, a process that may require long time in clay reach soils, but that is essential to ensure a single phase flow. Several runs are required to obtain a consistent average of saturated hydraulic conductivity value for a single sample and the number of samples to consistently describe the permeability of a soil is normally large due to the high spatial variability of this soil property (Warrick and Nielsen, 1980). Hydraulic conductivity is classified from very rapid (>25.4 cm/h) to very slow (<0.13 cm/h) (Hillel, 1998).

There are basically two types of permeameters: constant head and falling head. The most common are constant head, in which the inlet water level is kept constant throughout the run, therefore requiring a design to ensure constant water feeding of the circuit. Because of this, water expenditure is high in open circuit permeameters, a constraint overcome with close circuit models, in which the outlet water is conveyed to a tank and then pumped back to the inlet to pass again through the sample (Fig. 4.10).

Fig.3.10: Measuring erosion related soil properties: a closed-circuit laboratory constant-head permameter for 24 samples (left, Bragança), a torvane for for shear strengh (right).

In falling head permeameters, the water flow passing through the soil sample is under a continually decreasing head, because water is not added to the system as it drains. Soil hydraulic conductivity is computed considering Darcy law applied for the prevailing flow conditions in this case. This type of permeameter is less commonly used when compared with the constant head model, but it is better adapted to work in field conditions.

Infiltration volumes are measured in the field either indirectly under rainfall simulation or directly using infiltrometers (Hillel, 1998). These are metal rings inserted into soil surface, and filled in with water, set at constant, near constant or falling head conditions. In the second case, water level is allowed to drop to a certain extent inside de infiltration ring, meaning that water is added a regular interval to reach again the initial level, when it drops down to the lower level set for the measurement run. The water volumes added at recorded time intervals, allow computing the evolution of flow discharge and velocity (considering the infiltrometer cross section area) during the measurement run, the latter corresponding to the cumulative infiltration curve. Infiltration rate evolution curve are derived from the cumulative curve. The infiltration characteristics identified in a run are the initial and the final or steady infiltration rates, together with the declining function, all characterizing the infiltration behviour of soils.

Measurements with the ring infiltrometer are affected by subsurface lateral flow of water in the soil (Bouwer, 1986). In fact, especially in dry soil, the hydraulic gradient between the wet soil beneath the moist infiltrating area and the neighbouring dry soil, determines significant lateral flow and not simply the gravity-driven downward water flow, which corresponds to the parameter being measured. This effect is overcome by the double-ring infiltrometer, an apparatus that, besides the infiltration inner ring has an outer and much larger ring, with the single porpuse of obtaining a larger moist soil column, in the center of which measurements are performed, thus avoiding subsurface water flow outward the infiltrometer projection area along the soil profile. In double-ring infiltrometers, water is added to both rings but measurements are only done in the inner one.

Infiltration may be also computed from rainfall simulation runs, as simulated rainfall is known and runoff yielded by the bounded simulation target area is the main parameter of concern in these runs. The possibility of drawing the cumulative infiltration curve depends on the temporal resolution of runoff monitoring during a constant intensity rainfall simulation. There is an obvious and reliable assumption that evaporation and eventually evapotranspiration is virtually nil during the rainfall simulation, therefore allowing infiltration to be computed as the difference between rainfall and runoff.

5.3 Bulk density, porosity and compacity

Soil bulk density measurements are the basic step to assess soil porosity, a property that largely affects water infiltration and redistribution in soils, while providing an important indication of soil structural status. Furthermore, bulk density is an essential parameter to allow conversion of volume-based to mass-based soil parameters or properties.

Bulk density may be assessed by the cylinders method, the excavation method or even by nuclear methods (Blake and Hartge, 1986). The cylinders method is the most commonly used, consisting in a metal cylinder, with a sharp edged rim in one side that allows an easier insertion into the soil (Fig. 3.11). A guiding probe where the sampler cylinder is placed is hammered or forced to penetrate the soil by means of a motorized system. After full insertion, the probe is taken out of the soil leaning it laterally, an operation that requires soil excavation at least in one side of the probe, and with the help of a spade the soil is cut straight at the lower cylinder edge. The cylinder is removed from the probe and the sample is then oven-dried (105ºC) and weighted. Soil sample mass is divided by its volume (the cylinder volume) to obtain bulk density, and further division by the specific mass of water to obtain a relative, non-dimensional, bulk density.

Fig. 3.11: Sampling for bulk density (cylinders method) (Edroso, Vinhais).

The excavation method is based on the same principle as that of the cylinders method, meaning measuring the mass and the volume of the soil being sampled. The difference is that in this case the volume is neither fixed nor known at start. At the required site, a soil sample is taken digging a hole, the sample being later oven-dried and weighted in the lab. The volume of the hole has to be assessed and this can be made filling it with a measured volume of sand, or, when the hole has a regular prismatic or cylindrical form, take the required measures of its dimension to allow computing the volume.

A gamma-ray probe can also be applied to measure soil bulk density. It consists on a pair of parallel bars, inserted into the soil through a pair of tubes previously installed, one guiding the radioactive source and the other the detector probes. Gamma-ray attenuation is proportional to bulk density, following a calibration function that has to be derived for each site.

As they require special handling care (a radioactive source is part of the apparatus), gamma-ray probes are virtually out of fashion. Their main advantage was to allow monitoring bulk density changes through time in a given site, and this could be done as part of a monitoring scheme covering an area with several monitoring sites. The excavation method is more suitable in areas where soils have high rock fragment contents, a situation sharply limits the penetration of cylinders through the soil mass and leads to very many failed attempts to obtain a sample with the cylinders method (Poesen and Lavee, 1994). However, due to the difficulties in measuring the excavation volume, it is normally not so often applied.

Total soil porosity (P) is estimated from bulk density (BD), assuming a fixed soil particle density (real density, RD) with following expression (Costa, 2004):

P (%) = 100 (RD – BD) / RD

RD values assumed are normally 2.6 or 2.65, although in soils with high organic matter contents these might be corrected accordingly.

However, either total porosity or pore size distribution can be directly determined in lab by means of mercury porosimetry (Danielson and Sutherland, 1986). In this method, mercury (Hg), a non-hoisting liquid, is injected in the soil sample under controlled steps of increasing pressure, and the volumes of mercury spend to fill in the pores recorded. The pressure required to fill in a pore depends on its size, so that pressure steps can be assigned to the volume of pores with a given size class. The relationship between pore size and pressure comes from the Jurin law for describing capillary rise, assuming equivalent pore diameter, as soil pores are far from such a regular shape.

The same principle is applied when deriving pore size distribution from the soil moisture characteristic curve, which relates water content to pressure applied to extract water from a saturated soil sample in a pressure plate apparatus (Klute, 1986). The soil moisture characteristic curve is also known as the pF curve because the ordinate is then expressed as the Log10 of the water height equivalent to pressure applied, expressed as a positive value (Costa, 2004). Some characteristic points of the curve are actually measured ones, the rest of the curve being derived by interpolation or regression. Those are pF 0 (virtually saturation), pF 2.0, 2.54 and / or 2.7 (in the range of field capacity) and pF 4.2 (wilting point). Pressure plate method requires an apparatus consisting in a container resisting high pressure, in which a porous ceramic plate is placed saturated. Soil samples placed onto the plate, confined in rings or cylinders, are saturated with water together with plate. Once closed, the container is subject to a negative pressure that extracts soil water through the pressure plate and drains it outwards. When drainage ceases, the actual soil moisture content of samples is in equilibrium with the pressure imposed and so, the run stops to remove, weight, oven-dry and weight again samples, and soil moisture computed. Pressure plates differ according to pressure imposed. Variations of this general method include the apparatus named sand table, suitable for low pressure conditions, and the Tanner &Alrich apparatus in which the rate of water drainage from single and large soil samples, placed onto a similar plate and inside a similar container, is monitored.

Pore size distribution allows the deepest insight for understanding water flow through the soil, either during infiltration or redistribution. However,, for runoff erosion studies, only a part of the total pore size distribution is normally used and it addresses the larger pores. In fact, the most dynamic flow occurs in these pores, whereas in the finer ones flow is virtually nil at a rainfall event time scale.

Compacity is a soil property that represents the degree of compactness of soils andf gives an indication of the soil structural status and degradation due to compaction (Costa, 2004). Compacity (COMP) is computed directly from total soil porosity (P):

$$COMP = 100 - P$$

5.4 Soil resistance

Soil resistance to runoff erosion is basically a different way to describe soil erodibility, which is the susceptibility of soils to erosion, in this case not making any process separation, e.g splash, wash, and concentrated flow erosion (Morgan, 2005). On the contrary, when approaching the problem from the point of view of soil resistance, a distinction is normally made on what concerns processes involved, and the soil properties or those process parameters dependent on soil are usually selected for assessment, as factors conditioning process rates. As so, the following paragraphs deal successively with splash and aggregate disruption by raindrop impact, surface sealing and overland flow generation, incision by concentrated overland flow. A final note is also made about erodibility.

Aggregate disruption during rainfalls occur directly as consequence of raindrop impact, detaching soil particles, and from collapse as wetting proceeds and the air trapped in closed pores increases internal pressure. Resistance of aggregates in both cases, single or combined, may be assessed to infer soil resistance to the first erosion process occurring during a rainfall event, e. g. splash. Several methods were developed to assess Middleton, and they may divide in two categories: those based on immersion and those based on water drop impact onto aggregates (Le Bissonnais, 1996). In the first groups, the simpler approach consists in single aggregates immersed in water and the time required for their collapse is observed. The procedure is repeated with as many aggregates as required to adequately characterize a given soil, classifying aggregates according to time to disruption. Variations of this include also other liquids, e. g. ethanol that entirely dries out the aggregates, in a more or less complex lab protocol. In another complex protocol, requiring specific equipment, the method uses not a single aggregate but a soil sample and a set of sieves shaken under water for a certain time, the result being a granulometric curve that is compared with that of the dispersed soil (in sodium hexametaphosfate), meaning soil with all individual mineral particles separated. From results of these procedures, indexes of soil resistance may be extracted.

Water drop based methods require a dripper (burette) from which fall drops impacting an aggregate, the number of drops leading to aggregate disruption being counted, in a procedure repeated with as many aggregates as needed for a consistent characterization of soil resistance (e. g. Rousseva, 1989). Drop diameter, falling height and drop number determine kinetic energy applied to disrupt aggregate, the higher the more resistant is the soil. Drainage in target area largely condition results so that aggregates should be placed on a mesh instead of on an impervious target bench. The procedure may be changed for a fixed number of water drops (meaning a pre-determined kinetic energy) and the measurement of aggregate size at the end of the run, results from which soil resistance indexes may be derived.

Resistance to crusting or surface sealing is also an approach adopted to assess soil resistance to runoff erosion, as it affects runoff generation and overland flow development. While the crust formation process itself and the crusting susceptibility may be assessed also with some of the methods described above in this section, crust resistance to disruption is normally approached differently (Le Bissonnais, 1996). Crust resistance is an important surface ground erosion parameter as it affects infiltration, detachability under raindrop impact and shear resistance to rapid overland flow. Assessment procedures, applied in the field where crusts are identified, are based on mechanical properties of the crusted soil. Resistance to penetration, measured by a penetrometer (which can be more or less sophisticated, from a pocket penetromete to a penetrologger) is one of them, in which the force required for the penetration into the crusted soil of a conical or cylindrical edge of the apparatus guiding bar is measured by a incorporated dynamometer. Resistance to torque is another approach, performed with the torvame, an apparatus with radial blades, fixed in a circular plate, that are inserted in the crusted soil (Fig. 4.10). An axial pring measures the torque force required to infinitesimally move the blades, the higher the force the higher the resistance of the crust. Soil moisture content sharply conditions results and so, not only soil moisture content at the time of assessments should be known but also measurements should be made at comparable moisture content, namely that determining a fairly plastic soil consistency. These methods apply also for assessing soil resistance to incision by rapid overland flow, determining linear erosion.

Soil erodibility is nowadays most commonly assessed combining several erosion related soil properties in a single index, the K factor of the Universal Soil Loss Equation (Wischmeier and Smith, 1978). The involved properties are soil texture, organic matter content, structure and permeability and the result is an index that quantifies soil loss per unit rainfall erosivity. In spite of the precise quantitative evaluation that such a result indicates, which can hardly be validated in most cases, the K factor is very helpful in ranking soils according to their susceptibility to runoff erosion, meaning the bulk response in term of splash, interrill wash and incipient rilling. Moreover, relevant soil properties are all incorporated in this index in an empirically reliable combination. Routine soil laboratory analysis provide texture and organic matter results and it is not difficult to select a score for structure and permeability to be taken as an input in K determination, once the soils of the area are sufficiently known. The resulting K factor should be corrected when a significant proportion of rock fragments is present, using the known negative exponential relationship between rock fragment cover and soil loss (Figueiredo, 1990).

The original procedure to derive K required a nomograph that, again in the original form, comprised 2 chart blocks, an output in non SI units and an input with particle size classes that followed the USDA system, different from that internationally adopted (Atterberg scale). The original nomograph was adapted by Figueiredo (1990) to accommodate SI units output and the international (Atterberg) granulometric scale, for that requiring 3 instead of 2 chart blocks (Fig. 4.12). Besides the nomograph, at least up to 70% silt, a formula can be applied to compute K:

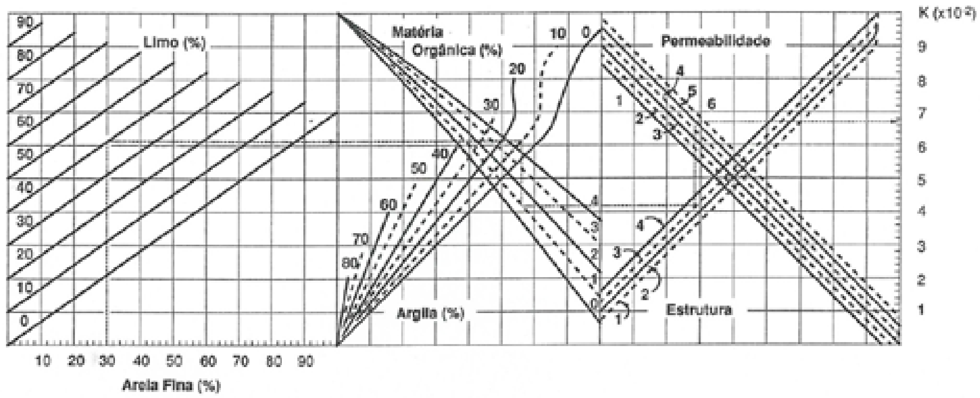

Fig. 3.12 – The erodibility (K factor) nomograph adapted to the international grain size scale and to SI units (Figueiredo, 1990). Note : Areia fina is Fine sand, Limo is Silt, Matéria Orgânica is Organic matter, Argila is Clay, Estrutura is Structure, Permeabilidade is Permeability.

5.5 Soil surface roughness

Surface roughness is decisive ground feature for the development of overland flow and runoff erosion, because it controls, at the micro-scale, water and sediment transfer along the slope (Morgan, 2005). More precisely, it determines the amount surface water storage, detention or retention during a rainfall event, therefore conditioning actual time to runoff generation and runoff amount. Moreover, once runoff is formed, development along the slope is conditioned by surface roughness as it affects the hydraulic friction and the length of flow path, therefore slowing down overland flow. Opportunities occur for enhanced sediment trap in surface depressions, due either to splash interception or to deposition of washed particles, as surface roughness increases. Surface roughness very much depends on soil management practices and it is a very dynamic feature in cultivated land, shown important changes with time (Morgan et al., 1998).

Methods for measuring surface roughness aim at providing a description of soil surface micro-topography. From which parameters or indexes may be extracted and used for runoff erosion interpretations or as input in erosion models. As detail in description of micro-topograhy increases, so increases sophistication of methods and equipment, and so increases complexity in data treatment and outcoming results. Full descriptions are 3D, while the simpler approaches provide a 2D description of soil micro-topography, meaning that a direction must be selected for the measurements performed along a straight line.

The simplest method is the chain method, consisting in a chain with a certain length that is placed along a line onto the soil surface, after adequate selection of measurement direction (Morgan et al., 1998). This is generally taken that of actual or expected overland flow paths. Once placed onto the ground, the straight line distance between the two edges of the chain is shorter the higher is surface roughness. An index is derived from the two lengths: that of the chain (normaly between 50cm to 1m) and that of the straight line distance between chain edges when resting over ground, named roughness index.

The profilemeter method provides a longitudinal profile of miro-relief, as a result of measurement of vertical distances from a reference level to ground surface, taken at regular horizontal distances. In the 3D approach the reference level is a plan and measurements are taken in two orthogonal directions in the plan. Reference plan or line are not necessarily leveled if their slopes are precisely known. The number of measurements taken depends on the required detail of assessment and it is limited by practical feasibility combined with instrumental capabilities.

In a needle profilemeter, a set of sticks, pins or needles, supported by a frame at regular short distances from each other (e. g. 5cm), are slide down to the ground and the heights to the reference frame measured or represented in a chart draw on the opposite edge of the sticks, pins or needles, to be later measured indoor. A very simple and straight forward variation of the method consists in a ruler along which, at regular distances, the distanced to the soil surface is measured with a tape (Fonseca, 2005). Either frame or ruler are fixed by two legs and leveled with a bubble level (Fig. 3.13). Data treatment consists in determining total length of ground surface and use it, together with ruler length, to compute the index above. Alternatively, random roughness, a second index is derived from data, which corresponds to the de-trended standard deviation of surface heights, meaning that a trend of data is obtained by regression, measured data subtracted from the trend line, and the standard deviation of the residuals around the trend line computed (van Wesemael et al., 1996).

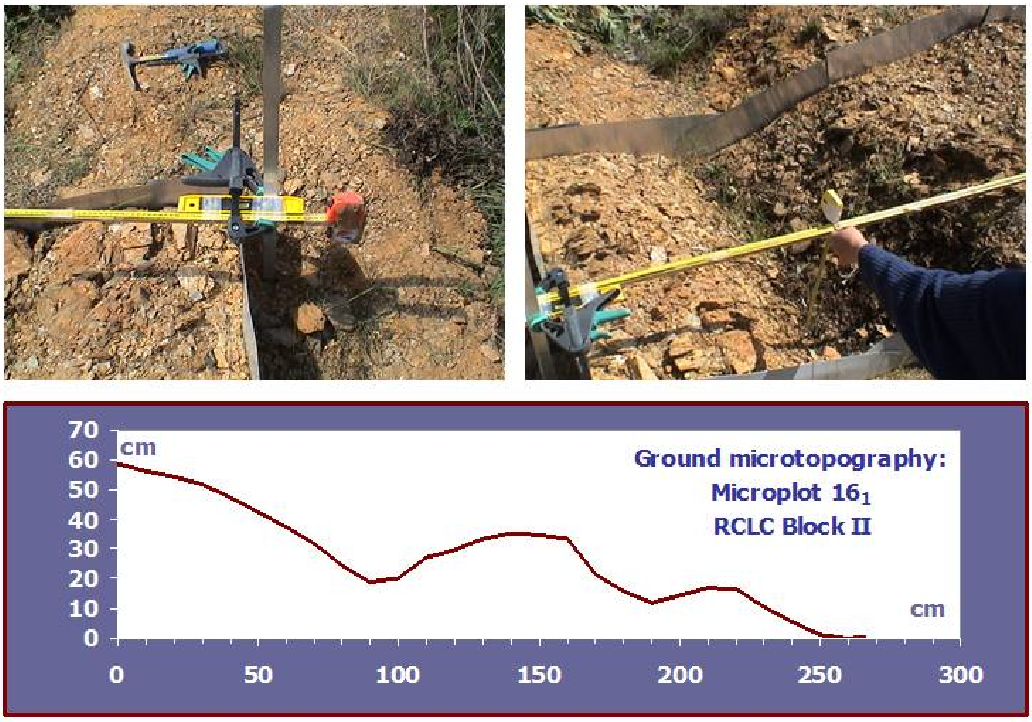

Fig. 3.13: Assessing surface roughness with a home-made device and the longitudinal surface profile of a micro-plot (Lamas de Podence, Macedo de Cavaleiros).

The procedures and instruments described above do not allow a very much detailed representation of soil micro-topography, as shortening distance between point measurements rises the number of measurements so as they become practically unfeasible. Besides, shortening distances is limited by instrumental capabilities. Laser profilemeters overcome these limitations and a provide non-contact measurements (van Wesemael et al., 1994; van Wesemael et al., 1996). They are sophisticated equipment, commonly placed in lab to work on simulated surfaces, but models exist to work in the field. Also models exist that allow working to output 3D results. The equipment consists in a frame supporting a laser source running over it at constant velocity by means of a motorized system. The laser beam is oriented to the soil surface and according to programmed operation, yet limited by equipment capabilities, measurements can be taken at very short distances along a line (0.1mm). Data is stored during runs and later transferred to perform data treatment and from which indexes may be derived. Due to the highly detailed data provided, complex approaches to deriving indexes are possible, as it is the case of using fractal analysis (van Wesemael et al., 1996).

In the case of 3D ground micro-relief surveys, more complexity of data treatment requires spatial analysis and GIS based methodologies. They can be applied with data sets issued form laser profilemeter measurements. However, approaches to this topic include also taking paired ortho-photos of surface ground, later treated with methods typical of aerial photo based surveys (Merel and Farres, 1998).

6. CONCLUDING REMARK

This overview on methods for assessing runoff erosion is expected to provide a consistent and comprehensive approach to the topic. However, in spite of the wish to cover the most essential cases that contributed to the development of erosion research, it was not meant to be a full review. As so, intentional or not intentional gaps may be found in this overview. Moreover, due the complexity of the object and of its dynamics, assessment methods require sometimes site specific solution to tackle with real world problems. Research innovative procedures are, therefore, ever present in erosion studies, while traditional methods keep their place in this field of knowledge, refining and consolidating protocols as well as adjusting their focus in terms of application conditions.

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CHAPTER 4

MODELLING RUNOFF EROSION

By

Márta Belényesi, Csaba Centeri, Niki Evelpidou, Varvara Antoniou

1. MODELLING RUNOFF EROSION

By Evelpidou N. & Antoniou V.

Over the years, considerable research and development has been accomplished for a series of suitable erosion models for estimating soil loss. These models vary in scope, from relatively simple empirical models or parametric models, based mainly on observations and statistical correlations, to physical models. The models also differ in terms of the complexity, the processes taken into account and the data required for their use. The accuracy and reliability of the estimates depend on the quality of the input data. As computer programmers say, "you put garbage, you get garbage" (Hudson, 1993).

The models can be classified in various ways. A useful distinction is between empirical (Empirical), natural (Physics-based) and other models.

1.1 Empirical models

Empirical models are usually simpler than all other types. They are mainly based on the analysis parameters which are determined by significant factors from field observations, measurements, experiments and statistical techniques that connect erosion factors with soil loss (Petter, 1992) and on the results and conclusions from them. Jakeman et al. (1999) reported that the characteristic feature of this category of models is the high level of the spatial and temporal data and their integration into a small number of variables. They are relatively easy and quick in predicting erosion, but they require long-term data (Elirehema, 2001). The constituted processes work towards the designed direction, meaning that the input data enter on one side of the equation, while the output data is on the other side of the equation, providing the estimates. In soil erosion studies, the most used models are empirical, with the USLE being the most popular (Universal Soil Erosion Equation - Global Territorial Loss Equation); it is more thoroughly discussed in the following section. In terms of data, USLE is one of the least demanding erosion models, and is widely applied in various scales (Wischmeier & Smith, 1978). The most frequently used empirical models are the following:

- **USLE:** In 1958, Wischmeier, a statistician of Soil Protection Agency in USA, was placed in charge in an effort to analyze and collect over 10,000 records of annual erosion in predefined small watersheds, in 46 selected fields in the Great Plains. The aim of Wischmeier and Smith (1978) was to develop an empirical model for the prevention of erosion based on climatic factors, morphological gradients and cultivation practices on agricultural land. This would enable specialists and policymakers to choose the type of measures which were necessary to maintain the erosion rates within acceptable limits. USLE (Wishmeier & Smith, 1978) is the most widely used model in predicting soil erosion. Due to its simplicity and clarity, it is used in education and research as an introduction to the understanding of erosion risk predictions (Hagos, 1998). It is based on deviation analysis of soil loss rates, in a defined framework of erosion studies in U.S.A., and was originally designed for the long-term estimate of the annual erosion rate in arable lands. The USLE equation is widely used mainly due to its relative simplicity (Desmet & Govers, 1995). The USLE model estimates the average annual soil loss and surface groove (sheet and rill) erosion. However, the model is unable to estimate both erosion for individual rainfall events and gully erosion (Foster, 1982; Keneth et al, 1991). An extensive analysis of the USLE model follows in the next paragraph.

- **MUSLE:** MUSLE model (Modified Universal Soil Loss Equation - Modified Global Territorial Loss Equation) is one of the modified versions of the USLE model, suitable mainly for estimates in countries with a tropical climate. In MUSLE model, the factor of rainfall energy was replaced with surface runoff. It is applied in individual rainfall events, without having to calculate the sediment transport rate, while the runoff factor represents the detachment energy and sediment transport. This is an equation for sediment yield (Roo, 1993).

- **RUSLE:** The RUSLE model (Revised Universal Soil Loss Equation - Revised Universal Loss Equation Territorial) is a revised version of USLE model, aiming to provide more accurate erosion estimates (Renard et al., 1994). This model includes the same factors as USLE, but all the equations used to determine the values of the factors have been revised. The most significant revisions were on vegetation cover C, cultivation practices, P, and the topographic factor LS. The C factor (land use variable) is now the product of four sub-factors: the most prominent land use, the degree of canopy cover, the soil cover and the surface roughness. A more detailed analysis of the RUSLE and comparison with the USLE follows in the next section.

- **MMF:** MMF model (Multiple Multiplicative Factor - Multiple multiplier) is an empirical model for the prediction of annual soil loss in a particular area and more specifically it is applied on slopes (Morgan, 2001). The MMF models separates the process of soil erosion in two phases, i.e. the water phase and the transported material phase. The model has been revised due to the increasing data availability and certain difficulties in assessing some parameters in the original version (Morgan, 2001).

1.2 Physics-based models

Physics-based models use data from the fundamental erosion processes and incorporate the laws of mass and energy conservation (Petter, 1992). They are based on the solution of fundamental physical equations and their validity lies on the fact that they represent the composition of independent variables affecting erosion, including the complex interactions amongst them and their spatial and temporal variations. These parameters are best suited for dynamic modeling (Jaroslav et al., 1996), because they have a physical meaning and can be measured in the field or determined approximately. In practice, however, the large number of parameters involved and the heterogeneity of fundamental characteristics, especially in drainage basins, means that these parameters will have to undergo calibration, which is an extremely complex process (Beck et al., 1995; Wheater et al., 1993). Physics-based models were developed to replace the conceptual distributed models since they provide a solid basis for understanding the erosion processes. Their main limitation is the high demand for data. Examples of such models are the following:

- **CREAMS** (Chemicals, Runoff and Erosion from Agricultural Management Systems). This model evaluates agricultural practices, regarding the transfer of soil contaminants due to surface runoff and soil water moving through the root system. Although it is a pollutant transport model, it is also considered as an erosion model because the pollutants are usually stored in sediments (Woodward, 1999).

- **ANSWERS** (Areal Non point Source Watershed Environment Response Simulation).

- **WEPP** (Water Erosion Prediction Project). WEPP erosion model (Nearing et al., 1994) was developed for the management and protection of soil and water and in environmental planning. It calculates the spatial distribution of net soil loss, by taking into account the spatial variability of morphology, roughness, soil properties, hydrology and land use (Nakos, 1983). WEPP describes a steady state of erosion and deposition caused by runoff. However, this condition is in reality rather rare, due to the relief and land cover, and also because of the roughness factor. In addition, the balance in the surface flow in the slopes of a basin is achieved at different times (Jaroslav et al., 1996).

- **EUROSEM** (European Soil Erosion Model). This model is adapted to the conditions of Central Europe. It uses extreme torrential phenomena in small areas or very small watersheds and simulates how water and sediment move on the soil surface. It is considered as quite demanding in respect of the primary data, and does not always give satisfactory results (Folly et al., 1999; Kinnell, 2004; De Vente et al., 2006).

- **AGNPS** (Agricultural Nonpoint Source Pollution Model). AGNPS is a deterministic model aiming to predict the quantity and quality of surface runoff, but also the transport of sediments and chemicals as a result of agricultural practices. It is based on extreme events and has been applied almost exclusively in the U.S.A., while there are a few examples from other regions, such as Italy and Germany. The model separates each basin in smaller test surfaces and the primary data derive from both maps and field work; it can be characterized as rather hard to use (Zhang et al., 1996; Lenzi & Luzio, 1997; Grunwald & Frede, 1999; Rode & Frede, 1999).

1.3 Other models

This category includes hybrid models, which combine a base model, a spatial model and other types of models. Specifically:

- **PESERA.** It is the most important hybrid model. It was developed for the quantification of soil erosion in environmentally sensitive areas at a regional or continental level and for policymaking soil conservation. It combines the effect of topography, climate and soil characteristics in a single integrated prediction of runoff and soil erosion. The data for each of these three factors are extracted from existing sources and are combined into a physics based model, resulting to a rational prediction of soil erosion. However, the model must be evaluated to ensure that its results are reliable, since further development, calibration and validation of the model is necessary (Kirkby et al., 2003). Finally, these results should be validated in low and high resolution and across different agro-ecological zones and also be compared with those from other models or methods used to estimate erosion risk.

- **“Black-box” or “mystery” model.** The term describes a model where the user is supplies certain informations and gets the result without having to know or understand what is happening in the "black-box". These models operate only in the direction assigned to them, meaning that all input data (inputs) are at one side of the equation, and the output lie on the other side of the equation. A common misuse of a "black-box"model is to try to apply it in reverse, aiming to solve the equation for one of the input data.

- **“Process-based” or “analytical component” models.** In these models, the user is able to explain mathematically each individual physical process and then to combine the separate effects. Because there are too many variables, and many mathematical calculations, these models can only work with the aid of computers.

- **Stochastic Models.** They simulate reality assuminh that the variables are depending on each other over time. They are particularly useful in hydrological studies and are based on the probabilities of return-period events.

1.4 Considerations in the assessment of soil loss

The estimation of soil loss is much more difficult than estimating runoff because there are many variables, deriving from physical processes, such as soil and rainfall, and also from the selected management practices. As a result, models of soil loss estimation, either empirically or "process-based", are complex, since their results are determined from a set of variables (Hudson, 1993).

However, models can produce essential and useful evaluations. A physics-based model is possible to calculate the individual rates of the erosion processes. For example, erosion from raindrop impact (raindrop splash) relates only to the energy of rain, ground cover, and soil type. Similarly, the sediment transport only requires to understand the effect of particle size and the flow rate. But an erosion model in case of a field, requires equations for both the aforementioned factors and equations for predicting the deposition and the transfer rate, which is not currently available.

Another factor that can be inserted into equations and estimates the soil loss, is the effect of soil conservation systems, such as levees and wells. The mass movement of soil and landslides are two more aspects of erosion that interfere with the assessment process. The only way to avoid these "interferences" is the use of hybrid models that combine the "physical process" with stochastic models based on the probability of the occurrence of such phenomena.

Furthermore, a basic difficulty in modeling the soil loss, is the variety of erosion forms, which have different causes and are affected by different factors. Thus, a model for predicting erosion in cropland uses a different set of parameters than those used for gully erosion, because the factors causing gullies can be defined qualitatively, but not quantitatively. However, models for estimating soil loss in agricultural land, and some of their depending factors, are easily estimated (Hudson, 1993).

Generally, the main problems of modeling lie in the following:

- the cost of the development,
- the complexity and friendliness to the user,
- the availability of data,

- ratification
- the value for end users,
- the scaling prediction from small to large areas,
- the performance.

Specifically:

The continuous use of USLE and not MUSLE or RUSLE ignores fundamental problems. It is clear, even in results from experimental fields, that USLE is unsatisfactory (Burwell & Kramer, 1983). At a European level, USLE is inappropriate because of the different types of rainfall, the hydrological processes and the relief, compared to the eastern U.S. (Jetten & Favis-Mortlock, in press) Also, USLE is unsuitable, after the importance of daily gully erosion and erosion due to tillage practices in Europe was recognized.

In order to achieve better erosion modeling in Europe, more sophisticated "process-based" models have been developed, such as WEPP and EUROSEM, although, there is little evidence that they are as useful as expected so far (Foster & Lane, 1987). Comparing the results of a physics-based model with those derived from field observations, some important differences can be noted: The model greatly exaggerates in erosion predictions, both in quantity and in scope. Also, the statistical distributions of data values are different. Thus, significant erosion is predicted in areas that have no particular problem (Evans & Brazier, 2005). The failure of such models is probably owed to their assumptions of runoff due to the Horton classification system or low intensity rainfall falling on saturated soils. Therefore, it is necessary to incorporate to the model erosion estimates from field work (Boardman, 2006).

Finally, according to the research conducted in several European countries by Van Rompaey et al. (2003) on the accuracy of estimates of soil loss by USLE and PESERA models, demonstrates that models generally give unsatisfactory results. The validation of the area of the selected basins using transported material in reservoirs is difficult because of the assumptions regarding the sediment delivery ratio. Gullies cannot be simulated in models and, therefore, particularly in Mediterranean countries, erosion is likely to be underestimated.

1.5 Comparison of erosion models used by european countries or research organizations

1.5.1 Overview

Several countries in Europe, aiming to address soil erosion, use a variety of assessment methods, from qualitative estimations of empirical modeling approaches to modeling, from which USLE and its derivatives are frequently used. Below, various methods used officially from European countries or research organizations are evaluated and compared, taking into account the **scale**, **transparency**, **complexity**, **cost efficiency** and **ambiguity**, with the final aim to draw conclusions about their reliability, flexibility and acceptance (Geraedts et al., 2008).

In order to compare RAM (Risk Assessment Methodology) objectively, they are assessed in terms of approach, target and according to the organizations using them. Then, they are represented in spider-charts consisting of five axes for the five different indicators mentioned in the previous paragraph (Geraedts et. al., 2008).

Most of the eleven RAM evaluated, aim at a quantitative approach, most of which uses the USLE model. Seven approaches are purely quantitative, one of them uses physics-based model, while the rest are based solely on estimates of (R) USLE. The main purpose of RAM is the development of hazard maps, except from the GLASOD method (Global Assessment of Human Induced Soil Degradation), which is universally used for mapping land degradation. More than half of the RAMs are executed by research institutes; these include universities and government agencies. All, except from two, are based on existing data, derinving mainly from databases, such as:

- **CORINE** (Coordination of Information on the Environment) or LUCAS (Land Use and Carbon Analysis System) for information on land use / land cover, soil maps at regional, national level, or large-scale (eg. European soil database).

- **Digital Elevation Models** (DEM) for information concerning the relief and the variety of climatic data (possibly deriving from MARS database).

Output data are expressed either in absolute terms (ton.ha-1.yr-1) or equivalent, relevant erosion classes. Finally, the size of the area or scale used by the models, is divided into regional and aggregate level (local, national, etc.) (Geraedts et. al., 2008).

1.5.2 Evaluation and comparison of different methods

As already mentioned, the comparison of different models (Geraedts et. al., 2008) was based on the following indicators:

- **Scale**: it refers to the map scale and is linked to the availability of existing cartographic data.

- **Transparency**: it refers to the applicability, clarity and logic of each method. The analysis from empirical models has the lowest transparency degree, while the physics-based models hold the highest.

- **Complexity**: it is associated with the processing of the input data and the amount of output data and it is calibrated based on the number of techniques used (e.g. laboratory experiments, application of GIS, RS (remote sensing), historical data, etc.).

- **Cost Efficiency**: it refers to both the financial cost and the means to achieve a goal. The high cost efficiency is achieved by using existing data and simple methods while the use of new data and complex modeling means low cost efficiency.

- **Ambiguous**: it refers to the uncertainty of calculations and estimates, which are related to the repeatability of the results. In general, physics-based models have relatively lower uncertainty (high - frequent repeatability of results) compared to methods based on the analysis of empirical models.

Each of these indicators has a scale from 0 to 10 and it is clear that there are large differences between the RAMs, even if the approach is similar, in some cases. The variety on map scales is the first reason for the large differentiation; the transparency of the methods ranges from clear, logical and comprehensible models to empirical relationships, but also factorial approaches and empirical models' analysis, which are less transparent.

In conclusion, physics-based models are transparent, complex, inefficient in terms of cost and their results have a relatively small indicator of ambiguity. Their uncertainty increases with increasing size of the study area. The approaches used for the empirical models' analysis or in combination with other methods, are very cost effective, but have a large uncertainty. Their transparency is generally low. This is overturned when the defined rules facilitate the repeatability of results.

In general, the aforentioned RAMs use relatively similar approaches for the estimation of soil loss, with the results expressed either in absolute terms or in relative categories. It should be oted that the accuracy of a model's output depends on the quality and resolution of the input data (Van der Knijff et al., 2000; Grimm et al., 2002; Kirkby et al., 2004; Gobin et al., 2006). The application of data from GIS or other digital datasets must be done with causion, given that a very low spatial resolution is inadequate for estimating erosin in small scales (Grimm et al., 2002).

Specifically, the Scandinavian RAMs use the USLE parameters, and their main objective is to simulate nutrients transport (especially phosphorus), since the ex-situ pollution is one of the main problems in these countries (Harrod, 1993; such reported in Boardman, 2006; Evans and Brazier, 2005).

The empirical model-based approaches, such as the Polish method and the GLASOD approach, and the methods of risk assessment based on factors, such as the French INRA approaches and CORINE, are largely based on estimations from empirical models, but they lack in detailed knowledge and are rather subjective (Grimm et al., 2002). The detailed estimation and evaluation of land use or climate change effects is difficult, as only a relative risk assessment is provided (Van der Knijff et al., 2000; Gobin et al., 2006). The INRA approach provides a relative repeatability of results, by setting established rules. The result of the qualitative and subjective evaluation of these methods is their low robustness. Furthermore, the map developed though the GLASOD method in Spain, proved to be the least accurate for erosion estimation in comparison with other methods (Sanchez et al., 2001). The CORINE and INRA approaches are based on the same methodology. However, the INRA approach is more accurate (Grimm et al., 2002; Gobin et al., 2006), given that it is based on more detailed input data, takes into consideration more realistic erosion processes (e.g. the existence of soil crust) and presents the results with higher resolution. Nevertheless, the disadvantages of qualitative approaches remain, since no absolute estimates of erosion are provided, and therefore the assessment of errors is hindered (Grimm et al., 2002).

The PESERA model is capable to simulate erosion at various levels (e.g. watershed, regional, national and European), demonstrating its flexibility. Although it is a hybrid physics-based model, the equations are simplified and the data required are relatively limited (Grimm et al., 2002). However, its suitability on particular local conditions remains doubtful because of this simplification of equations and data types (Gobin et al., 2006). The model has been developed for analysis of land use and climate change scenarios, having the potential to be used in the future as an excellent tool for policymaking, however more data of actual erosion rates are necessary for the validation of the model (ENVASSO, 2007).

At present, USLE model and its derivatives are, as aforementioned, the most commonly used models for estimating erosion risk. Their main advantage is their usability, which is owed to their relatively simple formulation-composition of the model where data are distributed in six factors. Moreover, a large database (associated with the long history of the model) and the integration of GIS lead to the easy processing of parameters (Van der Knijff et al., 2000; Morgan & Quinton, 2001; Drake & Vafeidis, 2004; Lewis et al., 2005). Consequently, the model can be used to simulate erosion over large areas, without extensive additional work to determine its factors, while taking into account the disadvantages, as mentioned in a previous paragraph. However, in case of large areas, the empirical models perform as satisfactory as physics-based models. Moreover, the USLE model approach provides detailed information also for small areas.

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2. MODEL USE AND BUILDING

By Belényesi M. & Centeri Cs.

2.1 Short description of ArcGIS^{tm}

2.1.1 What is GIS?

This chapter is based on: What is ArcGIS^{tm}" by ESRI, 2001

A geographic information system (GIS) integrates hardware, software, and data for capturing, managing, analyzing, and displaying all forms of geographically referenced information.

GIS allows us to view, understand, question, interpret, and visualize data in many ways that reveal relationships, patterns, and trends in the form of maps, globes, reports, and charts.

A GIS helps you answer questions and solve problems by looking at your data in a way that is quickly understood and easily shared.

There are many GIS software available nowadays, in this project we are going to use ArcGIS^{tm} system.

2.1.2. What is ArcGIS?

ArcGIS^{tm} system is an integrated geographic information system (GIS) consisting of three key parts:

- ArcGIS^{tm} Desktop softvare, an integrated suite of advanced GIS application
- ArcSDE^{tm} gareway, an interface for managing geodatabases in a database management system (DBMS)
- ArcIMS^{tm} software, Internet-based GIS for distributing data and services.

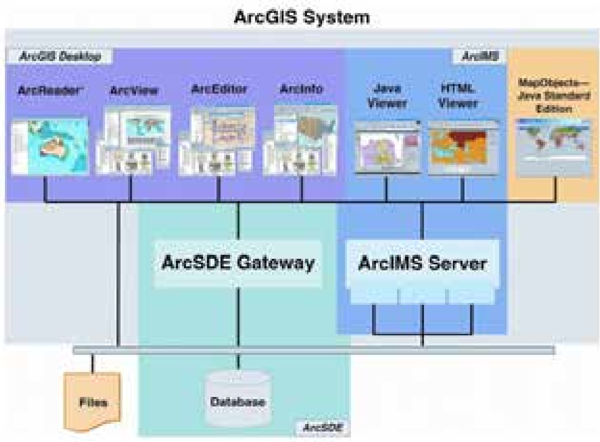


Fig. 4.1: ArcGIS System (Source: What is ArcGIS^{tm}*(2001))*

The ArcGIS^{tm} Desktop includes a suite of integrated applications: ArcMap, ArcCatalog, and ArcToolbox. Using these three applications together, you can perform any GIS task, simple to advanced, including mapping, data management, geographic analysis, data editing, and geoprocessing.

<u>ArcMap</u>

ArcMap is the central application in the ArcGIS Desktop. It is the GIS application used for all map-based tasks including cartography, map analysis, and editing. In this application we work with maps. These maps have a page layout containing a geographic window, or view, with a series of layers, legends, scale bars, North arrows, and other elements.

ArcMap offers different ways to view a map – a geographic data view and a layout view – in which you can perform a broad range of advanced GIS tasks.

<u>ArcCatalog</u>

The ArcCatalog application helps you to organize and manage all of your GIS data. It includes tools for browsing and finding geographic information, recording and viewing metadata, quickly viewing any dataset, and defining the schema structure for your geographic data layers.

<u>ArcToolbox</u>

ArcToolbox is a simple application containing many GIS tools used for geoprocessing. There are two versions of ArcToolbox: the complete ArcToolbox that comes with ArcInfo, and a lighter version that comes with ArcView and ArcEditor software. Tools are logically organized in themes based on the functionality, and are available to use without any customization. These operations can be very powerful and achieve great results.

ArcMap, ArcCatalog and ArcToolbox are designed to work together to perform all GIS tasks. For example you can search for and find a map document in ArcCatalog, then open it in ArcMap by double-clicking it in the Catalog. You can then edit and enhance your data through the tools available in the ArcMap editing environment.

In this project we are going to use all the applications listed above: ArcMap, ArcCatalog, and ArcToolbox.

2.2 The model builder

ModelBuilder is an application in which you create, edit, and manage models. It means, that this application allows you to graphically define geoprocessing operations. Using a simple tool, a user can drag and drop operations and data on a canvas, and connect them with directional arrows (signifying direction of data).

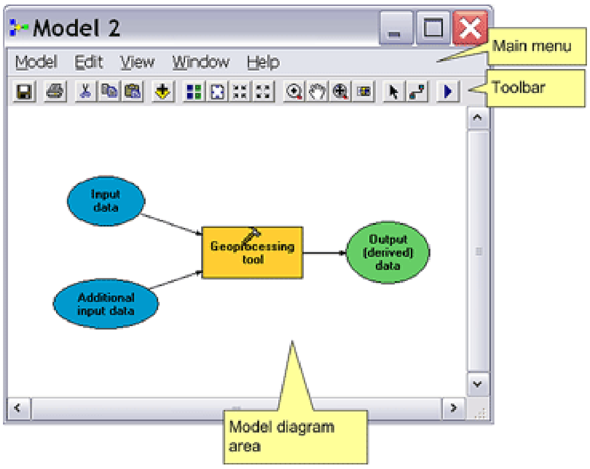


Fig. 4.2: Example from the model builder of ArcGIS (Source: http2)

We are going to build our erosion model using the Model Builder application!

2.3. Thematic layers and datasets

GIS organizes geographic data into a series of thematic layers and tables. Because data in a GIS are referenced to geography, they have real-world locations and could overlay one another. GIS links the location to each layer (such as people to addresses, buildings to parcels, or streets within a network) to give a better understanding of how the features interrelate.

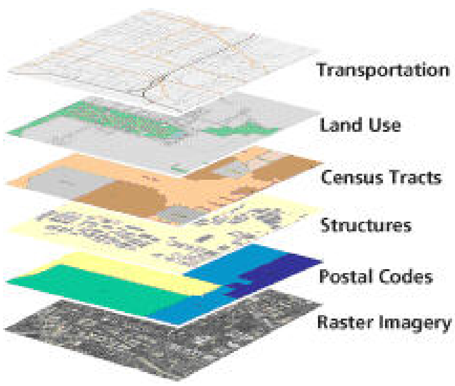


Fig. 4.3: Layers to be used in ArcGIS (Source: What is ArcGIS^{tm}*(2001))*

In a GIS, collections of geographic features are organized into data sets, such as land parcels, fire locations, buildings, orthophoto imagery, and raster-based digital elevation models (DEMs). Many of the spatial relationships between layers can be easily derived through their common geographic location. The basic data types used in any GIS are the next:

- Vector layers (points, lines or polygons)
- Raster layers (images, grids)
- Attributes (features describing spatial data, these attributes are organized in tables, these tables are – or can be – connected to spatial data under certain circumstances)

2.3.1 Vector layers

In a GIS, geographical features are often expressed as vectors, by considering those features as geometrical shapes. Different geographical features are expressed by different types of geometry:

Points

A simple vector map, using each of the vector elements: points for peaks, lines for contours or rivers, and a polygon for the lakes, settlements, etc.

Zero-dimensional points are used for geographical features that can best be expressed by a single point reference—in other words, by simple location. Examples include wells, peaks, elevation points, points of soil profile or other features of interest. Points convey the least amount of information of these file types. Points can also be used to represent areas when displayed at a small scale. For example, cities on a map of the world might be represented by points rather than polygons. No measurements are possible with point features. Point attribute tables can contain for example: elevation values of a peak or any elevation point, or all information of a given soil profile.

Lines or polylines

One-dimensional lines or polylines are used for linear features such as rivers, roads, railroads, trails, and topographic lines, or elevation contours. Again, as with point features, linear features displayed at a small scale will be represented as linear features rather than as a polygon. Line features can measure distance. Attributes like name of a river or elevation value of a contour can be also connected to line features.

Polygons

Two-dimensional polygons are used for geographical features that cover a particular area of the earth's surface. Such features may include lakes, park boundaries, buildings, city boundaries, land uses, landcover or soil maps. Polygons convey the most amount of information of the file types. Polygon features can measure perimeter and area.

Each of these geometries is linked to a row in a database that describes their attributes. For example, a database that describes lakes may contain a lake's depth, water quality, pollution level. This information can be used to make a map to describe a particular attribute of the dataset. For example, lakes could be colored depending on level of pollution. Different geometries can also be compared. For example, the GIS could be used to identify all wells (point geometry) that are within one kilometre of a lake (polygon geometry) that has a high level of pollution.

Vector data can also be used to represent continuously varying phenomena. Contour lines and triangulated irregular networks (TIN) are used to represent elevation or other continuously changing values. TINs record values at point locations, which are connected by lines to form an irregular mesh of triangles. The face of the triangles represents the terrain surface.

2.3.2 Rasters

Raster data represent features as a matrix of cells within rows and columns in continuous space. These cells are formed by pixels of a specific dimension size, and can be described as either "cell-based" or "image-based" data.

Cell-based data

Cell-based raster data sets (images and grids) are especially suited to representing traditional geographic phenomena that vary continuously over space such as elevation, slope, and precipitation. They can also be used to represent less traditional types of information such as population density, consumer behavior and other demographic characteristics. Rasters are also the ideal data representation for spatial modeling and analysis of flows and trends over data represented as continuous surfaces.

A grid is a raster data storage format native to ESRI. There are two types of grids: integer and floating point. Use integer grids to represent discrete data and floating-point grids to represent continuous data.

Attributes for an integer grid are stored in a value attribute table (VAT). A VAT has one record for each unique value in the grid. The record stores the unique value (VALUE is an integer that represents a particular class or grouping of cells) and the number of cells (COUNT) in the grid represented by that value. For example, if 50 cells have a value of 1 representing a forest, then the VAT would show a VALUE = 1 and COUNT = 50 for each of the 50 cells.

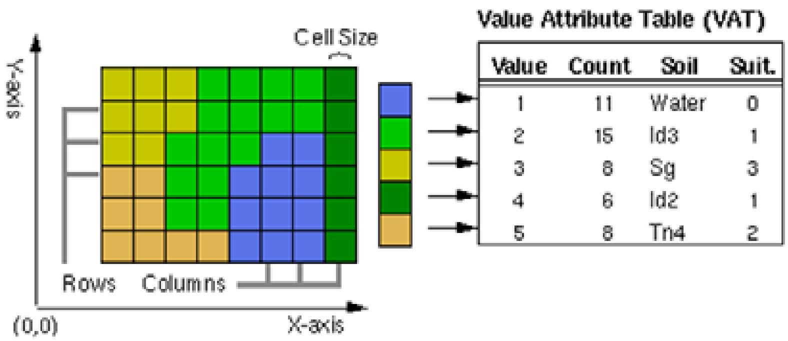


Fig. 4.4: The grids and their values in ArcGIS (Source: http2)

Floating-point grids do not have a VAT because the cells in the grid can assume any value within a given range of values. The cells in this type of grid do not fall neatly into discrete categories. The cell value itself is the attribute that describes the location. For example, in a grid that represents elevation data in meters above sea level, a cell with a value of 10.1662 indicates that the location is about 10 meters above sea level.

The range of data values that can be stored as grid values are:

- Floating-point grids can store values from -3.4^{38} to 3.4^{38}.
- Integer grids can store values from -2147483648 to 2147483647 (-2^{31} to $2^{31}-1$).

For integer grids, this information applies only to the VALUE item. An integer grid may have other INFO items added to its VAT whose range of values depends on the item definition.

The coordinate system of a grid is the same as that of other geographic data. The rows and columns are parallel to the x- and y-axes of the coordinate system. Since each cell within a grid has the same dimension as other cells, the location and area covered by any cell is easily determined by its row and column. The coordinate system of a grid is thus defined by the cell size, the number of rows and columns, and the x,y coordinate of the upper left corner. Grids also carry additional information, such as the coordinate system associated with the grid.

Image-based data

Image data ranges from satellite images and aerial photographs, to scanned maps that have been converted from printed to digital format.

2.3.3 Non-spatial (attribute) data

Additional non-spatial data can also be stored along with the spatial data represented by the coordinates of a vector geometry or the position of a raster cell. In vector data, the additional data contains attributes of the feature.

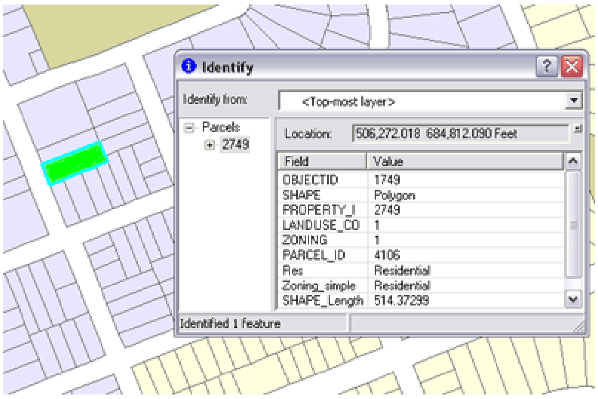


Fig. 4.5: Other data stored in ArcGIS (Source: http2)

For example, a forest inventory polygon may also have an identifier value and information about tree species. Polygons of a soilmap may have information on soil type, pH, water management type, etc. Polygons of landcover may contain type of vegetation, area, or C factor value, which represents cover management factor in erosion calculation. In raster data the cell value can store attribute information, but it can also be used as an identifier that can relate to records in another table.

In this project we are going to use vector data, raster data and descriptive (attribute) data as well. Calculations will be performed on raster layers, so we have to transfer all vector data to raster layers.

2.4 Calculations performed on grids

2.4.1 The spatial analyst extension and map algebra

The Spatial analyst extension and map algebra

ArcGIS Spatial Analyst provides a robust environment for advanced raster data analysis. This environment enables density mapping, distance analysis, surface analysis, and advanced analysis with map algebra, grid statistics, spatial modeling, and surface creation.

Map algebra is the main analysis language for ArcGIS Spatial Analyst. This language defines a syntax for combining map themes by applying mathematical operations and analytical functions to create new map themes. In a map algebra expression, the operators are a combination of mathematical, logical, or Boolean operators (+, >, AND, tan, and so on), and spatial analysis functions (slope, shortest path, spline, and so on), and the operands are spatial data and numbers.

Another useful tool is the Raster Calculator. The Calculator provides you a powerful tool for performing multiple tasks. You can perform mathematical calculations using operators and functions, set up selection queries, or type in Map Algebra syntax.

Fig. 4.6: Using the Raster Calculator in ArcGIS

Inputs can be raster datasets or raster layers, coverages, shapefiles, tables, constants, and numbers.

2.5 Exercise: calculating soil loss estimation on a test area

2.5.1 Preface

Soil is a non-renewable natural resource. Accurate estimate of soil loss is essential to make agricultural policy decisions and to plan land use at all scales from farm to national levels. A wide range of erosion models, most of them combined with GIS, varying in purpose and details are available for predicting soil loss. The well-known Universal Soil Loss Equation (USLE) (Wischmeier & Smith, 1978) was used to elaborate a map of the soil erosion risk assessment in this exercise.

2.5.2 Our test area

Fig. 4.7: The test area (Nemessándorháza, Hungary) (Source: Inst. Of Geodesy, Cartography and Remote Sensing)

2.5.3 Methods and data: The USLE Model

Methods and data: The USLE Model

The USLE is based on the following factors that are used to compute the mean annual soil loss:

$$A = R \times K \times L \times S \times C \times P \qquad (1.1)$$

where:

A = Mean annual soil loss

R = Rainfall erosivity factor

K = Soil erodibility factor

L = Slope length factor

S = Slope factor

C = Cover management factor

P = Agricultural Practice Factor

The procedure for calculating the mean annual soil loss consists of the next steps:

1. **build a Digital Elevation Model, if it is not accessible**
2. **determine the R factor**
3. **determine the K value**
4. **calculate the LS value from the Elevation Model**
5. **choose the crop type (C) factor for the crop to be grown**
6. **select the P factor**
7. **multiply the 5 factors together to obtain the soil loss per acre per year.**

2.5.4 From where can we get the data needed?

R-factor

The R-factor can be determined for example by applying a regression analyses on the data from meteorological stations stored in a meteorological database, if we are calculating the soil loss for a large area.

In our case there is no detailed R factor map for the test area, the R factor was taken as constant (415 MJ * mm * ha^{-1} * h^{-1} * y^{-1}).

K factor

Statistical analysis of rainfall simulation studies showed that K factors vary not only by soil types but each soil type's K factor varies by soil moisture content. Since soil moisture content is mostly varies by season, seasonal variability of K factors can be investigated by rainfall simulation studies. We can state that for erosion modeling more than one K factor is necessary during the year for proper calculations.

Other way of determining K factor is using estimates of different authors. We follow this way now. K factor estimations for soil types are made by Stefanovits (1966) and can be seen in Table 1 and Table 3. (see Annex I).

Soil map is provided for the test area, soil type information is included in the map's database. K factor data from Table 1 can easily attached to the map's database.

LS factor

This factor can be calculated from the Digital Elevation Model. The method of Hickey et al. (1994) and Desmet and Govers (1996) were used for calculating LS factor with GIS Arc/Info by Pataki (2000). The program uses the DEM and the derived inclination to calculate the length of slope and LS factor.

We provide the LS factor map of the test area for you.

C factor

C is the crop/vegetation and management factor. It is used to determine the relative effectiveness of soil and crop management systems in terms of preventing soil loss.

For classifying the C factor of land cover and vegetation the USLE manual (Wischmeier and Smith, 1978) was used. Land Cover map is provided for you, and C factor data (Table 2., see Annex) can be easily attached to the map's database.

P factor

Agricultural Practice Factor (dimensionless). „It is the ratio of soil loss with a specific support practice to the corresponding loss with upslope-downslope cultivation" (Wischmeier and Smith, 1978).

In our case P factor was constant (=1), since upslope-downslope cultivation was widespread in the area.

2.6 Workflow

All data you need is located in **Erosion/input** folder. Save all your data to **Erosion/output** folder!

Task 1.: Visualize your data

- Start ArcMap!
- Start ArcCatalog (1)
- Navigate to the folder, where the data are! (2)

  Erosion/Input
- Open the vector layers (3), take a look on the attributes! (4)
  - Source of the soil map: erosion/input/shp/soilmap.shp
  - Source of the vegetation map: erosion/input/shp/landcovermap.shp
- Open the LS grid (5), examine the attributes!
  - Source of the LS grid: erosion/input/grid/ls_grid
- Open (6) and examine K factor table, and C factor table!
  - Source of the K factor table: erosion/input/table/k_factor
  - Sourceof the C factor table: erosion/input/table/c_factor

In details:

1. Start ArcCatalog with  icon!

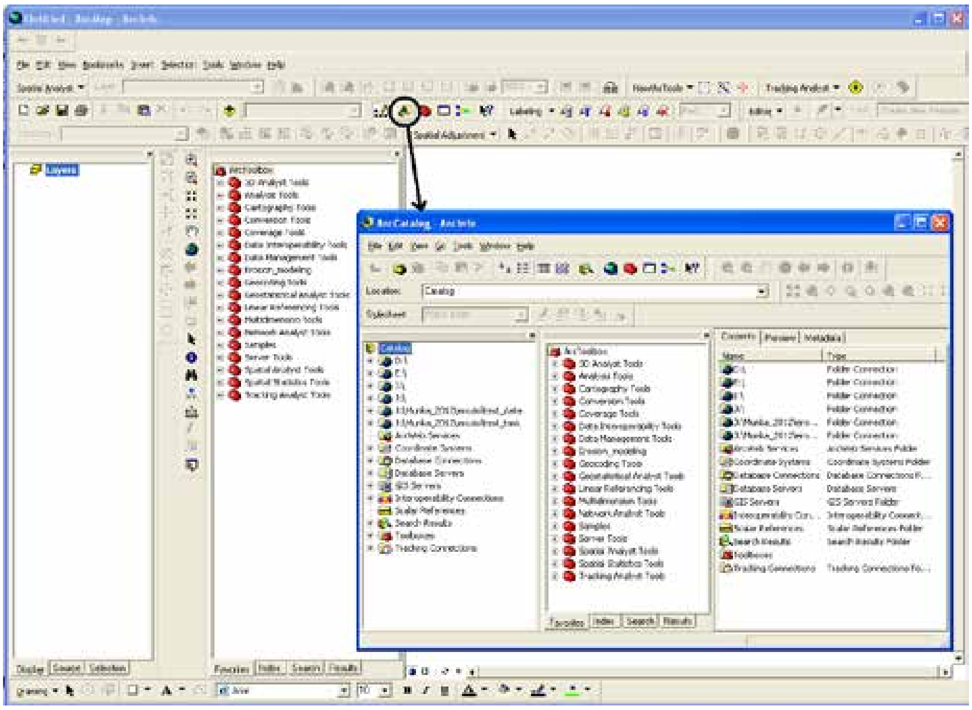


Fig. 4.8: How to start the ArcCatalog in ArcGIS

2. Navigate to the folder, where the vector data are (erosion/input/shp/)! Use the „Connect to Folder” icon  in ArcCatalog!

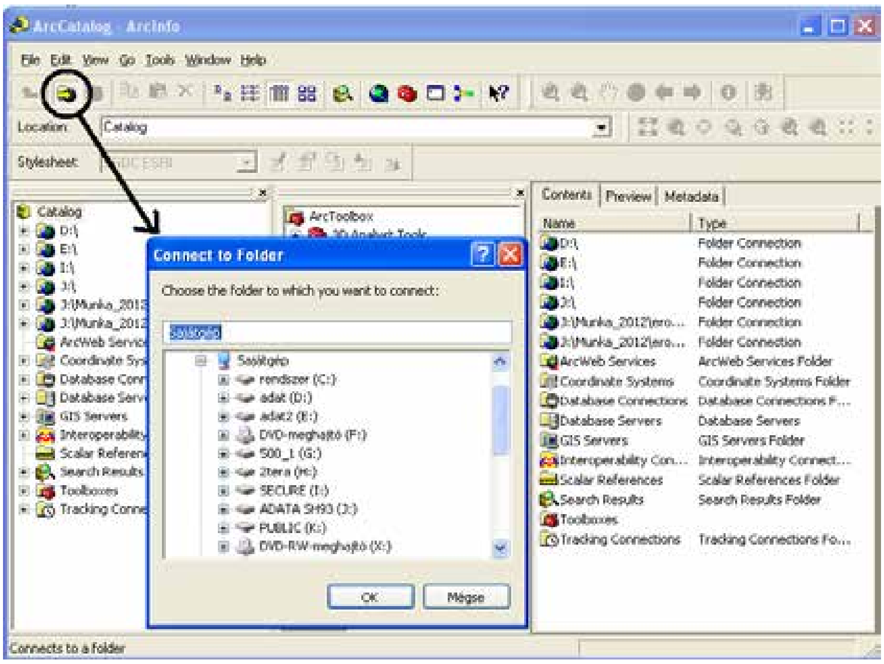


Fig. 4.9: How to find our input data in ArcGIS

3. Double click on the folder! You will find „solimap.shp” and „landcovermap.shp” vector files in it! Click ONCE on „Landcovermap.shp”! Press „Preview” in ArcCatalog. Open the vector layer in ArcMap by dragging it, and pulling it to the Layers section! Tick the layer, and it will become visible!

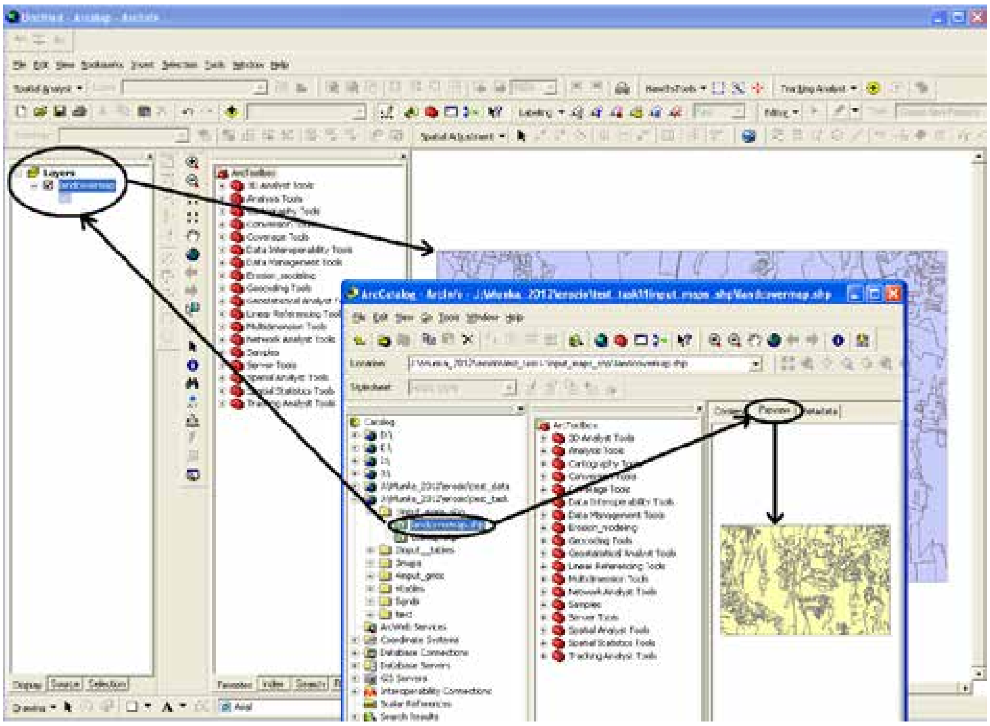


Fig. 4.10: Making a preview in ArcGIS

4. Examine the attributes of the polygons! Choose the  icon, and select a polygon on the map! Click on it! Examine the information box! As the next step, open the landcover.shp layer’s attribute table: right click on the layername, choose „Open Attribute Table” menu! Choose a polygon on the map with the selection icon , the connected row in the attribute table is highlighted! Select a row in the attribute table, the connected polygon is highlighted! Open and examine the landcover map as well!

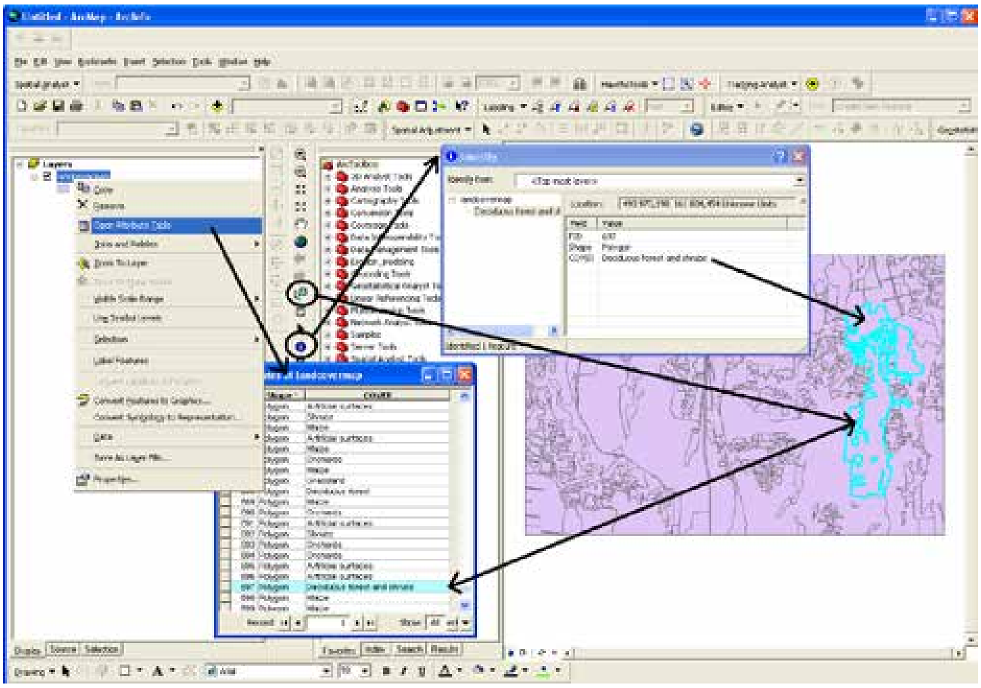


Fig. 4.11: Opening a map in ArcGIS and checking its content

5. Open and examine the „ls_grid” map the same way as you did with the vector files! You have to navigate to the folder: **erosion/input/grid/**

6. In ArcCatalog find in the folder **erosion/input/table/** „C factor.dbf” table, and press „Preview”! As you can see, in this table C factors are ordered to the different vegetation types. If you want to open the table, you have to drag and drop it to the „Layers” section (the same you did with the vector files). Right click on the tables, and open it! Open and examine „K_factor .dbf” table on the same way! Connecting the tables to the maps using the common features (vegetation type or soiltype) the factors can be joined to the vector's polygons!

Task 2.: Draw a workflow

Draw a workflow, using the information provided before! Use the symbols below! Write into the elements the name of the database or the operation by your words. Don't forget: before multiplying the data, vector layers have to be converted to grids.

	Input or output data (maps)
	Input or output data (tables)
	Operation
	Direction of procedure

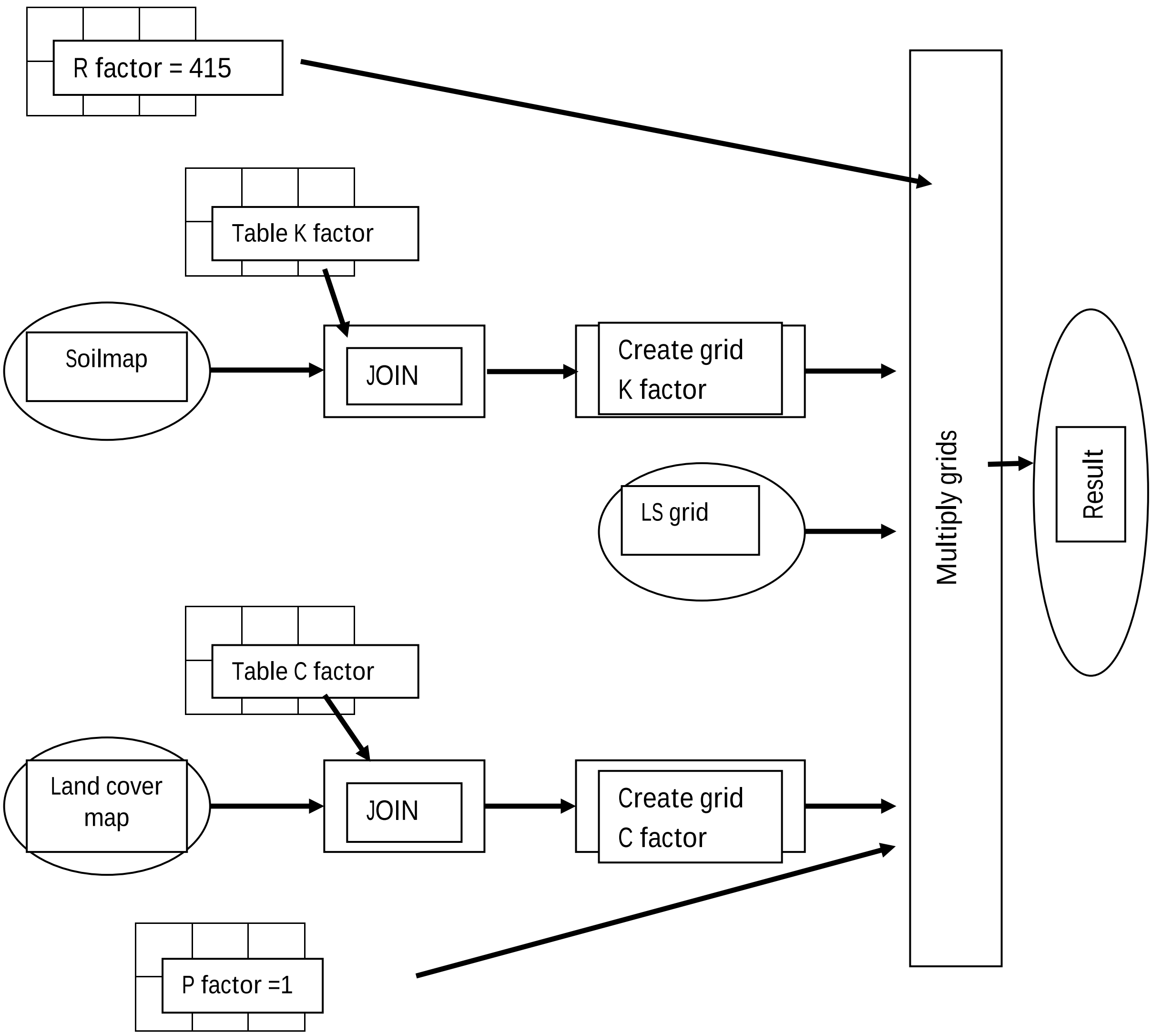


Fig. 4.12: Data workflow for calculating soil erosion with the USLE model in ArcGIS

Task 3: Join tables to vector layers

First join C factor table to the landcovermap! Open the landcovermap in ArcMap, if it is not opened yet, then open the landcover's attribute table! Open the „C_factor.dbf" table as well, as it was learnt in Task 1!

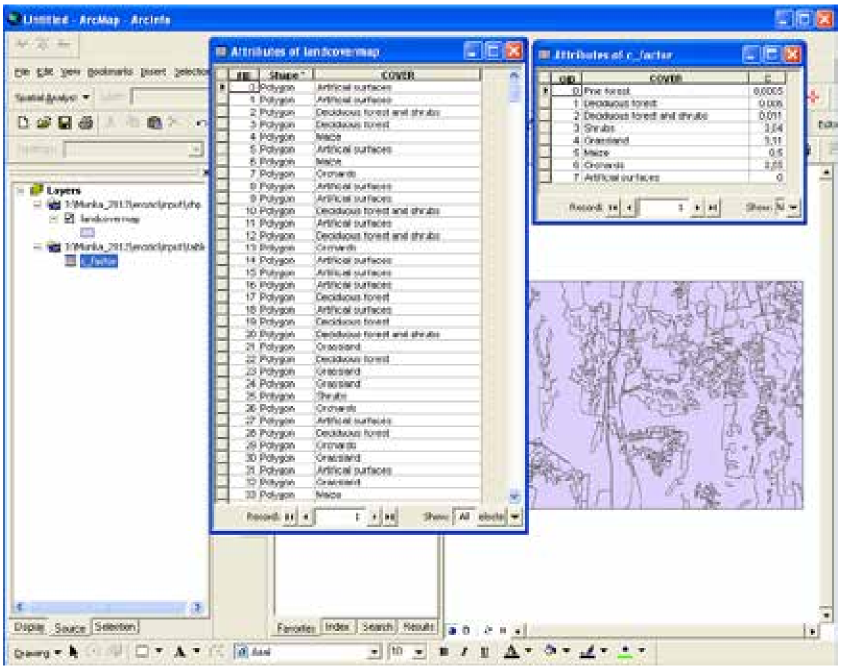


Fig. 4.13: Joining a table to a map in ArcGIS

Look for the „Join" tools in the „Options" menu in the „Attributes of landcovermap" window!

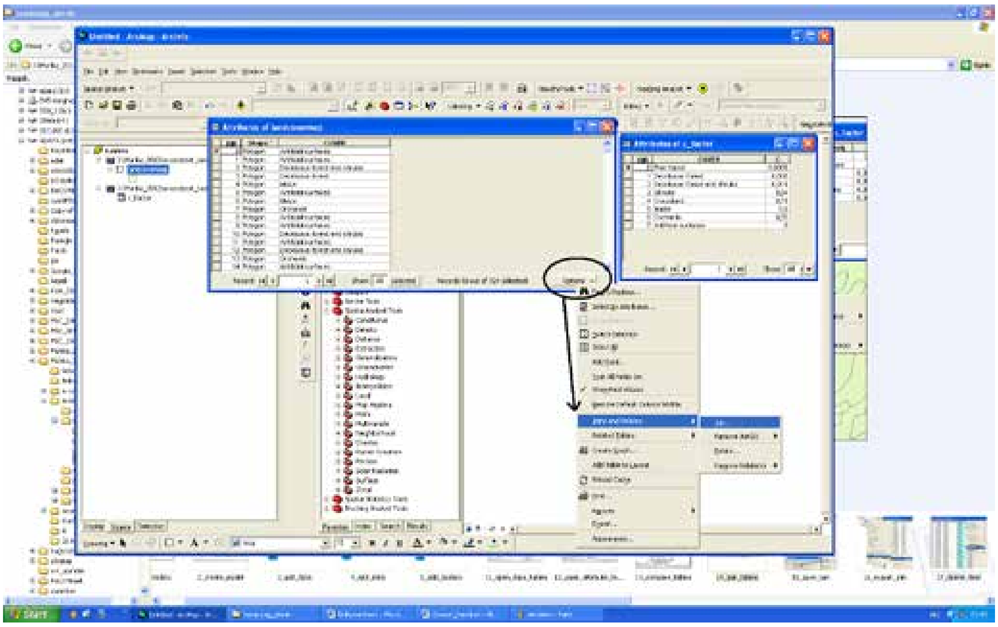


Fig. 4.14: Where to find the "Join" tool in ArcGIS

In the „Join Data” window you have to give the details of the procedure:

§ choose the field in the landcovermap layer the join will be based on,

§ choose the table to join to this layer,

§ choose the field in the table to base the join on,

§ keep all records!

Check the settings in the figure below!

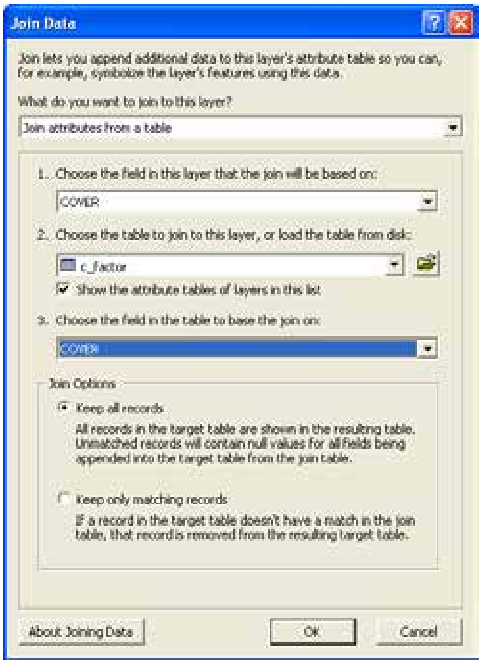


Fig. 4.15: Find and join data to a specific map in ArcGIS

The vector layer's attribute table and the „C_factor.dbf” is joined now. This join can be easily removed (Options/Joins and relates/Remove joins). If we want to keep the data joined we have to export it on a different name! Right click on the landcovermap vector layer and choose „Data/Export Data! Choose a name (we suggest landcovermap2) and the location (erosion/output/shp).

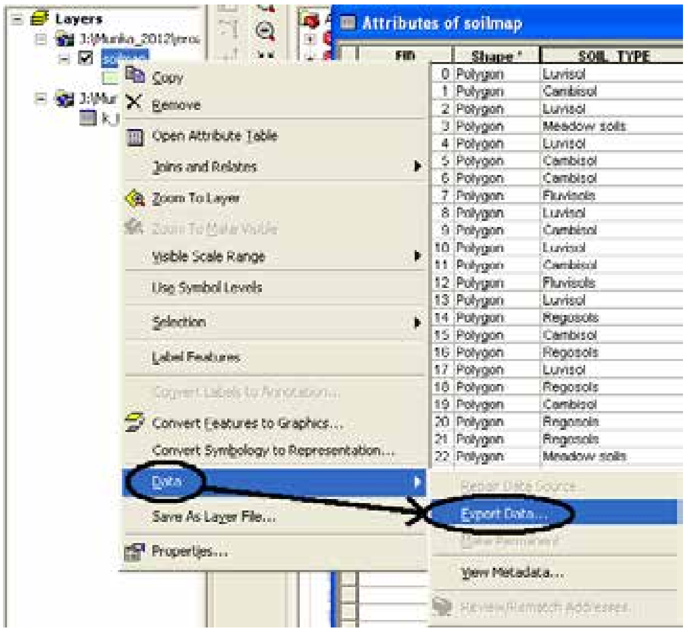


Fig. 4.16: Export data in ArcGIS

Open the created file, landcovermap2.shp, open it's attribute table, and delete those fields from the attribute table, which are not necessary! Choose the field (it will be highlighted) right click on it, and choose the option: „Delete Field"!

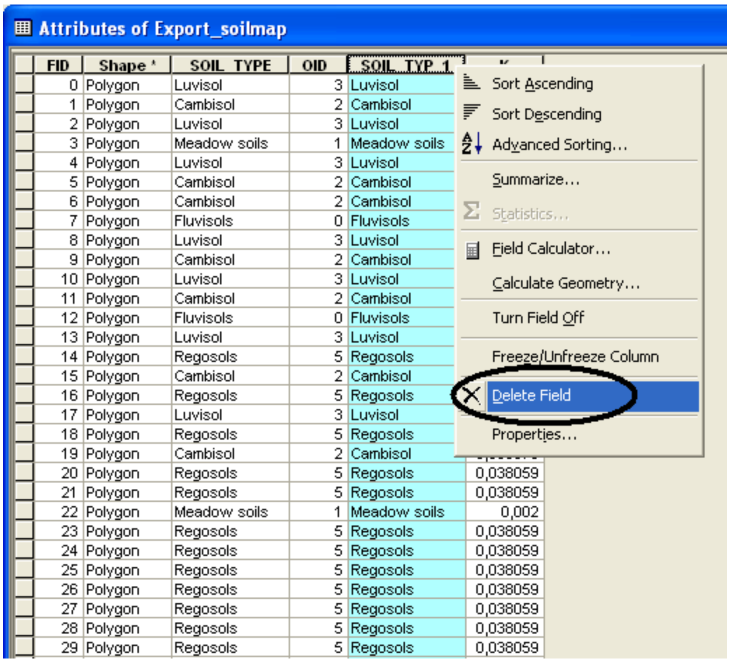


Fig. 4.17: How to delete the unwanted duplicate field in ArcGIS

Now join the K_factor.dbf table to the soilmap.shp! Save the result as „Soilmap2.shp to the folder erosion/output/shp.

Task 4.: Convert vector layers to grid

In the ArcToolbox section find the „Conversion Tools”! In this group look for „To Raster”, then „Polygon to Raster” tool! Double click on it! First convert „landcovermap2.shp” to a grid called „landcover”! Save it to the folder: erosion/output/grid!

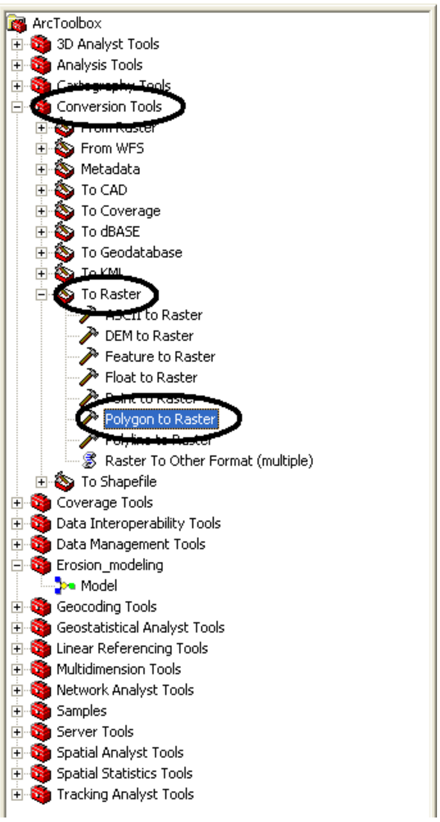


Fig. 4.18: How to make a raster from polygon in ArcGIS

In the example below you can check the parameters of the process! Pay attention to the „Value field”! This field is important, these values are going to become the pixel’s values in the grid. In case of landcover map we need „C” value, in case of soilmap we need „K” value!

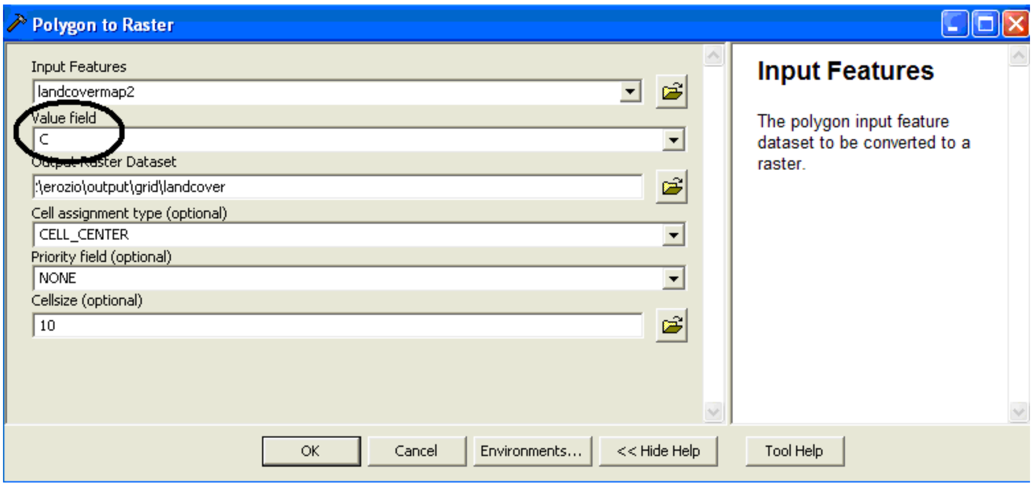


Fig. 4.19: Where to find and how to set the value field in ArcGIS

Now convert the soilmap2 vector layer to grid, name it to „soil”! „Value field” is „K”. Save it to the folder: erosion/output/grid!

Task 5: Multiply the factors using Mapcalculator

Now we arrived to the final calculation: by multiplying the factors of the USLE model, we will get the result: the amount of erosion (t/ha/year) for each pixel of the final grid.

In the below table you can see a summary of the factors we have to multiple:

$$A = R * K * L * S * C * P$$

Factor		***Source (raster data)***
Rainfall erosivity factor	R	415 MJ * mm * ha^{-1} * h^{-1} * y^{-1}
Soil erodibility factor	K	**Erosion/output/grid/soil**
Slope length factor	L	**Erosion/input/grid/ls_factor**
Slope factor	S	
Cover management factor	C	**Erosion/output/grid/landcover**
Agricultural Practice	P	1

Result: Erosion	A	Erosion/output/grid/erosion

In ArcMap open the grids signed with bold letters in the figure above! Choose „Raster Calculator" from the Spatial Analyst menu!

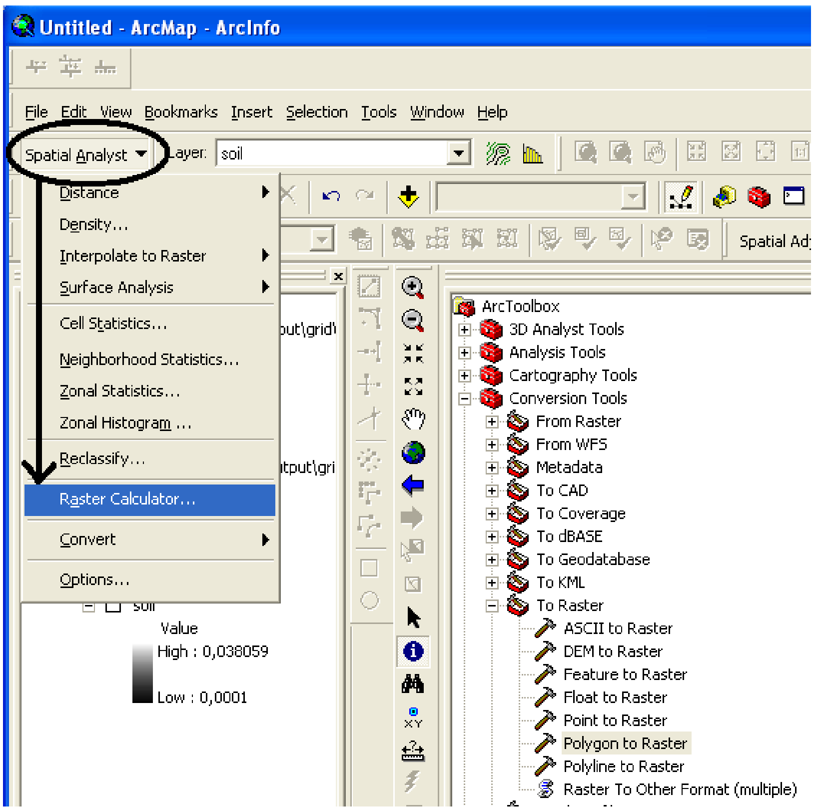


Fig. 4.20: How to open the raster calculator in ArcGIS

Build the expression by double clicking the factors in the „Layers” section of the window, and double click on the operators you want to use as can be seen below. The result will appear in ArcMap as the top most layer after evaluating the expression. It’s name is „Calculation”. It is a temporary grid so it is better to save it on a different name.

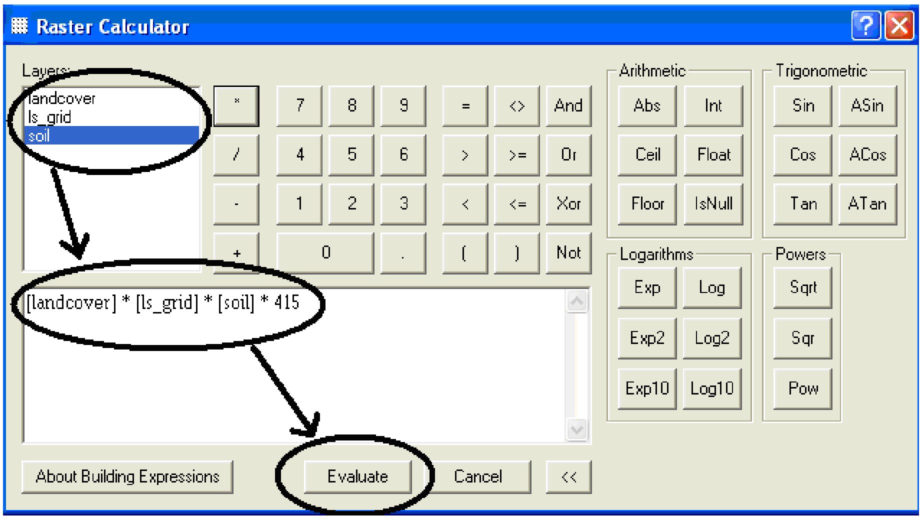


Fig. 4.21: How to make the calculations in the model in ArcGIS

Right click on the result (called Calculation), choose Data/Export Data. Follow the settings below, and export the data to the final grid called „erosion”! Use the folder erosion/output/grid!

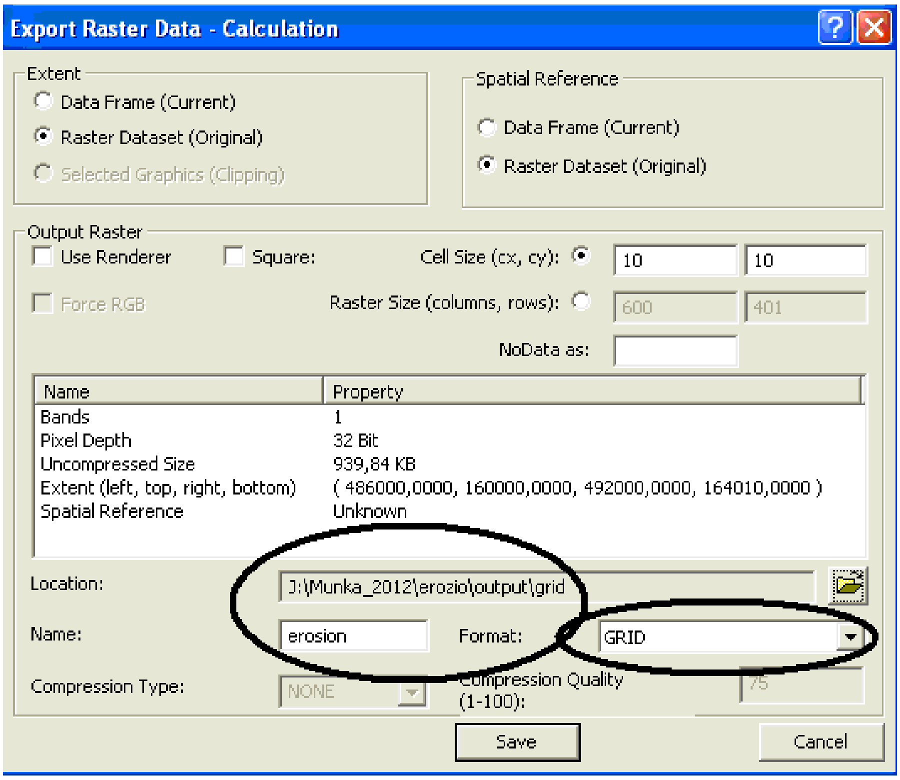
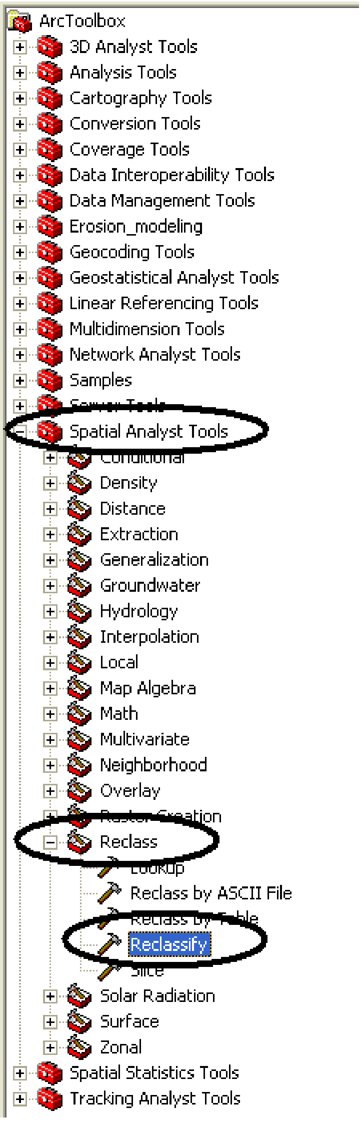


Fig. 4.22: How and where to export the results of the calculation in ArcGIS

The result is a floating point grid. We would like to classify the resulted values as follows:

- 0-2 t/ha*year: Sustainable areas. The upper limit of this class reflects the average rate of soil formation on intensive agricultural land. If average soil formation rate is about 2 t/ha*year than there is a real chance for soil degradation.
- 2-11 t/ha*year: The maximum allowable soil loss. This is the maximum rate of soil formation under optimal conditions.
- 11 t/ha*year < the predicted soil loss is above acceptable limits even for USLE. Still there is farming and farmers are using unsuitable methods of soil and crop management.

From the ArcToolbox menu choose: Spatial Analyst Tools/Reclass/Reclassify.



Set the parameters in „Reclassify” window, as you can see here. You can save the settings for the reclassification! Press the SAVE button, and give the name „Reclass” to the file. You will use it later! For the output grid choose a new filename: erosion2, use the folder erosion/output/grid and press OK!

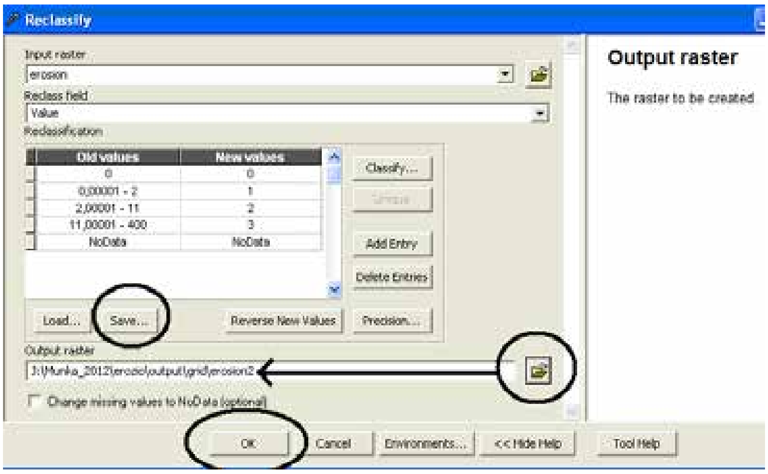


Fig. 4.23: Explanation for reclassification in ArcGIS

Task 6: Visualize the results

The result is on the screen now. Let's create a legend for it! Right click on the „Erosion2" layer, and choose „Properties" menu.

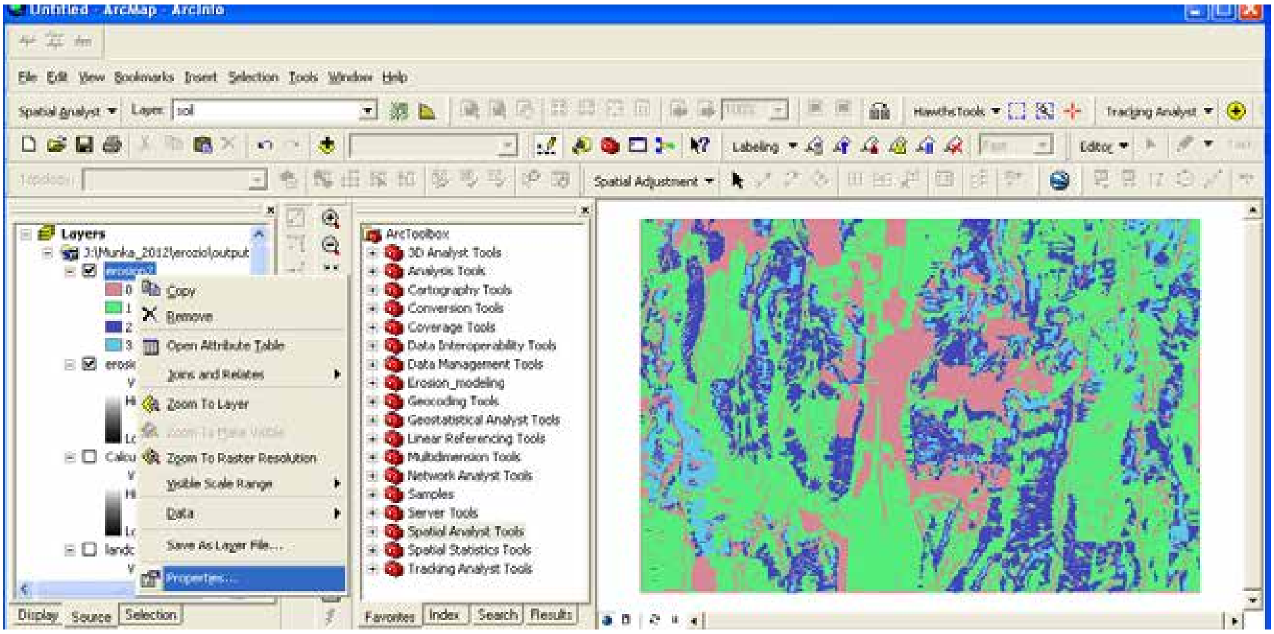


Fig. 4.24: Where to find the properties of our map in ArcGIS

Choose „Symbology"! Set the colors by double clicking on the symbols, and write explanations to the categories in the „Label" column! Press OK!

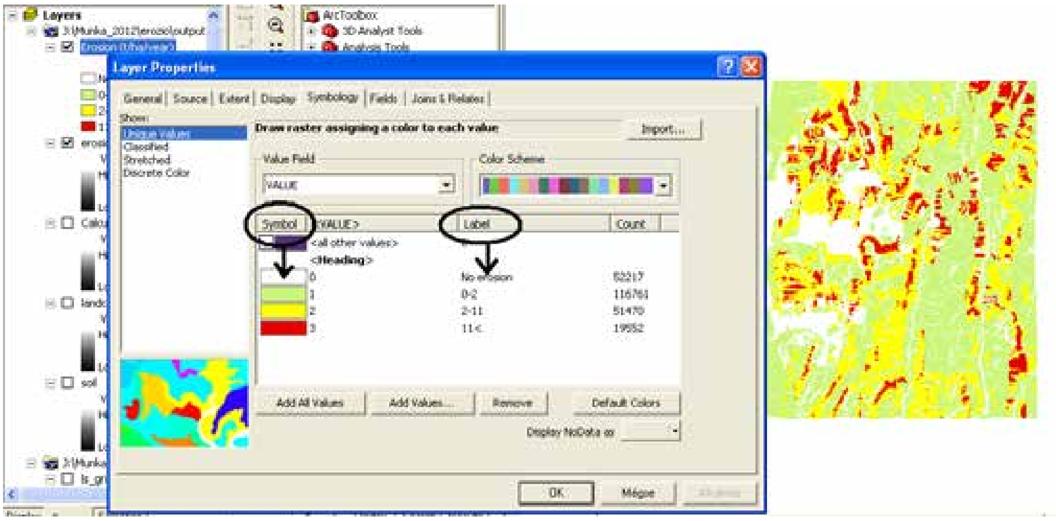


Fig. 5.25: How to set the symbols on the map in ArcGIS

Check the result in ArcMap. Now you can see where the most endangered areas are (red). We also would like to get the erosion statistics of the area. Right click on the layer „Erosion2" in ArcMap and choose „Open Attribute Table". In the column named „Count" you can see the number of pixels for each category. As you remember, the pixel size is 10x10 meters. Now you can calculate the area of the categories! Fill the table!

Category	Area (ha)
0	
0-2	
2-11	
11<	

Task 7: Build a model

If you want to calculate erosion for many areas, it is better to build a model. It is easy to change the model parameters to new ones later. We use the same dataset, but from another folder.

Data are in: **Erosion/Model/input**. Save your data to: **Erosion/Model/output**

First, in ArcCatalog you have to create a new toolbox. Name it to „Erosion modeling"!

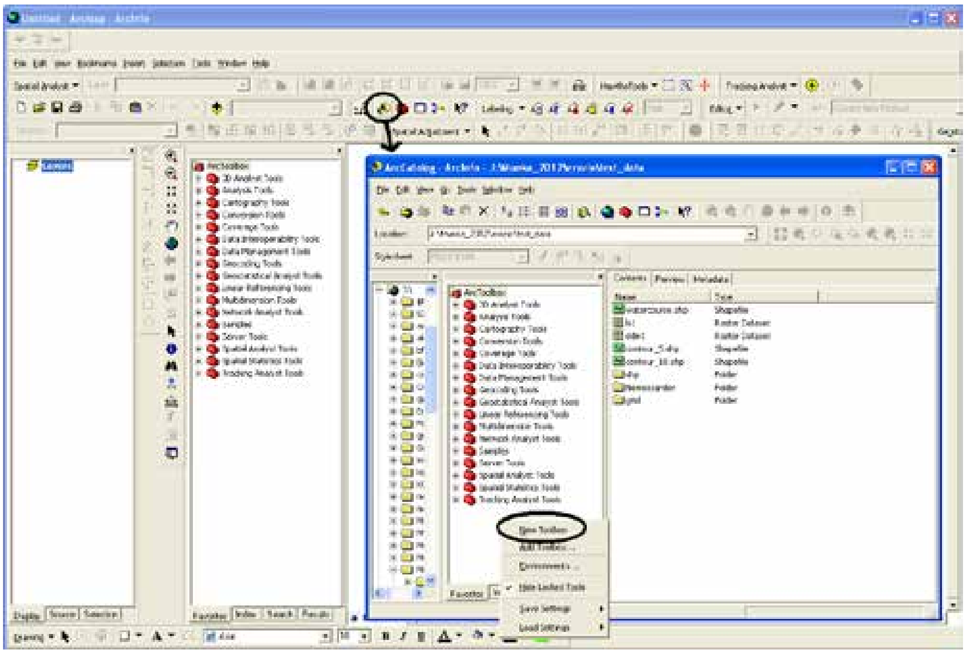


Fig. 4.26: Creating a new toolbox in ArcCatalog in ArcGIS

In the new tool create a new model! Right click on the „Erosion modeling" tool and choose: New Model!

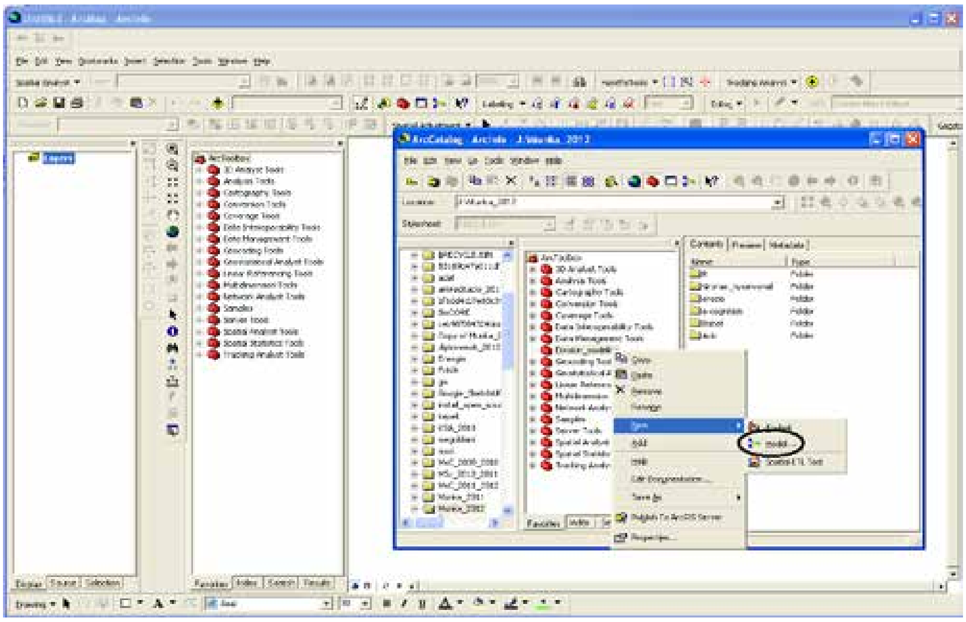


Fig. 4.27: Creating a new model in ArcCatalog in ArcGIS

In the Model window press Add Data or Tool  icon! Here you can choose the input data. Go to the „Input” folder, choose „shp” folder! Add „landcover.shp” and „soilmap.shp” to the Model. After this add „K_factor.dbf” and „C_factor.dbf” from the table folder! Add ” ls_grid” from the grid folder! Drag and drop the elements as you can see below!

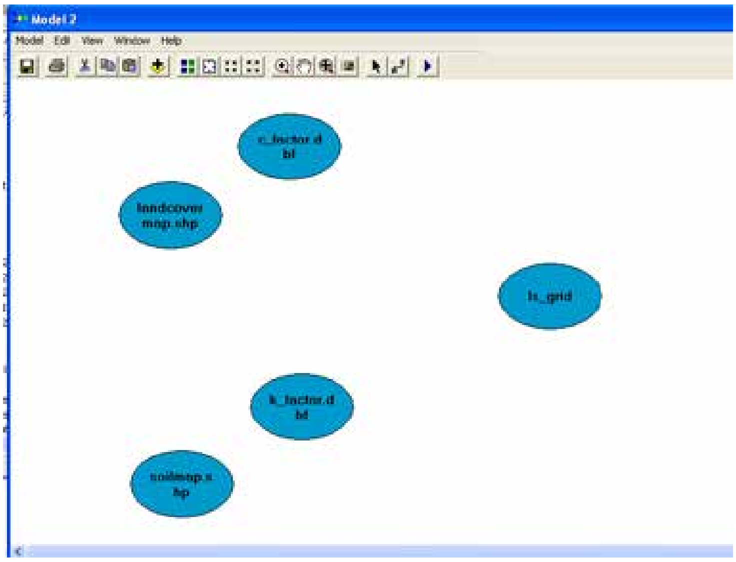


Fig. 4.28: Creating the model in ArcGIS

Now you have to join the tables to the maps. Now we give a tool to the model! Choose Add Data or Tool  icon again! Choose Toolboxes menu. From this menu choose System Toolboxes/Data Management/Joins/Join field tool. Press the ADD button! You have to add this tool twice. Double click on the first „Join Field” tool and set the parameters for landcover map! Repeat it with the tool for soilmap as well!

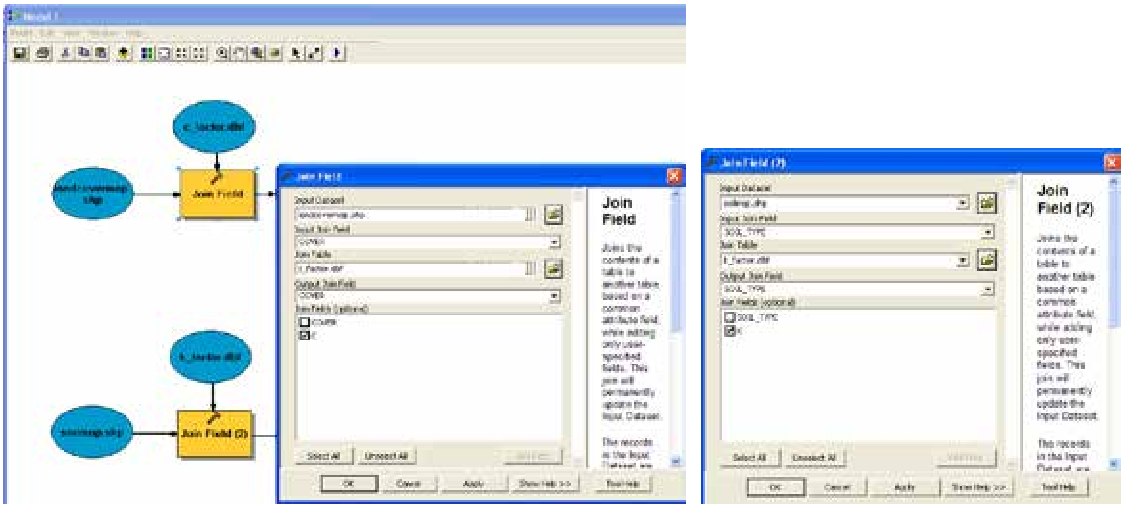


Fig. 4.29: Join tables for the map in the model builder in ArcGIS

Right click on the „Join Field” model elements and press „Run”!

Now we have to convert the vector files to grid! Choose Add Data or Tool icon again! Choose ConversionTools/To Raster/Polygon to Raster! Click: ADD! Double click on the new rectangle „Polygon to Raster”! Set the parameters, first for the landcover map! Run the processes! Repeat it with the soilmap layer!

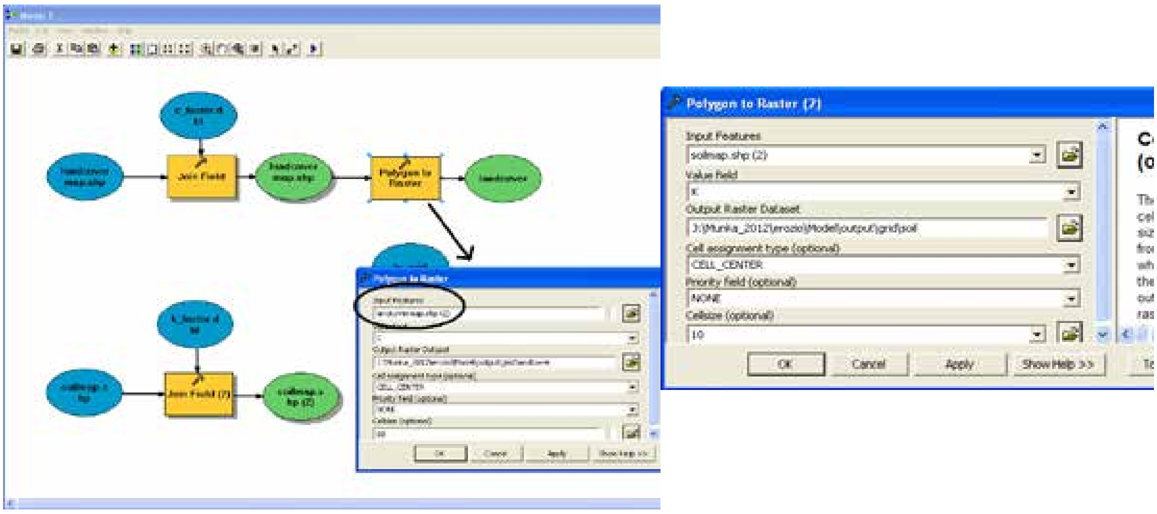


Fig. 4.30: Converting the vector files to grids in ArcGIS

As the last step grids has to be multiplied, and we also have to calculate with the R factor! Choose Add Data or Tool icon again! Look for the tool: Spatial Analyst Tool/Map Algebra/Single Output Map Algebra. Click ADD! First set the datasets’ list! You are going to use the „Input raster or feature data to show in Model Builder” section (red arrow in the figure below). After setting the list, drag a layer and pull it to the „Map Algebra Expression” window. Leave a space and press the „*” sign. Pull the next layer. Build the expression you see in the figure below!

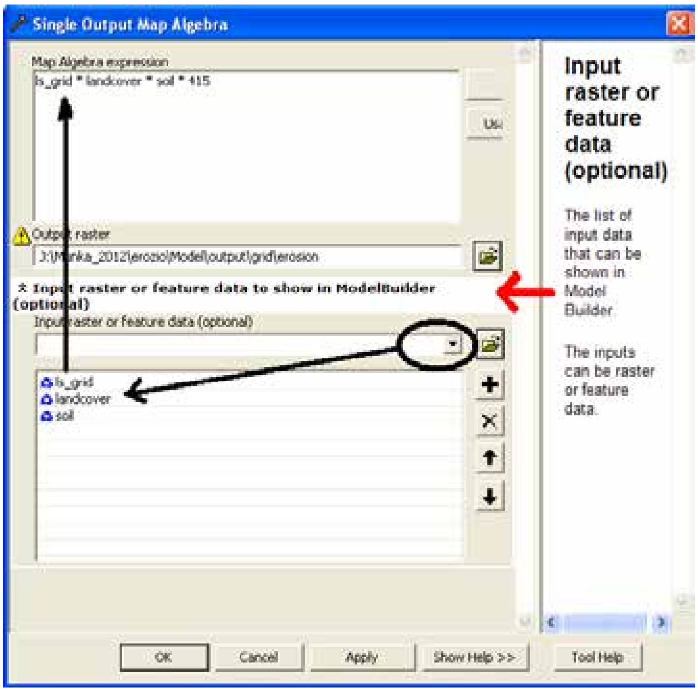


Fig. 4.31: Using the “Single Output Map Algebra” tool in ArcGIS

The last step is the reclassification. Choose Add Data or Tool  icon again! Look for Reclassify tool: Toolboxes/System Toolboxes/Spatial Analyst Tool/Reclass/Reclassify. Press ADD! Double click on the new tool to set the parameters! Now you need the reclass table you have saved before! Press the „Load” button and look for your reclass table! Set the output filename (name it Erosion2), and click „OK”!

Save the model, and run it! Check the results in ArcMap!

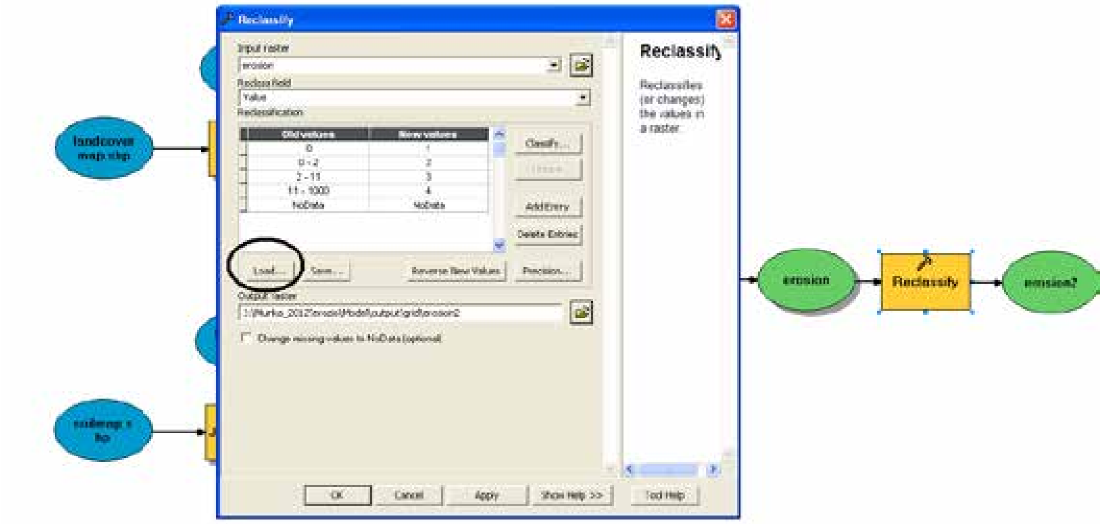


Fig. 4.32: Loading the formerly set reclass table in ArcGIS

References

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http2 http://webhelp.esri.com/arcgisdesktop/9.2/

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http://courses.washington.edu/gis250/lessons/raster_analysis1/index.html#an_props

Annex

Soil type	K factor
Fluvisols	0,0001
Meadow soils	0,002
Cambisol	0,009675
Luvisol	0,009821
Chernozems	0,018368
Regosols	0,038059

Table 1. K factor values

Land cover	C
Pine forest	0.0005
Deciduous forest	0.006
Deciduous forest and shrubs	0.011
Shrubs	0.04
Grassland	0.11
Maize	0.5
Orchards	0.55
Artificial surfaces	0

Table 2.: C factor values

Texture	**Highly eroded soils**	**Chernozems**	**Brown Forest Soil (BFS)**	**BFS with clay accumulation in B horizon**
Sand	0.45–0.55	0.35–0.45	0.35–0.45	0.40–0.50
Sandy Loam	0.50–0.60	0.35–0.45	0.30–0.40	0.30–0.40
Loam	0.50–0.60	0.30–.040	0.25–0.35	0.25–0.35
Clayey Loam	0.45–0.53	0.25–0.35	0.25–0.35	0.25–0.35
Clay	0.40–0.50	0.25 0.35	0.25–0.35	0.30–0.35

Table 3. K factor estimation for soil types (Stefanovits 1966)

CHAPTER 5

RUNOFF EROSION AND HUMAN SOCIETIES

By

Agustín Merino

THE INFLUENCE OF LAND USE AND MANAGEMENT PRACTICES ON SOIL EROSION

1. SOIL EROSION IN MANAGED SOILS

1.1 Introduction

More than 75 billion tons of soil are lost annually from terrestrial ecosystems via erosion. In Europe, almost 90 % of this type of soil loss is the result of anthropogenic impacts. Erosion mainly takes place in agricultural land and abandoned land, where losses of up to 40 Mg/ha/yr have been recorded. Erosion rates are also high in soils affected by severe wildfire, intensively managed forest plantations and construction sites (Boardman and Poesen, 2006; Verheijen et al., 2009).

Erosion increases the loss of organic matter and nutrients from soils and also reduces the water retention capacity of soils. This not only has an important impact on crop productivity, but also reduces the quality of the water as a result of the effects on sediment loads and agro-chemicals in waterways. Erosion also affects the hydrological cycle, and even the composition of air.

The Mediterranean area is considered for a long time (see chapter 3) to be particularly prone to erosion because it is subject to long dry periods (which hinders plant development), followed by heavy bursts of intensive rainfall. The precipitation mainly affects steeply sloping land, i.e. soil that is poor in organic matter and with low plant cover (Yaalon, 1997). For this reason, soil losses of 20-40 t/ha are frequently recorded after heavy rainfalls in such areas (García-Ruiz, 2010 and García-Ruiz and Lana-Renault, 2011).

Arable soils accounts for 70 % of soil erosion in Europe. Mediterranean landscapes have been subject to a large degree of erosion, especially since the 16th century, coinciding with the expansion of cereal crops and livestock farming. From the second half of the 19th century onwards, the high demographic pressure in rural areas has led to deforestation and farming of steeply sloping land. Other activities that have contributed to soil erosion include mining and the deforestation resulting from the expansion of the naval industry and railway construction. Wildfires have proliferated in recent decades, for different reasons, leading to extreme soil degradation. (Boardman and Poesen, 2006).

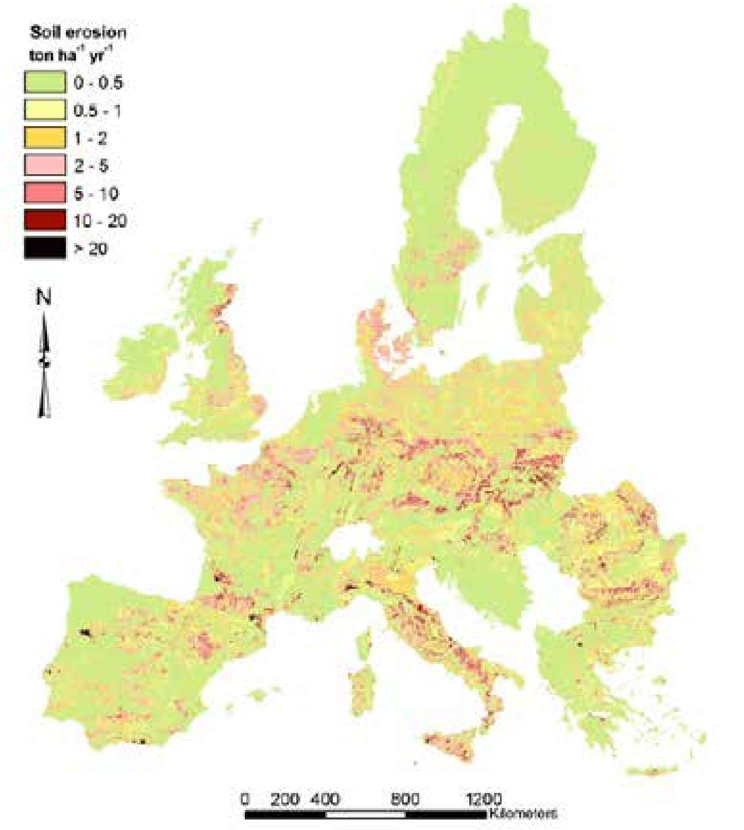

Fig. 5.1: Estimated sheet and rill erosion rates (t ha-1 yr-1) calculated for the areas of Europe covered by the CORINE database (Cerdan et al, 2010).

In many of these type of land, the soil is lost at a faster rate than the rate of renewal, which ranges from 0.3 to 1.4 t/ha/yr, depending on the factors that drive weathering (parent material, climate, land use) and dust deposition. However, in certain cases, some problems lead to the tolerable rates of soil erosion (e.g. any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur) being lower than the upper limits determined by soil formation rates. This occurs when high surface water turbidity affects aquatic wildlife or siltation of reservoirs (Verheijen et al., 2009).

1.2. Land use management, management practices and soil erosion

1.2.1 Croplands

Fifty percent of the terrestrial environment is devoted to agriculture, with one third being dedicated to crop growing and two-thirds to grassland. The first type is particularly susceptible to soil erosion (Fig. 5.2.) because the vegetation is removed and the soil left without the protective plant cover after sowing. In addition, intensive tillage reduces the amount of soil organic matter, leading to lower soil porosity and higher runoff. Land devoted to annual crops, such as cereals, is particularly vulnerable to soil erosion because of the lower degree of protection offered by this type of vegetation cover. Because of the growth cycles of these crops, the soil is left unprotected for a large part of the year, generally between October and February, coinciding with the season of heaviest rainfall in Mediterranean areas. When land is left fallow, the period during which the soil is poorly protected is even longer.

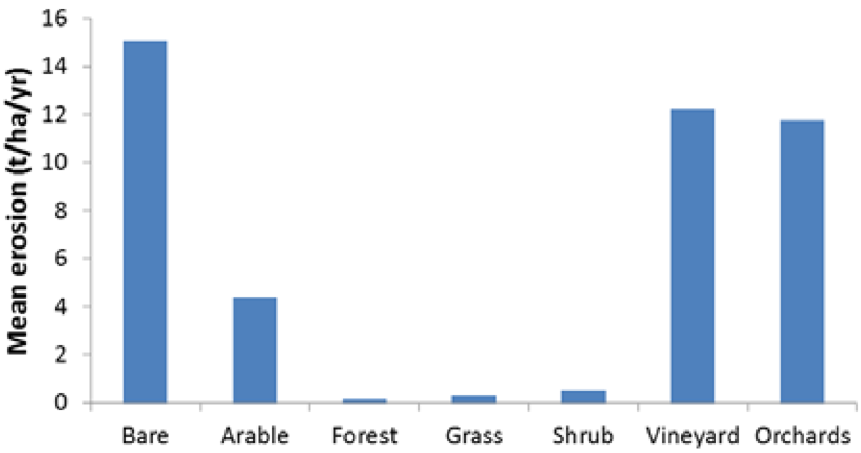

Fig. 5.2: Plot erosion rates for different land uses in Europe (Cerdan et al., 2010).

Soil conservation practices, such as reduced tillage and keeping crop residues, help maintain the organic matter content of soil, and therefore favour infiltration and reduce erosion (Fig. 5.3).

Some types of land dedicated to olives, vines and almonds also require frequent removal of vegetation by tillage or use of herbicides. All of these crops are also very important to the economy of the region and consequently occupy large areas. Due to the relief in Mediterranean areas, cultivation is often carried out on sloping land. Intensive tillage is commonly applied (some times several times per year) and herbicides used to reduce weeds. In addition, expansion of these crops on steep slopes makes the construction of new terraced fields obligatory. In this type of land, the surface soils often have extremely low contents of organic matter and gullies are often formed when the terraces are not properly designed (Durán-Zuazo and Rodríguez-Pleguezuelo, 2008).

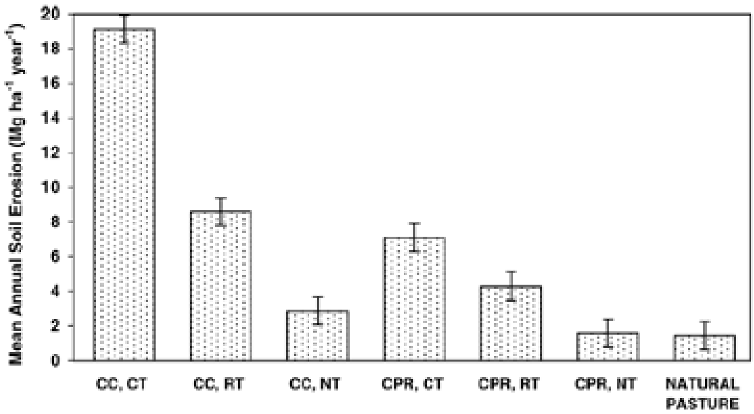

Fig. 5.3: Soil erosion of contrasting cropping systems (CC= continuous cropping; CPR: crop-pasture rotation) and soil tillage (CT= conventional; RT= reduced; NY= no tillage). (García-Préchac et al., 2004).

Another important factor leading to increased erosion rates is the change from traditional sustainable management practices to intensive management practices in which mechanization is used. This and the replacement of manure by mineral fertilisers has greatly reduced the soil organic matter content of these soils, thus reducing the porosity and infiltration rate. Compaction by machinery and the formation of soil surface crusts decrease the infiltration rate, favouring run-off, in addition to the erosion caused by water (rills, gullies, etc,) and by wind. Other types of erosion must be considered in agricultural land, i.e. tillage erosion, crop harvesting and land levelling.

Erosion of these areas also leads to high concentrations of suspended sediments and dissolved nutrients in waters. Some studies (Smith et al, 2011; Carpenter et al., 1998) have shown that nutrient losses may represent a significant proportion (10-60 %) of the amounts of nitrogen, phosphorus and potassium applied annually. Economic losses due to nutrient loss and damage to infrastructures by soil erosion and landslides may be as much as 14 % of the annual income.

Another problem detected in some Mediterranean cropland is excessive irrigation, sometimes with irrigation water of low conductivity. This leads to export of the excess water towards waterways. Agricultural activities lead to large increases in the concentrations of suspended sediment and solutes (chloride, sodium, sulphate, calcium and magnesium). In some cases the high concentrations of sulphates and calcium in the soil water cause result in piping development (García-Ruiz, 2010).

1.2.2. Land abandonment

Since the beginning of the 20th century populations have declined in many rural and mountainous areas, leading to the abandonment of thousands of hectares of land. In some areas where precipitation favours plant re-colonization of abandoned cropland, cover by shrub species improves the soil organic matter and reduces runoff and erosion. However, shrubland areas are often subject to repeated fires, and thus to soil erosion processes. In some cases the abandonment of bench terraces may also have important geomorphological and hydrological consequences. Abandonment of these structures usually leads to collapse of the walls and gully erosion, and extreme erosion rates are usually recorded (García-Ruiz and Lana-Renault, 2011).

Fig. 5.4: Gully erosion in abandoned terraces in arid climate (Burgos, Spain).

1.2.3 Rangelands

In semi-arid areas, rangelands are prone to soil erosion even under natural conditions. One of the most frequent problems is overgrazing, which leads to deterioration of the plant cover. In poorly managed livestock grazing, grass species are replaced by scattered shrubs. Cattle trails and ruts produced by off-road vehicles also channel run-off and favour the formation of gullies. The accumulation of cattle around the water sources often leads to complete denudation of the soil.

1.2.4 Intensively managed forests

In stable forests losses via erosion are very low, generally less than 1 t/ha/yr. In these ecosystems the litter layer covers the soil, providing very effective protection again the impact of raindrops and favouring infiltration. In addition, tree leaves and branches intercept the rain and moderate its effects. However, these protective mechanisms disappear in intensively managed forest plantations, which are often are cleared for production. Harvesting activities reduce surface cover and compact the soil, which increases erosion, at least temporally (Edeso et al., 1999). Perturbance due to wildfires or prescription burning also increases erosion rates.

Removal of felled logs by use of wheeled of tracked forwarders or skidders leads to a high degree of perturbance. Displacement of the ground cover and compaction may severely affect the mineral soil, especially when the soil is wet. Compaction not only increases soil erosion, but also reduces the porosity, thus affecting the productivity of the next forest rotation.

Intensive site preparation including management of logging residues and ploughing produce a large degree of disturbance and in certain cases, increased erosion. Moreover, extreme losses of soil occur when logging residues are removed or burned.

Tree harvesting also involves the removal of large amounts of nutrients. Forest soils are generally nutrient-deficient, and high loss of nutrients may occur when the intensity of timber harvesting is high.

Some measures can be adopted to prevent severe erosion. Clear-cutting, which involves the removal of all trees, and especially whole-tree harvesting (removal of stems, branches and leaves for energy biomass) should only be used in flat areas and on gentle slopes. To avoid compaction, the entry of machinery should be restricted to times when the soil is dry.

In managed forests, high rates of soil erosion occur as a result of poorly designed roads. These roads collect and channel large volumes of water, leading to severe gullying. Poorly designed logging roads may cause the loss of as much as 100 Mg/ha of soil by erosion. Cross channels or water bars should be used in logging roads to prevent excessive accumulation of water. Forested buffers along streams protect the waterways from sediments and logging debris.

Fig. 5.5: Poorly designed forest roads generating large amounts of run-off (Bizkaia and Lugo, respectively, Spain).

1.2.5 Construction sites

Sediment is often produced in construction sites (Pitt et al., 2007). Although the surface area affected is much less than in agricultural and forest land, building land accounts for a substantial sediment load production (in some countries 10 % of the national sediment load). Measurements made in these drastically perturbed areas show erosion rates that are 3 -100 times higher than in cropland.

One of the most important factors affecting the accelerated erosion rates in these areas is the high erodibility of soil that is freshly disturbed by excavation. The high K values in these soils are mainly due to the presence at the surface of a deeper soil layer with a very low soil organic matter content. In addition, the complete lack of plant protection makes these soils highly prone to erosion. The exposed material is also highly susceptible to the formation of large gullies, which can ruin pavements and foundations.

Fig. 5.6: The low content of soil organic matter of the freshly exposed soil material leads to extreme erosion rates in construction sites (Lugo, Spain).

1.4. Changes in soil properties affecting runoff and erosion

Unlike natural landscapes in which most of the organic matter produced by the vegetation is returned to the soil, in cultivated areas, large amounts of plant material are removed. Conventional agricultural practices involving extensive soil tillage leave the soil bare and unprotected from erosion. In addition, intensive tillage decreases both soil nitrogen and organic matter contents (Conant et al., 2007). soil tillage favours aeration and breaks up organic residues, making them more accessible to microbial decomposition.

Conversion of natural forest land to cropland produces rapid loss of Soil Organic Matter (SOM). The loss of organic matter due to intensive tillage has a huge influence on soil structure and soil water movement. Soil organic matter favours the formation of stable aggregates, and therefore the porosity and hydraulic conductivity in soils poor in organic matter are generally low.

The lower infiltration rate favours the generation of run-off in intensively managed cropland. Thus, when annual precipitation is more than 700 mm, run-off may exceed 24 % of total precipitation. As a consequence, soil erosion increases exponentially when the average annual precipitation is higher than 400 mm.

Surface crusts may form on bare soil and poor in SOM, causing a further decrease in surface soil porosity and infiltration. One of the mechanisms leading to crust formation is the dispersion of particles from aggregates when raindrops fall directly on the soil surface.

2. SOIL ORGANIC MATTER IN MANAGED SOILS

In managed soils the high erosion losses are closely related to soil organic matter (SOM) losses as a consequence of intensive tillage and crop production (García-Ruiz, 2010). Since this is a key factor, in the chapter we summary the main factors leading to preserve the SOM in the managed soils.

Although SOM only constitutes a small fraction of the total mass of soil, it affects the physical, chemical and biological properties of soil (Brady and Weil, 2007). It provides most of the cation exchange capacity in the surface horizons of soil, and some of its components are responsible for the formation and stabilization of aggregates. Organic matter contains large amounts of nutrients and regulates the inputs of these, especially N, to soils. Organic matter also stimulates plant growth. On the other hand, SOM plays a critical role in C balance, which is considered the most important factor affecting global warming, or the greenhouse effect (Intergovernmental Panel on Climate Change, 2007).

2.1. Influence of organic matter on soil propeerties (summary)

Organic matter (OM) has numerous positive effects on soils, and therefore maintaining the organic matter content of soil is one of the main objectives of soil management.

Influence on physical properties of soils. Humus provides dark colours, which increase the absorption of solar energy, and therefore the temperature of the soil. Humus also improves soil aggregation and the stability of aggregates. In clay soils it reduces the plasticity and cohesion, which facilitates manipulation and lessens compaction by traffic. It also favours water availability in soils, by increasing infiltration and the water retention capacity.

Influence on chemical properties of soils. The CEC of humus is between 2 and 30 times higher than that of clays, and usually provides between 20 and 90 % of the soil's capacity to adsorb cations. The organic acids associated with the humus also accelerate the release of K, Ca, Mg from mineral structures. Elements such as N, P, S and some micronutrients are stored as constituents of the organic matter until they are released via mineralization.

Some components of humus bind to metal ions to form complexes known as chelates. Some of these metals are micronutrients (Fe, Zn, etc.), which are present at low concentrations in alkaline soils because they precipitate as inorganic compounds under alkaline conditions. Reaction of these metals with soil organic matter to form chelates makes them more available to plants as they are maintained in a soluble form. In acid soils, chelates reduce the concentration of Al in the soil solution, thus reducing its toxic effects.

2.2 Amount of organic matter in soils

The amount of organic matter varies widely between different soils, from low contents in arid and sandy soils, to as much as 30 % in the A horizons of forest soils. Organic soils contain even higher quantities of OM. The organic C contents of the subsurface horizons are generally much lower than those of the surface horizons.

Influence of climate. Temperature affects the production of organic residues (plant growth) and destruction of organic matter (microbial decomposition).

Influence of natural vegetation. Climate and vegetation act together on soil organic matter. In climate zones where the natural vegetation includes forest and pasture, the organic matter content is higher in pasture areas. Pasture contains a high density of roots, which supply large quantities of organic residues to the soil.

Influence of texture and drainage. Soils that are rich in clay and silt contain more organic matter than sandy soils. On one hand, the amount of organic residues is higher in fine textured soils, in which the higher soil moisture favours plant growth and therefore the presence of plant residues. On the other hand, the higher soil moisture and the lower macroporosity reduce aeration in these soils, which in turn decreases the rate of decomposition.

Influence of cultivation and tillage. Apart from drainage of soils in humid zones, the activities that have the greatest effect on the organic matter content are cultivation and tillage. Transformation of a forest soil to a crop soil brings about a rapid loss of organic matter in the form of CO_2. This loss of organic matter is illustrated in Fig. 5.7, which shows the changes in organic matter content throughout the period of 30 years since a natural meadow was tilled and a continuous crop was planted. Even higher losses than those shown are observed in tropical forest soils, in which high temperatures favour the loss of organic matter even further.

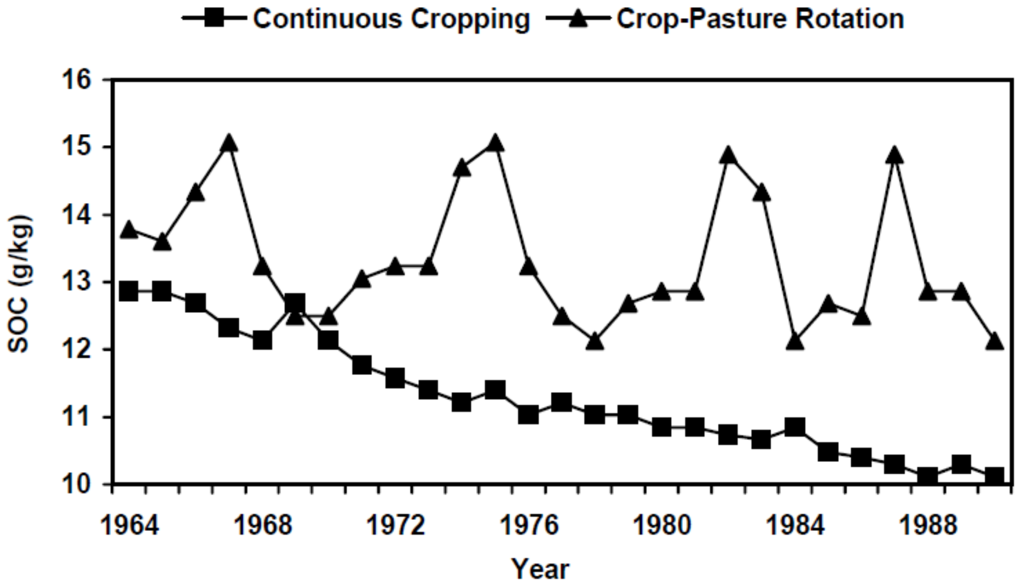

Fig. 5.7: Change in soil organic carbon concentration (0-20 cm depth) in two contrasting systems with conventional tillage (García-Préchac et al., 2004). The figure shows a) the loss of carbon after transformation of a forest soil to an agricultural soil under continuous cropping. The inclusion of a pasture helps to maintain the soil organic matter; similar effect is obtained when manure is regularly applied to the soil.

Influence of rotations, plant residues and nutrients. The sequence and type of crops, as well as the use of liming agents, fertilizers and manure affect the levels of organic matter. This is illustrated by the changes in organic matter in soils at the experimental station of Morrow (Illinois, USA; Fig. 5.8). The original soil was a forest soil, which was transformed to a cropped soil, in which different crop systems and fertilization programs were used. a) Crop system: The continuous crop with corn (CC) led to a progressive loss of organic matter, whereas in the rotation of corn and oat (CO/S) and corn-oat-hay (COH) this loss was considerably lower. A similar effect can be found in the example of the figure 5.7.

b) Fertilization: The lower crop production in the unamendment soils (UTC) led to a further SOM loss. The fertilization (manure-lime-phosphate) not only supply organic residues, but also increases crop production, which contributes to increasing the levels of organic matter via the generation of plant residues.

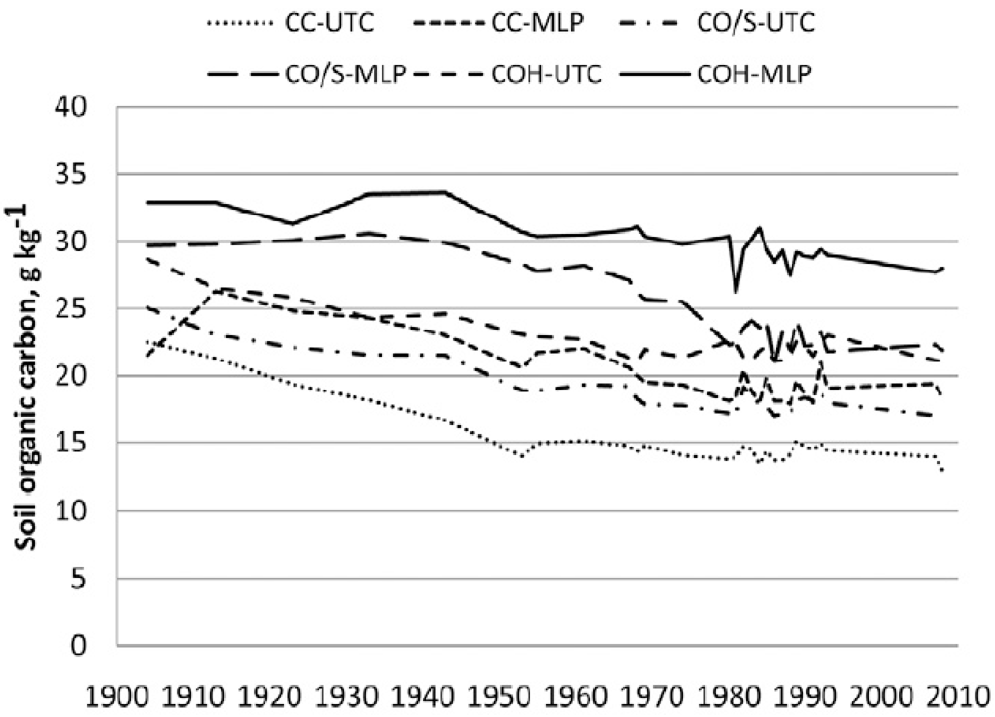

Fig. 5.8: Soil organic carbon trends over the 1904 to 2008 period in selected treatments in the Morrow plots. CC is continuous corn; CO/S is corn-oat rotation until 1967 and COH is the corn-oat-hay rotation. UTC plots have had no soil amedments, and MLP plots received manure-lime-phosphate before each corn crop during this period (Nafziger and Dunker, 2011).

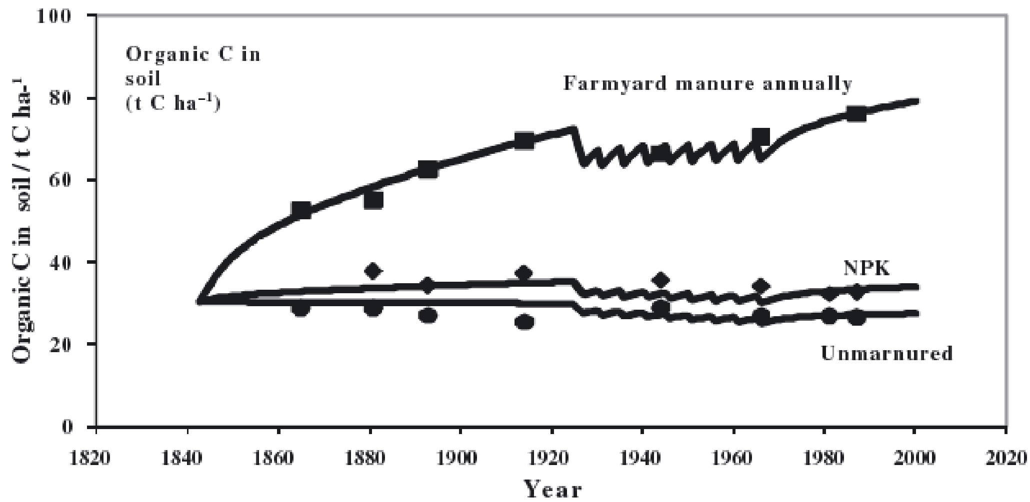

Fig. 5.9: Changes in soil organic carbon content in the plough layer (0-23 cm) in three treatments at the Rothamsted Research station (Powlson et al., 2011).

At the Rothamstead experimental station (Fig. 5.9), the original soil was a crop soil with a low organic matter content. Under continuous exploitation the soil organic matter content changed very little, even when inorganic fertilizer was added. The addition of manure greatly increased the level of organic matter until it almost reached equilibrium.

In comparison with traditional tillage, conservation practices such as mulching and no tillage leave a higher proportion of residues on the soil surface (Machado et al., 2003; Fig. 5.10).

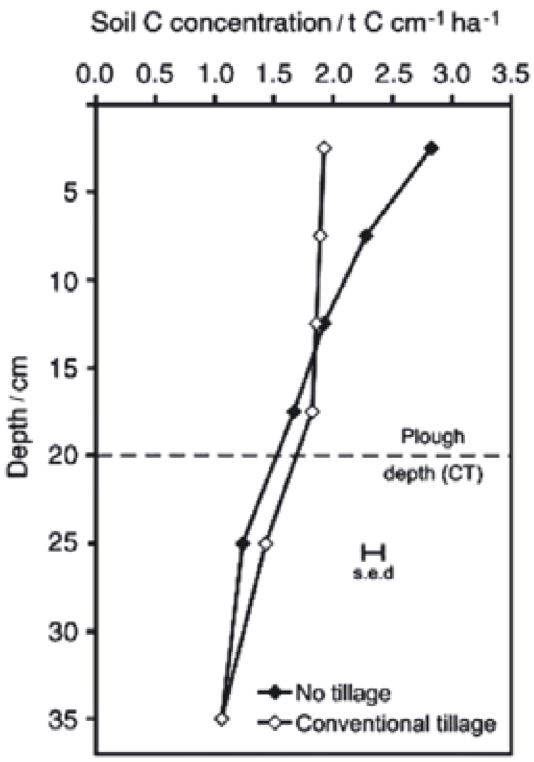

Fig. 5.10: Trends in carbon concentration with depth after 21 yr of no tillage or conventional tillage in southern Brazil (Machado et al., 2003).

2.3. Soil management and som quality

Soil organic fractions

Conservation of soil organic matter requires consideration of the amounts of SOM and also a good understanding of how the different components change over time. This is what we understand by the term ‘organic matter quality’, which can be defined as the distribution of different ‘pools’ of organic C that vary in their susceptibility to microbial metabolism (Fig. 5.11). The following three fractions are distinguished: active, slow and passive.

-The **active fraction** of the organic matter is formed by materials with a low C/N ratio and short half-life (i.e. half of the material can be metabolized in a few weeks or a few years). This fraction comprises microorganism biomass, fine organic detritus (mainly polysaccharides), non humic substances and fulvic acids. The active fraction supplies most of the nutrients required by plants, most of the readily mineralizable N and most of the structural stability, which favours infiltration of water and resistance to erosion.

-The **passive fraction** of the organic matter is formed by very stable material that remains in the soil for years, or even hundreds of years. This fraction includes most of the humus protected by clay-humus complexes, most of the humin and a large proportion of humic acids. The fraction represents between 60 and 90 % of the soil organic matter and the amount increases or decreases very slowly. The colloidal properties of humus are determined by the passive fraction, as this fraction provides most of the CEC and the water retention capacity.

-The **slow fraction** has properties that are intermediate between those of the active and passive fractions. This fraction often includes lignin-rich particulate plant tissues and chemically resistant components. The half-life of this fraction is in the order of decades. The slow fraction constitutes an important source of mineralizable N and of other nutrients.

Despite the existence of this conceptual model, there is no effective method of isolating and measuring the different organic matter fractions. The model of distribution of fractions explains why there is an initial rapid decrease in the amount of organic matter, followed by a much slower decrease, during the cultivation of forest soils (Fig 5.11). It also explains why the first effect of the loss of organic matter is the loss of structural stability, followed by a reduction in the capacity to supply nutrients. The properties associated with the passive fraction, such as the CEC and water retention capacity do not disappear from the soil.

From all of the above, we can establish a series of soil management principles that will help maintain the organic matter content of a soil. Maintaining an appropriate proportion of active organic matter in the soil is as important as maintaining the amount of organic matter. The soil should thus receive a continuous input of organic matter. The main sources of this fraction are plant remains, manure, slurry, compost and organic waste.

As tillage tends to lead to loss of organic matter by oxidation and erosion, minimal tillage and application of plant remains help maintain the levels. Greater plant production tends to increase the organic matter content, by the input of aboveground remains and roots. Because of the close relationship between the N content and organic matter, input of N is essential for maintaining adequate levels of soil organic matter. In this way, the inclusion of legumes in the systems and the appropriate use of N fertilizers tend to increase production, and therefore the input of organic remains to the soil. Likewise, some measures should be taken to minimize N loss via leaching (NO_3^-), erosion, volatilization (NH_3) and denitrification (N_2O, N_2).

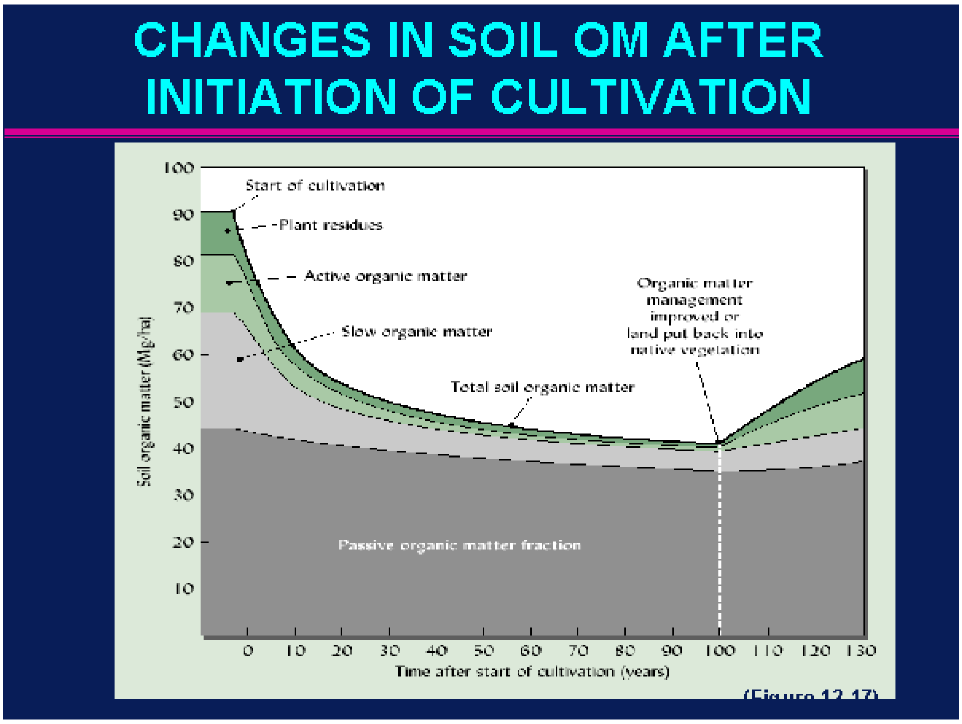

Figure 5.11: Changes in different organic matter fractions in the upper 25 cm of a representative forest soil transformed to arable soil. The soil initially contained 90 t of organic matter per hectare. The passive fraction represented about 42 t/ha (almost half of the total amount). The active fraction represented about 15 t/ha (16 % of the soil organic matter). After 40 years of cultivation, the passive fraction was reduced to 39 t/ha (i.e. a loss of 6 %). In the same time, the active fraction lost 90 % of its mass, and was reduced to 1.5 t/ha. Note how the loss of organic matter from the soil mainly affects the active fraction. Loss of this fraction explains how small changes in the organic matter content have large effects on certain soil properties such as the structural stability and N mineralization, which are associated with the active fraction of the organic matter (Brady and Weil, 2007).

2.4. Effects of agricultural activities on soil microorganisms

2.4.1 Types of soil organisms

One handful of undisturbed soil contains thousands of organisms of different types, including different vertebrate species, half a dozen earthworm species, 20-30 arachnid species, 50-100 insect species, dozens of nematode species, hundreds of fungal species, and possibly thousands of bacterial species, including actinomycetes.

This wide variety is possibly due to the almost unlimited variety of sources of energy and the wide range of habitats within soils. A single handful of soil will contain areas with good and poor aeration, low and high acidity, different temperatures, different moisture contents, areas with higher and lower concentrations of nutrients and of organic substances, etc.

Microorganisms (bacteria -including actinomycetes- fungi and algae) are the most numerous type of organisms in soil and make up the largest proportion of live biomass in soil. Earthworms and microflora also contribute to the biological activity in most soils. It is estimated that 60-80 % of soil metabolism is directly due to the activity of microflora, although the activity is favoured by the actions of microfauna (protozoans, nematodes, collembola and earthworms).

2.4.2 Beneficial effects of microorganisms

Soil flora and fauna are essential for soil functioning because of their effects on production and the ecological functions of soil. The following are the most beneficial effects:

Plant tissue's decomposition. Degradation of organic matter is accompanied by release of nutrients and organic substances that help in assimilation of the nutrients and in stabilization of the structure.

Degradation of toxic compounds. This mainly occurs in the surface horizon. The soil receives different types of toxic compounds, some produced by the microorganisms themselves, and other added via agricultural or industrial activities. Some bacteria and fungi play an important role in degrading toxic substances, and are therefore used in strategies aimed as recovering contaminated soils (biorecovery). In fact, some species of fungi accumulate extraordinarily high amounts of heavy metals, and are used as indicators of soil contamination.

Inorganic transformations. Mineralization of soil organic matter, mainly due to the action of microorganisms, provides most of the N, S and P ions available to vegetation (as NO_3^-, SO_4^{2-} and PO_4^{3-}). Likewise, microorganisms can reduce the uptake of Fe, Mn, Se and Cr by chemical transformation of the oxidation state of the metal ions.

Nitrogen fixation. In forest soils, N is fixed via the action of actinomycetes of the genus *Frankia*; in flooded soils it is fixed by the action of cyanobacteria, and in agricultural soils by the action of bacteria of the genus *Rhizobium*. Most N fixation takes place in nodules or in association with plants.

2.4.3 Effects of soil management on soil organisms

Environmental changes affect both the amount of soil organisms and their diversity. In this way, deforestation modifies both the soil moisture content and temperature and reduces the input of organic residues to the soil, which is usually accompanied by a reduction in the amount of microorganisms. Monoculture leads to less variability in plant residues and environmental conditions than in forest or pasture soils.

Agricultural activities have different effects on different organisms. Some practices, such as extensive monoculture and tillage, reduce the diversity of organisms and populations (Table 5.1). Liming and the addition of fertilizers or manure to infertile soils potentiates the activity of microflora. Low tillage tends to increase the role of fungi at the expense of bacteria. The addition of pesticides may drastically reduce the number of soil organisms. However, the application of some pesticides favours growth of specific microbial populations that can utilize the components as a source of energy.

However, in some cases the application of different products may have rather unpredictable effects. Thus, in some areas the application of insecticides was found to lead to a lower rate of litter decomposition. This occurred because death of the insects led to proliferation of nematodes that feed on bacterial decomposers.

	Fungi	Bacteria	Microarthropods	Earthworms
No tillage	360	260	1.31	60
Tillage	240	270	0.49	21

Table 5.1: Effects of tillage on the microbial biomass C and on soil faunal groups (data expressed in kg C/ha).

3. SOIL PHYSICAL PROPERTIES AND SOIL CONSERVATION

The physical properties of soil deeply influence the functions of the soil in the system and how the soil can be managed. These properties determine not only the movement of water and solutes through soil and the susceptibility of the soil to erosion, but also the growth of crops. In this chapter the main knowledge protecting the soil against the physical properties degradation is reviewed. For the main concepts on physical properties specific handbooks (Hillel, 2003; Brady and Weil, 2007) are recommended.

3.1. Texture, structure and porous space

Soil texture is the relative proportion of sand, silt and clay particles in a mass of soil. Soil texture determined the amount and size of porous, and therefore determined directly the hydraulic properties of the soil. Clay soil drains slowly and tends to remain saturated. Clay compacts easily making it difficult for water to infiltrate and for root to penetrate. On the contrary, sandy soil drains quickly, and as a result, plant are not given enough time to absorb water and nutrients.

Texture classes do not change easily in the field. The texture of given soil only can only be modified by mixing the soil with a different texture. For example, the incorporation of a large amount of sand to clay soil for use in a greenhouse.

The addition of organic or peat does not alter the texture of the soil, which refers only to the mineral particles. In the short term, soil management practices do not change the texture of the soil. However, a long or medium scale soil texture can be modified by erosion, clay illuvium or alteration of minerals.

The term structure refers to the grouping of particles of soil into aggregates. The structure changes important soil conditions such as the movement of water, the transfer of heat, aeration, porosity. Managements on soil that alters its structure are tillage, lime, drainage and the contribution of organic fertilizers.

The structure of the soil formation is favoured by the following factors:

a. the push or compression of aggregates by the roots of plants during their development

b. the expansion and retraction of some clays (phyllosilicates 2: 1) when they are wet or dry

c. the action link by the hyphae of fungi

d. the secretion of sticky organic compounds by the roots of plants and microorganisms.

e. action of earthworms

f. the presence of clays silicates, oxides of Fe and Al, soluble salts and CaCO3

g. the bond for "bridge" cations (Ca, Mg).

The size and configuration of the porous space enters the solid particles is crucial for the growth of the roots of plants and the movement of the air, water and solutes into the soil.

The pore space of a soil is the ground occupied by air and water volume. The amount of pore space is closely related to the Organization of the soil particles. If the particles are very close, as in a compacted soil, the porosity is low. If these particles are arranged in aggregates, as in soils of medium texture and rich in organic matter, the pore space will be large.

Pore Size

In the soil, there are two types of spaces, the macro and micropore. The size that separates both is around 0.06 mm in diameter. In most of the pores the micropore are in the interior of the aggregates and the macroporos between the aggregates. In this way, the structure, along with the texture determine influence macro and micropore distribution.

The macroporos of characteristic shape allow easy movement of air, as well as the percolation of water. On the contrary, the micropore are usually filled with water, so do not allow the exchange of air between the inside and outside of the ground. Through these, also the movement of water is slow. In this way, although a sandy soil has lower porosity than texture intermediate or clayey soil, the movement of water is very fast, due to the dominance of macroporos.

On the contrary, fine textured soils, allow a slow movement of water and air, despite the high proportion of pore space. In these soils aeration is often inadequate for the development of roots and microbial activity.

The pore space varies widely between soils. Sandy soils have a range of pore space between 30 and 50%, while the soils of fine texture, between 40 and 60%, and even more in the soils with high organic matter content.

Bulk density

The bulk density is the mass of a unit volume of dry soil. This volume includes both solids and porous spaces. The determination of bulk density is knowing the dry weight of a quantity of soil from which its volume is known. One of the reasons for measuring bulk density is that this value can be used to calculate the pore space. The pore space is inversely related to the bulk density. Thus, in soils of similar texture, the greater is the bulk density, minor is the pore space. Porosity is calculated in the following way

% Pore = (1-(DA/DR)) * 100, where DA and DR are the apparent density and real density, respectively.

Soils with a high proportion of pore space have values of apparent density lower than soils that are more compact. Consequently, any factor that influence the pore space affect the apparent density.

Fine textured soils gernerally have a lower bulk density than sandy soils. The bulk density of clay soils, clay loam and silt loam typically range between 1.00 g cm^{-3} and 1.55 g cm^{-3}, depending on its condition. Most frequent values of sandy soils range between 1.20 g cm^{-3} and 1.75 g cm^{-3}. Very compact soil reach values of 2.0 g cm^{-3} or even higher. In these compact soils the macropores have disappeared, therefore the roots present limitations for its development. This impediment takes place from values superior to 1.6 g cm^{-3}.

Subsurface horizons typically present values of bulk density above the surface. This is due to the lower percentage of organic matter, lesser degree aggregation and penetration of roots.

3.2. Soil compaction

Compaction is the increase in the density of a soil as a result of applied loads or pressure. In nature, the compaction can also take place through natural processes, such as the precipitation of carbonates or crusting of internal Fe. Tillage operations, especially those that use heavy machinery, tend to break the aggregates and to encourage soil compaction. The bulk density of the soil can be modified by the system of tillage used. Bulk density increases normally mean a worsening of the properties for the growth of roots, as well as undesirable changes of soil hydraulic properties. Soil strength is also used as a measure of soil compaction because it reflects soil resistance to root penetration. Extensive reviews about soil management and compaction are given by Hamza and Anderson (2005) and Batey (2009). In this chapter selected studies are shown.

Mechanical operations and soil compaction

Trafficking by wheeled farm machinery is common in most agricultural operations. Soil compaction by wheels is characterized by a decrease in soil porosity in the zone beneath the wheel and rut formation at the soil surface. Wheel load, tyre type and inflation pressure play an important role in soil compaction (Horn et al., 2001). An indication of the effect of the heavy machinery on the bulk density and soil penetration resistance are shown in Figs.5.12 and 5.13, respectively.

After cutting forest soils are also subjected to important compaction (Fig. 5.14). This is favored by mechanical harvesting, using heavy tractors or specialized felling (harvester) and logging (skidder, forwarder) machines.

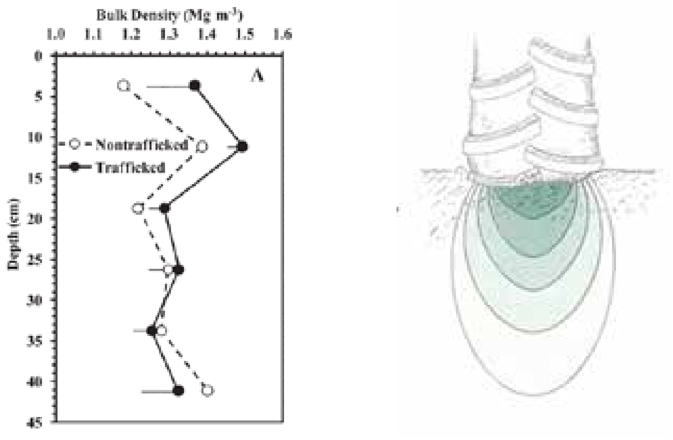

Fig. 5.12: Tractors and other heavy equipment compact the soil to a considerable depths, increasing bulk density and reducing plant and crop yields. The effect are especially damaging if the soil is wet. Depth distribution of soil bulk density under non traffiched and trafficked row (Blanco-Canqui et al., 2010).

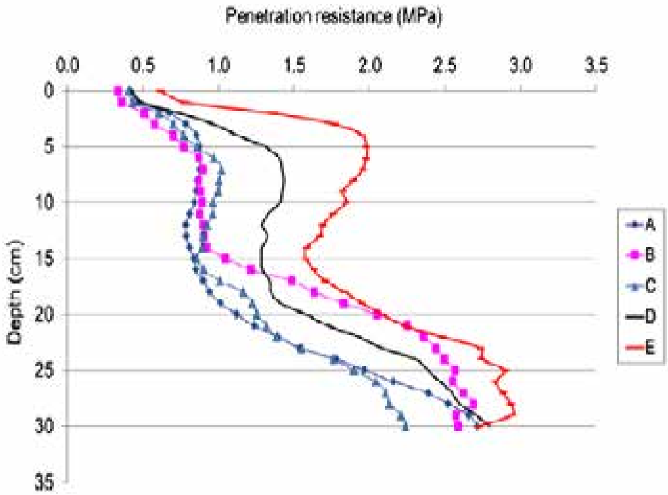

Fig. 5.13. Soil penetration resistance A = low pressure with press wheels, B = normal pressure, C = high pressure, D = tractor traffic, one pass, E = tractor traffic, three passes (Arvidsson et al., *2012).*

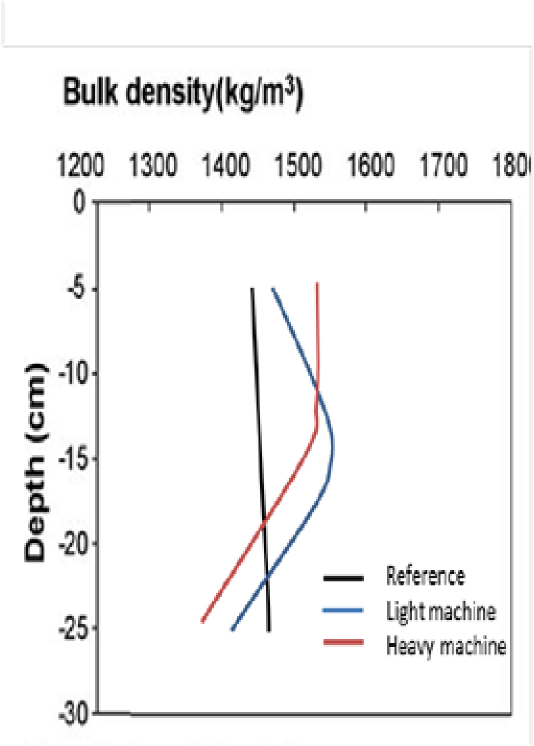

Figure 5.14: This graph illustrates the increase in bulk density, especially of the surface horizon, that occurs after logging as a result of the machinery (Ampoorter et al., 2010).

Influence of soil water content on soil compaction

Soil water content is the most important factor influencing soil compaction processes. The most important compaction damage occurs when the soil is wet. Thus, the increase in bulk density of soil is due to the effort of compaction and the water content in in the soil. Figure 5.15 shows the following:

a) the density of soil increases exponentially with an increase of applied force,

b) the force required to compact a soil decreases exponentially with the increase in humidity. Under a given constant effort, the effect increases with increasing humidity. It increases up to a maximum and then decreases. This maximum value is used in engineering to compact the soil, but for agriculture it is the most negative value.

c) the moisture content of "optimal" decreases as the compaction force grows.

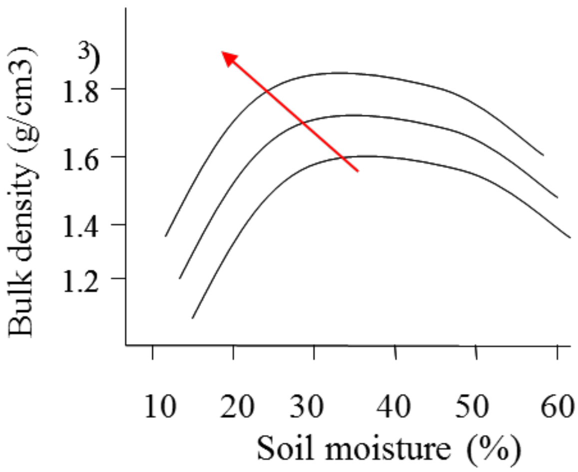

Fig 5.15: The increase in soil bulk density is because both of the effort of compaction as well as the water content in the soil.

<u>Effect of tillage on the size of pores</u>

Tillage has favourable and unfavourable effects on aggregation. If during tillage soil is not too wet or dry, the short-term effect is usually beneficial. Tillage breaks aggregates, incorporates organic matter to the soil, kills weed seeds and creates favorable conditions for germination or plantation. In addition, it reduces evaporation and enhances mineralization of organic matter, releasing nutrients.

In the long term, however, tillage operations have a negative effect on the structure of the soil. Firstly, during tillage soil mixture encourages oxidation of organic matter, reducing in this way the positive effects of this component on the structure of the soil.

Continuous cultivation, especially on soil which was originally high in organic matter, often leads to a reduction of the area of macropores. Data from table 6.2 illustrates this effect. The cultivation with tillage reduces the content of organic matter, as well as the total pore space. The most marked effect is, however, the effect on the size of pores: the amount of macroporos, necessary for the movement of water and air, is reduced to half the original value. This effect tends to reduce the infiltration and to restrict root growth.

History of the land	**SOM (%)**	**Total Porous Space (%)**	**Macropor. (%)**	**Micropor. (%)**	**BD (g/cm^3)**
	0-15 cm				
Uncultivated	5.6	58.3	32.7	25.6	1.11
Cultivated (50 years)	2.9	50.2	16.0	34.2	1.33
	15-30 cm				
Uncultivated	4.2	56.1	27.0	29.1	1.16
Cultivated (50 years)	2.8	50.7	14.7	36.0	1.31

Table 5.2: Effect of continued tillage on the porous space, distribution macro/micropores and bulk density. Note that the effect is most evident in the decline of macropores in the total porous space (Brady and Weil, 2007).

In recent years, many farmers are adopting **conservation practices** that reduce tillage and the handling of the soil. These practices keep crop residues on the soil surface. They tend to reduce the soil bulk density and increase the macroporosity.

Modern advances on herbicides and planting equipment allow adventitious vegetation control and the establishment of non-tillage vegetation, however, many producers still consider tillage as standard practice in many soils.

Plow pan

Some practices, such as the mouldboard plow, compact the soil in depth. The continued use of these practices promote the development of a dense layer below the tilled layer, called plow plan (plow pan, Fig. 5.16). The employment of rippers tend to loosen the soil in depth, although in some cases the effects are not lasting (Fig. 5.17).

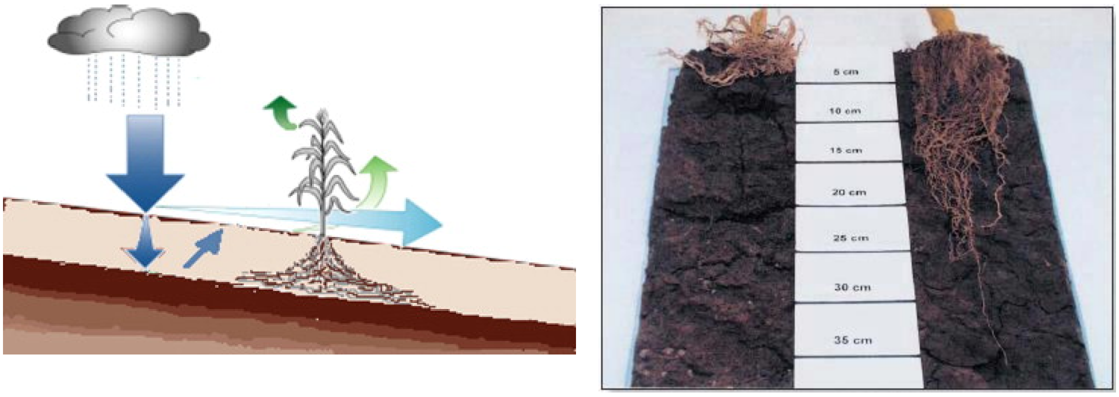

Fig. 5.16: A plow pan prevent the develop of root and the water infiltration in the soil.

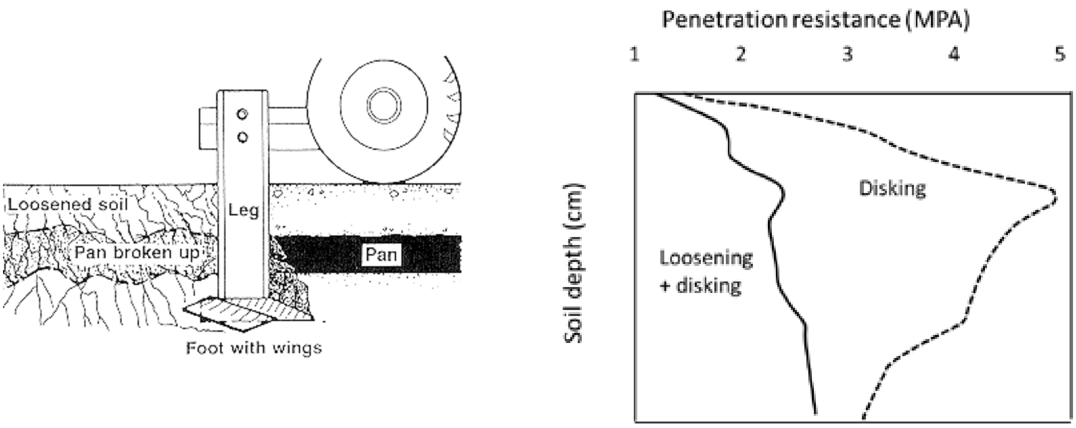

Fig. 5.17: Subsoiler can break up a plowpan reducing the resistance to penetration. However, the difference between the subsoiled area and the non subsoiled disappears over time. (Birkás et al., 2004).

3.3. Stability of aggregates: soil crusting

Stability of aggregates of soil is of great practical importance. Some aggregates are easily destabilised by the dripping of rain and by the action of the tillage. Other aggregates, on the contrary, resist disintegration, so they keep the structure (Fig. 5.18). The three main factors that are more important influences on the stability of the aggregates are:

A.- The mechanical action in the short term of microorganisms, particularly the hyphae of the fungus. The filamentous fungi that live in association with the roots of the plants are especially important for the formation of large aggregates (> 2mm) that appear in the non-cultivated soils (table 5.3).

B.- The action of cementing the intermediate products of the synthesis of microorganisms and decomposition. In this way, the bacterial polysaccharides meet clay particles. The hairs of the roots and the hyphae of fungi also produce polysaccharides that help unite particles of soil.

C.- The action of cementing some components of the humus, aided by the similar action by some inorganic compounds such as iron oxides. These components are mainly responsible for the stability of the smaller aggregates(< 0.25 mm).

Additionally, to be considered is the action of inorganic elements that favor the flocculation (Ca, Mg, Fe, Al) and cementation ($CaCO3$, oxides of Fe).

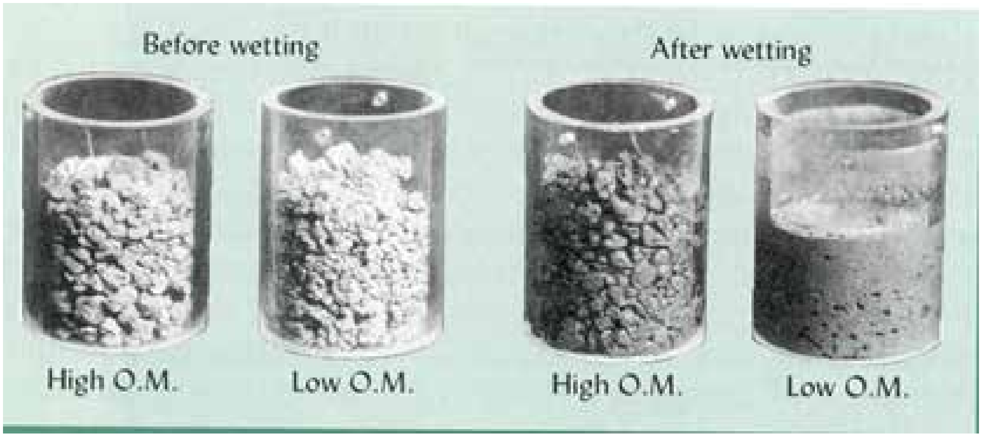

Fig. 5.18: The aggregates of soil with high organic matter content are much more stable. When the soil has low organic matter content, soil aggregates collapse, forming a layer of lower porosity. When this occurs under field conditions, it produces a surface crust (Brady and Weil, 2007).

The measure of stability is provided by wet sieving. This is determined, through weight, the amount of soil that remains in a sieve after flooding.

Table 5.3: The data in the following table illustrates the importance of different crop systems to maintain the stable aggregates. The injurious effect of the intensive cultivation it is clear (Shepherd et al., 2001).

	Water-stable aggregates (%)	
Crop	**Big (> 1mm)**	**Small (< 1mm)**
Pasture	52	48
Corn (11 yr)	15	85
Maize (11 yr) + Pasture 10 yr)	39	61

Soil crusts

Aggregates exposed on the surface of the soil are very vulnerable to destruction by rain. The smallest particles tend to enter the pores and seal them. Deposition of these particles within the pore spaces creates a compacted layer, of a few millimetres thick, at the surface. Other mechanisms of soil crust formation include compaction by the impact of drops and the deposition of fine particles via overland flow.

The result is the formation of a sealed surface layer that prevents infiltration (Fig. 6.19). Soil crusts are rather non porous and impermeable, with the subsequent implications for runoff and erosion. Furthermore, the presence of crusts greatly reduces the ability of seeds to germinate, thus prolonging the time during which the soil remains unprotected.

In some areas the soil crusting and sealing of the soil have disastrous consequences related to the low infiltration. On the one hand this phenomenon, favours the amount of water runoff and erosion power. In arid or semiarid soils, it reduces the amount of water available to crops. In this situation the emergence of seedlings is hampered, sometimes in their entirety. The crusting after planting allows only a small number of seedlings to emerge, so it is generally necessary to perform a new planting.

Fig. 5.19: Photograph of soil crust. Seedlings must break the surface crust, and most of the times, do not succeed if the crust is hard.

Crusting can be minimized when a plant cover is maintained to reduce the impact of rain drops. Also a reduced tillage when the soil is slightly damp can help prevent the formation of crusts. Management systems that reduce tillage needs are developing over recent years, it is the so-called conservation tillage. Some of these practices are the direct seeding on the stubble of previous crops (Fig. 5.20).

Fig. 5.20: Direct seeding equipment reduce soil disturbance and keep a layer of residues in contact with the ground. This protects the soil from the formation of surface crusts

4. HYDRAULIC PROPERTIES IN MANAGED SOILS

The study of the dynamics of the water in soil has much interest in understanding the development and properties of soil, and thereby the possibilities of use and conservation. On the other hand, we have to consider that most of inland waters that we use and consume have fluid on the surface or are drained through the soil. For this reason, the preservation of a natural resource so scarce, such as water, requires a management and rational use of soil. In this chapter that follows the movement of water through soil is studied: reviewing the basic principles, things affecting infiltration, saturated flow, non saturated flow and evaporation. Also the most usual methodology is explained. For the general principles of physical and hydraulic properties the students are addressed to general handbooks (Hillel, 2003; Brady and Weil, 2007).

4.1. Soil water balance

The amount of water in soil and its variation during the year is known as soil moisture regime. The regime of humidity is determined by the quantities of water which are provided, removed and retained by the soil, i.e. the water balance, and results from the combination of physiographic, climatic and soil factors, without forgetting the influence of soil management (Fig. 5.21).

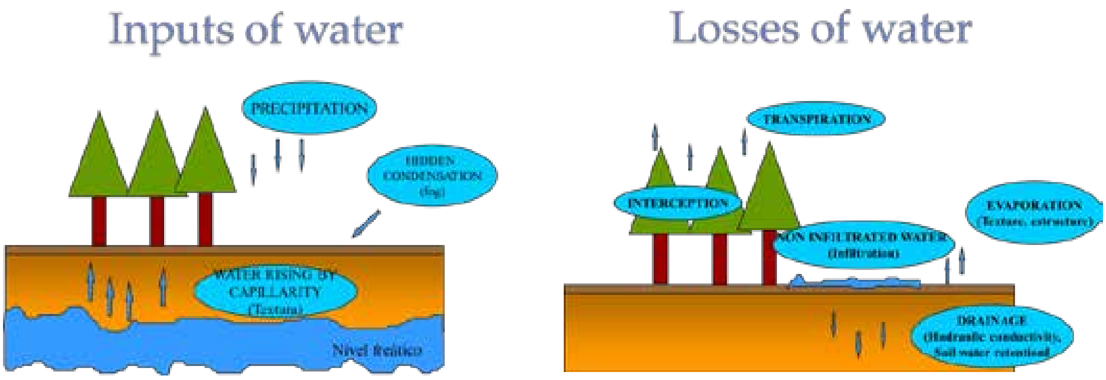

Figure 5.21: Water balance in the soil: Inputs and losses of water in natural systems

The factors that contribute to increase water content in the soil are:

1. The water of rain or irrigation, which represents a clear contribution.

2. Hidden condensation water.

3. Water that rises from the deep horizons in accordance with the laws of the capillarity.

On the other hand, the components that contribute to reduce water content in soil are as follows:

1. Non Infiltrated water, which remains on the soil surface without being absorbed by the soil.

2. Water intercepted by vegetation.

3. One of the most important losses of water occurs by evaporation.

4. Water of transpiration of plants.

5. Finally, it is necessary to consider water of internal drainage. Infiltrated water that circulates in descending sense and abandons the density of considered soil. This quantity is controlled by the hydraulic conductivity and the capacity of water retention for the soil.

4.1.1 Factors affecting the water balance in the soil

The balance of water in the soil is the combination of physiographic, climatic and soil factors.

Physiographic factors. The influence of the topography cannot be separated in the context of climate and plant. However, taking into account the position of the soil in the landscape, we can find three situations:

(a) the first case is **soils located in depths of valleys and humid depressions** where, as a result of the presence of a layer of water, the profile is completely saturated during the greater part of the year. The most common type of soils are Histosols and Gleysols.

(b) A second group of soils are the ones that are in **low-lying areas and have a layer of water located at intermediate depths**. The upper horizons wet by the capillary rise of water from the phreatic layer. Usually in these areas Luvisols and gleyc Cambisols are the most characteristic soils.

(c) The third situation, the most frequent case, refers to **hillside soils that are free draining**. In them, the water regime is completely dependent on the amount and distribution of rainfall and soil properties.

Climatic factors. Climatic conditions, and in particular the amount and distribution of the precipitation and evapotranspiration, constitute the main factor affecting the water regime of free draining soils.

In the **humid temperate regions**, during the greater part of the year precipitation exceeds evapotranspiration, therefore there is a prolonged period of excess water, while the drought period is short and of low intensity. On the other hand, the period of drought **in the semi-arid region** is prolonged and intensive, meanwhile, there is no period of excess water or it is of short duration.

Soil factors (Fig. 5.22). The balance of water in soil is heavily influenced by hydrodynamic properties of the soil. In this way, the infiltration determines the amount of water that can be absorbed by the soil. The hydraulic conductivity and that of retention of the soil condition determine the quantities of water loss by drainage. On the other hand, the evaporation is also conditioned by certain soil properties. These aspects are treated with greater attention in the following sections.

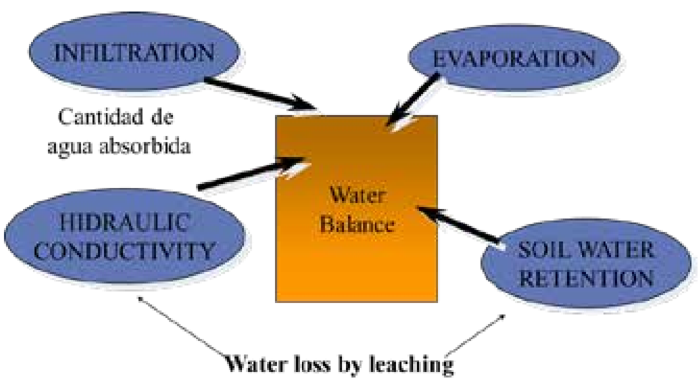

Figure 5.22: Soil properties determining the water balance in the soil.

4.1.2 Movement of the water in soil

When the water reaches the surface of the land through irrigation or rain, part of this water is infiltrated by the soil, while another part remains on the surface or flows over it (Fig. 5.23).

The water which infiltrates penetrates the porous spaces of the soil and replaces the air contained in them. If the amount of water is sufficient, the pore space is filled in its entirety and the excess water will move down through a physical process called **saturated flow**. If, on the contrary, the application of water is limited, the pores of the soil are only partially filled and the movement is governed by the physical process known as **non saturated flow**.

On the other hand, although the movement of the water in the soil is mainly produced in liquid state, it also can move in the form of steam. From the surface, the water is lost to the atmosphere, a phenomenon known as evaporation. Nevertheless, the movement of water in the soil can be of three types 1) saturated flow, 2) non saturated flow and 3) evaporation.

In this course we are going to address the processes more directly linked to run off generation: infiltration and water movement under saturated flow.

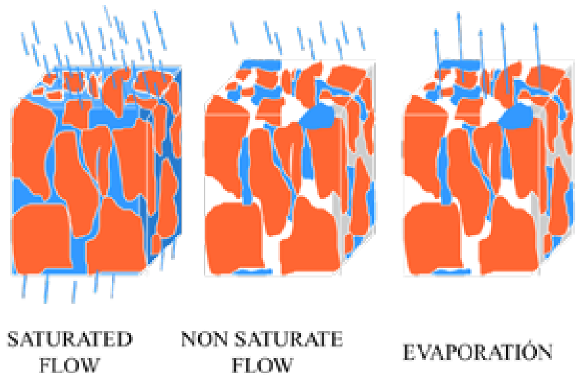

Figure 5.23: Water movement in the soil according to the soil moisture.

4.2. Infiltration

Infiltration is the term used to describe the process of entry of water into the soil surface. Greater or less infiltration capacity will determine a) how much water will enter in the root zone and b) how much will circulate by surface runoff. Thus, this property determines not only the risk of erosion but also affects the economy of water for plants, but it also affects.

The factors that affect the rate of infiltration are the following:

(a) **water content in the soil**. The infiltration rate varies widely between different soil types and within a single type, it will depend on the content of water. The increase in moisture in the soil reduces infiltration. In fact during the first hours of rain water content is determined by the rate of infiltration (Fig. 5.24).

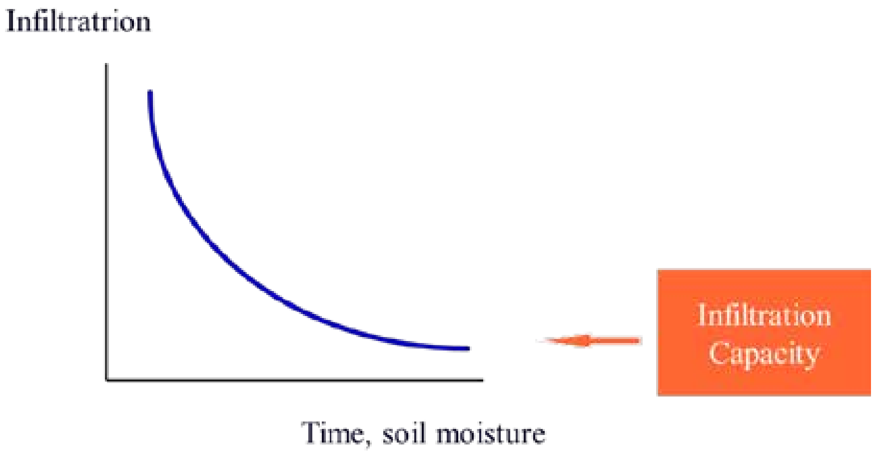

Figure 5.24: Evolution of infiltration depending on the soil moisture infiltration.

(b) **Properties of the soil**. All the soil properties that are involved in the transmission and storage capacity of the water modify infiltration. It is the case of the texture, structure or the presence of expanding clays. Thus, minor infiltrations were observed in soils of fine texture and massive structure.

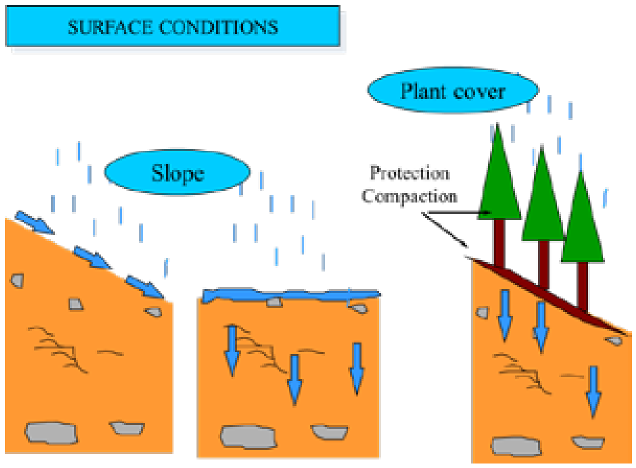

Figure 5.25: The infiltration is less outstanding on hillsides devoid of vegetation. On the contrary, vegetation favours infiltration in the same conditions a) because it slows down the speed of the water, increased the time for infiltration and b) the topsoil protects the soil from compaction.

(c) **Surface conditions and plant cover**. The slope of the field indirectly affects the rate of infiltration. In areas of high slope, the rapid movement of water only allows a short time for infiltration. The cover tends to increase infiltration because it slows the surface flow, increasing the available time for water to penetrate into the soil (Fig. 5.25). In addition, vegetation cover protects the surface of the soil compaction by the impact of raindrops. Thus infiltration varies widely between different uses of the land. In general, cultivation soils have lower rates of infiltration to meadows and forests (Fig. 5.26).

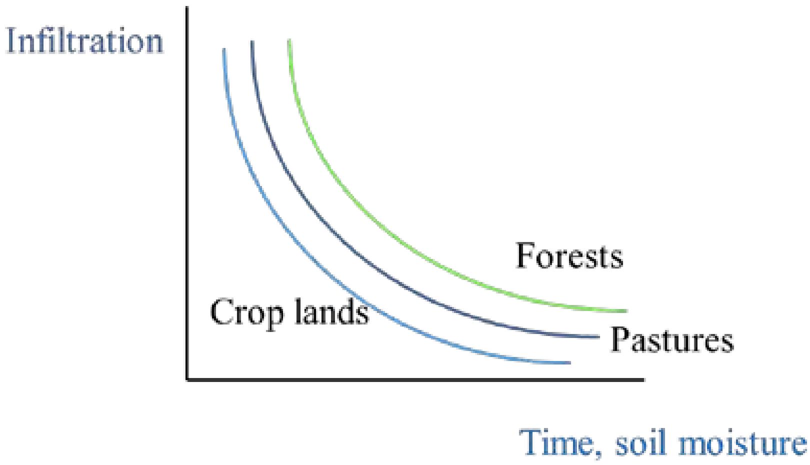

Figure 5.26: Forest soils have higher infiltration compared with grassland and crops

(d) an important aspect is that **the type of management of soil** can affect the infiltration of the soil. The land management must promote infiltration of water and avoid the runoff. Intensive tillage can oxidize organic matter and destabilize the structure, so in some cases low infiltration surface crusts may form. It has been proved that the soil conservation practice (no till) increases the stability of the aggregations of some soils and in turn it might improve the infiltration However, the net effect of this technique depends on the type of soil. In agricultural soils, surface water retention capacity can increase the time of infiltration. Another possibility is establishing a dense vegetation during periods of high rainfall, it is the protection cultivation, which favor the structure activity of the earthworms. The extensive information about this subject has been reviewed by Strudley et al. (2008)

Some studies, such as Gómez et al. (1999), show how the infiltration is reduced in soil subjected to intensive tillage (Fig. 6.27). The compaction by farm machinery has also important effect on the infiltration rate (Fig. 5.28). In these studies the determination of infiltration was made with the double ring infiltrometer (Fig. 5.29), in which infiltration capacity is determined by measuring the decrease of water per unit of time. Another field technique usually employed is the rain simulator. Models that simulate the infiltration of soil are Green and Ampt, Horton, Kostiakov and Philips (these models are not subject of this course)

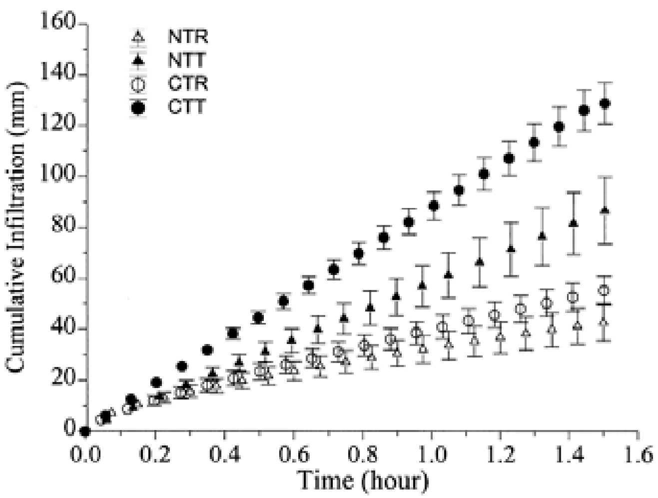

Fig. 5.27: Cumulative infiltration from rainfall simulation by position and tillage treatment (from Gómez et al., 1999). NT= No tillage. CT= conventional tillage. T= below-tree. R= in-row.

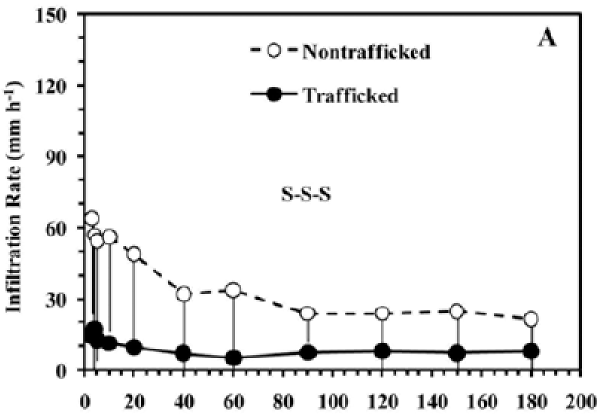

Fig. 5.28: Differences in water infiltration rates between nontrafficked and trafficked rows (Blanco-Canqui et al., 2010).

Fig. 5.29: The infiltration is measured in the field by a double ring infiltrómetro.

4.2.1 Soil water repellency

Some soils show hydrophobic dry properties, which slows the infiltration of water into the matrix of the soil (Fig. 6.30). In certain situations, this property can manifest itself in intense form and can go to far as to change the water balance: as in summer thunderstorms, after a fire, in mine dumps or sports cespeds.

Although the reasons for the development of the hydrophobic soil properties are not clear, it is believed that they are mainly related with the formation of functional hydrophobic films on the soil particles, however, it is not known with accuracy neither the origin nor the composition of these compounds. Different substances and radical chemicals, such as phenols, amines or fatty acids have been recently proposed.

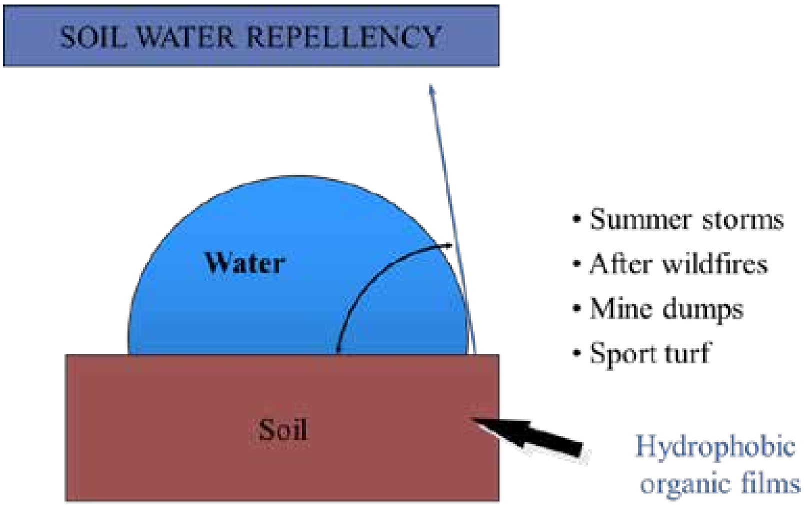

Fig. 5.30: Sometimes the soil develops hydrophobia on the surface. This effect is due to the formation of organic films that repel water, preventing its infiltration.

4.2.2 The case of fire

The heat of fire, although it is brief and affects a few mm of soil, causes significant changes of organic matter affecting the dynamics of water (Shakesby and Doerr, 2003). Fire causes the rupture and movement of organic compounds in the soil. From 125 °C some compounds of the organic fraction is vaporized and move towards the interior of the fire (Fig. 5.31). When they encounter cooler mineral particles, they condense on its surface and clog the pores of the soil. Some of these compounds which condense are repellents hydrocarbons to water. The conductivity after this process is reduced considerably, increasing the water runoff and, consequently, the rate of erosion. Mud flows caused by this process occur in some soils.

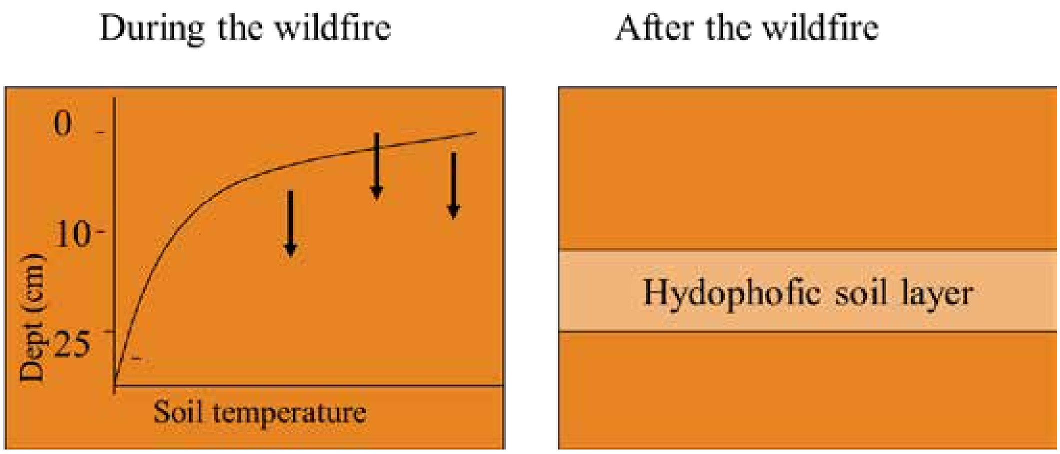

Fig. 5.31: Formation of a repellent layer of water during a fire (Brady and Weil, 2007).

4.3 Soil water flow: hydraulic conductivity

4.3.1 Darcy's Law

In a saturated soil, the movement of water through the soil is similar to the flow that occurs through pipes. However, due to the tortuosity of the pores, the flow of water through the soil is substantially reduced with respect to which would take place through pipes.

Consider the flow of water that takes place through the column of uniform soil saturated with water that is shown in this image (Fig. 5.32). On the one hand, the water that enters the **column is a pressure**, which is the exerted by the weight of the column of water above the ground. This force is called water potential, Hp potential. In this vertical column flow is also determined by the effect of gravity, the **gravitational potencional**, Hg. The force acting on the water so that it moves through the soil, is composed of the sum of these two potential, the potential of pressure and gravitational potential, is the **total hydraulic potential, H**.

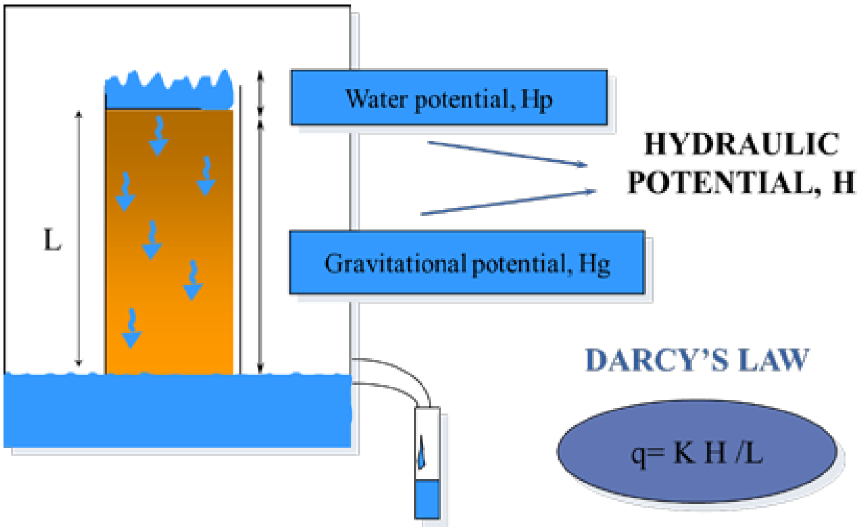

Fig. 5.32: Darcy's Law and hydraulic conductivity.

In this system, the flow of water per unit of time and surface is proportional to the hydraulic potential total and inversely proportional to the length of the water column and responds to the following expression, known as the Darcy's Law:

$$q = KH/L$$

The constant of proportionality of the Darcy's Law, K, is known as **hydraulic conductivity** and expresses the capacity of soil to transmit water. This parameter has different applications and used to meet the water flow coming out of the ground, in the planning of irrigation and drainage, as well as in the assessment of risks of erosion.

4.3.2 Hydraulic conductivity

In general, the hydraulic conductivity of the soil is between 0.10 and 25 cm h^{-1}. The characteristics of the soil that affect the hydraulic conductivity are basically the size and tortuosity of the pores (Fig. 5.33). As shown in the Fig. 5.34, the route of the water flowing through the pores is greater than the length of the column. As a result, the hydraulic conductivity will be heavily influenced by the soil properties which determine the geometry of the pore space, such as the texture, structure, as well as by certain physical, chemical and biological processes.

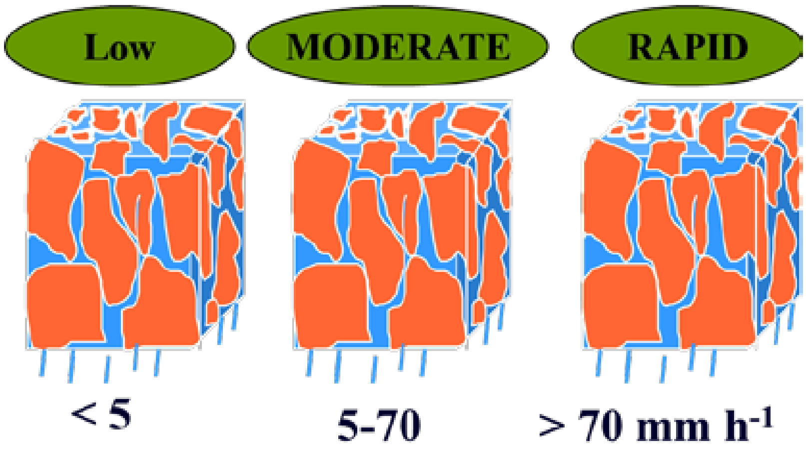

Fig. 5.33: The hydraulic conductivity varies in the different soils as a consequence of the texture and structure.

1. Effect of texture. Soils of thick texture, of high macroporosity, present high conductivities While the clay soils, with fine pores, will have lower conductivities.

2. Effect of structure. Granular structure or loose soils show rapid conductivity. On the contrary soils with structures laminar or massive show slow conductivities. In soils with weak structure, the withdrawal of organic waste can lead to the development of surface crusts, which reduce the infiltration (Fig.5.35).

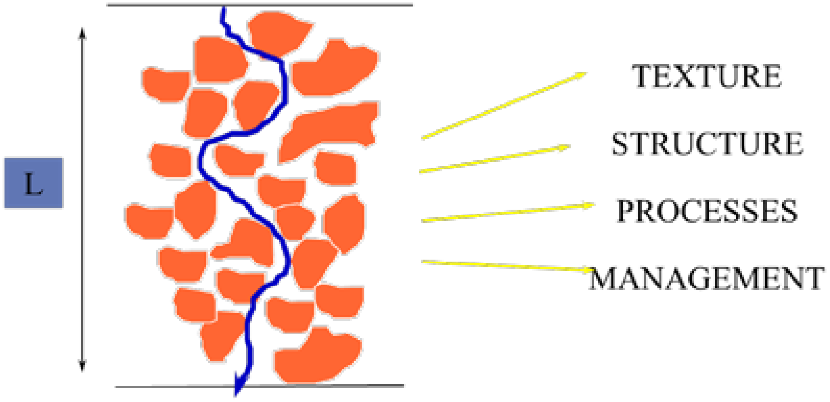

Fig. 5.34: The journey of a drop of water through the soil is very tortuous, so much more complex is the system of pores, which is influenced by the texture, structure, some soil processes (salinization, presence of clays expandable) and soil management.

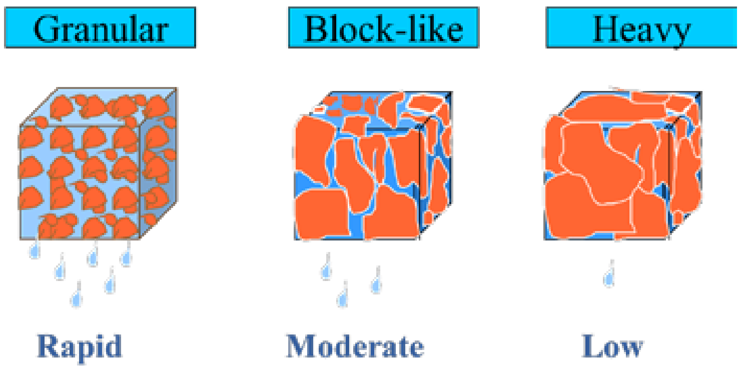

Fig. 5.35: The granular structure of soil, by virtue of its high porosity, has a conductivity superior to structures in blocks or the massive conductivity.

3. Physical processes: expanding clays. In the Vertisols, the hydraulic conductivity is reduced after or during a shower as a result of the expansion of clays.

4. Chemical processes, the presence of salts. Soils with a high content of sodium have a very low hydraulic conductivity, so it is frequent find water standing on its surface, as is the case in many saline soils, such as the Solonchack. This reduction is due to the ion sodium dispersing the clay, which favours the blockage of the porous space of the soil. On the other hand, the intake of calcium increases the conductivity of water due to an improvement in the structure, as shown in saline soils treated with gypsum.

5. T**he land uses and the soil management** affects greatly the soil hydraulic conductivity. Tillage, residue management and mechanical compaction alter most of the physical and hydraulic properties of the soil.

Tillage

An illustrative example is the study of Mahboubi et al. (1993), in comparing a soil subjected to traditional tillage and the technique of non tillage over 28 years (Fig. 14). As you can see, the intensive tillage reduced the hydraulic conductivity. The measurement of K was done by a laboratory method, a constant load permeameter (Fig. 15), which can point out most important limitations, the lack of spatial representativeness and the existence of flows preferred among the soil column and the cylinder that contains it.

Table 5.4: Soil organic matter, bulk density and hydraulic conductivity in different soil management regimes (Mahboubi et al., 1993).

	SOM (%)	**BD (g cm^{-3})**	**K_S (m d^{-1})**
No tillage	2.3	1.34	13.0
Moldboard plowing	1.5	1.31	0.9
Chisel plowing	1.0	1.31	0.2

The study of Gómez et al. (1999) show how the hydraulic conductivity can be reduced in soils of orchards subjected to intensive tillage (Fig. 5.36).

Mechanical compaction

Another aspect related to the management of agricultural soils which affects the hydraulic properties of the soil is heavy machinery traffic. Mechanical loading of soils can compact the soil, causing increased bulk density and altered pore shape and size distribution. However, the amount of soil compaction depends on the applied load, soil type and moisture status (Strudley et al., 2008). The study of Coutadeur et al. (2002) shows that the effect of heavy machinery as important both in the conventional management and tillage (Fig. 5.38).

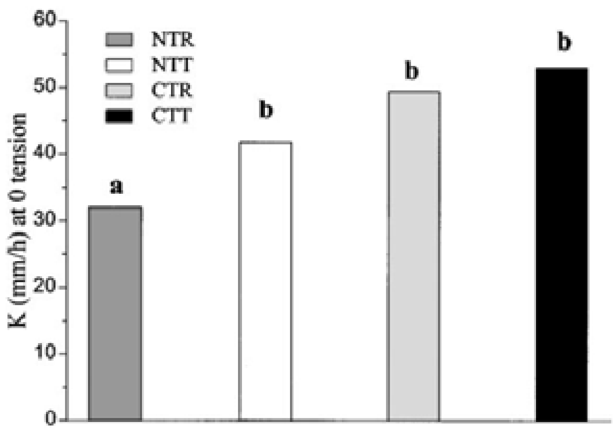

Fig. 5.36: Hydraulic conductivity (mm/h) at 0 cm water tension in a olive ochard. NT= No tillage. CT= conventional tillage. T= below-tree. R= in-row. (from Gómez et al., 1999).

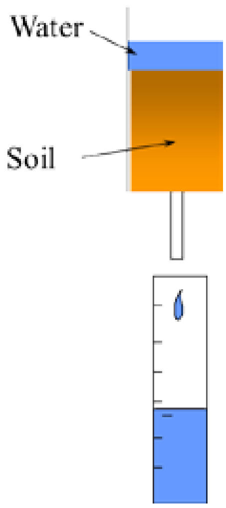

Fig. 5.37: The hydraulic conductivity is measured with a constant load permeameter.

The forest soils can also be affected. The study of Schack-Kirchner et al (2007) shows how the the effect of heavy machinery on both bulk density and hydraulic conductivity in a forest soil after harvesting (Fig. 5.39).

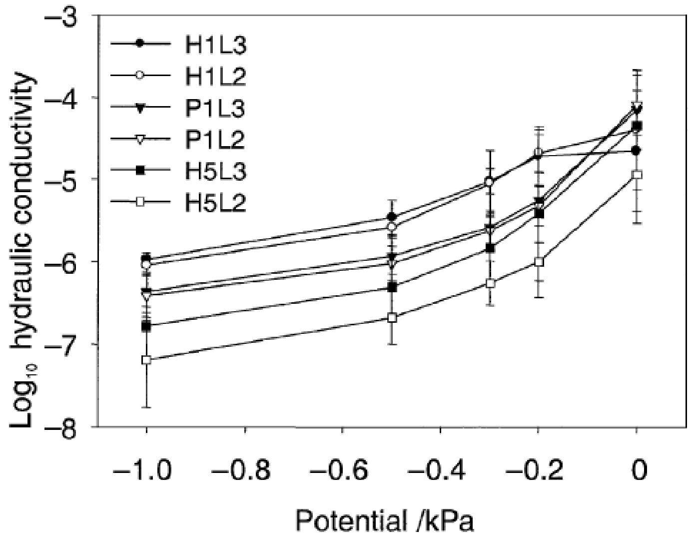

Fig. 5.38: Effect of compaction. Comparison of the log-transformed hydraulic conductivity of the seed bed (H1), the ploughed layer (H5) and the untilled layer (P1), either below the wheel tracks (L2) or between the wheel tracks (L3) (Coutadeur et al., 2002).

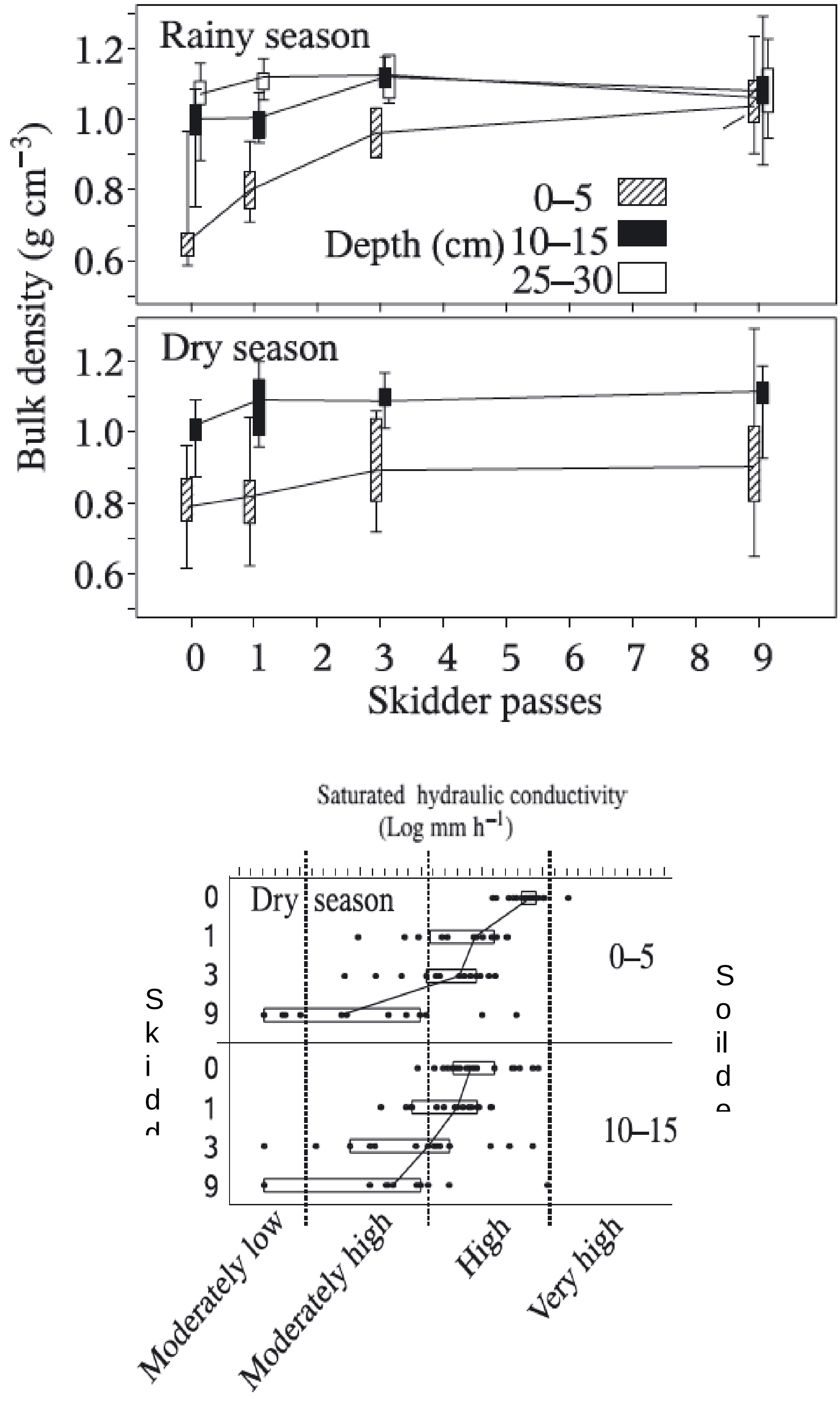

Fig. 5.39: Intensive tillage and the introduction of heavy machinery a) increases the bulk density and b) reduces the hydraulic conductivity.

The permeameter of Guelph (Fig. 5.40), based on the principle of the dry hole (water quantity that it is necessary to add to a hollow carried out in the soil so that the water level should be supported inside the hollow) was used in these works.

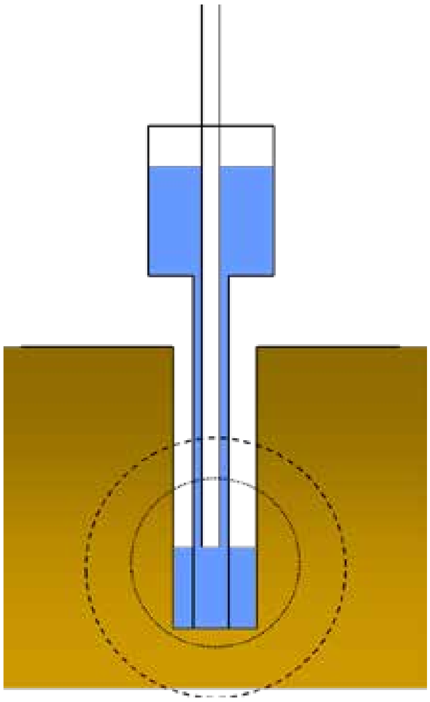

Fig. 5.40: The hydraulic conductivity is measured in field through a permeámetro of Guelph (method of hole).

5. CONCLUSIONS: MINIMIZING RUNOFF AND EROSION THROUGH MANAGEMENT OF SOILS

The previous information shows that soil degradation is one of the major problems facing current agriculture and forestry. Overuse of machinery, intensive cropping, short rotations, intensive grazing and inappropriate soil management leads to SOM loss, soil compaction and therefore runoff and erosion.

Although each soil presents particular problems, you can take the following scheme to preserve the quality of the soil and avoid erosion:

1. Reducing tillage, especially that resulting from the mixture of horizons, reduces the loss of organic matter stabilizing agent in aggregates

2. The effects of traffic and tillage machinery are less pronounced when performed in the periods in which the soil is with optimum moisture.

3. Maintaining the surface of the ground covered with vegetable residue provides organic matter to the soil, at the same time that protects the aggregates from rain drops.

4. The incorporation of manure helps meet the organic compounds that have been decomposed and help stabilize the soil aggregates.

5. The inclusion of meadow in the rotation process favours the soil organic matter content and to maintain the aggregation.

6. MEASURES TO CONTROL EROSION IN MANAGED SOILS

Water erosion selectively removes fine particles from the soil, leaving the larger particles. The fine particles include clay or fine silt as well as organic compounds, which are the components that have the greatest influence on nutrient and water availability. The sediment lost from an eroded soil may be up to 5 % times richer in organic matter than the soil left behind. Because of the lower soil porosity, and nutrient and water availability, the selective loss of these fertile components has a deleterious effect on crop production. Severely eroded soils may reduce crop yields by up to 30 %.

Restoration of the SOM in degraded soils is necessary not only to recover crop productivity, but also to increase the capability of C sequestration, thus contributing to reducing atmospheric CO_2.

The main measure that can be used to prevent soil erosion is to assign the most appropriate use in accordance with the limitations of the land. The land capability classification scheme provides a valuable tool for promoting sustainable use of land resources.

Most practices that prevent soil erosion are focussed on reducing soil perturbation and retaining plant cover. Such measures (e.g. conservation tillage) prevent run-off by intercepting raindrops and increasing SOM and soil aggregate stability.

Another strategy aimed at controlling erosion is to reduce the amount and velocity of run-off. Certain practices, such as contour cultivation and terracing, lower the potential of the water to generate erosion and also provide more time for water to infiltrate.

6.1 Land planning as a basic guide to soil conservation

The land capability classification system (USDA) is particularly useful for identifying land uses and management practices that can minimize soil erosion. The system uses eight land classes to indicate the degree of limitation imposed on land use, where Class I is the least limited and Class VIII the most limited.

6.2. Soil management

The different systems considered as conservation tillage maintain at least 30 % of the crop residues on the surface and enable reduction in the intensity of tillage. Farmers often use herbicides to kill weeds rather than applying conventional inversion tillage (e.g. with a mouldboard plough). The use of a chisel plough is also preferred because it stirs the soil but leaves a large proportion of the crop residues on the soil surface.

No-till: Soil undisturbed prior to planting, which is done in a narrow seedbed, 2.5-7.5 cm wide. Weed control primarily by herbicides.

Ridge till: Soil undisturbed prior to planting, which is done on ridges 10 to 15cm higher than row middles. Residues moved aside or incorporated in about one-third of soil surface. Herbicides and cultivation used to control weeds.

Strip till: Soil undisturbed prior to planting. Narrow and shallow tillage in row using rotary tiller, in-row chisel, and so on. Up to one-third of the soil surface is tilled at planting time. Herbicides and cultivation to control weeds.

Mulch till: Soil surface disturbed by tillage prior to planting, but at least 30 % of residues left on or near soil surface. Tools such as chisels, field cultivation, disks, and sweeps are used (e.g. stubble mulch. Herbicides and cultivation used to control weeds).

Reduced till: Any other tillage and planting system that retains at least 30 % of residues on the surface.

In many cases these systems produce the same or even higher crop yields, while saving time and soil. In recent years conservation tillage has become increasingly popular, in some cases due to encouragement via policies such as the Common Agricultural Policy in Europe.

6.3. Agronomic measures

Contour cultivation: Plants are cultivated in rows following the contours across the slope gradient to slow the flow of run-off water.

Strip cropping: In this system the field is laid out in narrow strips across the slope, alternating tilled crops, such as corn or potatoes, with hay and small grains. This system is useful for shortening the effective slope length on long slopes subject to sheet and rill erosion.

Crop rotation: In this practice a series of dissimilar types of crops are grown in the same area in sequential seasons. Growth of legume as a green manure in sequence with cereals and other crops allows the replenishment of organic matter and nitrogen.

Close-growing crops: These crops provide soil protection, seeding protection, and improve the soil between periods of normal crop production, or between trees in orchards and vines in vineyards. When ploughed under and incorporated into the soil, cover crops may be referred to as green manure crops.

6.4. Mechanized practices

The construction of various types of terraces reduces the effective length and gradient of a slope. Different types of terraces are usually used:

-Broad-based terraces: These permit the entire surface to be cropped and are used in areas where farmers use large machinery.

-Flat channel terraces and bench terraces: These are used on steep slopes and when properly designed allow large volumes of water to move from the soil without erosion.

Figure 5.41: Some examples of flat-channel terraces in Mediterranean environments.

6.5. Techniques to control erosion and sediment in construction sites

In highly perturbed areas, such as mines, road cuts and other construction sites, specific plans to control run-off and erosion are usually applied (Pitt et al., 2007). The high erosion rates make it necessary to sow fast-growing grass species adapted to the specific conditions. Mulch and erosion blankets are usually used to provide instant soil cover and prevent the seeds from being washed away. In steep unstable slopes, such as road cuts, hydroseeders are usually used. These machines spray a mixture of fertilizer and seeds wrapped in mulching material and sticky polymers. Removal and stockpiling of the original upper organic matter rich soil horizon prior to grading is also common practice. When the activity has been completed, the material is used to provide topsoil for landscaping the ground around the structures.

In these areas, the flow of runoff must be carefully controlled by means of grading, terracing, channel construction and retention basins. In channels subject to high water velocities, the soil should be protected by armors and gabions. Silt barriers made from straw and woven fabric are commonly used to trap sediments, and in large construction sites, sediment-retention ponds or artificial wetlands are used to remove sediments from the water.

Specific bioengineering techniques are carried out with vegetation and natural biodegradable materials. Brush mattresses and live stakes can be used to provide protection from scouring waters.

6.6. Control of gully erosion and mass wasting

Gullies are common in steep slopes partially covered by vegetation. They can also occur in concentrated water flows formed in arable land, as well as in poorly designed terraces and roads. In desert climates, the appearance of gullies can have devastating effects.

In small gullies, grass species can be sown to create grassed waterways. In more active gullies, a series of check dams can be constructed with materials available on site, such as rock or logs or brush.

Figure 5.42: Construction of a series of dams prevents the gully increasing in size.

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PART II - CASE STUDIES

CASE STUDIES – INTRODUCTION

By

Varvara Antoniou & Niki Evelpidou

1. RUNOFF EROSION IN MEDITERRANEAN AREA

Soil erosion is the most significant risk of land degradation in Mediterranean environments. It fills the valleys and reservoirs and dramatically reduces soil productivity, due to deterioration of the soil, its nutritional depletion, and the reduction of water retention capacity leading to the limited growth of vegetation and extensive desertification.

Although it is clear that land degradation in the Mediterranean is a product of human activity, it was significantly worsened by physical factors. Therefore, the observed high rates of erosion are attributed to climatic conditions (Langbein and Schumm 1958), existing poor vegetation cover and land use management (Douglas 1969; López-Bermúdez et al. 1984; Bryan and Campbell 1986).

The Mediterranean environment, in terms of climate, is characterized by strong seasonal contrasts. Annual rainfalls with high intensity and low frequency are the leading cause of soil erosion in Mediterranean Europe, and when low levels of annual rainfall are combined with maternal rocks with high vulnerability to erosion, the erosion potential is high (Romero Díaz et al. 1988). Given that the topography of a region is a very important element in development plans and land use practices, and also a factor contributing to soil erosion, the physiography of the Mediterranean region includes a number of different landforms, a large percentage of which is dominated by scattered vegetation zones. Such a physiographic variety presents the ideal conditions for slope erosion, which can lead to decreased soil productivity and eventually to desertification. The orientation of the slopes significantly affects the temperature of the local environment, which in turn affects evaporation, and hence the growth and resilience of vegetation. A highly inclined slope also affects the infiltration rate and accelerates runoff and sediment loss. The most inclined slopes bring about mass movements, such as subsidence and mud flows.

The vegetation type, covering a landscape, affects all soil dynamics, including the redistribution of water above and in the soil, and also microbiological activity, with biotic interactions that preserve soil structure in the upper layers through the process of accumulation, which is a powerful determinant of hydrological and biological soil characteristics (Imeson, 1984), and affects the response to erosion. Mediterranean areas with extensive crops watered only by rainwater, such as areas with cereals, vines, almond and olive trees, are mainly confined to hilly areas with shallow soils, very susceptible to erosion. These areas become vulnerable to erosion and desertification, because the reduced vegetation cover provides less protection from the raindrop impact during strong rainfalls (Faulkner 1990), the reduction in infiltration rates due to compaction by heavy agricultural machinery (Fullen 1985) and the formation of a surface crust.

Deforestation of arid areas on a large scale, combined with intensive farming and overgrazing has led to the acceleration of erosion. The erosion rates, measured in different regions of the Mediterranean, range from 0.4 to 1.7 mm/yr (Benito et al., 1992). However, even greater erosion rates of about 2.4 mm/yr or even 12 mm/yr have been reported in other areas (Lopez-Bermudez, 1990). These values can easily increase during intense rainfall events that occur occasionally (Alias-Perez and Ortiz-Silla, 1986; Lopez-Bermudez et al., 1991).

The Mediterranean appears to be the world's region affected in the ugliest way from degradation caused by man, for hundreds of years. Examples of degradation are all around, with indigenous woodlands being the unique remaining areas along with areas that can no longer support crops.

Accelerated soil erosion is as old as agriculture. Two leaders of the U.S. Conservation Service, Hugh Benntt and Clay Lowdermilk, wrote in 1938 the Yearbook of Agriculture: "*Soil erosion began when the first big rain stroke the first groove, which was carved/engraved by an early application of crop in the hands of the prehistoric man. Since then it continues, wherever man leaves the soil bare against the rain and wind*" (USDA, 1983). However, the first mention of erosion was made by Homer, in the *Iliad*. The Greek slopes were covered with forests and fertile topsoil, which, however, was rather shallow and vulnerable to erosion. Grazing and agriculture probably started in the mid-second millennium and constituted the beginning in the destruction of forests. Several thousand years of agricultural exploitation contributed significantly to a dramatic decline in agricultural productivity in the region, which had already been reported by Plato, who, speaking in Athens in the 4th century BC (Judges III) noted the occurrence of massive floods and landslides, deforestation and denudation due to grazing. This description provides one of the oldest recorded examples of degradation and desertification, which also involves climatic and anthropogenic causes. Two centuries earlier, Solon had already defended the discontinuous cultivation of grains in the sloping lands of Attica, and counter-proposed the cultivation of olive trees and vineyards. His advice was heard in the 4th century BC by Theophrastus, in his book *"The Cause of Plants"*. Considering the impact of land use on erosion and especially, the positive influence of olive groves, one can realize how appropriate the change of land use was, as proposed by Solon. However, none of the aforementioned advice located the source of the problem, which was not the choice of the crop itself, but the actual process of erosion and controlling the phenomenon. Additional historical information relating to the effects of degradation on vegetation can be found in Roman times, where land degradation led to the development of large pastures. Wherever the Romans consolidated their domination, they repeated the same pattern of extensive forest cleaning, overfarming and overgrazing of the land, in order to satisfy the greed of the emperor (Hillel, 1991).

In the Mediterranean, land use changes in recent history are mainly owed to physical and technical factors, as well as socio-economic reasons. Specific land uses were associated with specific behaviors of populations, with changes in spatial distribution and pressure on natural resources. The area underwent significant changes from the mid 19th century, when rural development began. The mismanagement of the land, which was stimulated by the demographic dynamics, contributed to the shift of rural population and activities in marginal areas that were unsuitable for cultivation. The human influence on the landscape was increasingly negative through the conventional, extended and large-scale agriculture that had a negative impact on soil properties and intensified erosion processes. The expansion of cultivated areas at the expense of forests marks major ecological changes because of the deforestation and the rupture of the original balance between farming, grazing and forestry.

Greece, in particular, appears highly degraded with many areas facing a significant risk of desertification. According to recent studies, 35% of Greece is at high risk of desertification or has been desertified, while 49% is considered at moderate risk (Fig. 1). "High risk" areas include the Aegean islands, part of Thessaly, eastern Central Greece, eastern Peloponnese, and Crete -particularly the South and East- as well as several areas of the mainland.

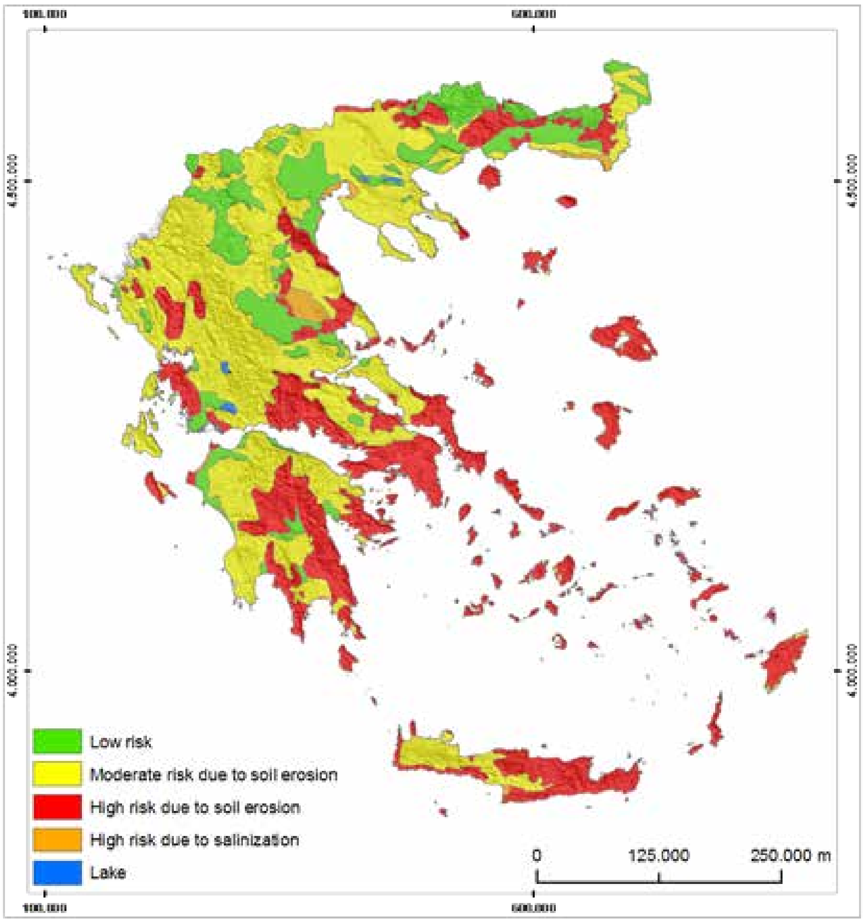

Fig. 1: Potential desertification map of Greece (based on Yassoglu et al., 2009).

In terms of climate, there is a gradual transition towards greater periods of droughts and periods of intense and major rainfalls. The high frequency and low intensity rainfalls, combined with high morphological reliefs, result to intense surface runoff that is usually accompanied by soil loss and flood events. The removal of the already scarce soil mantle by the rare, intense rainfalls, leads to the degradation of vegetation in sensitive areas and creates a vicious circle, since the reduction of vegetation intensifies erosion.

From the three climatic zones found in Greece, semi-arid and dry sub-humid (UNEP, 1997, Nastos et al., 2011), favor soil erosion on sloping areas and decelerate the recovery of vegetation, where it has been damaged, while they also contribute to land salinisation and drying. These zones are characterized by low annual rainfall, high potential evapotranspiration, uneven distribution, high intensity and high erosivity of rainfall, high deficiency in humidity and high temperatures during the germination period of plants.

The Greek morphological features and especially the high morphological gradients and their frequent changes, contribute to increased surface runoff, which is also intensified by land use and anthropogenic interventions. The main factor is the expansion of agricultural activities in the mountainous and semi-mountainous zones of the country, especially when they are not combined with erosion protection measures.

Large areas are covered by Mesozoic limestones, which form shallow soils and along with their high water permeability, they create a xeric microclimate, particularly on the southern slopes, that acts negatively on the development of vegetation. An important part of agricultural land is also located in hilly areas of tertiary marls, which are also highly vulnerable to desertification due to both the large erosivity of the soil they form and because of the dry soil environment that is developed during periods of reduced rainfall. Currently, they do not show extensive desertification because of their large soil depth, however, the recent intensification and mechanization of tillage and the use of heavy machinery led many of these lands at threshold levels, especially in Thessaly (Central mainland Greece) and Crete (southern Greece). Rocks, such as acidic rocks and some volcanic igneous rocks, play a secondary role as they form soils at slow rates and create coarse texture, like in the case of west Lesvos.

In recent years, abandonment has been observed in sloping areas that were cultivated and maintained terraces that offered protection to the slopes from runoff erosion. Characteristic examples are found in the largest part of coastal Greece and the islands, with the Cyclades being a typical case (Fig. 2). The latter case is mainly owed to the change of activities of the local population (shift from agriculture to tourism) and has led to a vicious circle; the abandonment of cultivation terraces in recent years has accelerated erosion in these areas and the loss of the scarce soil mantle has made their reuse for cultivation difficult.

Significant degradation of mountainous and semi-mountainous areas is also caused by livestock. The number of animals that are usually bred in this zone is often much greater than the capacity of the pasture. The results are the destruction of soil structure, the differentiation of the floral composition and the reduction of herbaceous vegetation’s density. A typical example of intense erosion from overgrazing is the island of Ikaria.

Fig. 2: Well preserved cultivation terraces at Syros island (Cyclades) (upper image) and abandoned cultivation terraces at the same island (lower image).

Last but not least, one of the most important factors of erosion in Greece is forest fires. Statistical data from the Forest Service show that forest fires have quadrupled, in the last decades. During the period 1964-1975, the average of burned areas was 129,000 acres per year, while during 1976-1986 was 378,000 acres and reached 520,000 acres in the decade 1986-1995. Although fires may lead to regeneration of natural vegetation when not preceded by intense erosion, the high incidence in recent years has strongly degraded a large percentage of forest lands in the country. When the frequency of fires in the same areas is large and accompanied by grazing, the destruction is irreversible for soil and vegetation.

The disastrous human interventions were, until today, numerous and are mainly summarized to the following:

(i) the non-rational management of water resources (poor irrigation practices, overexploitation of underground water, lack of drainage), leading to salinization of aquifers and soil salinization, (ii) unaccepted agricultural practices (intensive cultivation of agricultural areas, overuse of fertilizers, non-rational tillage, lack of protection of agricultural land through the use of resources offered by modern technology),

(iii) the concentration of population in the lowland plains and coastal areas due to favorable conditions for agriculture, commercial, industrial and touristic development,

(iv) grants for crops and livestock in areas with production levels that do not ensure sustainability, resulting in the destruction of vegetation cover,

(v) very long periods of strong economic and social activity, with frequent periods of political instability and long-term occupations by foreign powers that contributed to the degradation of forests and natural resources in general,

(vi) unsustainable forest management practices that led to the destruction of forests by repeated fires, excessive logging, overgrazing and changes of their use, and especially, in recent years, by large transportation projects, power lines and other engineering works that occupy extended forest lands, break through the cohesion of the ecosystem and cause significant disruption with negative consequences for the soil, the vegetation, wildlife and water conditions.

Many early civilizations recognized the connection between accelerated erosion and the reduction of agricultural productivity on their land. Terraces appear to be among the first erosion control practices or are probably the most widespread practice of configuration for the relief. Chinese, Phoenicians, Greeks, Romans, Incas and Mayans were among those that built integrated systems of terraces. Such efforts are now taking place on some islands, such as the island of Naxos (Cyclades), where the maintenance and protection of terraces is succeeded through EU grants.

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CASE STUDY 1: SOIL EROSION RISK AND SEDIMENT TRANSPORT WITHIN PAROS ISLAND, GREECE

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ABSTRACT

Paros Island is part of the complex of Cyclades Islands, situated in the central Aegean Sea, Greece. The climate of Paros Island is representative of the Mediterranean type, with abrupt rainfalls and lower temperatures during winters and long term sunshine accompanied by dry periods during the summer, conditions able to cause severe problems on the land. The island's low vegetation leaves the ground exposed to erosion. Also, the recent change of the local people from agricultural activities to touristic ones – especially the abandonment of the agricultural terraces during the last 50 years – has influenced the soil cover of the island in an unfavorable way, leading to total soil loss and exposure of the bedrock in many areas. Aggravating this fact, the island's steep slopes render soil regeneration almost impossible. The transferred soil is either moving towards the small alluvial plains or, in most cases, is being deposited directly to the sea. In the following paper an effort is made to depict the currently existing situation on the island by delineating the areas still appearing to be at high risk for erosion and to estimate the average amount of soil loss. For the later, the Universal Soil Loss Equation (USLE) was tested on the data of the island. The application of the USLE was implemented via MapInfo and ArcGIS Tools Software. The main results of the present study is that it outlines the areas where severe erosion might occur and also gives a good example for land users on the best

scenario to protect their land against erosion and which plant they should use or avoid to protect their soil from erosion.

Keywords: Paros Island, erosion risk, soil loss, modeling, USLE

1. INTRODUCTION

Soil erosion is the natural process of soil material and sediment transfer due to runoff. Erosion depends on a series of different factors such as the climatic status of an area, the vegetation, the soil structure, the topography. An accelerating factor to erosion is usually human activities, like in the case of land use change or inappropriate land use (Evelpidou, 2006; Hacisalihoglu, 2007; Podmaniczky et al., 2010).

From the aforementioned factors, the climatic status is the most direct parameter to erosion (Podmaniczky et al., 2010), since rainfall or snowfall produces water runoff in streams and rivers, or in areas between streams in the form of sheet flow or rill flow.

One main problem caused by erosion and concerning human activities, is land degradation. Land degradation occurs mostly because of sheet or rill erosion, processes very common in areas such as cultivated lands. The eroding capacity of these processes progressively increases downslope and is strongly related to the vegetation cover (Langbein and Schumm, 1958).

A factor that aggravated the erosion rates during the last 50 years in Paros Island, as in many other Cycladic islands, was the turn of the local people into touristic professions, thus abandoning the cultivated land of the more disadvantaged areas. This case is more common in the cultivated areas on steeper slopes, where the need to create terraces to uphold the soil from eroding, the costs of their preservation, and the inability to cultivate by mechanical means, often led the farming systems to operate close to the margins of viability. The result from the abandonment of the terraces is the progressive loss of the soil's resources leading to soil thinning and shrub growth. The decreasing landscape diversity makes the area more prone to erosion and fire events (Marin-Yasseli and Martinez, 2003). This is of major importance since soil erosion leading to surface rocks is a very active process in the Mediterranean environments (Poesen and Hooke, 1977).

The effort to overcome the problems caused by erosion, has led to the need for calculating methods of the soil loss amounts and their comparison (Centeri et al., 2009), in order to introduce the necessary soil conservation methods at the time and place most needed. The soil loss equation used broadly for most erosion cases, is the USLE model (Universal Soil Loss Equation). The USLE was firstly introduced by Wischmeier et al. (1958) and was announced to its present form by Wischmeier and Smith (1978).

The USLE uses factors of Rainfall Erosivity, Soil Erodibility, Slope Length and Slope Steepness, Vegetation and Conservation Techniques as input parameters and calculates the amount of Annual Soil Loss per Hectare. USLE only predicts the amount of soil loss that results from sheet or rill erosion on slopes and does not account for additional soil losses that might occur from gully, wind or tillage erosion.

In this paper the USLE was used in the entire area of Paros Island to locate the areas with the major problems, as well as to propose solutions especially for the arable parts of the island.

2. METHODOLOGY

2.1 Area description

Paros belongs to the complex of Cyclades islands in the Aegean Sea (Fig. 1). It is situated between Naxos and Sifnos islands and is within a distance of 90 sea miles from Athens.

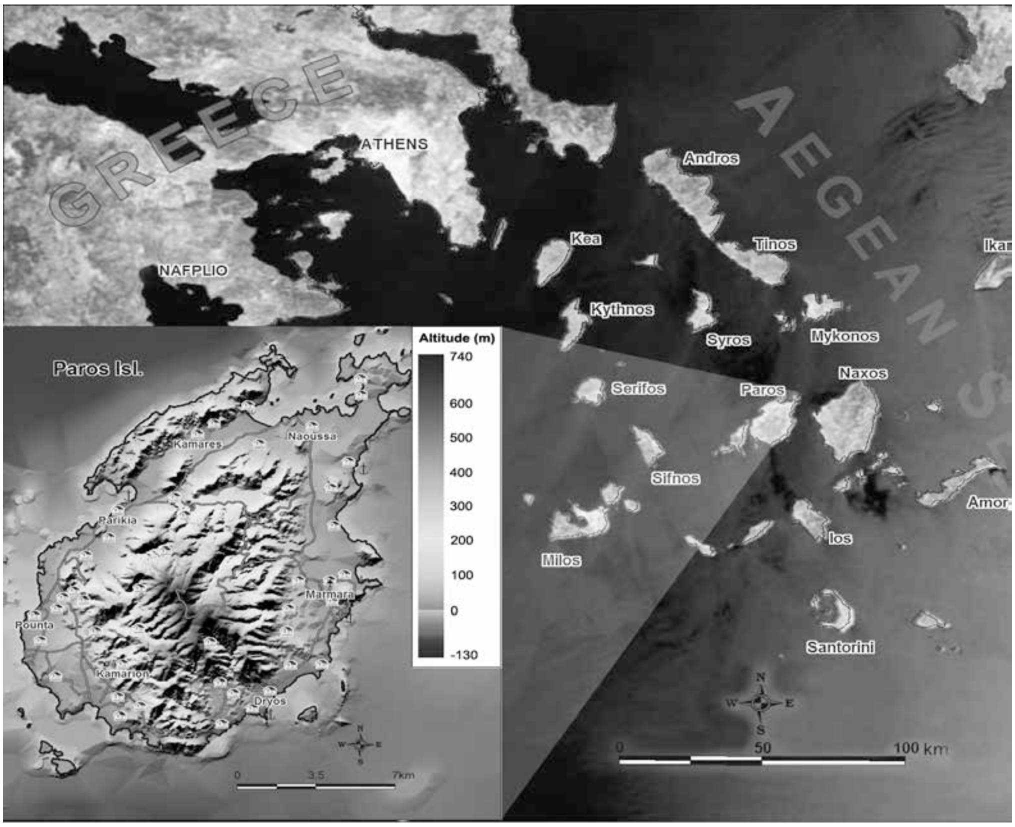

Fig. 1: The examined case study area of Paros Island.

Paros has an area close to 192 km^2 and the landscape is mountainous towards the center of the island, with a highest altitude of 771m. The areas close to the coastline are more flat and the coasts in the NE and the NW are strongly dissected. The largest plains of the island are those of Naoussa in the North, Marmara in the East, Dryos in the South-East and Pounta in the West.

The vegetation on the island is poor and exists mostly on the mountainous part and the south-eastern plains, where water is more abundant. The south-eastern plains are the most fertile in the island. The most usual crops are vineyards and wheat, while there are also plantations of olive trees.

The climate is mild Mediterranean, with light winters and very rare frost or snowfall. The summer is cool, with strong winds and extended rainless periods (Table 1).

Table 1. Mean monthly rainfall values for Paros Island in mm (years 1975–1995)

Jan	Feb	Mar	Apr	May	Jun	
83.0	63.1	50.5	20.9	8.6	2.2	Year
Jul	Aug	Sep	Oct	Nov	Dec	431.6
0.5	0.4	6.6	40.1	65.0	90.7	

The prevalent winds are North with a frequency of 55% and are very intense (7 Beaufort). The summer is characterized by north and north-east winds, while spring is characterized by south-east winds. The mean annual temperature is 18.5^{o}C (HNMS, 1999).

2.2 Geomorphology

As it is already mentioned, the slopes are steeper towards the mountainous part at the center of the island. However, the main reason for the distribution of the intermediate slope values in Paros seems to be the lithology and the tectonic status.

Likewise, these two parameters influence the drainage network. The directions of the drainage characteristics (streams, watersheds, knick points are often related to the tectonic lines of the area (Evelpidou, 1997).

The coastlines of the island are mostly rocky and characterized by steep slopes usually more than 20%. The flat coasts are met within the bays and generally within the areas with loose material. The sandy coasts appear strictly within the small bays of the island and possess the shortest part of the coastline.

2.3 Land use

In the mountainous area the vegetation is poor and according to the Corine maps the prevalent land covers are sclerophyllus bushes and natural vegetation (Table 2). Generally Paros is a non-agricultural island and the only organized cultivations appear in the East and North plains of the island. In the eastern part along with the arable lands, olive tree plantations and some vineyards are present.

Table 2. Corine types on Paros Island

Code	**Area (km^2)**	**In % of total area**	**Description**
111	0.47	0.23	Continuous urban fabric
112	3.72	1.87	Discontinuous urban fabric
131	0.40	0.20	Mineral extraction sites
211	16.71	8.42	Non irrigated arable land
221	0.91	0.46	Vineyards
223	1.93	0.97	Olive trees
242	51.68	26.04	Complex cultivation patterns
243	54.00	27.20	Principally agriculture, with natural vegetation
321	32.43	16.34	Natural grasslands
323	33.27	16.76	Sclerophyllus vegetation
334	2.97	1.50	Burnt areas
Total	**198.48**	**100**	

The human activities dramatize a very important role on the island throughout history, since from very old times it was inhabited for the extraction of its perfect quality marbles.

This led to the creation of many extraction sites throughout the entire island. Another characteristic human intervention on the landscape is the construction of innumerous terraces along the mountain slopes. Terraces can be found almost everywhere on the island and were also created during past years, but are still preserved until today as the only way of soil stabilization on the slopes.

2.4 Using the universal soil loss equation

The application of the USLE took place within GIS Environment. ArcGIS and MapInfo GIS software were both used for the necessary digitization, the maintenance of the geospatial database and the processing analysis of the input data in order to produce the expected results. GIS technology was combined with GPS technology in order to update during fieldwork, the mapped information.

The primary data were acquired in the form of maps and digitized layers provided by the corresponding Greek Organizations. The Geological map, scale 1:50000 (IGME, 1996) and the Land Resources and land Compatibility maps, scale 1:50000 (NAgReF, 1999) were acquired for Paros Island. Also the topographic map, scale 1:50000 (HMGS, 1970) was acquired as well as the digitized information concerning the Corine Land Cover data of Paros, scale 1:100000 (HeMCO, 2002). The accurate perimeter of Paros Island was digitized by the LANDSAT satellite image of the island.

The georeferencing and the digitization of the maps and the satellite image was accomplished within the MapInfo GIS environment and the projection system used was the UTM WGS 84, zone 32 northern Hemisphere. All the digitized data were imported into ArcInfo/ ArcGIS software, maintaining the same projection system, in order to administrate the database and proceed with the analysis to extract the necessary factors for the USLE.

The USLE uses the factors of rainfall erosivity, soil erodibility, slope length and slope steepness, vegetation and conservation techniques as input parameters and calculates the amount of annual soil loss per hectare. USLE only predicts the amount of soil loss that results from sheet or rill erosion on slopes and does not account for additional soil losses that might occur from gully, wind or tillage erosion. The USLE is characterized by the following equation (Wischmeier and Smith, 1978):

A = R * K * L * S * C * P

where A is the average yearly soil loss (measured in $t \cdot ha^{-1} \cdot y^{-1}$).

Therefore the USLE, in order to operate has an input need for factors concerning:

R = Rainfall erosivity factor [MJ mm $ha^{-1} \cdot h^{-1} \cdot y^{-1}$],

K = Soil erodibility factor [$t \cdot ha \cdot h \cdot ha^{-1} \cdot MJ^{-1} \cdot mm^{-1}$],

L = Slope length factor [no dimension],

S = Slope steepness factor [no dimension],

C = Crop management and vegetation cover factor [no dimension],

P = Efficiency of anti-erosion measures [no dimension].

Within ArcGIS, a separate information layer was developed for each one of the aforementioned factors by retrieving the needed information mostly from the prototype maps.

The next step was the creation of a new information layer containing a grid 5x5m that covered the entire area of Paros Island, where the corresponding USLE input factors were calculated for each cell.

The final step was the calculation of the A factor in each cell of the grid, while each input factor as well as the output factor are depicted in maps.

3. RESULTS

The first results were the preparation of the basic maps of the USLE model in order to be used in GIS environment.

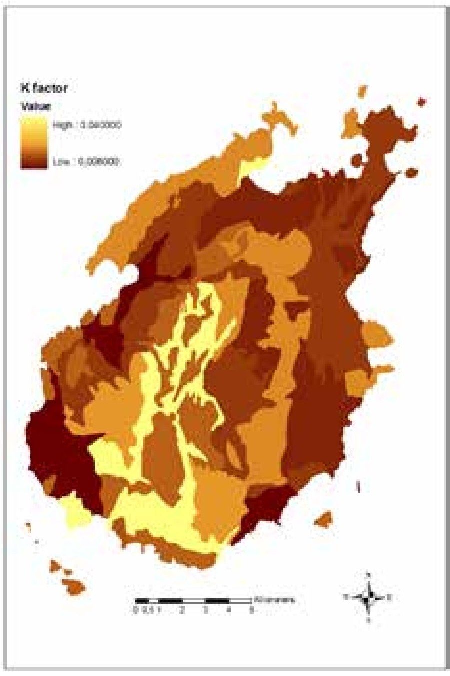

Fig. 2: Spatial distribution of the K factor.

Fig. 2 presents the soil erodibility map of the island. Soils with the highest erodibility can be found around the steepest slopes of the middle of the island.

The Digital Elevation Model (Figure 3a) of the island was needed and used to produce the LS factor map (Figure 3b), during the preparation of the USLE model.

The four different scenarios for crop rotations resulted in four C factor maps (Figure 4 a-d), which correspondingly provide with four different scenarios of soil loss for the study area.

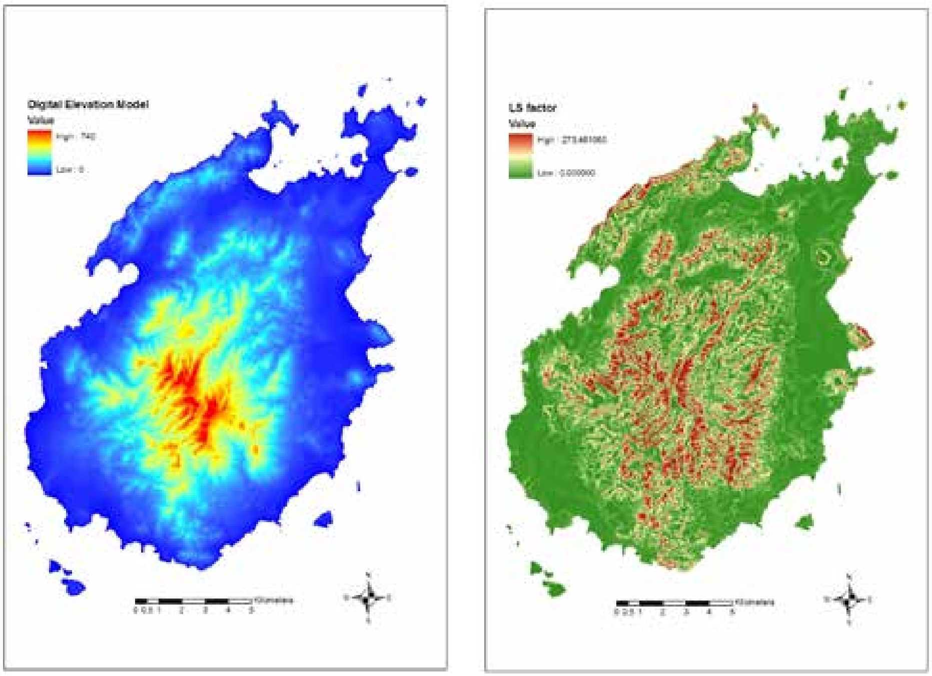

Fig. 3: a) The Digital Elevation Model, b) Spatial distribution of the LS factor.

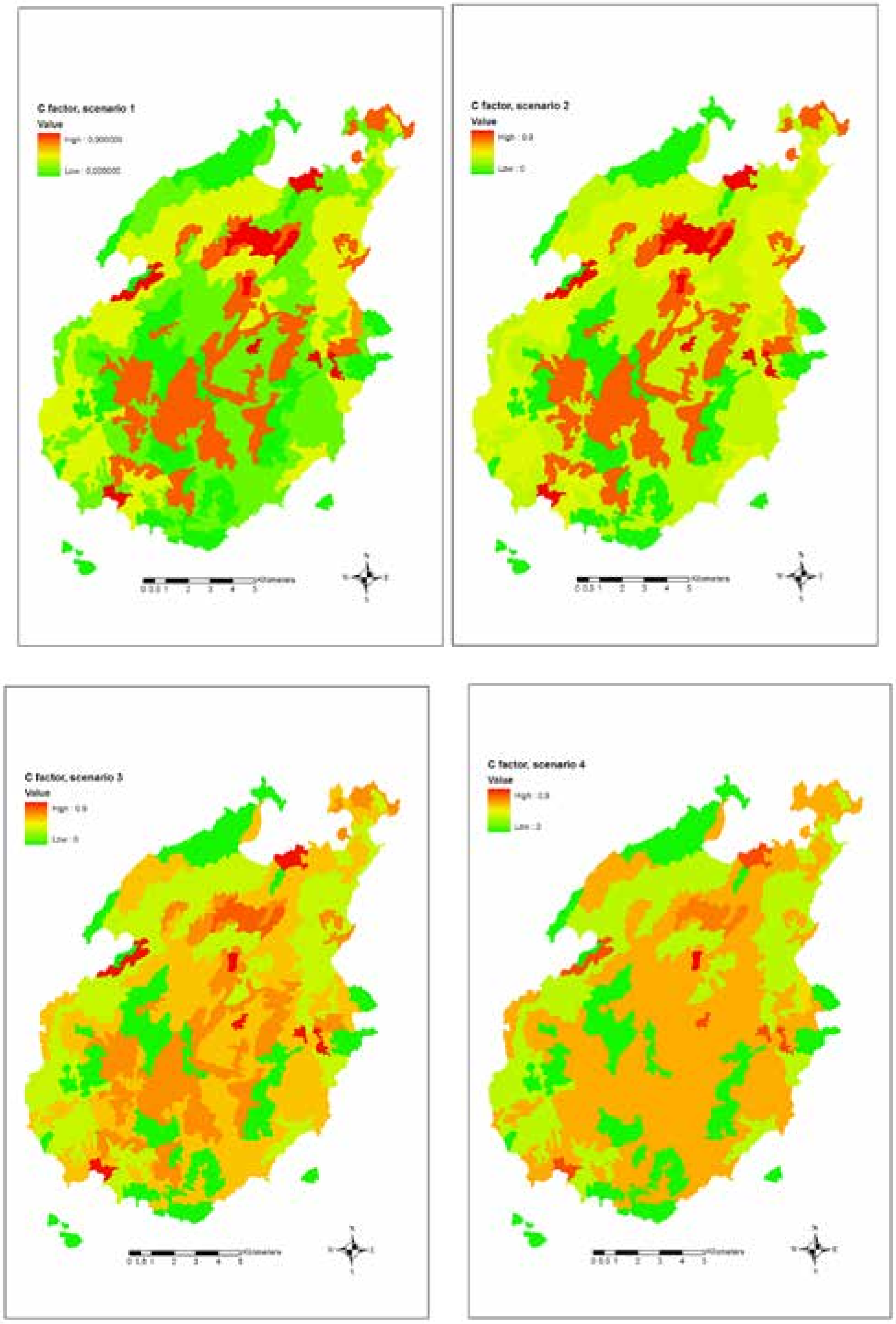

Fig. 4: Spatial distribution of the C factor according to a) the 1st scenario, b) the 2nd scenario, c) the 3rd scenario, d) the 4th scenario.

For the production of C factor maps, scenarios had to be established for land use since it is impossible to create a database or a map about the exact extent and time of the various crops to be produced on arable lands. This is why it was needed to prepare a scenario (Table 3).

Table 3. C values for different crop scenarios in arable lands		
Scenarios	**Type of Crop**	**C**
1st	a. Maximal value for winter wheat or other similar plants (barley, oat etc.). 4-5 years crop stability, with residues worked into the soil; or b. peas, beans, soybean, potato after 1 year with residues left on the surface.	0.1
2nd	a. Maize, sunflower, tobacco, sugar beet with 1 year crop stability with residues left on the surface; or b. peas, beans, soybean, potato after 3 years with residues worked into the soil.	0.25
3rd	Maize, sunflower, tobacco, sugar beet monoculture after 3 years with residues worked into the soil	0.5
4th	a. Maize, sunflower, tobacco, sugar beet monoculture after 5 years with residues worked into the soil; or b. orchards.	0.6

Table 4 contains the results of the application of the USLE for the four scenarios with the C factors of four crop rotations (or single plants/surface cover, it can be chosen by the model user).

Table 4. Yearly amounts of soil loss in four different scenarios				
	A ($t\ ha^{-1}\ y^{-1}$)	**Cell Count**	**Area (km^2)**	**% of total area**
Scenario 1	0-2	3201961	80.05	40.33
	2-11	2106646	52.67	26.53

	11<	2630555	65.76	33.13
	Total	7939162	198.48	100.00
Scenario 2	0-2	2940242	73.51	37.03
	2-11	1865043	46.63	23.49
	11<	3133877	78.35	39.47
	Total	7939162	198.48	100.00
Scenario 3	0-2	2824400	70.61	35.58
	2-11	1668453	41.71	21.02
	11<	3446309	86.16	43.41
	Total	7939162	198.48	100.00
Scenario 4	0-2	2787736	69.69	35.11
	2-11	1635170	40.88	20.60
	11<	3516256	87.91	44.29
	Total	7939162	198.48	100.00

The spatial distribution of the soil loss rates for the four scenarios is depicted in the maps illustrated on Fig. 5(a-d).

According to these maps and Table 5.5.4, the lower the C factor is, the lower the soil loss rate. It is worth noting that it is possible to achieve low C factors even by choosing specific crop types, like in the 1st scenario, so the land use of the specific areas is not necessary to change, in order to prevent soil loss.

From the maps of Figure 5(a-d) it can be seen that the areas more prone to runoff erosion and soil loss are the highest areas with steep slopes. The four plains of Naoussa, Dryos, Marmara and Pounta show low soil loss rates at the first scenario, but some parts of Naoussa and Dryos plains are severely influenced as the scenarios are getting worse.

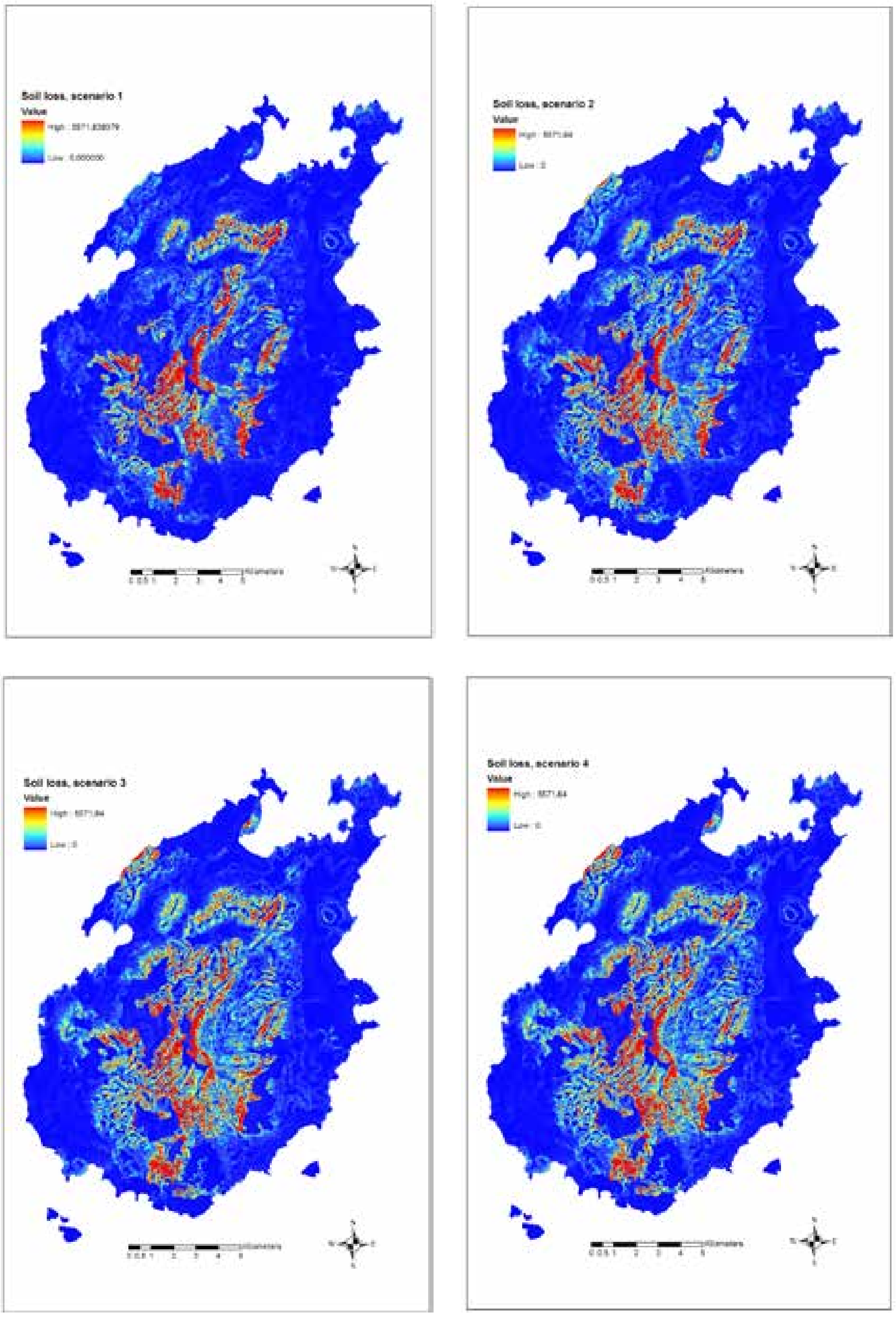

Fig. 5: Spatial distribution of the soil loss rates according to a) the 1st scenario, b) the 2nd scenario, c) the 3rd scenario, d) the 4th scenario

4. CONCLUSION

Paros today is a non agricultural island. The cultivations that existed for many centuries on the innumerous terraces of the island, have been notably diminished during the late 20th century due to the turn of local people to touristic and other modern professions. This fact aggravated the soil loss rates on the island since the terraces were abandoned driving the landscape diversity to degradation. The presentation of the soil loss rates with the use of the USLE and GIS software on the island shows clearly that the problem is very intense mostly on the higher slopes and less on the flatter areas of the island. The main active cultivations are located on the east part of the island. The introduction of different crop scenarios on the USLE's C factor for this case study, shows that selective crops can be really useful in the two main cultivated plains in the North and South-East part of the island, by decreasing the soil loss rates in some cases even less than 2 t/ha yearly. Along with the selective crops, the preservation and recreation of some terraces, even for non agricultural reasons, are proved to be effective solutions to the soil restoration on the island.

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CASE STUDY 2: THE SOIL EROSION IN THE GREATER URBAN AREAS (ATHENS - BUDAPEST)

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ABSTRACT

Climatic changes affect both coastal and inland areas. Keeping ecosystems at a balance, reassuring clean water and air along with making rational use of arable land, are of great essence for the human population. Soil degradation processes, especially aeolian and water erosion, are seriously affected by the climatic conditions and the corresponding changes. Studies were undertaken on highly eroded slopes to show the effects of sedimentation in Greece and in Hungary. Quantitative and qualitative analyses were applied on the examined sediment to indicate its deepness and nutrient content.

The Hungarian study area is situated approx. 40 km to the South-East of the capital (Budapest). The examined area is characterised by slopes of an average length of only 150-200 m, but the current farming practices and the climatic conditions already resulted in 2.6–3.2 m sediment at the bottom of the slopes. The nutrient content of the sediment is extremely high, reaching 2455 mg × kg^{-1} P_2O_5 content. Rainfall events of higher intensity are causing more runoff and much faster changes in the landscapes. Our aim is to present different scenarios for potential climatic changes that increase the possibility of more erosive rainfall events. Intense rainfalls highly increase the potential of soil loss and, thus, sedimentation.

The Greek study area is situated approximately 35 km to the North-East of the Capital (Athens), on Mountain Penteli, where intense erosion phenomena are met.

The main reasons for the increase of erosion phenomena are the fires and the climatic changes that have taken place in the recent years. Due to slope steepness, contour log terracing has been applied to protect the burned areas from erosion. A representative location has been chosen in the south-western slope of Mountain Penteli, in order to measure the erosion rate at the upper parts of the slope and the deposition rate at the lower parts.

Keywords: soil erosion, climatic changes, nutrient loss, Penteli (Greece), Galgahévíz (Hungary)

1. INTRODUCTION

The present change of climatic, economic, demographic and political conditions calls greater attention on the importance of protecting soil resources. One of the major soil forming factors is the parent material. Its research is important in order to understand the soil formation and soil degradation processes. During this research, two areas with very distinct geographical background were compared. On the Greek study site there was solid rock parent material while on the Hungarian site there was soft loessy material.

On the Greek study area (Mountain Penteli), the bedrock consists of metamorphic rocks, mainly marbles and schists (Lepsius 1893; Kober 1929; Avdis, 1990; Papadeas, 2001).The marbles are mainly composed of calcite (98%) and other minerals depending on the variety of the marble, such as muscovite, sericite and chlorite (Kleftakis *et.al.* 2000).

On the Hungarian study area geological (loess), pedological (soils formed on loessy material) and climatic conditions (high amount and intensive precipitation in June) result in a high amount of soil loss (Vona *et al.*, 2006; Centeri & Vona, 2006; Centeri *et al.*, 2006, 2008; Falusi *et al.*, 2007; Tóth and Centeri, 2008).

2. DESCRIPTION OF THE STUDIED AREA

2.1. Description of the Greek studied area

The mountain range of Penteli is situated at the NE part of the Athenian plain between Parnes and Hymettos. The elevation is 1,109 m and the mountain is mainly covered by forest (about 60 to 70%), although during the last twelve years the area has suffered three major fire events. The fires took place in July 1995, August 1998 and July 2007. These major fire events have reduced the forest coverage thus exposing the area to more intense water runoff and erosion.

Penteli area is characterised by shallow soils. The so called A-C soils (where A horizon is the humus rich layer and it is the only layer above the bedrock) are typical in Penteli. The soil forming factors do not allow the formation of thick soil layers and logging resulted in significant loss of the topsoil. Forest fires have degraded soil attributes and allowed further slimming of the soil layers. Thin soil layers appear mostly on less steeper slopes, as well as in shallow holes of the bedrock on slopes with higher inclination.

2.2. Description of the Hungarian studied area

The Hungarian study area is near Galgahévíz village. It is situated approximately 40 km from the capital of Hungary (Budapest). Galga is the name of a river that formerly flooded the area in a strip of 800-1000 m wide. This flooding delivered coarse sandy material on the lower parts of the area, while on the upper slopes thick loess material exists.

13 different soil types may be found on the area and most of them were formed on loess or loessy sand material. Most of the soils belong to sandy soils and chernozem brown forest soils and only a small amount of other soil types are present (Fluvic Vertisols 2%, water effected Vertisols 13%, salt effected soils 10%, moving sand 4%).

The climate is continental. The average annual temperature is 10-11°C, the coldest month is January with -2°C, and the warmest month is July with 21°C. The yearly temperature range is 23°C. Measurements by meteorological stations started in 1900.

The annual precipitation is 525-550 mm (<60% falls during the summer months, June has the highest amount of precipitation 65-67 mm). Only 10% of the precipitation arrives in solid form (mostly snow).

Thanks to the effects of the Mátra Mountain, the warming period takes longer in the springtime and the cooling period is faster in autumn. Abrupt temperature changes are frequent. In the first third of May, at the end of June and at the beginning of October sudden temperature falls are characteristic.

3. MATERIALS AND METHODS

3.1 Field measurements, data collection and analyses

Field work was carried out since December 2005. Based on contour log terracing that was applied in the area and with the aid of GPS technology, a set of metering devices have been installed. The location of the devices was accurately mapped and imported to the GIS (Fig. 1).

Fig. 1: Metering devices, Mountain Penteli, Greece.

Geological data was gathered from bibliographical references and fieldwork. Geomorphological data derived from interpretation of aerial photos and satellite images along with detailed field work. Satellite images were used as well, in order to determine the land use on inaccessible areas. All this primary data was imported into the GIS in order to be analysed and to create runoff models.

Empirical and bibliographical study led to the identification of the crucial parameters amongst the primary data, which are considered necessary for the outcome of the desired result. These parameters include lithology, vulnerability, slope, aspect and drainage density, which were calculated on a 100 x 100 m grid.

Lithology was estimated by grouping geological formations according to their resistance to weathering. Vulnerability expresses the complex value of resistance to erosion, taking into account chemical composition, tectonic strain and lithology parameters. Slope inclination parameters such as aspect and slope steepness define the direction and volume of soil movement. Drainage density determines the percentage of runoff.

The final estimation of erosion risk was accomplished by compiling all the aforementioned parameters with an approach of a Boolean set of logical rules, within GIS environment. The results of the GIS modelling procedure were then compared to field work observations for verification.

Due to the different environments, a different approach was used in order to address the soil loss issue. In the Greek study area, research was focused on soil volume loss. The thin soil layers on Mountain Penteli make the determination of soil volume transfer of great importance.

Measurements revealed a range of deposition justifying modelling data and estimations (see Fig. 8.1). The five parameters, climate, topography, parent material and time, described by Jenny (1941), when combined in different ways, sometimes give the same result through different paths.

3.2. Soil laboratory analyses

The distribution of P_2O_5, K_2O, $CaCO_3$ and soil organic matter content was examined in the upper and lower sections of the slope in Galgahévíz in 2004 and 2006. Average samples were taken from the upper 20 cm layer of the soil. Shallow drillings were made to examine sediment thickness in Galgahévíz (see Fig. 2). Soil samples were taken every 20 cm from the drilling. Hungarian samples were examined at the Dept. of Soil Science and Agricultural Chemistry.

Fig. 2: Shallow drilling, Galgahévíz, Hungary.

4. RESULTS

The aim of this research was to estimate the erosion risk and the soil deposition volume in the wider region of urban areas.

Table 1 demonstrates that the differences between the slope sections can be equalized by using a high amount of fertilizer. The only data which proves that erosion occurs on these slopes is the $CaCO_3$ content. It is a well-known phenomenon that loess material is mixing with the upper soil layers during cultivation and erosion is taking place, because soil is getting shallower, while loess is getting closer to the surface. Loess has a high amount of $CaCO_3$ thus above 5 % $CaCO_3$-content of the soil is a sign of erosion when this value at the bottom of the slope is only 3.85%.

Table 1. Laboratory results of the upper soil layer (0-20cm), Galgahévíz, Hungary

Sample site (Galgahévíz)	**2004 pH (KCl)**	**2006 pH (KCl)**	**2004 pH (H_2O)**	**2006 pH (H_2O)**	**2004 $CaCO_3$ (%)**	**2006 $CaCO_3$ (%)**
Arable land UTS	6.7	6.9	7.2	7.8	NA	7.57
Arable land LTS	6.9	6.9	7.2	8.1	NA	3.85

Sample site	2004 SOM	2006 SOM	2004 AL-P_2O_5	2006 AL-P_2O_5	2004 AL-K_2O	2006 AL-K_2O
	%	%	mg*kg^{-1}	mg*kg^{-1}	mg*kg^{-1}	mg*kg^{-1}
Arable land UTS	2	2,2	1524	819,9	218.4	185.9
Arable land LTS	1.5	2.4	1322	1653	218.4	197.8

UTS = Upper Third of the Slope, LTS = Lower Third of the Slope, SOM = Soil Organic Matter

The sampling of the deeper layers showed that soil layers have originated from the upper slope sections. This is the reason of the high amount of nutrients in the deeper soil horizons (Table 2). P_2O_5 content of the soil was reaching 2455 mg*kg^{-1} at the depth of 120-140 cm and remains above 2000 mg*kg^{-1} until the depth of 200 cm on the intensively cultivated slopes of Galgahévíz. The high amount of K_2O, however, does not mean nutrient overload as does the P_2O_5-content. The NH_4-N and NO_3-N has an interesting abrupt in its growth at the depth of 80 cm in both the western and eastern samples. In the eastern samples there are more abrupt changes because the depth of the sediment is thicker.

The aforementioned data prove that there were different amounts of nutrients in the upper soil layer when erosion occurred or it might as well prove the rainfall erosivity of the given period when the sediment was deposited.

Table 2. Laboratory results of the shallow drilling east of the cooperative

Code and depth (cm)	pH (KCl)	pH (H_2O)	SOM %	Ca %	P_2O_5 mg*kg^{-1}	K_2O mg*kg^{-1}	NH_4-N mg*kg-1	NO_3-N mg*kg^{-1}
W0-20	8.1	7.4	2.0	ND	ND	ND	ND	ND
W20-40	8.1	7.2	2.6	1.3	1266.0	383.0	6.9	6.9
W40-60	7.7	7.2	5.4	1.3	1283.0	430.0	6.9	6.9
W60-80	8.0	7.2	2.4	1.1	1371.0	272.0	10.3	10.3
W80-100	8.0	7.2	1.7	0.8	1540.0	294.0	3.4	6.9
W100-120	7.9	7.2	1.8	1.0	1304.0	392.0	3.4	3.4

W120-140	8.3	7.6	1.3	1.0	392.0	407.0	6.9	6.9
W140-160	8.2	7.5	ND	1.2	660.0	424.0	3.4	3.4
W160-180	8.4	7.7	ND	2.1	669.0	478.0	6.9	10.3
W180-200	8.5	7.8	0.3	3.4	1012.0	438.0	13.7	17.2
W200-220	8.5	8.0	0.9	6.1	863.0	329.0	<KH	10.3
W220-240	8.5	8.2	1.0	7.3	895.0	246.0	3.4	24.1
E0-20	8.0	7.3	3.1	2.3	1944.0	334.0	6.9	3.4
E40-60	8.0	7.2	1.8	1.7	1968.0	295.0	6.9	6.9
E60-80	8.0	7.1	2.7	1.2	2247.0	283.0	3.4	6.9
E80-100	8.1	7.3	2.4	1.3	1950.0	277.0	6.9	3.4
E100-120	7.8	8.4	1.0	0.9	2221.0	320.0	3.4	3.4
E120-140	8.0	7.3	ND	1.0	2455.0	357.0	6.9	3.4
E140-160	7.5	8.1	0.2	1.4	2284.0	353.0	3.4	3.4
E160-180	8.1	7.3	1.0	0.7	2204.0	379.0	<KH	<KH
E180-200	8.2	7.4	1.9	0.7	2040.0	355.0	10.3	6.9
E200-220	8.3	7.5	0.8	0.8	1291.0	312.0	3.4	3.4
E220-240	8.1	7.4	0.0	1.8	941.0	262.0	10.3	10.3
E240-260	8.3	7.4	ND	2.7	834.0	253.0	3.4	<KH
E260-280	8.3	7.4	ND	2.2	706.0	217.0	<KH	6.9
E280-300	8.3	7.5	ND	2.3	502.0	222.0	10.3	6.9
E300-320	8.6	7.6	0.2	7.5	182.0	107.0	3.4	3.4

ND = no data, W = west of the cooperative, E = east of the cooperative, KH = limit of measurability.

Nutrients are known to move downslope with soil erosion. In Galgahévíz, Hungary, high amounts of nutrients were found both at the upper and the lower sections of the intensively cultivated slopes of Galgahévíz.

Table 3. Laboratory results of the soil analyses in the Sósi Creek watershed								
No	**Vegetation**	**Slope**	**pH(H_2O)**	**pH(KCl)**	**$CaCO_3$ (%)**	**SOM (%)**	**AL-P_2O_5 (mg/kg)**	**AL-K_2O (mg/kg)**
1	Alfalfa	Upper	7.8	7.3	11.70	1.51	98.8	137.4
		Middle	8.4	7.3	9.26	1.54	128.4	153.0
		Lower	8.2	7.1	3.28	2.22	119.6	171.5
2	Deciduous forest	Upper	7.1	7.1	3.41	3.25	215.4	353.8
		Middle	7.1	7.1	4.14	3.48	159.1	444.2
		Lower	7.2	7.2	7.87	2.26	205.5	377.3
3	Arable land	Upper	7.1	7.1	1.55	1.49	160.1	230.6
		Middle	7.3	7.3	15.60	1.37	156.1	191.5
		Lower	7.2	7.2	7.41	1.76	169.9	235.6
4	Deciduous forest	Upper	8.1	7.2	4.97	3.07	47.4	175.1
		Middle	8.2	7.4	12.92	4.06	61.3	196.5
		Lower	8.0	7.3	10.77	2.83	110.7	248.4

SOM = Soil Organic Matter, AL = ammonium-lactate

Erosion risk and soil deposition volume in Mountain Penteli are illustrated in the form of G.I.S. generated maps. Fig. 3 shows the spatial distribution of erosion and deposition zones. As expected, erosion is more intense on more inclined slopes.

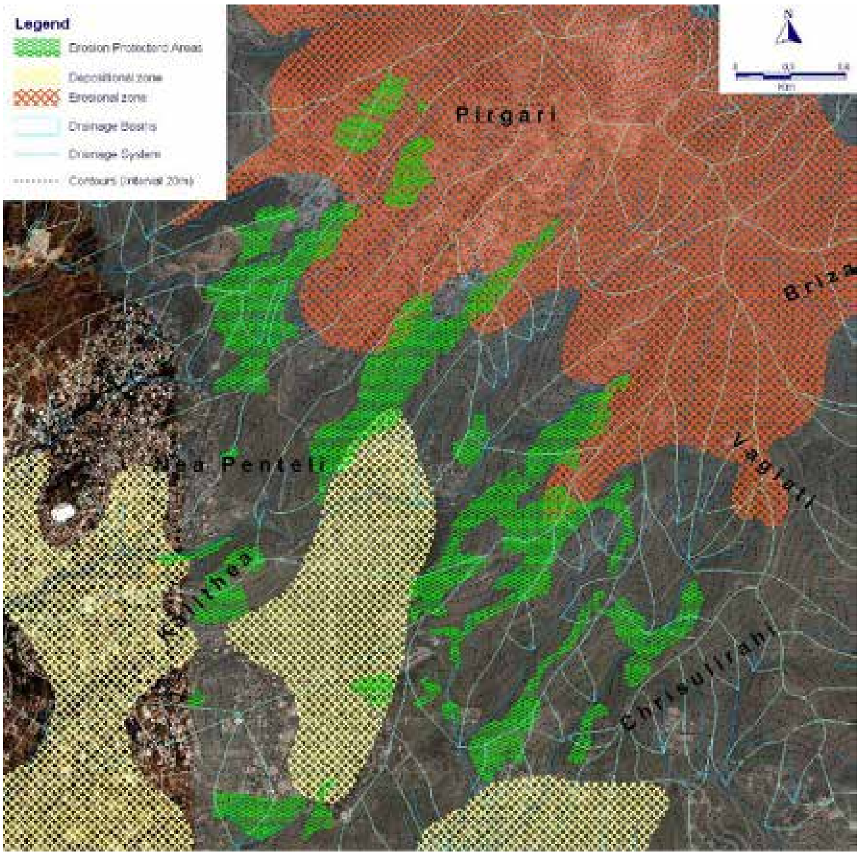

Fig. 3: The spatial distribution of erosion - deposition zones in the burned area and the zones with the tree trunks for the protection from erosion.

The map representing erosion risk (on a 100x100 m grid – Fig. 4) reveals that areas with greater slope inclination are more susceptible to erosion. It also demonstrates the effectiveness of contour log terracing. The areas where contour log terracing was applied show medium to low erosion risk. The qualities of soil movement overall, reveal that altitude is not the main decisive factor. The morphological attributes (surface runoff, aspect, and slope) affect the soil path greatly. However, contour log terracing has proven to be very effective if applied with caution.

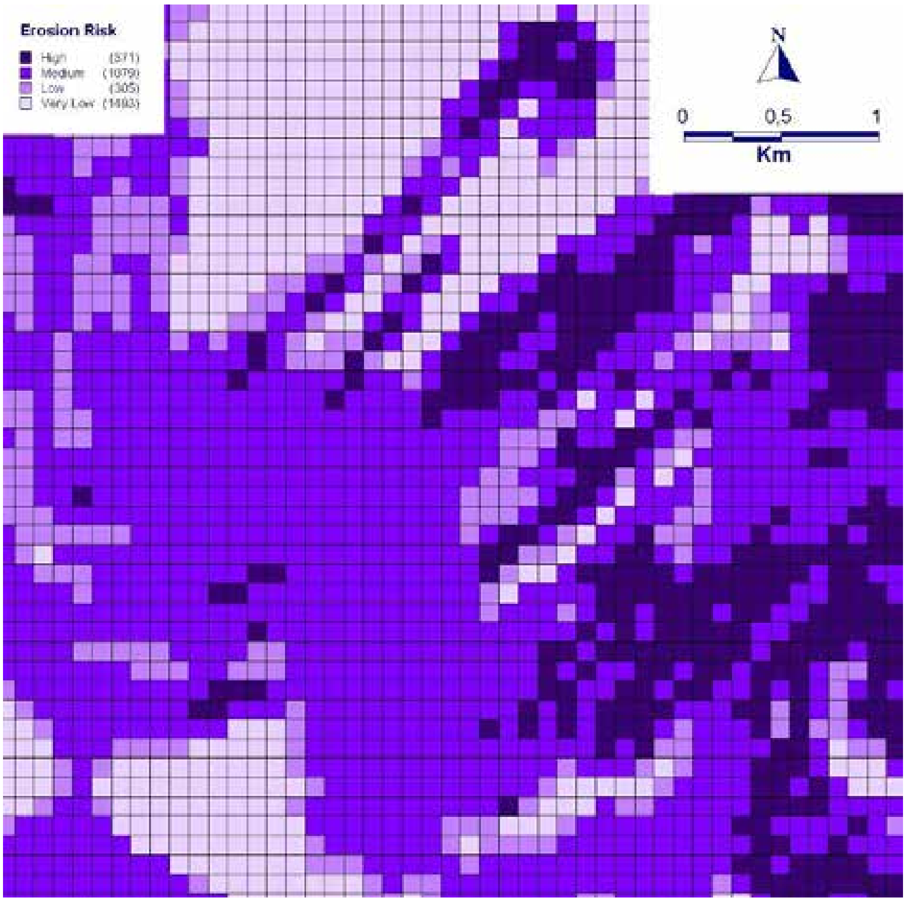

Fig. 4: The highest erosion risk areas have been calculated at the East part and the erosion protected areas now have medium to low erosion risk.

5. CONCLUSIONS

The two different study areas show different soil loss results. On the Hungarian site severe erosion could occur – as it is proven by the high amount of nutrients in the layers of the shallow drillings – on the Greek study site the parent material and the steep slopes did not yield thick soil layers, so that much erosion could not be seen since there is only little soil layer on the area.

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CASE STUDY 3: SITE PREPARATION IMPACTS ON PHYSICAL AND CHEMICAL FOREST SOIL QUALITY INDICATORS

Felícia Fonseca, Tomás de Figueiredo and Afonso Martins

1. INTRODUCTION

Forest soils in the Mediterranean region frequently have limited rooting depth, high coarse elements and low organic matter content, which tend to limit water storage in the soil profile. Accordingly, application of site preparation techniques is essential to enhance soil water storage and availability in these environments (Querejeta et al., 2001; Alcázar et al., 2002; Piatek et al., 2003; Imaz et al., 2010). However, site preparation for afforestation currently lacks accurate planning, based on sound experimental results driving to techniques most adequate to each situation and respecting stand productivity and ecosystem sustainability requirements. Improving soil quality is one of the most important factors for sustaining the global biosphere and fundamental in forest systems sustainability (Wang and Gong, 1998). A simple set of established soil properties can provide useful information on soil quality (Sparling et al., 2004). Soil quality has been defined as “the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health" (Doran and Parkin, 1994). Land use and management practices seriously impact the direction and degree of soil quality changes in time and space (Wang and Gong, 1998). Possibly the most significant impact of site preparation from a soil quality perspective is on rooting depth and soil hydrological processes, because increase the availability of resources that plants have access (water and nutrients). Runoff and sediment loss is commonly highest in the first few years after site preparation, for the reason that canopy cover is scarce and ground vegetation may be insufficient for controlling erosion (Lucci and Della Lena 1994; Figueiredo et al., 2011).

Taking in account the soil complexity and the high number of properties that can be monitoring, it is important to select the most appropriate to the objectives of soil management practices. For comparison different site preparation techniques (Schoenholtz et al., 2000) suggested as soil quality indicators, soil effective thickness, bulk density, soil penetration resistance, particle size, aggregate stability, organic matter content, nutrient availability, cation exchange capacity (CEC), base saturation (BS) and pH, because these variables are generally sensitive to soil management practices. Due to inappropriate land use and management, soil erosion, acidification, nutrient depletion, pollution and other natural resource problems have been affected human development (Lal, 1990; Figueiredo et al., 2011).

The objectives for this study were (i) evaluate changes in physical and chemical properties as indicators of forest soil quality, owing to mechanical site preparation; (ii) to compare soil quality using well-known indicators for that forest systems such as rooting depth, bulk density, particle size and nutrients availability; (iii) based on soil quality and productivity to indicate which techniques are most appropriate for the region under study.

2. MATERIAL AND METHODS

This study is a part of a large project, where were tested several site preparation operations in the environmental impacts (soil erosion and carbon storage), behavior of forest species installed (mortality, height and diameter growth, biomass production) and evolution of soil properties. Details of the design of this experiment are available in another published works (Fonseca et al., 2011; Figueiredo et al., 2011). Accordingly, only a succinct review is presented here.

The experimental field was established near Macedo de Cavaleiros, Northeast Portugal at 41^{o} 35´N and 6^{o} 57´W, 660 to 701m altitude, mean annual temperature 12^{o}C, mean annual rainfall 656 mm, with a typically Mediterranean seasonal distribution (INMG, 1991). According to FAO/UNESCO (1988), soils are dystric Cambisols and dystrict Leptosols developed on schist, in the area characterized as sandy-loam, with high stoniness, normally acid, moderate to poor in organic matter content, with low to very low P and low to moderate K contents (Agroconsultores e Coba 1991; Fonseca 2005).

Experimental design comprised three blocks, where six treatments representing different intensities of soil disturbance by mechanical operations for site preparation, were randomly distributed on experimental plots (Table 1). This table includes also a treatment without disturbance (TSMO), which corresponds to the original soil and is taken as a reference for comparison with the remainder treatments in what concerns the tillage effects on chemical and physical soil properties. Site preparation techniques under test were selected among a set of commonly applied in afforestation schemes, yet with no consistent experimental base for such options.

Treatment	Description of site-preparation operations
TSMO	Original soil control (no intervention on the original abandoned field)
	Slight disturbance
SMPC	No subsoiling, no ploughing, plantation with hole digger (60 cm depth)
RCAV	Subsoiling over the whole area, with covering shovel (around 60 cm depth)
	Moderate disturbance
SRVC	No subsoiling, contour bunds shaped by two plough passes (around 90 cm depth)
RLVC	Subsoiling in future plantation rows, contour bunds shaped by two plough passes (around 90 cm depth)
	High disturbance
RCVC	Subsoiling over the whole area, contour bunds shaped by two plough passes (around 90 cm depth)
RCLC	Subsoiling over the whole area, contour ploughing over the whole area (around 90 cm depth),

Table 1 : Treatments related to six soil preparation operations before plantation

The species selected were *Pseudotsuga menziesii* (*PM*) and *Castanea sativa* (*CS*), which were planted at a distance of 4 x 2 m in alternate rows, with two rows for each plot, 12 plants in each row with a total of 24 plants per species. The plantation was made by hand, in February 2002, using nursery seedlings, *PM* in plastic bags and *CS* of bare root.

Before site preparation operations 48 soil profiles were observed, and one year after planting 6 soil profiles per treatment were observed in the plantation line. Soil samples were taken at 0-20, 20-40 and 40-60 to assess the effects of site preparation on soil properties and possible relationships with plant response. Soil quality monitoring was based on morphological properties observed in soil profiles and laboratory analysis of soil samples collected in the profiles. The soil properties chosen in this study were selected taking in account the main factors limiting plant development in the Mediterranean region like water and nutrients. Thus, was considered to be relevant soil effective thickness, bulk density, clay content and soil organic C, because they have an important role in water storage, nutrient availability and root expansion. Chemical properties as total N, available P and K, exchangeable bases, cation exchange capacity and pH are relevant for biological activity and plant nutrition. Soil quality measures such as the ones proposed here can be usefully combined with measurements of soil loss (Sparling et al., 2004). Data on soil loss can be found in Figueiredo et al. (2011). In this work will be addressing the effects of soil erosion on soil properties, using an enrichment ratio (E), where E = nutrient content or other property in eroded sediment / nutrient content or other property in original soil (Gachene et al., 1997). Thus, whenever E > 1 means that the eroded sediment was enriched in the element analyzed, in relation to original soil, which may indicate the occurrence of natural soil fertility loss by erosion action. Also, in order to identify differences in soil properties between original soil (TSMO) and site preparation operations (SMPC, RCAV, SRVC, RLVC, RCVC and RCLC), was applied the same idea of enrichment ratio (E), but now E = soil property in each site preparation operations / soil property in original soil. When E > 1 means that the soil property analyzed was improved with site preparation.

Soil samples collected in soil profiles, were air dried, sieved (mesh size 2 mm) and analysed for organic C (Walkley-Black method), total N (Kjeldahl method), available P and K ((Egner-Riehm method). Exchangeable bases were determined by atomic adsorption (Ca^{2+} and Mg^{2+}) and by flame emission spectrophotometry (K^{+} and Na^{+}). Cation exchange capacity was calculated as the sum of exchangeable bases and exchangeable acidity. Soil pH was measured in a soil-water suspension (1:2.5 soil-water ratio). Effective soil thickness, defined as the set of layers most explored by plant roots, was measured in each profile during soil profile description. Bulk density was determined in undisturbed samples, weighting oven-dried soil (at 105 ºC), collected in 100 cm^3 cylinders in depths 0-5, 5-15, 15-30 and 30-60 cm. Particle size analysis was done by pipette method.

Statistical analysis of data comprised one-way ANOVA and multiple comparison of averages (Tukey, 5%), performed to assess the significance of treatment effects on results. Simple correlation analyses were applied to estimate relationships between soil properties and variables measured in plants.

3. RESULTS AND DISCUSSION

3.1. Physical soil quality indicators

In slight soil disturbance treatments (SMPC and RCAV), the soil effective thickness (rooting depth) remains very similar to the original soil (TSMO), but in treatments with moderate and intensive site preparation (SRVC, RLVC and RCLC) there are a significant increase ranging from 30 to 40 cm (Fig. 1). The soil effective thickness is a property that varies very little over time and reflects the soil physical conditions ability for plant growth (Wang and Gong, 1998, Schoenholtz et al., 2000).

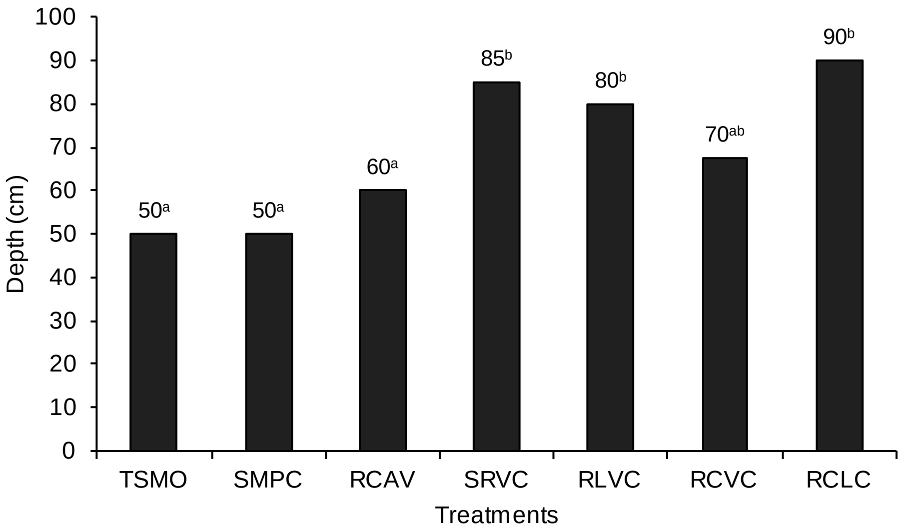

Fig. 1: Means of soil effective thickness one year after site preparation. For treatments, averages with the same letter are not significantly different ($P < 0.05$).

Given the climatic characteristics of the study area (high summer water deficit), the soil effective thickness proved to be a property determinant of survival and growth of forest species installed (Fig. 2), showing a good correlation at the end of 42 months with survival (r^2 = 0.942 and 0.745, respectively for *PM* and *CS* species) and with the height and diameter growth for *PM* (r^2 = 0.859 and 0.781, respectively) and *CS* (r^2 = 0.919 and 0.975, respectively). Moreover, in plots in which soil effective thickness did not exceed 60 cm, the mortality was above 95% for both species (Fonseca et al., 2011).

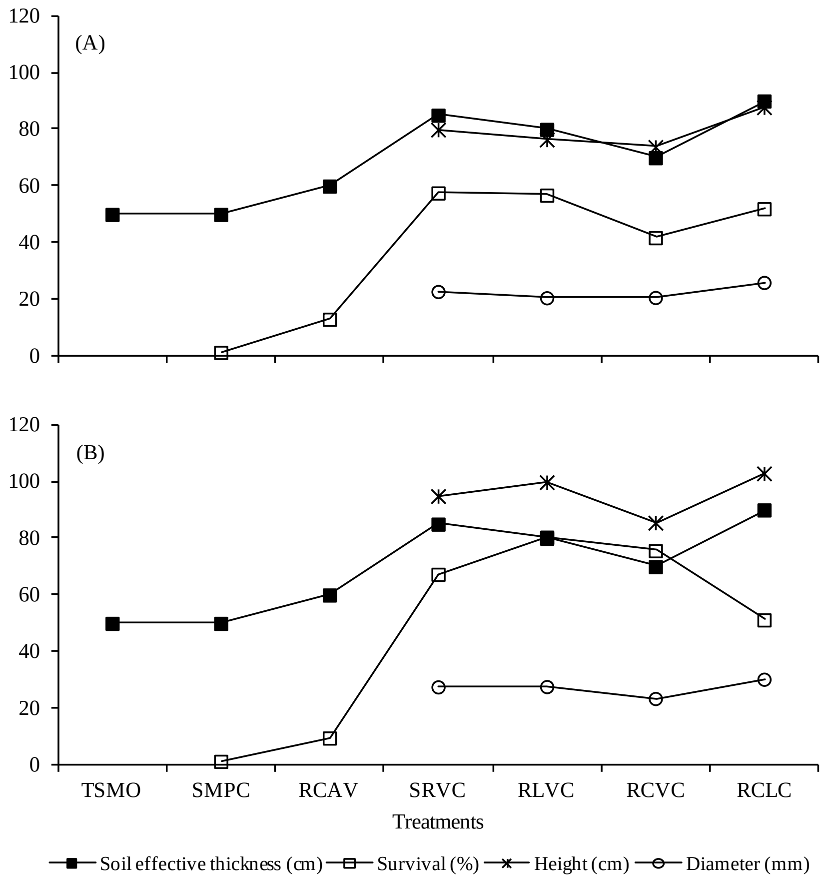

Fig. 2: Relationship of survival and growth in height and diameter of PM (A) and CS (B) species with the soil effective thickness, according to the treatments.

Also, treatments with moderate and intensive site preparation contributed to decrease bulk density in all depths considered. Bulk density values ranging from 1.44 to 1.76 g cm^{-3} in the original soil (TSMO) and 1.33 to 1.39 g cm^{-3} in RCVC treatment, respectively for 0-5 and 30-60 cm depth. The remaining treatments showed values intermediate to these (Fig. 3). Furthermore, intensification of site preparation also contributed to the reduction of spatial and vertical bulk density variability (see meaning letters of bars in Fig. 3), which favors water and oxygen availability (Schoenholtz et al., 2000) and root development (Logsdon and Karlen, 2004).

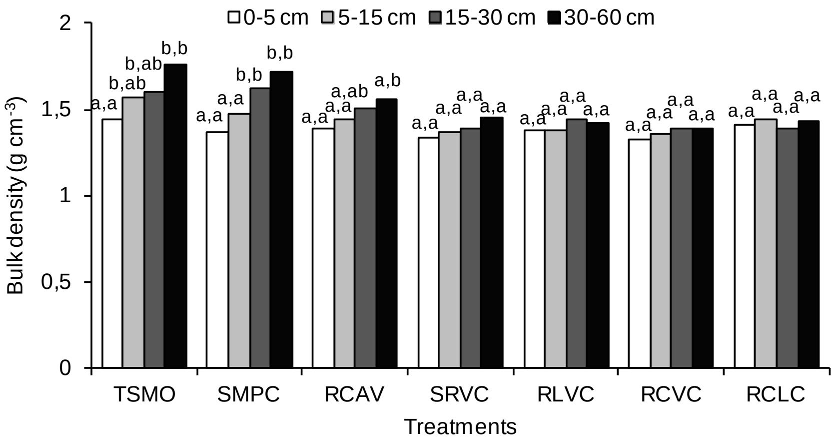

Fig. 3: Soil bulk density (g cm-3) per treatment and soil depth. The first letter indicates that for the same treatment, means followed by the same letter depths are not significantly different; the second letter indicates that for the same depth, means followed by the same letter treatments are not significantly different (P < 0.05).

Soil texture classes ranging among loam and sandy-loam (Table 2). However, there are variations in sand, silt and clay fractions, which can lead to differences in pore size and consequently cause differences in aeration conditions and soil water retention (Dexter, 2004). In general, there are no significant differences between treatments for the fine earth fraction (sand, silt and clay), but the treatments with furrow hillock surface soil (SRVC, RLVC and RCVC) show an increase in clay content in surface mineral layer (0-20 cm), showing a reverse trend in the deeper soil layers (20-40 and

40-60 cm), when compared with the original soil (TSMO). Once the young plants are essentially dependent on the moisture content of the surface layers (Fernandes and Fernandes 1998; Kanegae et al., 2000), these higher levels of clay can help increase water retention and nutrients absorption during summer rainfall, with positive impact on plantations' success.

Soil properties	Depth (cm)	Treatments						
		TSMO	SMPC	RCAV	SRVC	RLVC	RCVC	RCLC
Coarse sand (%)	0-20	37.9[a]	35.2[a]	32.6[a]	33.1[a]	27.7[a]	35.0[a]	37.6[a]
	20-40	36.7[a]	34.9[a]	34.2[a]	33.1[a]	33.0[a]	36.6[a]	33.1[a]
	40-60	42.1[b]	36.4[b]	33.6[ab]	22.8[a]	30.6[ab]	39.8[b]	35.6[b]
Fine sand (%)	0-20	33.0[a]	32.7[a]	33.8[a]	28.8[a]	34.1[a]	26.3[a]	27.8[a]
	20-40	29.4[a]	26.2[a]	32.7[a]	32.0[a]	32.4[a]	32.3[a]	32.7[a]
	40-60	25.5[a]	30.7[a]	22.2[a]	38.9[a]	33.7[a]	31.9[a]	29.7[a]
Silt (%)	0-20	18.6[a]	21.1[ab]	20.8[ab]	24.2[ab]	25.2[ab]	27.5[b]	25.9[ab]
	20-40	19.0[a]	19.9[a]	19.6[a]	24.6[a]	23.5[a]	20.8[a]	21.4[a]
	40-60	16,7[a]	14.7[a]	20.8[a]	26.1[a]	19.3[a]	16.7[a]	23.3[a]
Clay (%)	0-20	10.6[ab]	11.0[ab]	12.7[ab]	14.0[b]	12.9[ab]	11.1[ab]	8.7[a]
	20-40	14.9[a]	19.1[a]	13.5[a]	10.3[a]	11.2[a]	10.3[a]	12.5[a]
	40-60	15.7[ab]	18.3[ab]	23.5[b]	12.1[a]	16.4[ab]	11.6[a]	11.4[a]

Tabe .2 : Soil particle size distribution under different site preparation techniques (treatments). In the same line, means followed by the same letter are not significantly different (P > 0.05)

3.2. Chemical soil quality indicators

When compared with the original soil (TSMO), the treatments with moderate and high soil disturbance (SRVC, RLVC, RCVC and RCLC), contributed to a significant decrease in organic C concentration (59, 48, 35 and 47% for the SRVC, RLVC, RCVC and RCLC treatments, respectively) and total N (45, 33, 30 and 44% for the same treatments, respectively) in surface layer (0-20 cm) (Table 3 and Fig. 4), being the result of the disruption of the balance between formation and mineralization humus processes (Madeira et al., 1989; Islam and Weil, 2000; Saviozzi et al., 2001). In the same treatments, C and N tends to increase in depth, showing significantly higher levels in deeper layer (40-60 cm), which is associated with the migration of organic matter in the soil profile by the action of tillage. Despite the changes observed in the levels of carbon and nitrogen, Dick et al. (1998) reported that the largest variations in organic matter content occur during the first five years after site preparation, with little variation beyond that period.

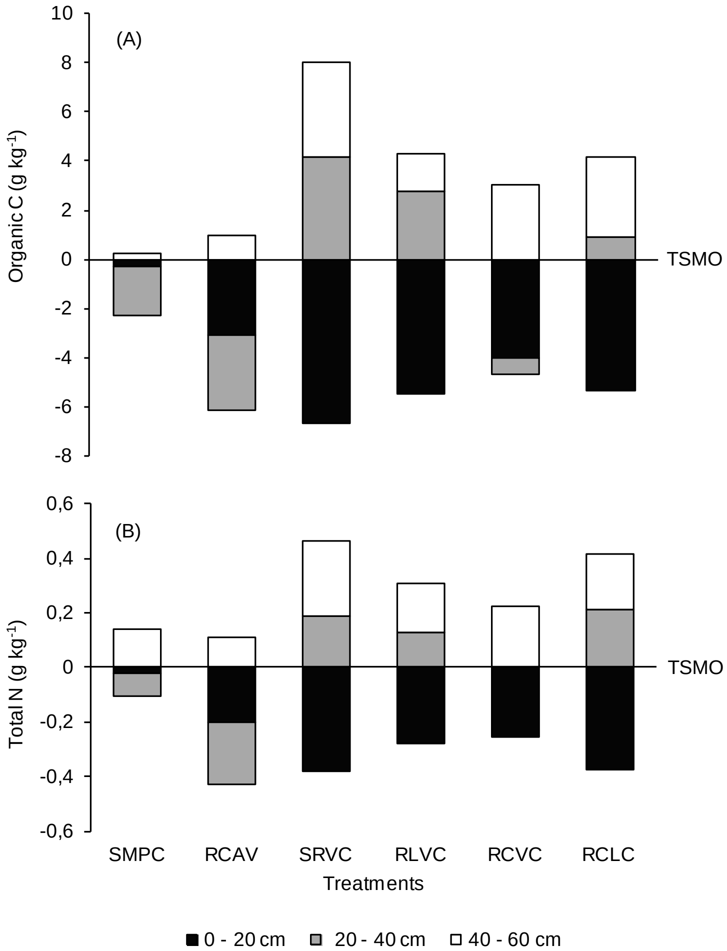

Fig. 4: Variation of soil organic carbon (A) and total nitrogen (B) in relation to the original soil (TSMO).

Soil properties	Depth (cm)	Treatments						
		TSMO	SMPC	RCAV	SRVC	RLVC	RCVC	RCLC
Organic C (g kg^{-1})	0-20	11.4^{b}	11.1^{b}	8.3^{ab}	4.7^{a}	5.9^{a}	7.4^{a}	6.0^{a}
	20-40	6.3^{ab}	4.3^{ab}	3.2^{a}	10.5^{b}	9.0^{b}	5.6^{ab}	7.2^{ab}
	40-60	2.5^{a}	2.7^{a}	3.5^{a}	6.4^{a}	4.0^{a}	5.5^{a}	5.7^{a}
Total N (g kg^{-1})	0-20	0.85^{b}	0.82^{b}	0.65^{ab}	0.46^{a}	0.57^{a}	0.59^{a}	0.47^{a}
	20-40	0.54^{ab}	0.46^{ab}	0.30^{a}	0.73^{b}	0.66^{ab}	0.54^{ab}	0.75^{b}
	40-60	0.29^{a}	0.43^{ab}	0.40^{ab}	0.57^{b}	0.47^{ab}	0.51^{ab}	0.49^{ab}
SEB (cmol kg^{-1})	0-20	1.52^{a}	1.39^{a}	1.77^{a}	3.32^{b}	1.57^{a}	1.19^{a}	2.26^{ab}
	20-40	1.71^{a}	1.67^{a}	2.02^{ab}	3.35^{b}	1.78^{a}	1.32^{a}	2.59^{ab}
	40-60	1.81^{a}	2.82^{ab}	3.04^{b}	1.53^{a}	1.97^{a}	1.67^{a}	1.22^{a}
CEC (cmol kg^{-1})	0-20	3.31^{a}	3.26^{a}	3.92^{a}	6.57^{b}	3.63^{a}	3.30^{a}	4.59^{a}
	20-40	4.24^{a}	4.43^{a}	4.16^{a}	5.35^{a}	4.68^{a}	3.29^{a}	4.31^{a}
	40-60	5.26^{a}	5.32^{a}	6.42^{a}	4.66^{a}	4.30^{a}	5.34^{a}	4.24^{a}
BS (% CEC)	0-20	47.1^{a}	42.5^{a}	43.0^{a}	44.8^{a}	43.2^{a}	36.7^{a}	50.2^{a}
	20-40	42.8^{a}	41.5^{a}	48.9^{a}	48.8^{a}	38.6^{a}	39.3^{a}	45.2^{a}
	40-60	39.8^{a}	50.6^{a}	51.4^{a}	34.5^{a}	46.3^{a}	36.8^{a}	32.0^{a}
pH (H_2O)	0-20	5.1^{a}	4.9^{a}	5.1^{a}	5.0^{a}	4.9^{a}	4.8^{a}	5.0^{a}
	20-40	5.0^{a}	4.9^{a}	5.0^{a}	4.9^{a}	4.9^{a}	4.9^{a}	5.0^{a}
	40-60	5.0^{a}	4.9^{a}	5.0^{a}	4.8^{a}	5.0^{a}	5.0^{a}	4.7^{a}

SEB - sum exchange bases; CEC - cation exchange capacity; BS - base saturation

Table 3: Chemical soil properties under different site preparation techniques (treatments). In the same line, means followed by the same letter are not significantly different ($P > 0.05$)

Available P values (Fig. 5A) range from very low (0-25 mg kg^{-1}) to low (26-50 mg kg^{-1}) (Santos, 1991). These values can be related to phenomena of retention and fixation, tending to increase with depth and tillage intensity, which is very important because Portuguese soils are very poor in this nutrient (Arrobas and Coutinho, 2002).

However, these concentrations do not necessarily mean that there are deficiencies of phosphorus given the presence of mycorrhizal fungi (commonly observed during root systems observation) to promote plant nutrition, including phosphorus absorption, even if it is in low concentrations in soil (Binkley, 1986; Honrubia, 1992). In relation to available K (Fig. 5B), this shows values considered moderate (51-100 mg kg^{-1}) (Santos, 1991). Site preparation techniques influenced positively the potassium availability owing to changes in other factors such as aeration, temperature and moisture content (Sharma et al., 2005), but without a consistent trend.

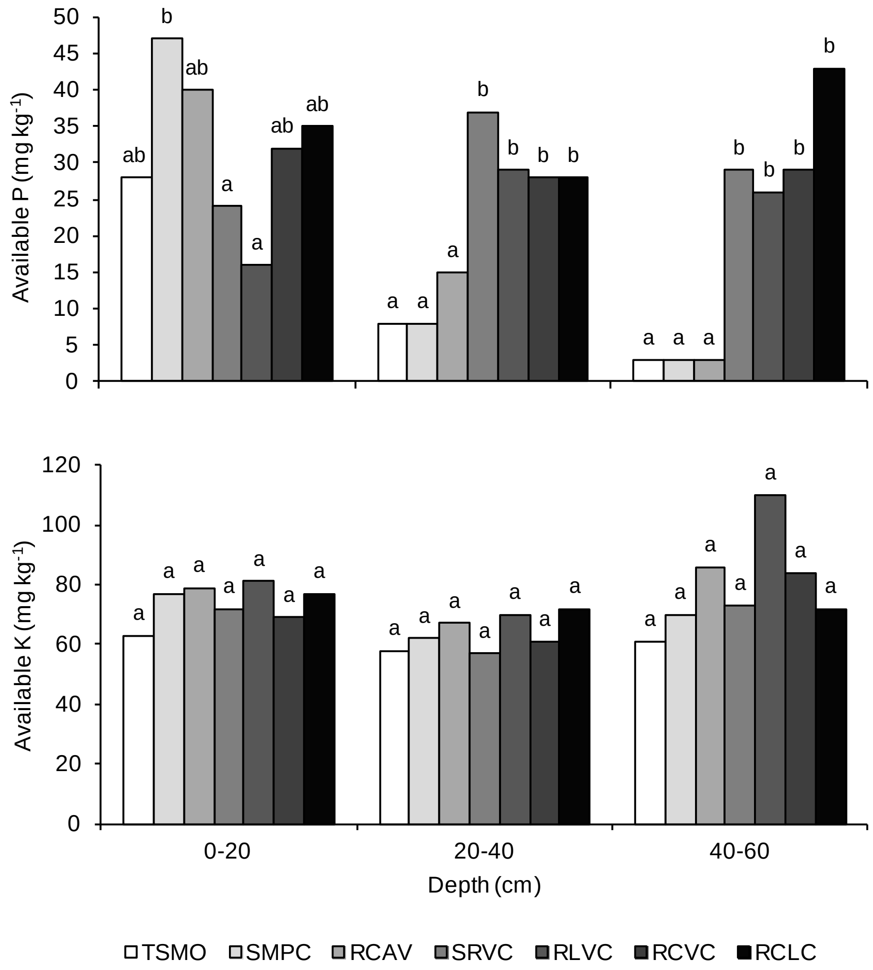

Fig. 5: Available phosphorus (A) and potassium (B) one year after site preparation. For treatments, averages with the same letter are not significantly different (P < 0.05).

In surface soil layer, sum of exchangeable bases increase, although only significantly in SRVC (Table 3). These values are considered very low (Metson, 1956), which is justified by the lithology of the parent material and leaching phenomena. Calcium is the most important basic cation in the soil, contributing around 70% for the sum exchangeable bases.

3.3. Enrichment of surface soil layer and eroded sediment after site preparation

Values of enrichment ratios (E) for clay, soil organic C, total N, available P and K, exchange bases (Ca^{2+} and Mg^{2+}), cation exchange capacity (CEC) and pH (H_2O) of the surface soil layer (0-20 cm) and eroded sediment are depicted in Table 4. Data for RCLC treatment are missing because the high surface ground roughness after site preparation was such that erosion monitoring was discarded.

Parameter	Treatments				
	SMPC	RCAV	SRVC	RLVC	RCVC
Surface soil layer (0-20 cm) (TSMO values equal 1)					
Clay (%)	1.13a	1.31a	1.44a	1.33a	1.25a
Organic C (g kg^{-1})	0.98a	0.73a	0.41a	0.52a	0.65a
Total N (g kg^{-1})	0.97a	0.76a	0.55a	0.67a	0.70a
Available P (mg kg^{-1})	1.68b	1.43b	0.86a	0.57a	1.14a
Available K (mg kg^{-1})	1.22a	1.25a	1.14a	1.29a	1.10a
Ca^{2+} (cmol kg^{-1})	0.95a	1.25a	2.21b	1.10a	0.84a
Mg^{2+} (cmol kg^{-1})	0.75a	1.13a	2.84b	0.94a	0.56a
CEC (cmol kg^{-1})	0.98a	1.18a	1.98b	1.10a	1.00a
pH (H_2O)	0.96a	1.00a	0.98a	0.96a	0.94a
Eroded sediment (Treatment surface soil values equal 1)					
Clay (%)	0.60a	0.62a	0.53a	0.67a	0.84a
Organic C (g kg^{-1})	1.50b	1.05a	1.40b	0.91a	1.46b
Total N (g kg^{-1})	2.02b	1.35a	2.29b	1.42a	1.14a
Available P (mg kg^{-1})	0.82a	0.48a	0.56a	0.64a	0.73a
Available K (mg kg^{-1})	1.57b	0.96a	1.24a	1.68b	1.57b

Ca^{2+} (cmol kg^{-1})	1.43b	0.96a	1.11a	0.82a	1.05a
Mg^{2+} (cmol kg^{-1})	2.20b	1.02a	1.55b	1.69b	1.11a
CEC (cmol kg^{-1})	0.95a	0.65a	0.75a	0.77a	0.73a
pH (H_2O)	1.06a	1.06a	1.05a	1.07a	1.04a

CEC - cation exchange capacity

Table 4: Enrichment ratios (E) of surface soil layer (0-20 cm) and eroded sediment according to the treatments t. In the same line, means followed by the same letter are not significantly different (P > 0.05)

When compared with original soil, site preparation increased clay content and nutrients availability in soil (usually E > 1) with the exception of organic C and total N, as a result, site preparation can have several positive effects on soil quality in Mediterranean region. In eroded sediment E were higher than unity for almost all sediment elements, being particularly high for total N, available K and Mg^{2+}. Clay content showed E < 1, which was unexpected because majority of soils preferentially lose clay particles in eroded sediment. Gachene et al. (1997) obtained similar results. Soil pH E is slightly above unity, which highlights the enrichment of the sediment in exchange bases in relation to original soil. Soil organic C and total N losses associated with site preparation techniques and soil erosion could be the major environmental problem in this region and could have adverse effects on physical and chemical soil properties. However, the low sediment production, about 1 t ha^{-1} yr^{-1}, (Figueiredo et al., 2011) observed during this study showed that the damage to the system by erosion was hardly noticeable.

4. CONCLUSIONS

The site preparation techniques applied to install a mixed forest stand in a Mediterranean region, as reported in this paper, contributed to the increase of soil effective thickness and decreased of soil bulk density. In slight soil disturbance treatments (SMPC and RCAV), the tillage effects are not so evident, approaching the characteristics of these soils to the original situation (TSMO), resulting in high mortality of forest species planted, so they are not recommended for the region under study. In treatments with moderate disturbance (SRVC and RLVC), despite the significant reduction of the carbon and nitrogen, contributed to an overall improvement of soil quality indicators (clay, soil effective thickness, bulk density, available P and K, etc), as well as, to the highest rates of survival and a good height and diameter growth 42 months after stand installation. Treatments with the highest intensity of site preparation operations (RCVC, RCLC), associated with rising production costs and possible worsening of environmental impacts, did not compensate these drawbacks with improvements in soil properties and plantations' success. Sediment loss due to site preparation was low (about 1 t ha^{-1} yr^{-1}), consequently the soil fertility loss associated with soil erosion was of little relevance. Thus, in the region under study, the treatments SRVC and RLVC proved to be more sustainable for forest stands installation. Finally, it is important to note that the type of site preparation technique is extremely important in soils from areas where annual rainfall distribution shows a misfit with the growing season, as occurs in the Mediterranean region.

ACKNOWLEDGMENTS

The authors wish to thank the Program AGRO measure 8.1, project Agro-156, which financially supported this study, the Regional Forestry Services that took charge of site preparation mechanical operations and Mr. João Xavier, the owner of the experimental area, for his agreement on the establishment of the experiment.

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CASE STUDY 4: INTEGRATED FARM-SCALE APPROACH FOR CONTROLLING SOIL DEGRADATION AND COMBATING DESERTIFICATION IN ALENTEJO, SOUTH PORTUGAL - AN EXAMPLE OF GOOD FARMING PRACTICES TOWARDS A SUSTAINABLE LAND USE IN A HIGH DESERTIFICATION RISK TERRITORY.

Eugénio Menezes de Sequeira, Artur Lagartinho, Esmeralda Luís and Rita Alcazar

1. INTRODUCTION

In general, the Northern Mediterranean climate is characterized by:

- Arid, semiarid-arid and dry sub-humid conditions, with a high inter-annual variability and pronounced seasonal variations, with sudden and high intensive rainfall concentrated with a few months followed by seasonal droughts with high temperatures;
- Poor and highly erodible soils, prone to develop surface crusts;
- Inadequate agricultural practices used in the past, which led to the deterioration of soil and water resources;
- Unsustainable water resources exploitation (chemical pollution, salinization, sodization, exhaustion of aquifers);
- Concentration of economical activity in coastal areas as a result of urban growth, industrial concentration, tourism and unsustainable irrigated intensive agriculture with inland land abandonment;
- Inland depopulation, especially after 1960, with the lowest demographic density (17 inhabitants by square kilometre);
- High vulnerability to climate change.

Portugal presents a high prevalence of poor soils (more than 66% of the territory) and great risk of potencial erosion (68% of the territory), with only less than 8% of high quality soils (Program CORINE - Soil Erosion Risk and Important Land Resources).

Soil degradation in Portugal was enhanced by inadequate land use, mainly sealing high quality soils, farming overexplotation with improper occupation or practices, which lead to chemical degradation (salinization, nitrate and heavy metals contamination), through physical degradation caused by the destruction of organic matter and erosion (in particular erosion caused by water).

There were also other irreversible mistakes on land management, such as the “Wheat Campaign” established by the Portuguese Government in the 1930 decade, whose purpose was to transform Alentejo in “the barn of Portugal”, by implementing cereal single-crop cultivation to make Portugal self-sufficient. Areas that were uncultivated started to be sowed, despite slope, and the use of chemicals was promoted. Besides loss of soil and decreased fertility, soils also lost their vital function of regulating the hydrological cycle, because permeability and water retention was afected by changes in soil thickness, porosity, organic matter content and structure.

Degradation and soil loss are visible in schist areas of Portugal, and erosion risk is higher in areas where vegetation cover is reduced, soil stability is diminished, organic matter and permeability are less and are more prone to surface crusting (which prevents infiltration and increases runoff and erosion). Furthermore, degradation and erosion inhibits availability and quality of water in the soil, which causes vegetation cover reduction, decreases soil protection, reduces organic matter content and worsens soil structure, further accentuating erosion. Therefore, it is a vicious cicle.

In Castro Verde (Alentejo, South Portugal), the League for the Protection of Nature (LPN) recognized the impact of desertification and has implemented several projects to promote best farming practices, towards a sustainable land use in a high desertification risk territory with outsanding biodiversity values dependent upon extensive farming practices (Fig. 1).

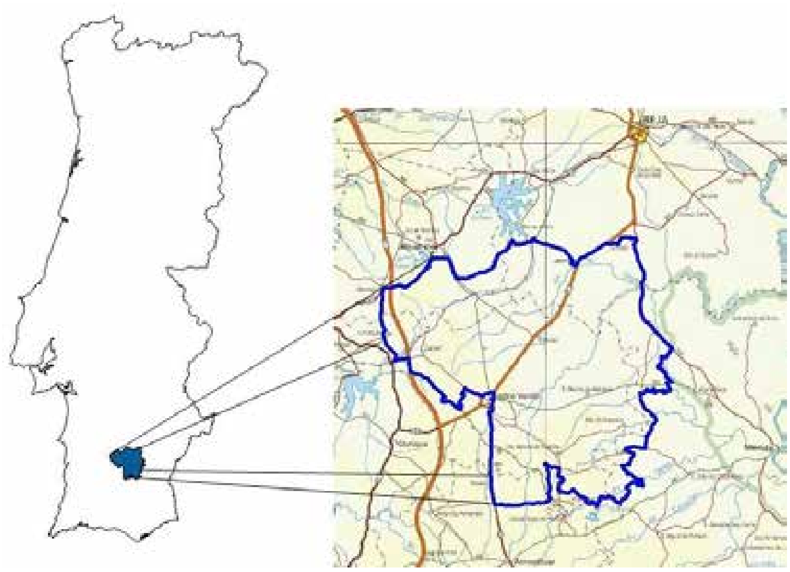

Fig. 1: Location of the Castro Verde Special Protection Area (SPA), classified under the European Nature Areas (Natura 2000 Network).

Work carried out by LPN in Castro Verde to combat desertification

The Castro Verde region has shallow and stony soils (Lithic Leptosoils in shales), 80% with less than 20 cm depth, with more than 40% stones and gravels and less than 40 mm maximum water holding capacity, with high risk of erosion (Fig. 2). Considered one of the high desertification areas of Portugal is also identified as very vulnerable to climate change.

Fig. 2 : Example of the Lithic Leptosoil in Castro Verde. Talvez o Eugénio tenha uma imagem mais representative?

However this area is considered as a high valuable natural area, recognized initially, in 1995, by Birdlife International as an Important Bird Area (IBA) and by the

European Commission of the European Union as a Special Protected Area (SPA) in the Natura 2000 Network, in 1999 and expanded in 2008.

The Castro Verde SPA covers around 85000 hectares and is the most important and representative area for the conservation of steppe birds in Portugal (Fig. 3), due to the diversity and abundance of species as the Great Bustard (Otis tarda), the Little Bustard (Tetrax tetrax), the Lesser Kestrel (Falco naumanni), the Black-bellied Sangrouse (Pterocles orientalis), the Montagu’s Harrier (Circus pygargus), the Calandra Lark (Melanocorypha calandra), amongst others.

Fig. 3: Some of the steppe birds of the Castro Verde SPA (Great bustard – on the left; Lesser kestrel – on the right).

In the early decade of 1990 land use was going to shift from extensive farming to intensive forestry with a non-native species (eucalyptus), which would jeopardize the pseudo-steppe ecosystem and probably lead to the local extinction of globally threatened species. Concerned with this threat LPN, in partnership with Castro Verde City Council and the local Farmers Association, engaged a process in 1992 to avoid and promote a solution for this threat, aiming also to contribute to the maintenance of environmental concerns as part of local heritage and the preservation of this Mediterranean agro-ecosystem, where humans are included and have a important role.

With the European Union support from the LIFE Nature Program, LPN was able to acquire 1800ha of land, divided by 6 farms (considered as Biodiversity Reserves), identified as the most important areas in the region for steppe birds. These farms are managed accordingly to research studies on steppe bird ecology that identified habitat management practices needed. Among the management measures established in LPN farms are: annual percentages of land farmed for cereals (less than 40%), grazing pressure allowed (0,2 Livestock Units of sheep or cattle), sowing specific crops for bird feeding (approximately 2 % of leguminous for grain), establishment of farming activities calendar (without farming machinery activity during birds breeding season, specially for the protection of ground nesting species), creation of nesting sites for hole-nesting species, plowing according to contour level and slope, protection of riverbeds, encourage direct seeding or no tillage system to minimize soil erosion.

The existence of the pseudo-steppe area depends on the maintenance of traditional farming practices, based on extensive dry farming of cereals (wheat, oats or barley) rotating with fallow lands, in which land "rests" in order to recover its fertility and is simultaneously used as pasture (Figs. 4, 5 and 6).

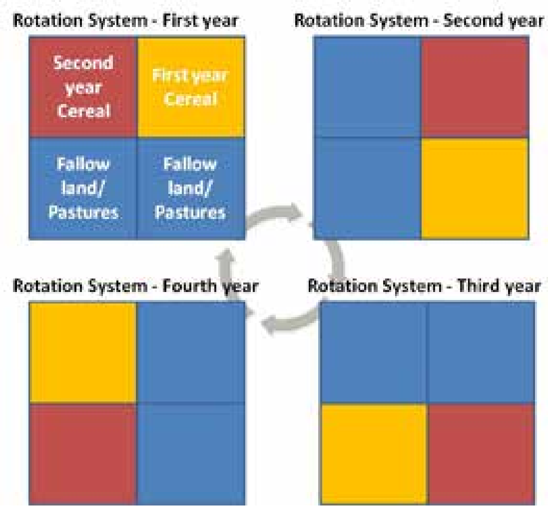

Fig. 4: Rotation system applied in the traditional extensive farming scheme, which rotates parcels of dry cereals with grazing, in periods of 2 to 5 years (orange square examples the first year of sowing with a dry cereal, such as wheat, red square is the second year of cereal, such as oat, and blue squares represent fallow land used for grazing).

Fig. 5: A fallow land field in the Spring and traditional livestock (sheep).

Fig. 6: A cereal field in the Spring and the stubbles in the Summer.

To avoid the increase of afforestation in agricultural land of this region, in 1995 started an Agri-Environmental Scheme, formulated within the Common Agricultural Policy (named Castro Verde Zone Plan)[1], to preserve this ecosystem and to protect the exceptionally birdlife associated. This agri-environmental measure aims to compensate farmers for the loss of income caused by the application of measures to protect bird's habitat. Farmers may voluntarily apply for a minimum period of 5 years and commitments are very similar to those applied by LPN farming management. This measure has been strongly supported by local farmers and has been the principal factor responsible for the preservation of traditional extensive agriculture, birds and landscape. In 2004 a new measure, complementary to this, has included, supporting farmers to apply minimum tillage techniques.

In this sense, LPN has tried to develop from 1992 an approach of coherent long term projects, within a broad plan named "Castro Verde Sustainable Program", aiming the promotion of local sustainable development, including the protection of endangered bird species, combat to the desertification process, improving soil process formation, soil carbon sink, promotion of ecotourism, environmental education, amongst others. The work of LPN has been recognized by several national and international prizes, which awarded the Program and its Projects.

LPN envisages the promotion of a sustainable agriculture, which protects natural resources and biodiversity and simultaneously assures farmers income that includes ecosystem services provided. Therefore, new farming technologies must be developed that avoid soil depletion through erosion and increment soil formation processes. The main projects carried out by LPN in Castro Verde within the scope of the

[1] Renamed to Integrated Territorial Intervention (ITI) of Castro Verde in 2007.

desertification combat and sustainable farming practices are listed above. In terms of desertification research, a long term approach and funding would be fundamental, which it is not always possible.

Projects of the Castro Verde Sustainable Program focusing in sustainable farming practices and the desertification combat

2. PILOT PROJECT TO COMBAT DESERTIFICATION (1999-2004)

Agronomic technology to promote a sustainable agriculture in Mediterranean dry lands must improve soil water holding capacity, infiltration rate, increase soil depth, soil organic matter content and decrease runoff and erosion rate. The Pilot Project to Combat Desertification, developed between 1999 and 2004, within the framework of the National Program of Desertification Combat, was implemented by LPN in partnership with the Alentejo Environmental Administration.

Project objective was to demonstrate the negative impacts of some traditional farming practices and test, demonstrate and reproduce techniques to combat desertification, namely direct seeding and the appliance in the subsoil of sewage sludge from wastewater treatment plants (WWTP). These two techniques were studied also in other LPN subsequent projects and their appliance have been perfected until the most recent project (Rural Value), which will be presented in full detail further ahead.

In this project four plots with 0.5 hectares each, were installed in one of LPN farms in Castro Verde, reproducing two traditional farming practices (TF) of the region and two innovative techniques (IT) to combat desertification:

- Plot 1 – TF - cereal sowing with conventional plowing of the top soil (20 cm);
- Plot 2 –TF - fallow land with natural vegetation;
- Plot 3 – IT cereal sowing with direct seeding;
- Plot 4 – IT - sewage sludge subsoil appliance followed by direct seeding cereal sowing.

A collection system for water runoff and sediment yield was also installed on each plot, which drains towards a pond where monitoring of flows and sediments takes place (Fig. 7).

Fig. 7: Construction of the system to collect water runoff and sediments of a plot in the Pilot Project to Combat Desertification

Results revealed an erosion rate diminishing with the increase of soil structural stability (Table 1). That is, in the plot were direct seeding was applied and the plot with both sewage sludge subsoil amendment and direct seeding, sediments were 10 times less, comparing with erosion in the plot with plowing. Runoff erosion was also high in fallow land due to soil crust effect (Fig. 8) which inhibits water infiltration and increases water velocity and drags out soil fragments of bigger dimension.

Fig. 8: Example of the soil crust due to compactation.

With this project it was possible to demonstrate to farmers the amount of soil loss from erosion if different farming techniques were applied.

Table 1 : Sediments collected on Plots of the Pilot Project to Combat Desertification.

Sediments collected **(Kg ha year)**	**Plot 1** cereal sowing with plowing and harrowing by the higher slope[1]	**Plot 2** fallow land with natural vegetation – with no treatment or crop in the soil[2]	**Plot 3** direct seeding with no tillage	**Plot 4** sewage sludge subsoil appliance followed by direct seeding
2002/2003	132 (±10)	80 (±10)	12 (±10)	20 (±10)
2003/2004	148 (±10)	160 (±10)	74 (±10)	73[3] (±10)

[1] Significant tillage, though the year did not have torrential rains

[2] Fragility of the made to drain the watercourse

[3] Considerable error due to horses invasion of the plot

3. PROJECT AGRO 140 – SUSTAINABILITY EVALUATION OF SOME CULTURAL SYSTEMS IN BAIXO ALENTEJO

The AGRO 140 project which took place between 2001 and 2005 was implemented in cooperation between LPN and National Agronomic Station, to pursue the result obtained in the Pilot Project and was funded by Agro funds of the Common Agricultural Policy.

A field trial was established in one of LPN farms, in an area of 26ha divided in 13 plots (Fig. 9), of 2ha each, submitted to different forms of soil use: coated fallow, wheat, barley and fallow. In each plot three treatment were applied (1/3 of the 2 hectares): rippling, rippling with waste water sewage sludge appliance and a control treatment (without intervention). On each treatment of each plot an erosion box (1m x 2m) was placed, connected with a collection deposit, for water and sediments and the water volume and amounts of sediments were quantified annually (Fig. 10).

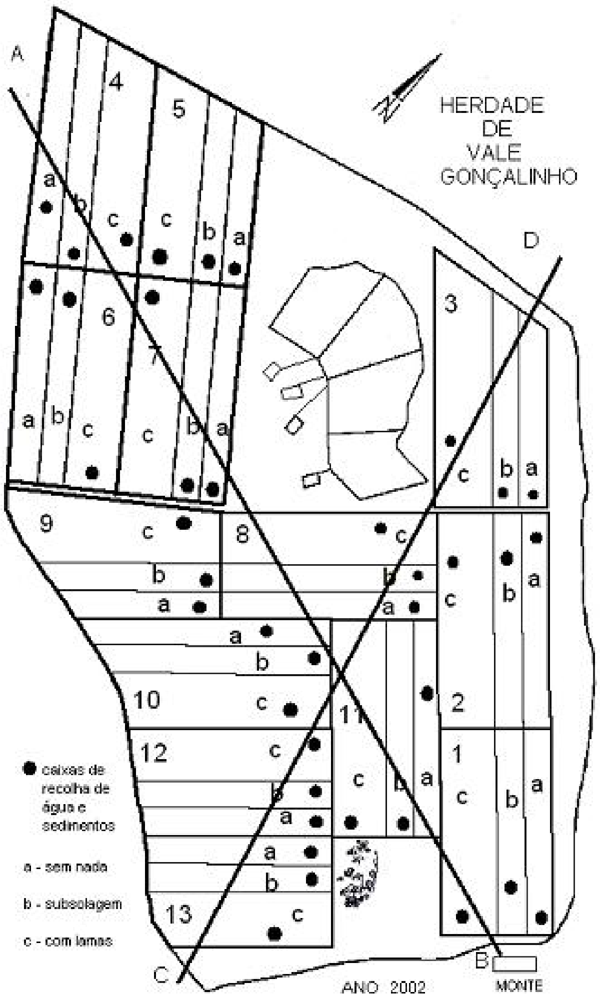

Fig. 9: Plan of the 13 plots of 2 hectares, each one with three different treatments (a. control, b. rippling, c. rippling with waste water sludge).

Fig.10: Collection of soil particles and water runoff from the plots.

In the mean of the years and the three treatments, there was only significant difference between the volume of water collected from the fallow (41,8L) and from the coated fallow (19,5L). As for the treatments, and concerning the volume of water collected, the control presented values significantly greater (51,5L) than the rippling (25,9L) or the rippling along with waste water sewage sludge appliance (33,6L). Concerning the years, the mean values of runoff water in the third year were significantly the lowest (17,9L), in relation to the previous years (Castelo Branco, 2006).

As for the amount of sediments, the greater value was observed in the plots cultivated with wheat (293,9kg/ha), and the lowest in those submitted to fallow (84,3kg/ha). Concerning the treatments, the amount of sediments in the control (260kg/ha) was significantly greater than the recorded for the other treatments (56kg/ha). Concerning the mean values for the different types of soil use and the treatments, the amount of sediments did not varied along the years (123,7kg/ha).

The results obtained are explained by the fact that the deep and vertical mobilization that the soil was submitted to, with the rippling and the rippling with waste water sewage sludge appliance, allowed greater infiltration of water and, consequently, the runoff of lower amounts of water and sediments.

Results obtained in cereal production (amount of grain produced) indicated that higher productions were achieved with the appliance of waste water sludge injection in the soil.

4. PROJECT RURAL VALUE – SUSTAINABLE DEVELOPMENT OF THREATENED EXTENSIVE FARMING SYSTEMS

Project Rural Value was implemented between 2008 and 2011 and was funded by EEA Grants. LPN was the coordinating beneficiary in partnership with promoter and the other partners of the project were Instituto Superior Técnico, Instituto Nacional de Recursos Biológicos and the Municipality of Castro Verde.

Rural Value aimed to promote the sustainable development of threatened extensive farming systems, namely in the Castro Verde cereal plains, by promoting innovative techniques and exploring several themes associated with sustainable farming, biodiversity and the society.

One of the Activities of RuralValue project was devoted to the Mitigation of Drought and Desertification. This task aimed to test, at the farm level scale, the effect of rippling with waste water sludge injection to increase soil formation process, reverse soil erosion, increase water retention and organic matter content in soil. The expected result of these interventions is the increase of production along with greater local agriculture yields, contributing therefore to combat desertification and making farming activity more sustainable.

Sewage sludge appliance was tested as a technique to increase the rate of soil formation (increase weathering by rock surface exposed to hydrolysis and organic matter), improve fertility and increase water retention, thus, a way to combat desertification. The main difference from previous LPN projects (Pilot project and Agro 140), in which this technique was tested, was the scale of the field trial, now with a dimension closer to the farm management reality. The logistics of this appliance and the monitoring process were also more elaborated. Comparisons between common tillage practice and the direct seeding technique were also made, in areas with and without the previous appliance of sewage sludge in the subsoil.ß

Sewage sludge was applied in dry-farming cereals areas of the rotation system, but because it is not allowed to perform any tillage during Spring due to the bird's breeding season, the only months when the appliance is possible is during the Summer (after mid July) until Autumn (when sowing is carried out). Sewage sludge injection on the soil was applied just before first year cereal sowing, which starts the rotation period and the process can be repeated on the following rotation period (Fig. 11).

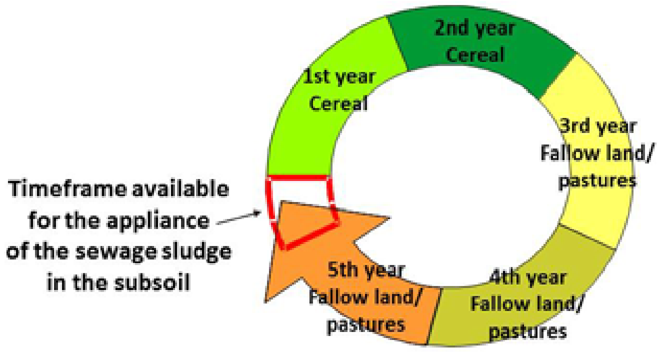

Fig. 11: In soils of the agricultural rotation system the period in which sewage sludge has applied was before sowing the first year cereal after land stayed has fallow for at least two years.

Sewage sludge was collected from the local urban Waste Water Treatment Plant (WWTP), transferred to an intermediate tank and then transferred to a specific machine (named injector), developed by LPN, which has a storage bin and two hooks that allow the injection in depth in the soil (Fig. 12 and 13). The injector is pulled by a tractor.

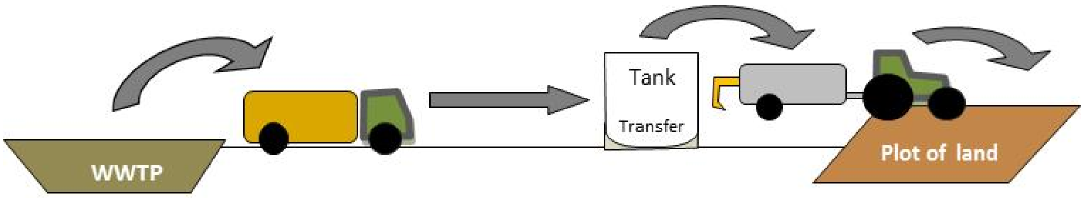

Fig.12: Scheme exemplifying the process of collection of sewage sludge from the Waste Water Treatment Plant until it is applied in the soil.

Fig.13: Tractor with the injection storage bin operating

The injector applies the sewage sludge in two furrows at each time through the two hooks in the rear, with one meter distance in between, reaching a depth from 30 to 60 cm depth. Like this it is possible to reach R horizon (C horizon is quite rare in the large majority of the soils in this region), to facilitate the processes of weathering and soil formation. Between each passage the space is of about 2 meters (Fig. 14).

In this project the demonstration area involved four farms with over 100 hectares sowed. Before sewage sludge appliance tests were made to control their quality and avoid soil heavy metal and microbiological pathogenic contamination.

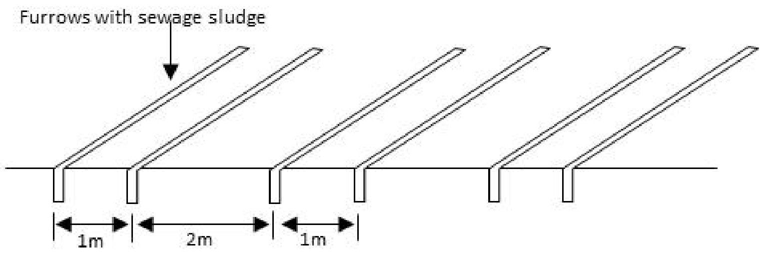

Fig. 14: Method of applying sewage sludge in the soil.

Sewage sludge and sub-soiling increases rock exposed to weathering, increases soil water aggressivity therefore increasing hydrolysis of rock minerals, and soil forming process, clay formation. Hence, the recovery of degraded soil, to increase soil depth from 15/20 cm till 50/70 cm could be reduced from more than 5.000 years to less than 100. Meanwhile the production, soil fertility, soil water retention and aquifer recharges is increased. At the same time, the injection in R horizon of organic matter and mineral elements (such as Phosphorus, Nitrogenous and Potassium) from sewage sludge, increases root growth.

The assessment of this technique was made in areas were sowing of cereals was made by direct seeding or conventional tillage practice was used. The most common sowing practice in the region involves previous soil tillage, in which soil is wrought and revolved two times in the top soil layer (20cm). Direct seeding consists in using a seeding machine that introduces seeds and fertilizer into thin strips in the soil without the need to revolve it (Fig. 15). With this technique soil loss is much lower and, by keeping soil structure intact, this process increases water infiltration rate, decrease runoff, and also avoids carbon loss by mineralization of organic matter and promotes soil carbon sequestration by means of the roots and contributes to increase water retention.

Fig. 15: Direct seeding machine working in cereal sowing

Despite the need to continue monitoring the effects on the medium-long term, due to the long time soil formation process, Rural Value first results of assessed impacts in the soil, water, cereal production and biodiversity, gave positive indications of the success of the sewage sludge subsoil appliance technique. Briefly, the main results were:

- No contamination of soil, water or plants with heavy metals and pathogens;
- Increase of the microbiologic activity in the soil, leading to a better balance in carbon and nitrogen present in the microbial biomass, conditions usually linked to a rise in fertility;
- For the cereal production, no conclusive results could be extracted, because data represent a single agricultural year and the particularly adverse weather skewed data (waterlogging of fields hindered crops development and productions in the region were reduced);
- Trend of increased plants biodiversity after sewage sludge;
- No conclusive results for the fauna bio-indicator analyzed (spiders), despite a greater abundance in areas where sewage sludge was applied.
- The life cycle analysis identified a greater environmental performance of applying the technique, due to the avoided costs of placing this waste in landfill.

5. *PROJECT PRACTICE – PREVENTION AND RESTORATION ACTIONS TO COMBAT DESERTIFICATION. AN INTEGRATED ASSESSMENT*

Practice is an ongoing project (2009-2012), supported by European Union Research Program (FP7) and coordinated by CEAM Foundation, having partners from 16 institutions of 12 countries, among which LPN, which participates with the Castro Verde site.

While the drivers and processes of desertification are given attention from science and a lot of research is being conducted, two major gaps stand out: systematic evaluation of practices to combat desertification is still lacking, and the valuable local knowledge of those influenced by land degradation is too rarely fully considered. PRACTICE addresses this challenge integrating evaluation of prevention and restoration practices and knowledge exchange in a process defined by both scientists and stakeholders in an interactive manner (Vallejo, 2010).

The project is a global initiative that gathers scientists and stakeholders from some of the most affected regions of the world, to combine local and scientific knowledge to help address the desertification challenge. The central goal of PRACTICE is to link science to society in order to share and transfer evaluation methods and practices to combat desertification. To pursue this goal, PRACTICE aims (Idem):

- to develop and implement an integrated evaluation tool to assess the effectiveness of prevention and restoration practices.
- to develop translational science strategies and education material addressing and involving stakeholders at all levels, from farmers and local organizations to national and international bodies.
- to create an international platform of long-term monitoring sites aimed at supporting future synthetic analysis, improving the accessibility and use of long-term data, and disseminating knowledge on best practices worldwide.

The methodology of the project comprises two key activities (Idem):

- the establishment of a coordination platform, linking a variety of field sites for the long-term assessment of sustainable management and actions to combat desertification;
- the definition of an integrated protocol for the assessment of these actions.

The assessment protocol will integrate ground-based and remote-sensing approaches, biophysical and socio-economic evaluation, and expert and local knowledge. It will be implemented in the field to consistently document and evaluate the effectiveness of prevention and restoration techniques and strategies (Vallejo, 2011).

The project will also link assessment and evaluation with training and education, adaptive management, and knowledge sharing and dissemination, by implementing innovative participatory approaches – involving all relevant stakeholders (Vallejo, 2010).

The project intends to collect and disseminate long-term experiences and datasets on land restoration and sustainable management in various involved countries all over the world. An integrated method for the assessment of prevention and restoration practices is developed to provide the basis for evaluating a wide diversity of actions to combat desertification, in a wide range of socio-economic, biophysical, and technical conditions. It also facilitates direct links between stakeholders through their involvement in the evaluation process, bridging the gap between research and practice. Integrated assessment of the efficiency of mitigation and adaptation techniques in the network of sites will foster feedbacks for proposing new actions to optimize prevention and restoration results, in the framework of adaptive management principles (Idem).

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CASE STUDY 5: THE ROLE OF NO-TILL AND CROP RESIDUES ON SUSTAINABLE ARABLE CROPS PRODUCTION IN SOUTHERN PORTUGAL

Mário Carvalho

1. INTRODUCTION

Under Mediterranean conditions the concentration of rainfall that prevails over winter results in waterlogging, erosion and the impairment of timeliness of field operations, while the scarcity of precipitation during the spring leads to water stress in crops. The general characteristics of Portuguese soils serve to aggravate the problems for crop production. Soil fertility is inherently poor (about 70% of the soils have an organic matter content that is less than 1% and only 4% have a cation exchange capacity that exceeds 20 meq/100 g of soil) and water infiltration and internal drainage are negatively affected by the instability of soil structure and the marked changes in clay content that occurs between soil horizons. Both climatic and soil constraints limit yield potential and the efficient use of the resources, such as fertilizer particularly nitrogen, whilst imposing agronomic limitations by preventing the correct timing of operations, which cannot be overcome by increasing labour input because of the need of farms to stay economically competitive. Any meaningful amelioration of the situation can only be achieved by a significant improvement in soil fertility and in soil-water relationships, which can only be acquired through increases in soil organic matter (Carvalho, 2006, Douglas et al., 1986).

The effect of no-till (NT) on soil organic carbon (SOC) seems to depend on the prevailing conditions of climate, soil and crop, with results in the literature varying from the absence of effect when the whole soil profile is considered (Dolan et al., 2006) to an increase over the depth of tillage (Martin-Rueda et al., 2007), and even to enhanced levels below the depth of tillage (Ordõnez – Fernandez et al., 2007). The positive impacts of NT on SOC have often been attributed to a reduction in the rate of organic matter mineralization in the absence of soil disturbance (Recolsky, 1997). There are also authors who state that the beneficial effects of no-till depend on the amount of the crop residues produced over the course of the crop rotation (Salinas-Garcia et al 2001; Halvorson et al 2002; Lopez-Bellido et al., 2010). However, it is generally recognized that beneficial effects of NT are derived from maintaining crop residues on the soil surface and the associated control of soil erosion (Towery, 1998). The relative importance of this aspect depends on the soil and on climatic conditions, but conventional tillage can result in soil loss through erosion that is more than 75 times greater than that from no till systems (Engel et al., 2009). Under such circumstances and over the long term, nutrient losses from the soil can be very large, being aggravated by the enrichment of organic matter, phosphorus and potassium on constituents of the soil sediments such as clay, (Sharpley 198,5). Consequently, whenever prevention of soil erosion is an important benefit derived from the adoption of no-till a significant increase in SOC would be expected.

No-till can also affect soil water relationships. Under no-till, especially when an adequate amount of residues is left in the soil surface, there can be a reduction in water lost by runoff (Lal and Van Doren Jr., 1990) and a concomitant increase in infiltration. The residues on the soil surface will also reduce evaporation of water from the soil surface, and both increased infiltration and greater conservation will tend to increase soil water content, especially under Mediterranean conditions (Morell et al., 2011). Therefore, waterlogging can be accentuated during the initial year of no-till, under soils with a small saturated hydraulic conductivity or a perched water table. However, structural stability and the number of vertical continuous biopores also increase under no-till, which contribute to an increase in the saturated hydraulic conductivity over time (Ehlers and Claupein, 1994). Under these circumstances trafficability would be expected to improve (Gruber and Tebrugge, 1990) and allow more timely field operations, a very important agronomic benefit under Mediterranean conditions.

The aim of this paper is to discuss the role of no-till and crop residues as means of overcoming some of the main constrains to arable crop production in Portugal.

2. MATERIAL AND METHODS

Runoff and erosion studies (Fig. 1) were evaluated over two seasons, using runoff frames. The conventional tillage system (CT) consisted of a pass with a plough in the summer and then disk harrowing before seeding the wheat crop. No till (NT) was performed with a triple disc no till seeder, with weed control being achieved with a pre-seeding application of Paraquat. The slope of the land was uniform within each replicate of the treatments and varied between 6 to 8% between blocks. A detailed description of the experiment can be found in Basch et al. (1990).

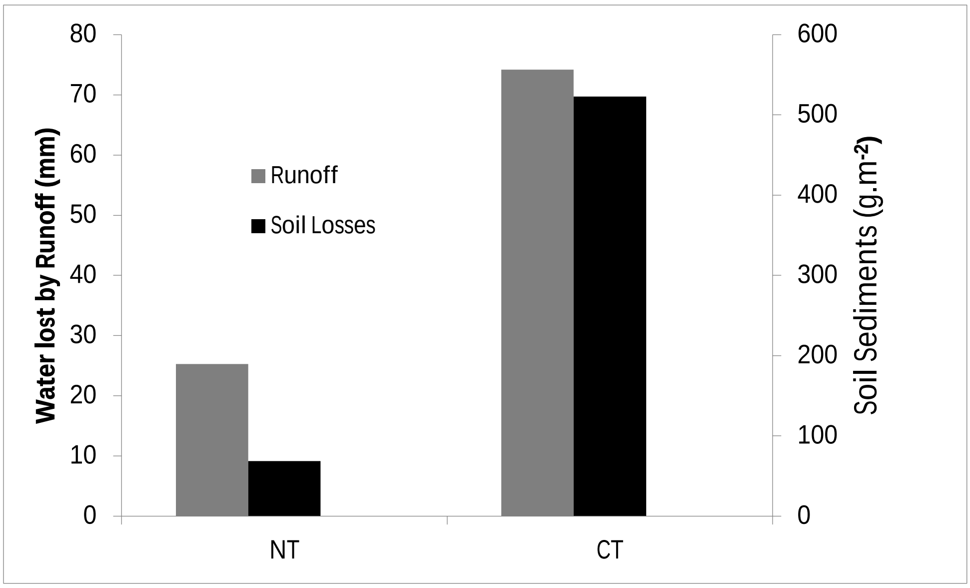

Fig. 1: Effect of the tillage system on runoff and soil losses by erosion during a wheat crop in the south of Portugal. Values are verage of two years. NT – No Till; CT – Conventional Tillage (Based on Basch et al., 1990).

Data collection on the Vertic Cambic soil (50% clay) took place 6 years after the tillage systems were put in place (1984/85 – 1989/90). The crop rotation was sunflower – wheat – barley. The tillage systems studied were no till (NT) for all crops of the rotation, and the conventional tillage system of the region, which is: summer plough (30 cm) + disk harrow (at least 2 passes) for the sunflower; tine sacrifier + disk harrow for wheat and barley. The experiment is described in Carvalho and Basch (1995).

Measurements on the Luvisol (31.1% and 46.8% clay in A and B horizons) were taken as part of a long term experiment comparing tillage system (1995/96 to 2007/08). The crop rotation was lupines – wheat – oat for forage – barley. The conventional tillage system consisted in one plough (25 cm) and disk harrows before seeding, and the straw of cereals was bailed. For the NT treatments weeds were controlled before seeding with glyphosate and crops were direct drilled. In one treatment the straw of cereals was kept on the soil surface (NTS), while for another treatment the straw of the cereals was bailed (NT).

3. RESULTS AND DISCUSSION

Under Mediterranean conditions the concentration of rainfall during late fall and winter, when soil cover by the crop is minimal, creates the opportunity for soil erosion under conventional tillage systems but no till can be very effective in reducing runoff and the consequent soil loss by erosion (Fig. 1). A reduction in erosion under no till was due to both a reduction in runoff and in the amount of soil sediment transported per unit of water volume (2.7 and 7.0 g of soil per litre of runoff water in NT and CT respectively), although the data was collected in the first year of imposing the treatments.

The results available in the south of Portugal indicate an increase of soil organic matter (SOM) under NT (Figs. 2 and 3), but the effect seems to be dependent on the soil and the amount of crop residues left on the soil surface. On the Vertic clay soil (Fig. 2), NT increased SOM over the depth of tillage, after 6 years under the same crop residue management. However, on the Luvisol, the effect of NT under the same residue management programme was much smaller and took longer in comparison to CT (Fig. 3). On this soil, NT could only improve SOM significantly when the straw of the grain crops was left on the field. The difference between the two soils could be explained by a greater effect of CT on the mineralization rate and the larger soil loss by erosion on the Vertic clay soil compared to effects on those values in the Luvisol.

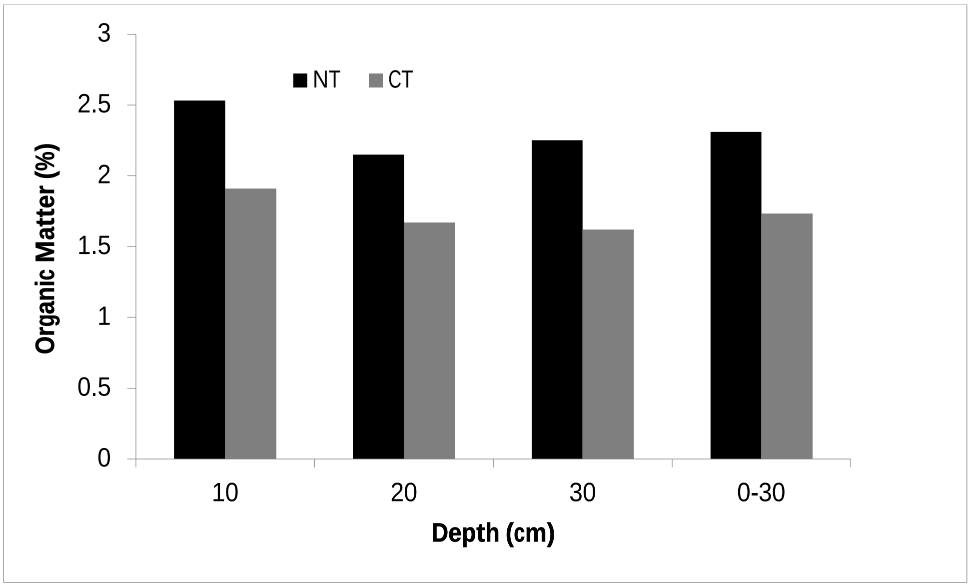

Fig. 2: Effect after six years of different tillage system (NT-No till; CT-Conventional Tillage) on soil organic matter over the depth of tillage, on a Vertic Cambic Soil in the south of Portugal. (Based on Carvalho and Basch, 1995).

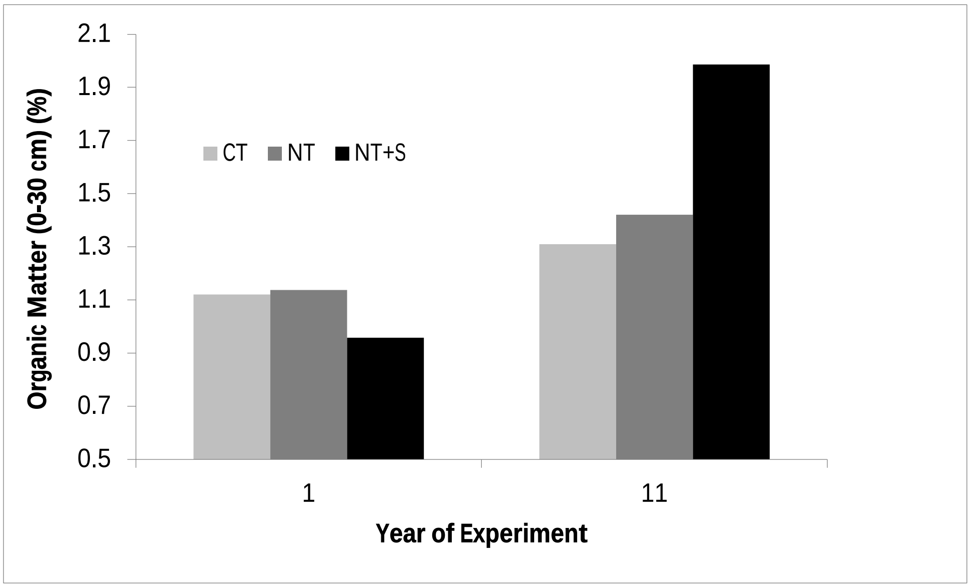

Fig. 3: Effect of tillage system and crop residues management on the soil organic matter content (0-30 cm) on a Luvisol in the south of Portugal. (CT-Conventional Tillage and straw bailed; NT-No Till and straw bailed; NT+S- No Till and straw kept on the field) (Unpublished data).

With time NT improved structure stability (Figs. 4 and 5) and further improvements in water infiltration (Lal and Van Doren Jr., 1990) and soil conservation would be expected. The improvement of structural stability under NT is more evident on the aggregates bigger than 0.5 mm. The effect of NT on improving structural stability appears to be more rapid than the effect on SOM (Figs. 3 and 5), suggesting that it is probably due to the enmeshment of soil aggregates by the fine roots of plants and the mycelium of associated fungi.

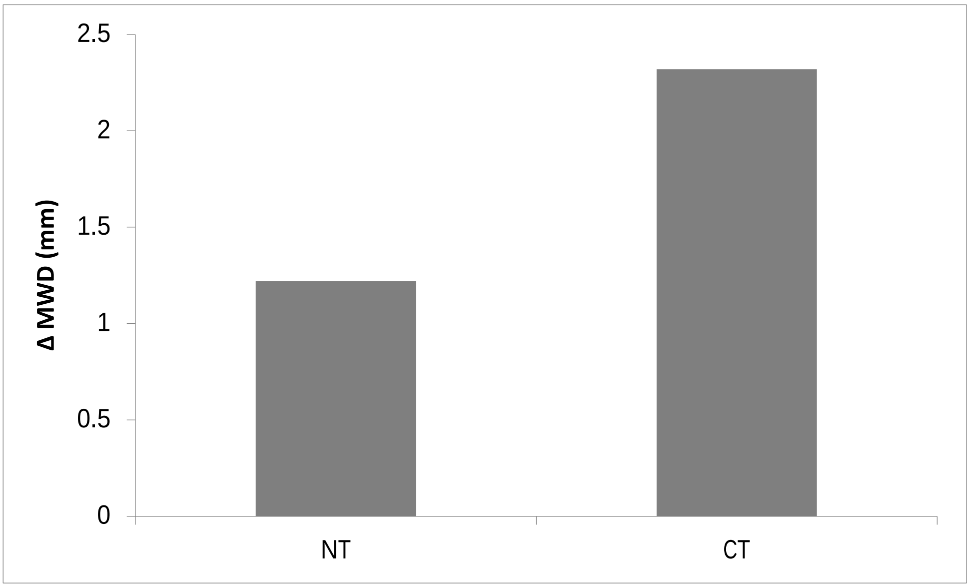

Fig. 4: Effect after six years of tillage system (NT-No till; CT-Conventional tillage) on aggregates stability (0-10 cm) on a Vertic Cambic Soil in the south of Portugal. The term Δ MWD signifies the change in mean weight diameter of aggregates after wet sieving compared to dry sieving, and therefore higher values

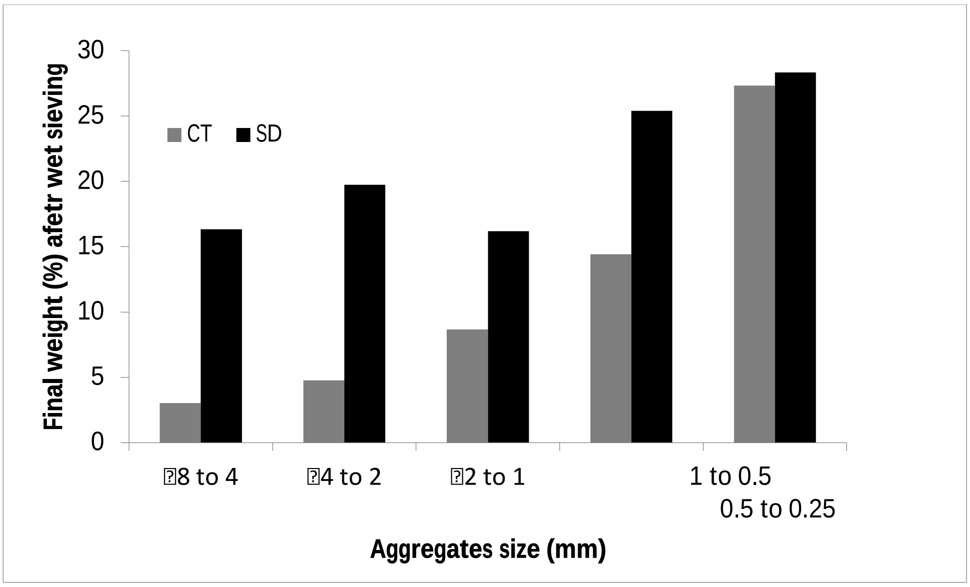

Fig. 5: Effect after three years of tillage systems (CT-Conventional tillage; NT-No Till) on aggregate stability (0-10 cm) on a Luvisol in the south of Portugal. The aggregate stability was evaluated by the final weight (as a percentage of initial weight) of the different classes of aggregates, after wet sieving, and therefore higher values correspond to a higher wet aggregate stability (unpublished data).

The development of a continuous network of biological porosity by NT due to the growth of roots and the burrowing of soil meso and macro fauna, such as earthworms, is well known (Goss et al., 1984), but under Mediterranean conditions the process of structure development can be quite rapid because of the rapid drying of the soil during the spring and summer. This drying can help to create vertical cracks that can be used by plant roots (weeds and crops) at the beginning of the next rain season (Fig. 6). This type of porosity together with enhanced aggregate stability are very effective in improving hydraulic conductivity, which is very important under the wet winter of Mediterranean climate (Fig. 7).

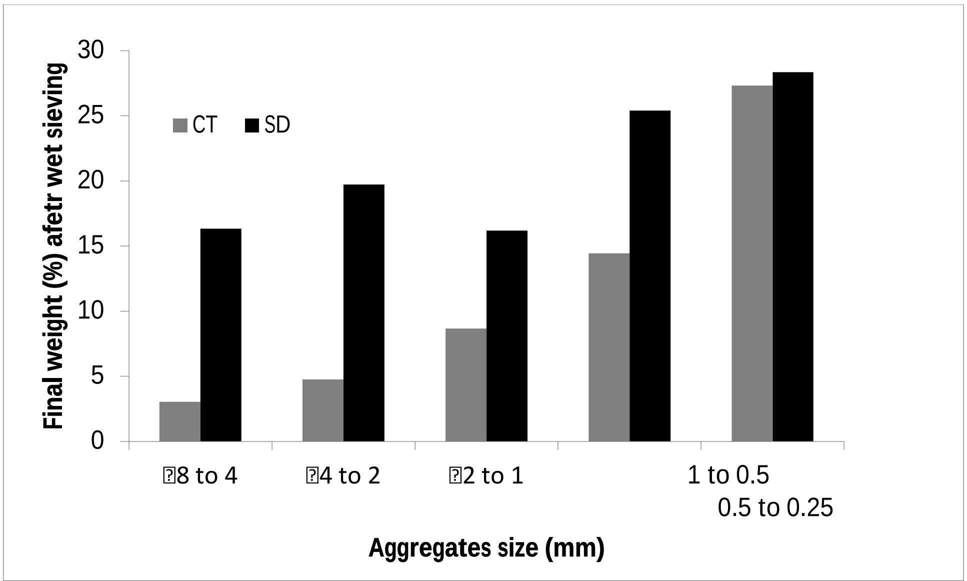

Fig. 6: Effect after six years of tillage system (NT-No till; CT-Conventional tillage) on biological porosity on a Vertic Cambic Soil in the south of Portugal (Based on Carvalho and Basch, 1995).

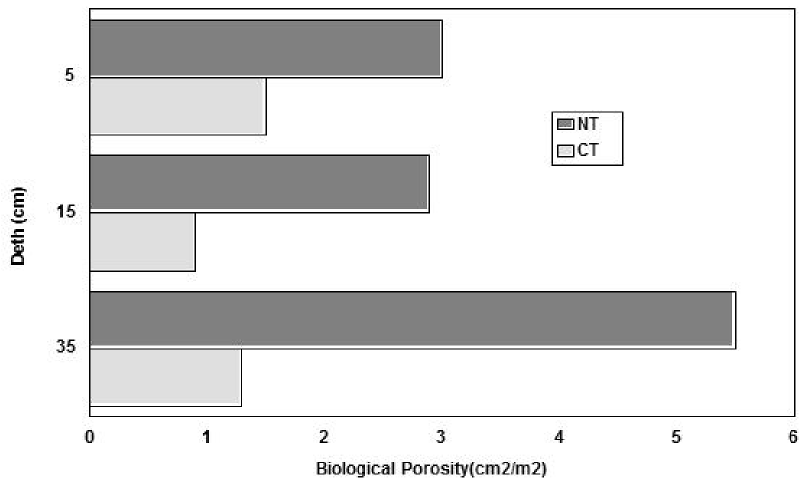

Fig. 7: Effect after six years of tillage system (NT-No till; CT-Conventional tillage) on saturated hydraulic conductivity on a Vertic Cambic Soil in the south of Portugal (Based on Carvalho and Basch, 1995).

The development of soil properties under NT as described above has important implications for arable crop production under Mediterranean conditions. The improved infiltration of water reduces the loss from runoff during the winter, which is particularly important during dry years, while the enhanced saturated hydraulic conductivity helps to alleviate waterlogging problems during wet winters. The better drainage associated with a higher soil cohesion under NT improves the trafficability of the soil, allowing a correct time for field operations, critical in the face of the variability of Mediterranean climate. The increase of SOM helps improve soil fertility. Consequently, an improvement in crops productivity should be expected together with an increase in the efficient use of soil resource, such as nitrogen.

Grain yield of wheat under NT, relative to CT, with and without the bailing of the straw, increased over time, and the average yield for the last four years of the experiment was 200 and 750 kg/ha greater under the two NT treatments (Fig. 8). These differences were consistent with the increments of SOM in soil under the two NT treatments (Fig. 3). The improvement of SOM was also related with an increase of the applied nitrogen use efficiency (NUE) (Fig. 9). According to the equation presented in Fig. 9, for 1% of SOM the most economical N fertilization (according current prices 4 kg of wheat per one kg of applied N) will be 160 kg N/ha and the yield obtained 3063 kg/ha (19.1 kg of grain per kg of applied N), which is a typical value for the region. However, for 2% of SOM the same variables will be 98 kg N/ha and 3587 kg/ha (36.6 kg of grain per kg of N). The explanation for this sharp effect of SOM on NUE can be the leaching losses of nitrogen during the winter. Under Mediterranean conditions critical crop development stages , such as tillering and spikelets differentiation take place during the winter, and any nitrogen deficiency will affect crop performance. Therefore nitrogen has to be available during the winter, and if the soil is poor in organic nitrogen, more mineral nitrogen must be applied as fertiliser.

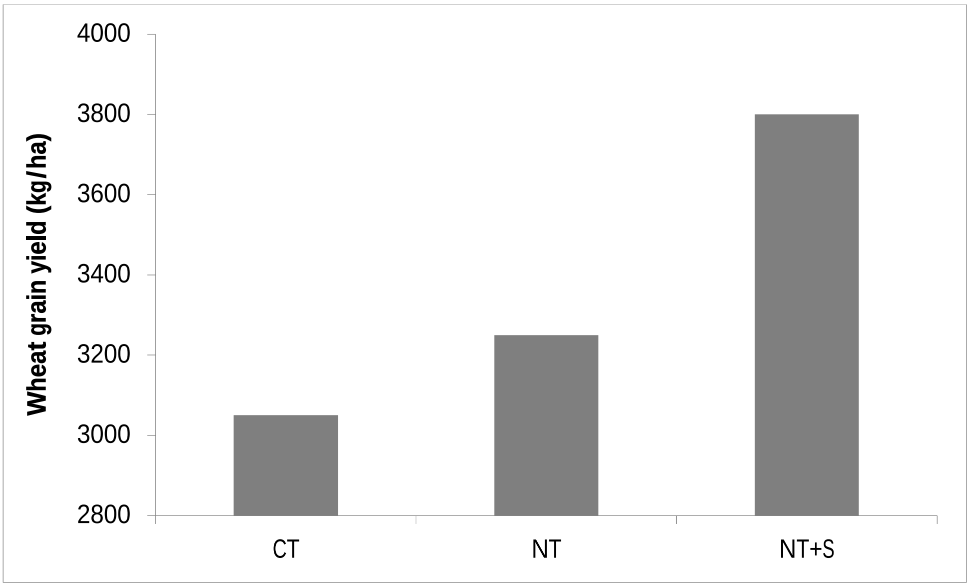

Fig. 8: Effect of tillage system and crop residues management (CT-Conventional Tillage and straw bailed; NT-No Till and straw bailed; NT+S- No Till and straw kept on the field) on the wheat grain yield (average of four years from 2005/06 to 2008/09) when the treatments were already in place from 1995/96, on a Luvisol in the south of Portugal (unpublished data).

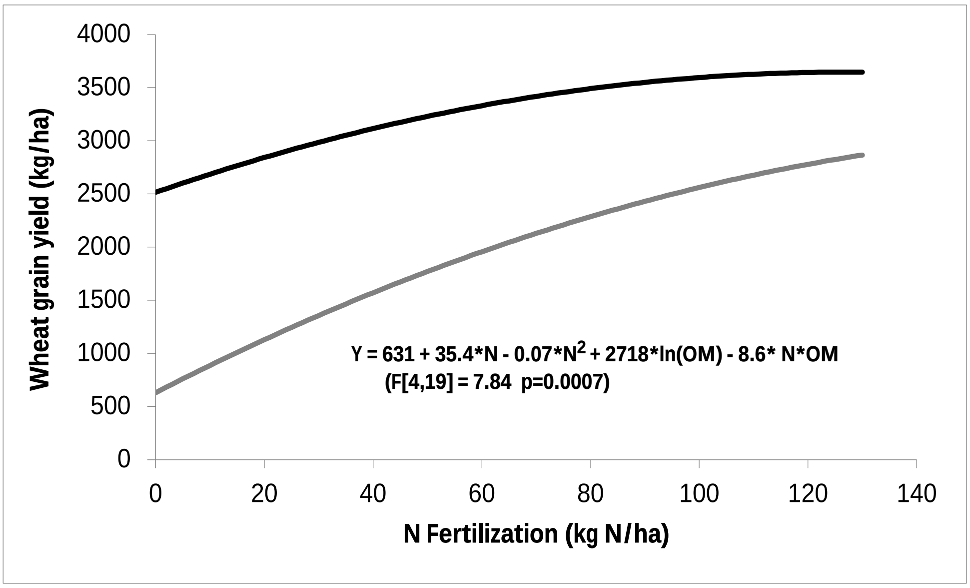

Fig. 9 : Effect of soil organic matter content (0-30) on the response of wheat to nitrogen on a Luvisol in the south of Portugal. In the presented equation N is the applied nitrogen (kg N/ha) and OM is soil organic matter. The grey line is relative to the wheat response to N when OM is 1% and the black line when OM is 2%. The different contents of the soil in organic matter were developed under two different treatments: Conventional Tillage and straw bailed; No Till and straw kept on the field) (Based on Carvalho, 2006).

The decrease of wheat yields during wet winters, which commonly occurs under conventional tillage systems, is due not only to waterlogging but also to the enforced delay in applying nitrogen top dressing and post-emergence herbicide. The pressure from weeds and the need for nitrogen increase with the amount of winter rainfall but often application has to be delayed until the beginning of March when the winter is wet, which is too late to benefit the crop. For nitrogen, the importance of an application in January (first top dressing at tillering for wheat) depends on the amount of rainfall from seeding to full tillering of the crop. If this period is dry, a later application of nitrogen (beginning of stem elongation – end of February/early March) is enough to achieved maximum yields. However, if the period is wet, an application of nitrogen in January is indispensable and cannot be fully offset by a later fertiliser application, even of 120 kg N/ha (Fig. 10). The negative consequences of a delay of post-emergence herbicide, either in terms of the dose of herbicide eventually required and the yield benefit to the crop, are also clear under Mediterranean conditions (Fig. 11). The better trafficability of the soil under NT is therefore key to maintaining cereal yields in wet years, by allowing applications of nitrogen and herbicides at the correct time. Experience in southern Portugal shows that, by adopting NT and using the correct equipment (light tractors and low pressure tyres) it is possible to apply fertilizer or herbicides without greatly damaging soil structure, irrespective of the amount of rainfall. Even if these benefits are nullified experimentally by hand application of nitrogen and herbicides to CT plots, the long-term commitment to NT and the maintenance of straw in the field have improved the economic benefits relative to CT (Fig.12). The improvement in the net margin of wheat production under NT is due to a reduction in costs associated with tillage (energy and labour) and nitrogen application, and the increase in SOM (Fig. 9) and its associated improvement in yields (Fig. 8).

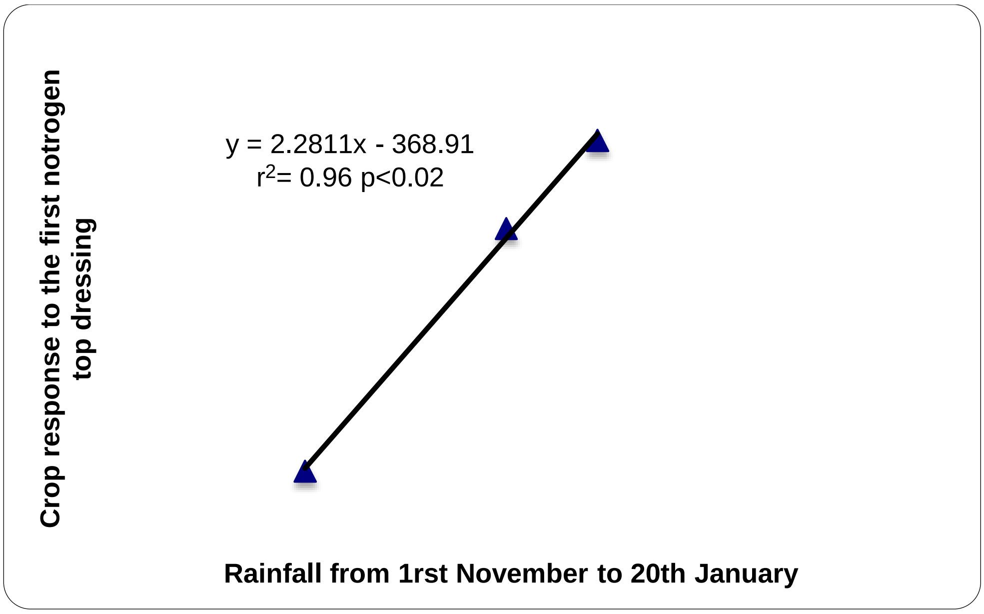

Fig. 10: Interaction between wheat yield increase to a nitrogen application (60 kg N/ha) at tillering (20th of January), for the treatments were an extra 120 kg/ha were applied at the beginning of stem elongation (28th February), and the amount of precipitation between seeding time and the first top dressing with nitrogen (Carvalho et al. 2005).

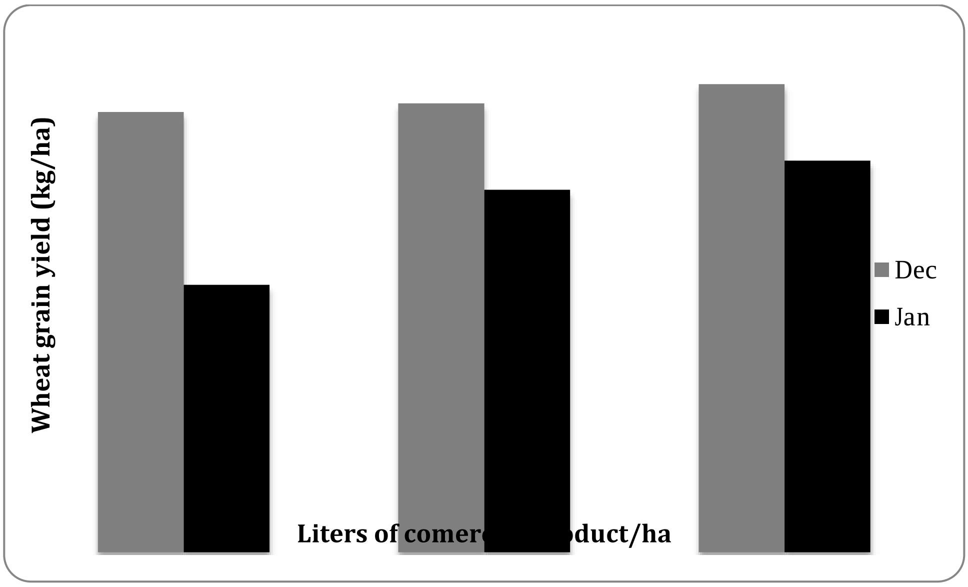

Fig. 11 : Interactions between the spraying time, the herbicide level and the wheat yield on a Luvisol in the south of Portugal. The herbicide tested was Dopler Plus ® (250g/l of diclofope-metil + 20g/l of fenoxaprope-p-etil + 40 g/l of mefenepir-dietil) (Based on Barros et al., 2008)

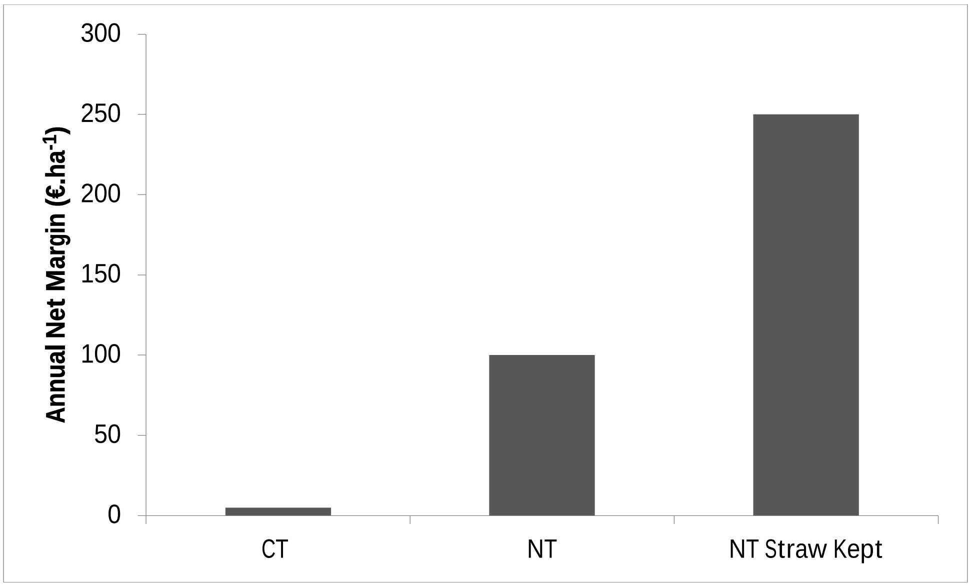

Fig. 12: Effect of tillage system and crop residues management (CT-Conventional Tillage and straw bailed; NT-No Till and straw bailed; NT+S- No Till and straw kept on the field) on the wheat net margin on a Luvisol in the South of Portugal (Based on Marques, 2009).

4. CONCLUSIONS

Over the long term, NT is improving the sustainability of arable crop production under the conditions prevailing in the South of Portugal. A reduction of soil erosion and its associated improvement in the SOM content, particularly if the straw of grain crops is kept on the soil surface, has improved soil fertility, crop yields, nutrient use efficiency and annual net margin of the wheat crop. NT has also improved water infiltration, drainage and the trafficability of the soil. These are important benefits to stabilize yields over time under Mediterranean conditions. A reduction of runoff is important to increase soil water storage in dry years, while an improvement in the timeliness of field operations associated with better internal drainage is crucial for improving crop yields during wet winters.

ACKNOWLEDGEMENTS

The author acknowledges the valuable assistance of Professors Michael J. Goss and Isabel Brito for the careful review of this article.

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CASE STUDY 6: RUNOFF AND SOIL LOSS FROM STEEP SLOPING VINEYARDS IN THE DOURO VALLEY, PORTUGAL: RATES AND FSACTYORS

Tomás de Figueiredo, Jean Poesen, Alfredo Gonçalves Ferreira & Dionísio Gonçalves

1. INTRODUCTION

Some of the most renowned vineyard regions in the world are located in mountain or sloping land, and they very much contrast with flat or gentle slope viticulture (Unwin, 1996; Pérez Verdú & Jofre, 2007; Damásio et al., 2010). The high potential erosion risk prevailing in such topographic conditions has for long been tackled with traditional soil conservation structures, as it is the case of dry-stone walled terraces (e. g. the Douro region, NE Portugal, or the Priorat region, NE Spain), which depended on the availability of cheap labour for constructing and maintaining those structures. Furthermore, with crop management operations manually performed, traditional vine plantation systems required high permanent labour inputs (Bianchi-de-Aguiar, 1987; Guimaraens & Magalhães, 2006; Pérez Verdú & Jofre, 2007; Martinez-Casasnovas & Ramos, 2009).

If these socio-economic conditions change, alternative vineyard plantation and management systems are needed, and the inherited landscapes require appropriate management. Both are very important challenges for sustainable development of such regions, as well as for erosion and soil conservation research (Borselli et al., 2006; Pérez Verdú & Jofre, 2007; Martinez-Casasnovas & Ramos, 2009; see also also CERVIM and its periodical publication Mountain Vine.growing. In spite of the potential erosion risk and probably because of the control that is effectively applied, soil erosion is a systematically cited topic of concern when describing plantation and cultivation systems in these areas. A review of mean soil erosion rates for different land use types in Europe revealed that vineyards have the highest soil loss rates in the Mediterranean, i.e. 16.6 ton ha^{-1} $year^{-1}$ (Cerdan et al. 2006). Several studies address the erosion impact caused by the implementation of new vineyard systems or as a consequence of the abandonment of formerly cultivated areas, and apply a wide range of methods for assessing erosion rates, or related land features, as shown by Rosa, 1981; Tropeano, 1983; Richter, 1991; Figueiredo & Ferreira, 1993; Martínez-Casasnovas & Sanchez-Bosch, 2000; Borselli et al., 2006.

Soil erosion is a well-known problem in Northeastern Portugal and has been systematically reported in regional soil descriptions (e. g., Martins, 1988; Agroconsultores e Coba, 1991; Figueiredo, 2001; Figueiredo, 2012). In the Douro Region (NE Portugal), where the grapes for Port Wine are produced, the problem is emphasized by its specificity and economical importance. In fact, the Douro Region was the world's first to be formally granted the statute of quality wine production region, dating back to the middle of the XVIII century, due to the originality of processes used for wine making and to the high quality of the wines produced. Since 2001, the Douro Region has become UNESCO World Heritage (Bianchi-de-Aguiar, 2000; see also UNESCO), a status that fostered socio-economic and cultural activities in the region but that highly increases responsibilities in terms of conservation and sustainable use of regional resources, as it is the case of soil and water. The Douro Region covers 250 000 ha, of which 46 000 ha are vineyards, producing several wine types including the Port, which is mainly exported (near 1 million liters annually in the last decade), with a very significant contribution to the Portuguese wine production, the wine export and the Gross Agricultural Product (Damásio et al., 2010; see also IVDP).

This Region, set along the Douro River and its tributaries, displays a strongly human-made landscape covered with vineyards cultivated on steep slopes. Heavy labour made the stabilization of these hill-slopes possible by means of traditional terracing, manually built according to models that changed through time (Bianchi-de-Aguiar, 2000). In the 1970's, in a context of growing socio-economic constraints, alternative vineyard installation and cultivation techniques were tested and progressively adopted in the 80's, namely row plantation perpendicular to the contour ("vinha ao alto") (Bianchi-de-Aguiar, 1987). Nowadays, the new system is an integral yet small part of the plantation schemes allowed under UNESCO World Heritage rules, which are understood as a recent relevant contribution to the exceptional living and active cultural landscapes that justifies the granted status(Bianchi-de-Aguiar, 2000; Damásio et al, 2010). As a high potential erosion risk system, especially considering the prevailing steep slopes, the impact of the vine row plantations in an up- down-slope direction on the stability of regional agro-ecosystems, soil and water conservation issues had to be experimentally assessed (Rosa, 1981).

It should be stressed that long-term erosion data series in Europe are quite rare (Boardman & Poesen, 2006), and they are widely recognized as highly valuable, either from the strictly statistical point of view or from that of an in-depth study of temporal variations in the occurrence and magnitude of erosion processes. This paper particularly aims at presenting the results of long-term runoff and soil loss data recorded in a set of meso-scale erosion plots installed in vineyards planted in rows perpendicular to the contour, in the Douro Region. It aims as well at identifying, and quantitatively deriving the significance, of erosion factors locally active, that help interpretations of results obtained.

2. MATERIAL AND METHODS

2.1. Study area

The study site is located in the heart of the Douro Region, near Pinhão, at Quinta de Santa Bárbara, a state experimental station (41˚10’ N, 7˚33’ W, 130 m elevation) (Fig. 1). The site represents the regional main agro-ecological features. Steep slopes dominate in the long hill-slopes draining towards the Douro River and its tributaries, the gentler slopes being found in the round crests dividing catchments of the higher order streams on schist areas. These Paleozoic schists form the largely dominant geological basement, with quartzites outcropping in narrow crests and Variscan granites in relatively wide surfaces. Soils developed under these conditions, for the most covered by shrubby Mediterranean vegetation, are shallow, with high rock fragment contents, corresponding to Leptosols, or to Regosols (rarely Cambisols) on the less steep slopes (FAO/UNESCO, 1988; Agroconsultores e Coba, 1991; Bianchi-de-Aguiar, 2000).

Fig. 1: Pinhão, the study site area in the Douro Region (UNESCO World Heritage): location and landscape, with steep slope vineyards, terraces of several types and shrubs

Besides, with strongly humanized landscapes, the Douro valley hillslopes are covered by a very large area of Anthrosols. These have a highly disturbed profile due to site preparation for vineyard plantation, associated with topographical changes caused by structural soil conservation measures, namely terraces of several types, since long applied in this Region,. Soils have a high content of fine sand and silt and a low content of clay in the fine earth; they are acid and poor in organic matter. Rock fragment content is very high, either at the surface or throughout the entire profile depth. However, organic matter contents tend to increase and rock fragment contents tend to decrease, in the surface soil layer, with time since disturbance due to vineyard installation (Agroconsultores e Coba, 1991; Bianchi-de-Aguiar, 2000; Figueiredo, 2001).

The climatic regime is Mediterranean, with rains concentrated in winter and autumn, and a typical severe soil water deficit in summer. The average annual precipitation over 30 years is 650 mm. The average maximum occurs in winter (December, 100 mm) and the minimum in summer (August, 10 mm). The mean annual air temperature is 16 ˚C and ranges from 8 ˚C in January, to 25 ˚C in July. Soil water deficit is very severe and lasts for 6 month, from mid-spring to early autumn (INMG, 1991; Agroconsultores e Coba, 1991).

Vineyards are normally planted in rows on the contour, and plantation requires deep ripper operations, that break the hard rock. Freshly outcropping rock fragments, mixed with fine earth, are further crushed by machine traffic during site preparation.

This may include also the installation of terraces, formerly with stone walls supporting a vertical raiser, nowadays with bare earthen raisers leaned at natural rest angle (Agroconsultores e Coba, 1991). Bench width depends very much on hillslope gradient but, where installed, benches are nowadays level narrow strips, less than 2.5 m wide, accommodating a single vine row, whereas in some of the older terraced areas benches were wide multiple-row strips, draining outwards (Bianchi-de-Aguiar, 2000; Queiroz et al., 2008). Alternatively, following a model spread in the Douro Region during the 80's of the XX century, some vineyards are plated in rows installed perpendicular to the contour. Changes in the plantation system with time have been pursuing economy in labour demand, increased mechanization and improved wine quality (Bianchi-de-Aguiar, 1987).

2.2. Erosion plots setup and operation

Runoff and soil loss were measured since 1978 in five plots installed in vineyards, planted in 1971, with rows perpendicular to the contour (Rosa, 1981). Plot slope gradient equals 45%, and plot area equals 1/60 ha (32.1 m long by 5.2 m width). The plots have a slightly different slope aspect. Also, plots were planted with different distances between plants and between rows, as shown in Fig. 2. The experimental design aimed at testing the effect of Treatments with different plant density, set on paired plots, except for the intermediate plant density tested on one plot only (Plot 5). The soil is a schist-derived gravely silt loam, classified as an Anthrosol (FAO/UNESCO, 1988), and, according to the analysis of soil samples taken from the surface horizon of all plots, contains on average 5 % clay, 41 % silt and 50 % fine sand, about 60 % by mass of rock fragments, and has low organic matter content (0.5 %) (Figueiredo, 2001). Estimated rock fragment cover is 55%, an average of assessments based on surface soil photograph analysis, taken in all plots as that shown in Fig. 2 (Figueiredo, 2001).

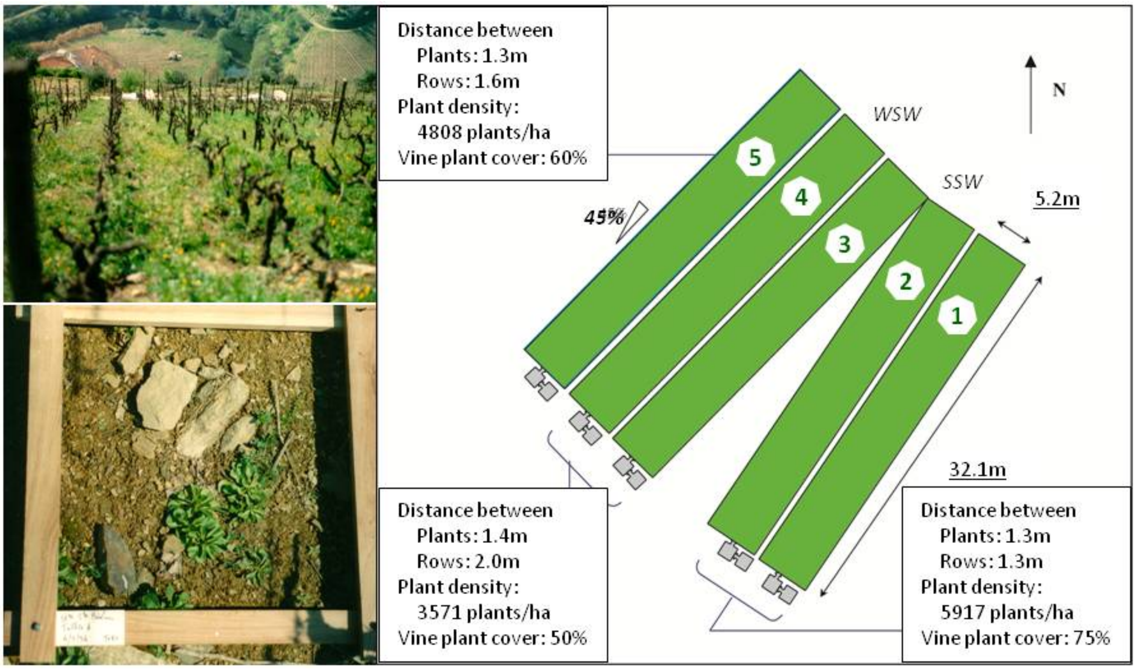

Fig. 2: Erosion plots in Douro Region vineyards at Quinta de Santa Bárbara, Pinhão: experimental setup and characteristic plot and soil cover conditions in Spring.

Runoff water and soil lost from each plot, were collected at the base of the plot, in a set of three tanks interconnected by means of 1/11 discharge slots. Each one of these allows, when the tank is full, that one eleventh of the incoming runoff passes to the next tank. During the 10-year observation period, the third tank never filled. Data were collected after each rainfall, or after several consecutive rainfall events, depending on the weather conditions, because cleaning of all tanks took about 6 h. Records of each data collection corresponded to what is hereafter named "event". The volume of runoff water collected in each tank was measured and, after stirring, a 0.5 L sample of water plus suspended load was taken, to determine the dry sediment mass. If all sediment particles were deposited on the tank bottom, the tank was drained and the mass of the deposited sediment was determined. Runoff and sediment samples were taken about 17 times, in average, each year (Figueiredo, 2001).

A weather station provided daily data on evaporation (pan), wind speed and direction, and precipitation (pluviometer and pluviograph).

Crop management operations performed in vineyards were recorded, together with qualitative information regarding plant development. These records were matched with quantitative field assessments of ground and canopy vegetation cover (based on the analysis of soil surface photos and canopy size measurements), so as to allow deriving the annual evolution of vine canopy cover, adventitious vegetation ground cover and soil surface disturbance by tillage operations (Figueiredo, 2001).

2.3. Data treatment

Precipitation, Runoff and Soil Loss from the 5 plots, measured during a 10-year observation period (1979-1988), were analyzed at annual, seasonal and event levels. The first soil loss observations recorded dated November 19th 1978 and the last considered in the analysis dated February 14th 1989, adding up to 167 events. For the calculation of seasonal and annual values, the official dates of such periods were followed, corresponding event values were summed, and data linearly interpolated with cumulative precipitation for events covering consecutive computation periods (Figueiredo, 2001).

When assessing the overall erosional response to rainfalls produced by a soil surface under such soil and topographical conditions, a single value representing that response was computed as the average runoff and soil loss from the five plots, assuming an average vine plant cover (62%). On the other hand, individual plot runoff and soil loss values were kept when analyzing the effects of specific characteristics (i.e. vegetation cover) on their erosional response. The former approach was also possible because plots had actually a parallel response to rainfall events and the coefficients of the correlation between plot data series (either annual or event runoff and soil loss) were all statistically significant (Figueiredo, 2001).

Data extraction from the rainfall records (pluviograph bands) was the first step of precipitation data treatment. This basic data set allowed computation of rain erosivity indexes with precipitation depth, intensity for several rainfall durations, and kinetic rainfall energy, estimated according to Wischmeier & Smith (1978), adapted to SI units (Tomás, 1997). Other than the recommended truncation criteria, concerning precipitation depth or intensity, were also applied for computing rain erosivity indexes, and their performance assessed (Figueiredo, 2001). Rainfall intensity-duration-frequency curves were also derived for Quinta de Santa Bárbara, after extreme rainfall analysis (Chow, 1964).

Besides descriptive statistics computed for all data sets, correlation and regression techniques were applied as well. Furthermore, analysis of variance was applied to identify statistically significant differences in results, caused by the plot and season effects and their correspondent interactions. For multiple comparison of means the contrast method was applied (Steel & Torrie, 1980).

3. RESULTS

3.1. Runoff and soil loss rates

Annual average rainfall depth in the 10 years of records was 573 mm, with around 40 % in winter, only 7 % in summer, with spring and autumn seasons having slightly over 25 % each, as expected in a typical Mediterranean environment (Fig. 3). Global average annual runoff and soil loss were 22 mm and 361 kg ha-1, respectively, the global average runoff coefficient being 4 %. Seasonal distribution of runoff was quite similar to that of rainfall but sharply different for soil loss. Actually, spring and summer accounted for more than 80% of the annual soil loss.

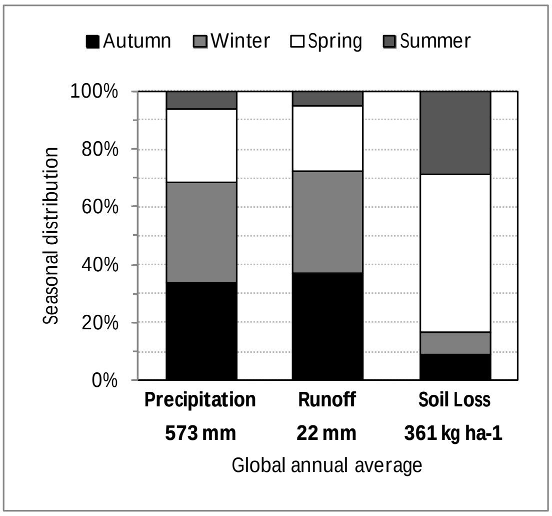

Fig. 3: Global 10-year averages of annual Precipitation, Runoff and Soil loss at Quinta de Santa Bárbara, and their seasonal distribution

Annual soil loss ranged from 50 kg ha-1 to 1906 kg ha-1, the year with the highest annual soil loss recorded contributing to 53 % of the total soil loss observed in ten years and the year with the second highest annual soil loss to 22 % (Fig. 4). Annual soil loss was in most years much lower than average (median = 122 kg ha-1) and some years (1988 and 1980) accounted for most of the soil loss recorded. Distributions of rainfall and runoff during the 10-year period were much less biased than that of soil loss. Furthermore, rankings of the three variables according to their magnitude do not match well, indicating the poor relationship between them found at annual scale (Fig. 4).

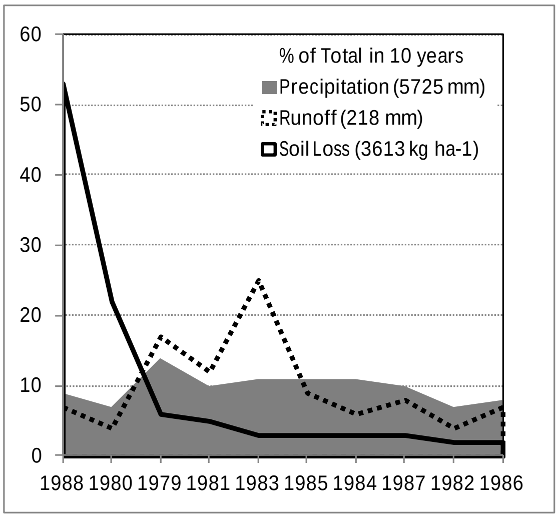

Fig. 4: Relative contribution of each year to total Precipitation, Runoff and Soil loss recorded during 10 years at Quinta de Santa Bárbara. Years are ranked from highest to lowest soil loss rates)

Annual and seasonal values presented reflect the plots' erosional response at event level. Actually, bias found in average seasonal soil loss distribution and in the annual series resulted from 4 outliers in the event data series. As depicted in Fig. 5, the event with the highest soil loss (on 27-6-1988) amounted to 833 kg ha-1 and accounted for 23 % of total erosion in ten years. The first and the second highest erosion events together accounted for 45 % of that 10-year total soil loss and occurred in two consecutive weeks (June 20 and 27, 1988). These events and the fourth event in soil loss magnitude occurred within a three-week period. The 4 events with the largest soil loss contributed for 72 % of the ten-year soil loss occurred in late spring and early summer.

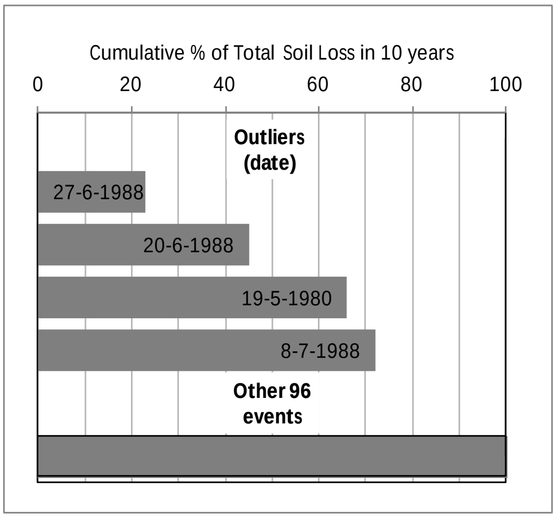

Fig. 5: Relative cumulative contribution of major erosion events to total soil loss recorded in 10 years at Quinta de Santa Bárbara. Total number of erosion events equals 100

Outliers contribute significantly to the total annual soil loss, in the case of 1980 representing virtually all the soil loss recorded (98%, Fig. 6). Annual soil loss in all the 10 years of records was, indeed, very much dependent on the contribution of the most erosive event or, of the two major events, which represented, on average, respectively 39% and 62% of the annual total soil loss. Coefficients of the correlation between annual total soil loss and that caused by the most erosive event is 0.90, increasing to 0.99 when the two most erosive events are considered.

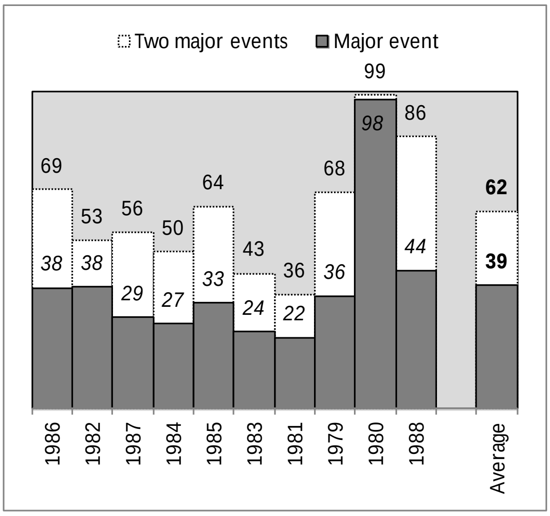

Fig. 6: Yearly and average % contribution of the largest and the 2 largest erosion events to annual soil loss (years are ranked from lowest to highest soil loss rate)

On the other hand, the 167 events analyzed include 67 producing only runoff (that may be labeled as non-erosive), while both runoff and soil loss were collected in the remainder 100 events (2/3 of the total). Therefore, a wide range of erosion conditions prevailed during the 10-year period, yielding from high-magnitude soil loss outliers to runoff events producing only clear runoff water. Consequently, the event soil loss data series shows very high dispersion around the mean value (378 % coefficient of variation, CV) and it is highly skewed (5.2 kg ha-1 median against a 7 time higher mean of 36.4 kg ha-1). The other two event data series show much lower dispersion and bias: precipitation – 128 % CV, mean 36.1 mm, 1.6 times higher than median; runoff – 255 % CV, mean 1.4 mm, 3.5 times the median.

3.2. Factors affecting plot erosional response

Rainfall erosivity is the primary factor affecting plots erosional response. From all tested rain erosivity indexes, incorporating rainfall depth, intensity and kinetic energy, in a single or combined way, the best performing one was EI30m (rainfall kinetic energy times the maximum intensity in 30 min, computed at event level, MJ ha-1 mm h-1), calculated with all rains and not with restrictions imposed in the original EI30 (Wishmeier & Smith, 1978). EI30m explains half of soil loss data variance at event level in a quasi-linear relationship (SL is event soil loss):

$$SL=0.306\ EI30m^{0.848}\ (r=0.713,\ N=100)$$

Hence, a large part of data variance is not explained by a single erosivity index or rainfall characteristic, as illustrated in Fig. 7 for the outlier events. These indeed resulted from uncommon rainfalls. However, not only differences in return period do not match differences in soil loss magnitude, but also duration for maximum return period is quite different when comparing the 4 events (the first and the fourth in magnitude were merged in Fig. 7). Other factors have to be considered for explaining recorded data.

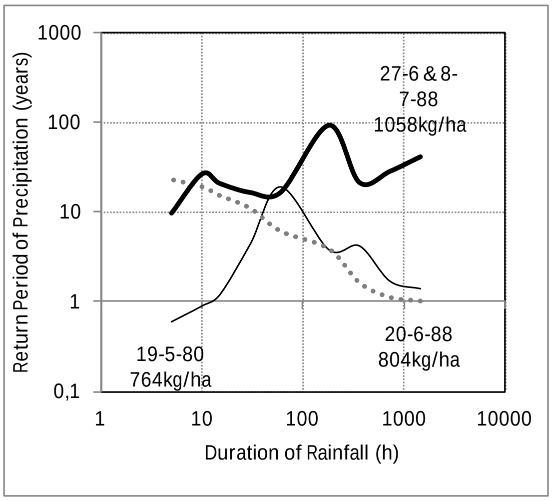

Fig. 7: Seasonal Return Period of Precipitation determining Outlier Events, as affected by rainfall duration

Average annual EI30m was 796 M ha-1 mm-1, an index very well correlated with EI30, the original version (r=0.997), but with a much poorer correlation with annual precipitation (r=0.632).

Average global estimates of vegetation cover , combining ground surface and canopy cover, reached a seasonal maximum in summer (132% of the whole-year average) and a minimum in winter (71% of the hole-year average), autumn showing a slightly higher cover than spring (106% against 91% of the whole-year average). Plots correspond to different plantation schemes and annual soil loss was significantly lower in plots with high cover (1 and 2, with 75% vines canopy cover), when compared with low or intermediate cover (plots 3 and 4, with 50% ad plot 5, with 60%) (Fig. 8). Regression analysis, applied with annual average values, show a typical negative exponential relationship between soil loss and vegetation cover (Fig. 9a). Plant density, a parameter representing vine canopy cover, relates similarly to average annual plot soil loss, yet with lower, non-significant, correlation coefficient (Fig. ??.9b). The quality of the relationships is much improved when considering median instead of mean soil loss values (Fig. 9).

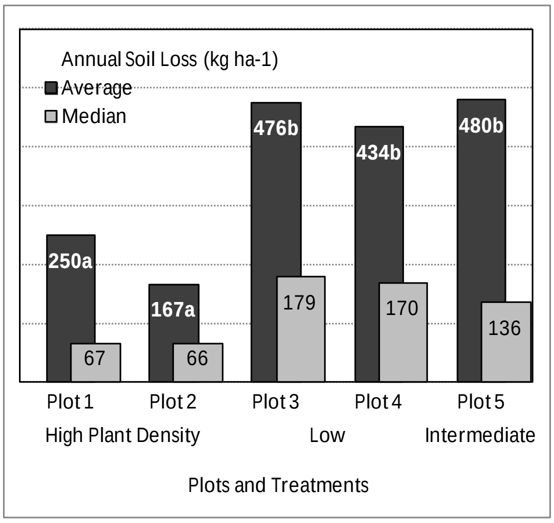

Fig. 8: Annual average and median soil loss from the erosion plots of Quinta de Santa Bárbara after 10 years of records

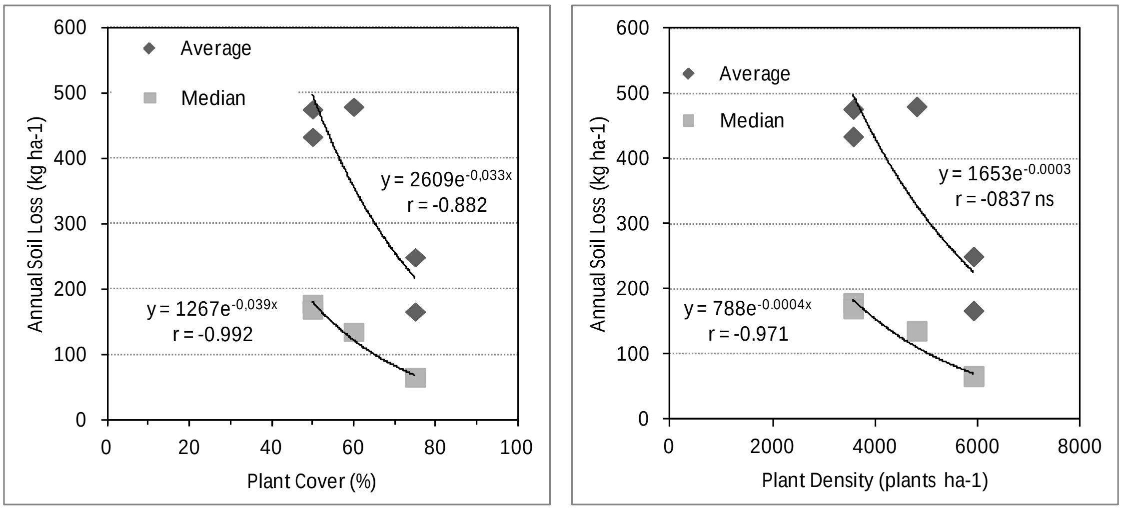

Fig. 9: Relationship between vegetation cover and soil loss mean annual values at Quinta de Santa Bárbara erosion plots: a. plant cover; b. plant density

Soil disturbance by tillage operations may result in an increased soil susceptibility to runoff erosion. The hypothesis was tested comparing event soil loss per unit erosivity index, corrected for the actual vegetation cover, prior to and after tillage operation dates recorded. These include tillage, performed in mid-autumn, and “descava”, a light operation for spring weed control. For tillage, the average change was a slight increase of +1.1 kg EI30m-1 ha-1 bare soil (with a median of 0.7), while for “descava” average and median changes were much lower (+0.1). In both cases, changes in soil loss rate were not statistically significant. For tillage, however, the range on observed changes was very large (from +42.8 to -25.1 kg EI30m-1 ha-1 bare soil).

Other factors affecting the erosional response are soil and topographical conditions prevailing in the 5 plots, which are similar in all of them (Figueiredo, 2001). Based on several studies (e. g. Hudson, 1981; Poesen 1993; Roose, 1994; Torri et al. 1997; Morgan, 2005) it can be expected that, a poorly structured, low permeability silt-loam soil, also poor in organic matter, has a very high erodibility. In addition, slope gradient is very steep in all plots (45%), thus representing a high potential erosion risk. In contrast, the very high rock fragment cover and contents of the soil in the plots effectively limits runoff erosion at the Quinta de Santa Bárbara experimental site, which is in line with results reported elsewhere (Poesen et al. 1994). Runoff and soil loss rates presented above clearly show that rock fragment cover is much more effective in controlling runoff erosion than top soil rock fragment content in promoting it in Quinta de Santa Bárbara plots.

4. DISCUSSION

Annual runoff was on average 22 mm and annual soil loss was less than 0.5 t ha-1 in all plots. Richter (1991) observed a much lower annual runoff depth, i.e. 3.8 mm year-1, for vineyards planted perpendicular to the contour in the German Mosel valley, where erosion plots were monitored on similar steep slopes and on soils having high contents of rock fragments as well but under different climatic conditions. On the contrary, average annual soil losses areis comparable to the rates reported by Richter(1991), but much lower than those measured in other wine-producing regions: i.e. 70 t ha-1 in NW Italy (Tropeano, 1983), more than 100 t ha-1 in Romania (Bianchi-de-Aguiar, 1987), 20 t ha-1 in the Ardèche Region, France (Augustinus & Nieuwenhuyse, 1986) and a mean soil loss of 16.6 t ha-1 for the Mediterranean (Cerdan et al. 2006). Under the prevailing conditions of slope gradient, vegetation cover and soil erodibility, these values highlight the strong protective effect of rock fragments (Poesen & Lavee, 1994; Poesen et al. 1994), which explains the overall behaviour of soil lost by water erosion in the Quinta de Santa Bárbara erosion plots.

Soil losses were concentrated in the spring-summer period, whereas autumn and winter showed significantly lower erosion rates in all plots. These results show a distribution of soil loss during the year which is different from that reported by most Portuguese studies on soil erosion (e. g. Ferreira et al., 1985; Coelho et al., 1990; Silva et al., 1995; Tomás, 1997), in which the main erosive season is autumn. These results reflect the effect of outlier events that occurred in late spring-early summer. They call the attention for the case of permanent crops, in which the traditional spring tillage operations may increase the soil susceptibility to erosion, in the semester when vegetation cover and precipitation characteristics were highly variable during the 10 years of records (Figueiredo, 2001), all factors that contribute to very erosive events. In fact, annual precipitation correlated well with wet semester soil loss (autumn and winter; r=0.726) and poorly with dry semester soil loss (spring and summer; r=-0.151) (Figueiredo et al., 1998), therefore indicating annual soil loss was highly dependent on the occurrence or non-occurrence of spring-summer rainstorms, while in autumn and winter smaller rain events regularly generated erosion every year. Moreover, not only short-duration low- frequency rainfalls were less intense in the wet semester when compared with the dry semester, but also the dead vine leaves protected the soil surface from direct rainfall impact (Fig. 10 and Fig. 11).

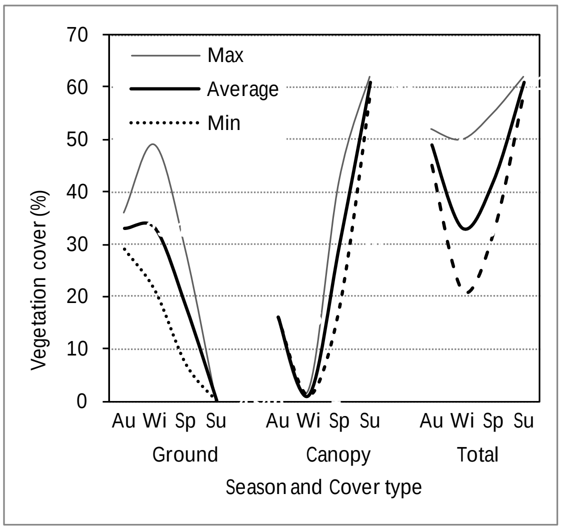

Fig. 10: Seasonal changes in mean, minimum and maximum vegetation cover at Quinta de Santa Bárbara.

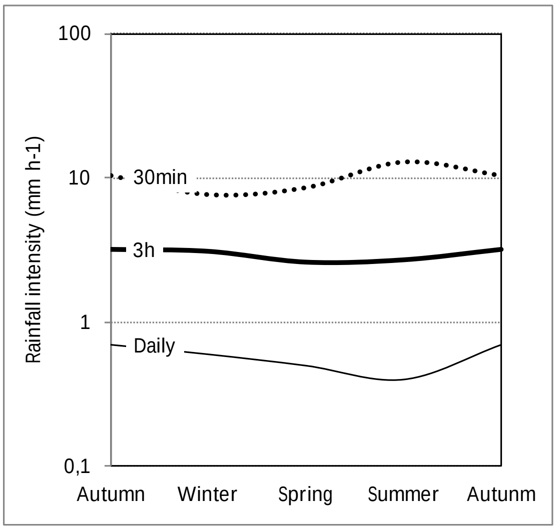

Fig. 11: Seasonal changes in average maxima rainfall intensity at Quinta de Santa Bárbara, for different rainfall duration

The four recorded outlier erosion events determined almost 75 % of the total soil loss in 10 years. This very high concentration on few events, together with the evidence of no sediment export during ca. 33 % of the total number of events, very much skewed and scattered the event soil loss data distribution. Longer term data series were treated by Tomás (1992 and 1997), regarding plots with wheat-fallow crop rotation in Vale Formoso, Alentejo, Southern Portugal. There, the most erosive year contributed to 19 to 37% of the total soil lossin 22 years and the two most erosive years accounted for 36 to 50% of that total (Tomás, 1992). Other long-term experiments (e. g. Edwards & Owens, 1991; Poesen et al., 1996; Larson et al., 1997; Shipitalo & Edwards, 1998) also show this soil loss concentration in few records (events and years); however, none of them reaches such a strong effect as that reported for the Quinta de Santa Bárbara erosion plots.

Outlier events were determined by very erosive rainfalls besides vegetation cover effects. In fact, vine canopy cover estimate was 43 % in the 1980 event, whereas in the 1988 events estimates were 71 % and 83 % in the earliest and latest event, respectively, meaning that the most erosive ones occurred when full cover was nearly reached (Figueiredo, 2001). Even though rainfall erosivity was very high (event EI30 accounting for 51 % and for 63 % of the annual values in 1980 and 1988, respectively; Figueiredo & Gonçalves, 2008), it is not possible to determine a typical rainfall duration and frequency associated to such events, as the duration corresponding to the maximum return period in each case was very different (Fig. ??.7). Moreover, no erosivity index adequately relates to the erosional response under such circumstances (Figueiredo, 2001).

As explored by Figueiredo & Gonçalves (2008), rainfall erosivity at Quinta de Santa Bárbara, in Douro, NE Portugal, was not as high as that computed by Tomás (1997) for Vale Formoso, Alentejo, SE Portugal, having the largest and longest Portuguese erosion plot data set (676 against 1038 average annual EI30, in MJ ha-1 mm h-1). In spite of the similarity in climatic regime (typically Mediterranean in both cases), Vale Formoso depicts stronger aridity (469 mm average annual precipitation) and the normally associated climatic features (e. g. rain intensity with a maximum 100 years return period and 30 min duration is 80 mm h-1, against 66 in Quinta de Santa Bárbara) (Figueiredo & Gonçalves, 2008).

As stated above, with the soil and topographical conditions prevailing at Quinta de Santa Bárbara, and considering that vine rows do not provide full cover of the soil surface, soil loss rates recorded are actually very low. This is explained by the effects of the very high rock fragment content and cover in significantly reducing soil erosion rates on such hill-slopes. Figueiredo (2001) developed an explanation for the low soil loss rates observed related to rock fragment cover. As rock fragments affect sediment transfer along the hillslope, due to their effects on surface roughness and overland flow tortuosity, their effects actually also interact with the effects of slope length on soil loss (Figueiredo et al., in prep.). This slope length effect adds to the effect of cover, which directly controls fine earth particle detachment due to raindrop impact (Figueiredo et al., 2004).

5. CONCLUSIONS

Results discussed above lead to the following conclusions:

a) Average annual soil loss, from the five erosion plots reported in this study, was below 0.5 t ha-1, which is very low compared to soil loss rates normally reported for Mediterranean vineyards;

b) The very high rock fragment contents and cover of the soil is the single mitigating factor that helps explaining the observed low erosion rates;

c) Plantation scheme, affecting plant density and vegetation cover, significantly contributed to explain differences in observed soil loss between plots;

d) Runoff erosion was concentrated in the spring-summer semester, when more than 75 % of average annual soil loss occurred, mainly due to high-magnitude erosion events that occurred in those seasons following low frequency rainfalls, and they contributed to almost 75 % of the total soil loss recorded in 10 years.

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CASE STUDY 7: RUNOFF EROSION IN PORTUGAL: A BROAD OVERVIEW

Tomás de Figueiredo

1. INTRODUCTION

Soils are generally a scarce natural resource in Portugal as far as productivity is concerned (CNROA, 1983; Agroconsultores e Coba, 1991; Agroconsultores e Geometral, 1995). In fact, the major part of the territory is potentially not suitable for agriculture, corresponding to areas with misuse and over-exploitation of the soil resource in cropland, and to typical marginal land cover by forests and shrubs (Fig. 1). The soils with moderate and high suitability are under agricultural use, more or less intensive, that take advantage of their productivity, but that partly experience incorrect management practices (Sequeira et al., 2012, this issue). Besides the crucial support of food and fiber producing activities, soil functions in ecosystems, either cultivated or natural and semi-natural, contribute in providing services and public goods that rise attention to management of large tracts of marginal areas, whilst tackling the problem of persistent population decrease and ageing in rural areas that lead to unmanaged land or abandonment, for instance with consequences for wildfire hazard and control (CIMO, 2009; Rosário, 2011). As so, poorly provided by good soils, enduring threat of several types all over the territory (with a large extent assigned to runoff erosion, CAN, 1980), Portuguese soils require knowledge and protection, to limit resource depletion, recover degraded areas and ensure sustainability pf actual or foreseen land uses and soil-based activities.

This opening chapter of the Portuguese case studies presented hereafter aims at providing a general picture of runoff erosion distribution in Portugal, more precisely in the country’s continental territory, adding further information in view calling attention to wider range assessments that rise awareness towards natural resources degradation trends, and sustainable land use and management.

A review on the topic of soil erosion in Portugal has been published by Coelho (2006). It is therefore a recent and comprehensive review of relevant research and broad approaches to the identification, quantification and distribution of the problem in Portugal, up to that date. The following text takes Coelho (2006) as a reference information base, that is summarized here and to which is added new information, issued from more recent research and other developments carried out at National level.

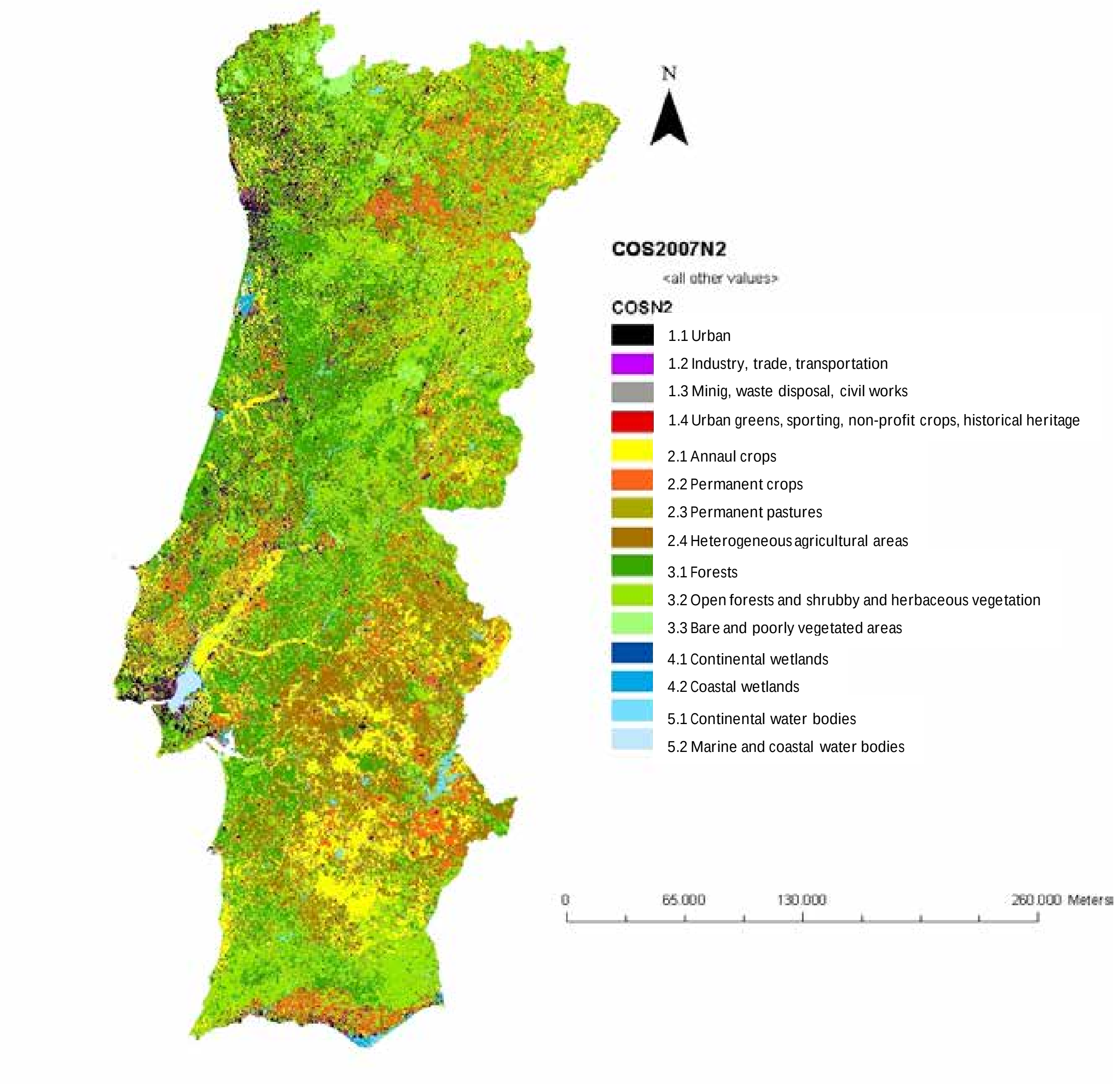

Fig. 1: COS2007 – Land use and cover map of Continental Portugal 2007 (IGP, 2007).

2. CONTINENTAL PORTUGAL: NATURAL SETTING AND LAND USE

The Portuguese continental territory is a land of contrasts, where the Tagus River splits apart the Northern hilly country from the lower lands of the Southern half (Fig.

2). In fact, north of the Tagus, 75 % of the territory is above 200 m elevation, with an average altitude of 370 m, while the southern tract of the country the mean elevation is 160 m, with 62 % of the area below 200 m (Medeiros, 1987; Medeiros, 2005). Very few mountain ranges exist in the South, the highest peak hardly surpassing 1000 m (Serra de S. Mamede, 1027 m), but the top of Serra da Estrela, north of the Tagus, in the Central Massif, approaches 2000 m (1993 m). The most impressive mountain ranges are aligned from NW, near the coast, to Southeast, inland. This general physiographic picture readily indicates a geomorphologically active natural setting, in which hill-slope processes as runoff erosion play an important role.

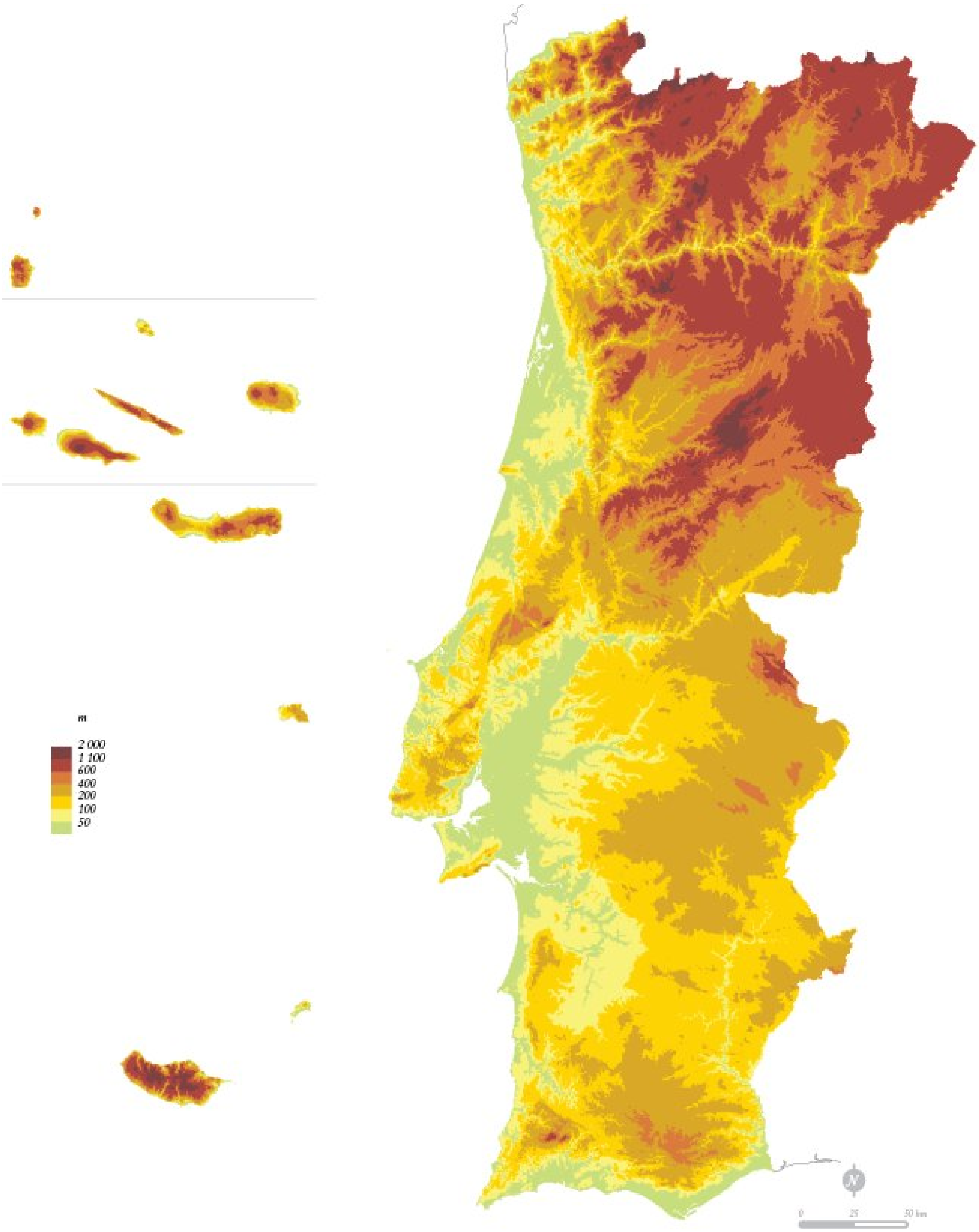

Fig. 2: Hypsometric map of Portugal, 1:1 000 000 (CNA, 1980).

Precipitation in Portugal ranges widely, from around 300 mm annual average in the NW near-coast mountain tops (the rainiest spot in Europe), to below 500 mm in the Southern region of the Algarve and in inland areas of the deep Douro valley (Ribeiro, 1986). Factors generally affecting rainfall spatial distribution are latitude, altitude and continentality, coupled to outcome quite contrasting areas a short distance apart. As typical of the Mediterranean climates, rainfalls are scarce in summer, and mainly concentrated in autumn and winter, and inter-annual variability is very sharp (coefficients of variation of annual precipitation reach 30% about the mean, Lencastre & Franco, 2006). As annual precipitation amounts decrease in the territory, rainfall concentration increases (Daveau, 1977) and, therefore, higher intensive rainstorms occur in drier areas when compared with the wetter ones, meaning more severe consequences in terms of runoff erosion. In fact, and in average terms, while for the more humid areas of Northern Portugal values of the annual rainfall erosivity index (R or EI30, in SI units) may be assumed roughly similar to that of annual precipitation (in mm), this relationship does not stand for the whole country, as in drier areas of Southern Portugal they may reach about the double of annual precipitation values (Figueiredo & Gonçalves, 1990; Figueiredo, 2001; Tomás, 1997).

In about 3/4 of the Portuguese continental surface the ancient geological basement (the Hesperian Massif) outcrops metamorphic rocks (Palaeozoic schists) and magmatic rocks (Variscan granites). A centre-western belt together with the south facing Algarve ones are Mseo-Cenozoic terrains, where Secondary limestone and more recent sedimentary rocks outcrop (including sandstones, marls and clays). The third main structural unit of Continental Portugal is the Tertiary Tagus basin, a depression of the ancient basement filled in with thick layers of loose sediments (Ferreira, 2005).

Soils derived from these parent materials are for the most acid and very acid, except in the calcium carbonate rich or basic lithologies (Varennes, 2003). Also, where hard rock basements of schist and granite dominates, soils are shallow, coarse (in granites) or medium texture (in schists) tony and incipiently developed (Leptosols, Regosols), especially in the highlands North of the Tagus river, corresponding to marginally suitable or unsuitable areas for agriculture (Fig. 3). In the higher altitudes, soils generally depict a much higher carbon content in the surface horizon than those of the lower, drier and hotter lands, while in gentler slopes deeper soils may develop (Cambisols, Luvi or Alisols) (Agroconsultores e Coba, 1991; Aroconsultores e Geometral, 1995). On the sedimentary lithologies of the Meso-Cenozoic belts and Tertiary Tagus basin, where slopes are gentler, under milder and drier climatic conditions, deeper and well developed soils dominate (Cambisols, Luvi or Alisols, and even Mollisols). This is also the case in the few volcanic or micro-crystaline rocks outcrop, as basalt (Lisbon) and micro-gabbros (Beja, Alentejo)), where finer textures are common, often with internal drainage problems (Vertisols), nevertheless highly suitable for agriculture. The deeper Portuguese soils are the incipient Fluvisols better represented in the main rivers flood plains (Mondego, Tagus and Sado), with very high suitability for agriculture (Cardoso, 1973; CNROA, 1983). In most of the country, soils are weakly structured and carbon contents are low, with the exception of highland soils mentioned above. In brief, soils with lower erodibility correspond to the coarse textured, permeable (granite derived in most cases), rich in organic matter (in the highlands), while the most erodible ones are silt loams, poor in organic matter due to drier and hotter climate and land use and management practices (typical of the schist areas of the Hesperian Massif but also of the sedimentary carbonate rich areas of the Western Meso-Cenozoic belt) (Ricardo, 1980; Figueiredo, 1990). Rock fragments are a common feature in Portuguese soils and they contribute to visibly reduce potentially high fine earth erodibility (Poesen & Lavee, 1994; Figueiredo, 2001).

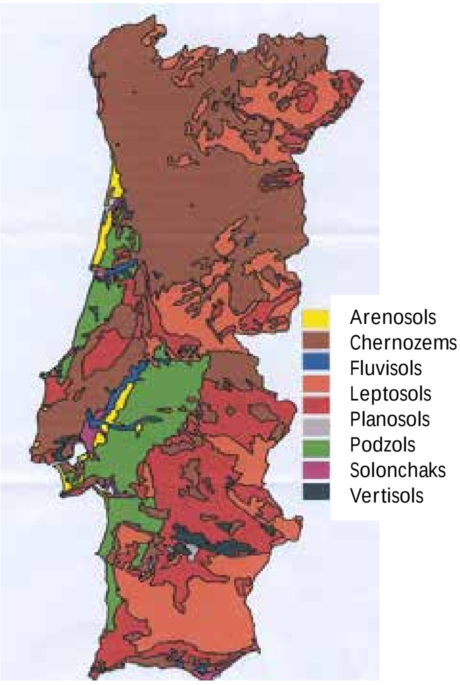

Fig. 3: Soil map of Portugal, 1:1 000 000, FAO/UNESCO legend (CNA, 1980), simplified.

As a Mediterranean country, Portugal depicts the typical natural vegetation communities that characterize this ecological region, and roughly follows the same land use patterns as other countries in the Mediterranean, even though largely contrasting vegetation cover are found in a deeper and more careful insight (Ribeiro, 1986). In 2000, the territory was roughly halved in forest uses (including agro-sylvopastoral and shrubs) and agricultural land use (including pasture land) (Painho & Caerano, 2006.). Native forest cover, based on the oaks, Quercus (Q. pyrenaic and Q. robur in humid and colder areas; Q. suber and Q. rotundifolia in the hotter and drier areas) is poorly represented and so, most forest land are planted stands, where Pinus pinaster and Eucalyptus globulus are the dominant. Even though forests provide a very effective soil protection, hazardous forest management practices are common (installation and, timbering operations, infrequent clearance inducing fire hazard, for instance), which turn forest land into important sediment sources instead of soil retention areas (Agroconsultores e Coba, 1991; Coelho 2006). On the other hand, the key Mediterranean crops are also present in large extent in Portugal – winter cereals, olive groves, vineyards – to which must be added mixed farming systems, where cattle, grazing areas, annual and permanent crops are arranged under a single farm yet split in parcels, as it is the case of large areas in the North of Portugal. Scrubland, included in the forest land use major type is very significantly represented in continental Portugal, the latest assessments indicating about 36 % of the total area (CIMO, 2009). Land cover is changing rapidly in the last decades, as in the last decade of 20th century agricultural land use proportion decreased and forest land use increase (less 84 thousand ha and more 12 thousand ha, respectively), a trend that persisted during the first decade of the 21st century (yet not quantified) (Painho & Caetano, 2006; see Fig. 1 for the 2007 assessment). Furthermore, abandonment and intensification go side by side, eventually in different environments, as shown by the increase in scrubland areas as well as that of new permanent crop plantations, namely olive groves. Signs are contradictory as far as soil protection is concerned, because the decrease in the less protecting areas as that of agricultural land uses is followed by more intensive cropping systems and, on the other, the global increase in forest land use means an increase in scrubland, invading former cropland and grazing areas, now non-managed land tracts where social control does not prevent natural or human induced hazards (as wildfires).

The indicated ranking of land use types according to the protection of soils they may provide bears on the assumption that regular management practices are associated with such land use types, at the farm or small rural community scales. However, soil and crop management practices play a decisive role in the actual soil degradation trends or soil protection status a certain land use type endures (Morgan, 2005). Furthermore, when wrong management practices are applied cropping systems over unsuitable soils, soil degradation is magnified (e.g. Agroconsultores e Coba, 1991) and an extreme example exists to prove it in the Portuguese recent history that was labeled as the wheat campaign (Coelho, 2006). The rapid increase of cereal production promoted in the 30's and 40's of last century, under a policy based on the self-sufficiency paradigm, lead to extended shrub replacement by cereal fields, in unsuitable sloping land, with disastrous consequences for land productivity and soil erosion, which still persist in the Alentejo inland, where degraded soils did not recover yet, while scrubland invaded back these areas.

Actual European agricultural policies their translation in national regulations accommodate a series of programmes directly or indirectly addressing runoff erosion control (as the package of agri-environmetal measures). Good management practices are promoted through financial incentives provide under cross-compliance schemes, focused on pasture land, grassed lanes in permanent crops, conservation tillage practices, integrated production and organic farming (INGA webpage). As well, incentives were given to convert abandoned farmland into afforested areas that are continued for long time after installation, to ensure application of good forest management practices persist during stands life, as it is the case of clearance and other sylvicultural procedures (IFADAP webpage,). Yet, not only adequate control on the application of such measures may lack but also a part of the territory falls out of the range of these measures. Furthermore, in cases policies do not encompass the fast land use changes and perspectives the Portuguese territory is experiencing. Actual major threats to soil resource are those associated with the hydrological consequences of wildfires in marginal lands, while in cropland, traditional intensive tillage operations cause structural degradation of soils, decline in organic carbon soil storage and increase runoff erosion risk (Coelho, 2006; Carvalho, 2012, this issue).

3. SOIL DEGRADATION ASSESSMENTS

This section summarizes information regarding soil degradation assessments, namely due to runoff erosion, performed for continental Portugal, most of which reported by Coelho (2006) in her review on the topic. Therefore, it follows closely this reference, but it adds also new information issued from more recent works. It addresses experimental results and maps, following presented, briefly commented; consistency of assessments and their interpretation is also shortly commented.

The Vale Formoso, Alentejo, erosion experimental centre has the longest record of soil los in Portugal (three decades), assessed in reference Wischmeier erosion plots, under traditional and alternative crop rotations, typical of the southern rainfed cereal based cropping systems (Roxo, 1994; Tomás, 1997). Average soil loss on plots cultivated with winter cereal did not exceed 2 t ha-1 y-1. A set of plots were also installed in another very important agricultural region: the Douro valley, with vineyards in very steep slopes and stony soils (Rosa, 1981). Records were deeply explored for the first ten years after plot installation and average soil loss did not exceed 0.5 t ha-1 y-1 (Figueiredo & Ferreira, 1993; Figueiredo et al. 1998; Figueiredo, 2001). These are the longer and most significant monitoring stations and they were set under a national programme for assessing erosion rates and erosion control measures efficacy in the main agricultural land use types, following evidences throughout the country of large areas were under soil degradation process or already degraded.

Apart from these, most of the field studies that provided data on soil loss rates are short duration experiments carried out within research project activities. They include field plots, normally small scale ones, gully erosion measurements, field surveys and field rainfall simulations, the latter being largely dominant. As expected, experimental conditions very much affect results obtained, but they allow interpretations on the erosional response of different land uses (crops, forests, shrubs, burnt areas). Coelho (2006) discusses these data, concerning the team from the University of Aveiro, mainly focused on wildfire effects in forests (authors involved are Coelho, Ferreira, Shakesby and Walsh in work indicated in the references list), and the team of Poesen, including the author of this text (Poesen, Vandaele, Vandekerkhove I references list), mainly focused in gully erosionin in arable land. Further research performed in Portugal includes more recent studies by Figueiredo et al. (2011). Bompastor et al. (2009) and Fleskens & Strosnider (2007), focused in young forest plantations, shrubland and olive groves, respectively.

All field studies considered, a consistent trend in results is that soil loss rates are normally below expected for the prevailing site erosion factors. However, very much higher soil loss rates are recorded under intense and less frequent rainfall events, pointing out the importance of thresholds in erosion processes. They also point out the need of further research, still lacking in certain land cover types, and to a more careful attention to forest areas in what concerns soil protection.

Maps at national scale show large tracts under severe risk, namely the inland belt from North to South, yet much wider in Alentejo. This risk regards directly or indirectly runoff erosion as some of the national assessments focuses on soils, climate, vegetation and desertification. Besides such surveys, model outcomes as that of PESERA (Kirkby et al, 2004) provide also a picture of the potential risk of the territory. Generally, map interpretations match with interpretation developed in the previous section, based on the natural and land use conditions. Figs. 3 to 6 support and illustrate these general comments.

Indications provided by maps are mainly qualitative but, nevertheless, they normally do not match with the experimental quantitative approaches. Besides methodological inconsistencies that might be found when comparing the different approaches followed to study the same problem, scale is critical issue in erosion research. Furthermore, as complex processes are involved, with dynamic thresholds, erosional response to certain experimental conditions may fall well apart from expectable. On the other hand, most data basis that feed information to built up maps neglect a crucial soil feature that largely controls erosion rates and that is widely represented in the Portuguese soils, namely in the areas where erosion risk is more severe: the coarse fraction of soils, composed by rock fragments, roughly qualified as soil stoniness. This feature is sometimes also neglected when discussing experimental results. It is important to note that, as rock fragment increase in soil surface due to selective erosion of fines, therefore indicating that soil is under threat, the lower can be the soil loss rates measured due to the increasingly higher soil protection they provide.

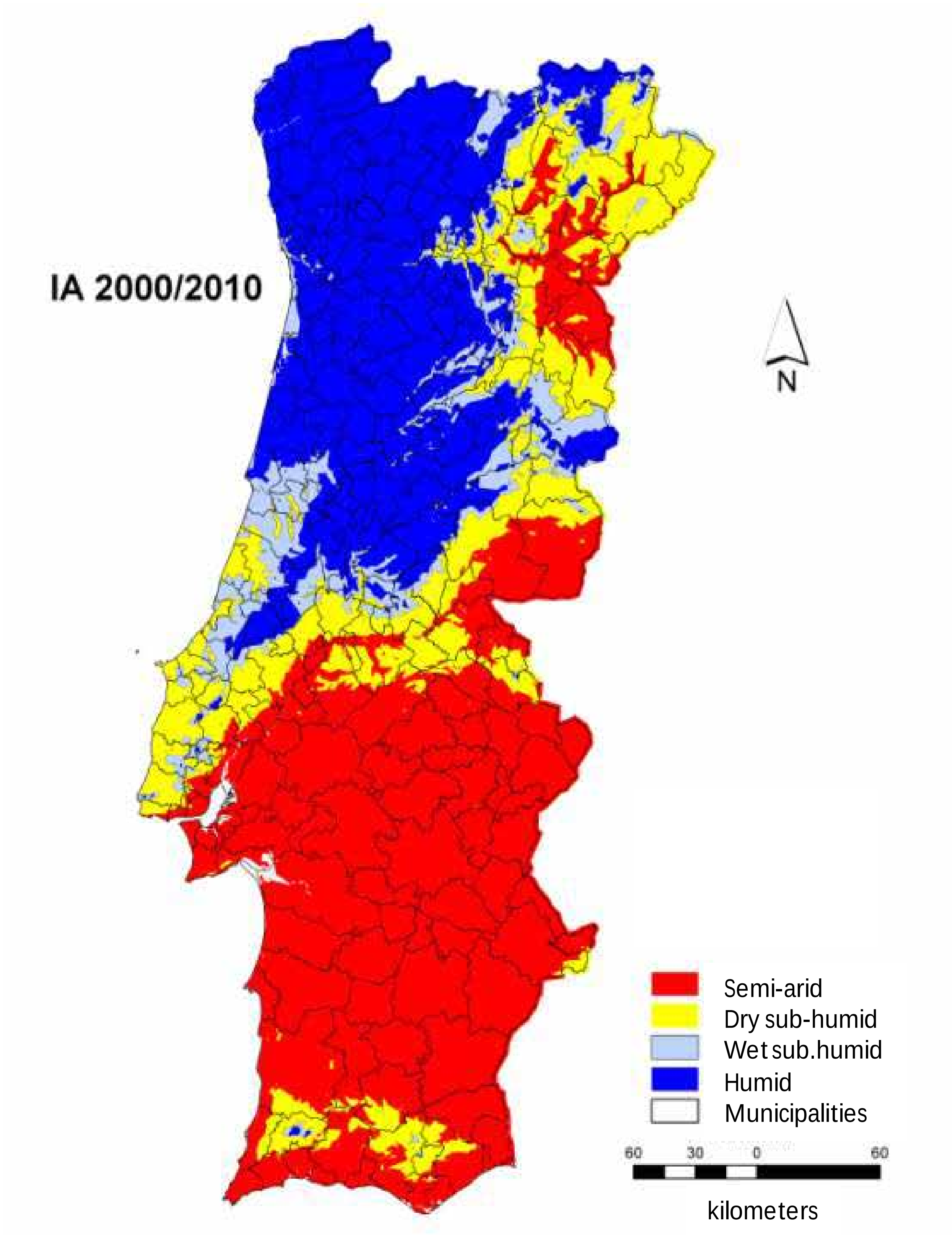

Fig. 4: Desertification susceptibility, assessed by the Aridity Index (IA) for Continental Portugal, 2000/2010 (PFNCNUCD, 2011).

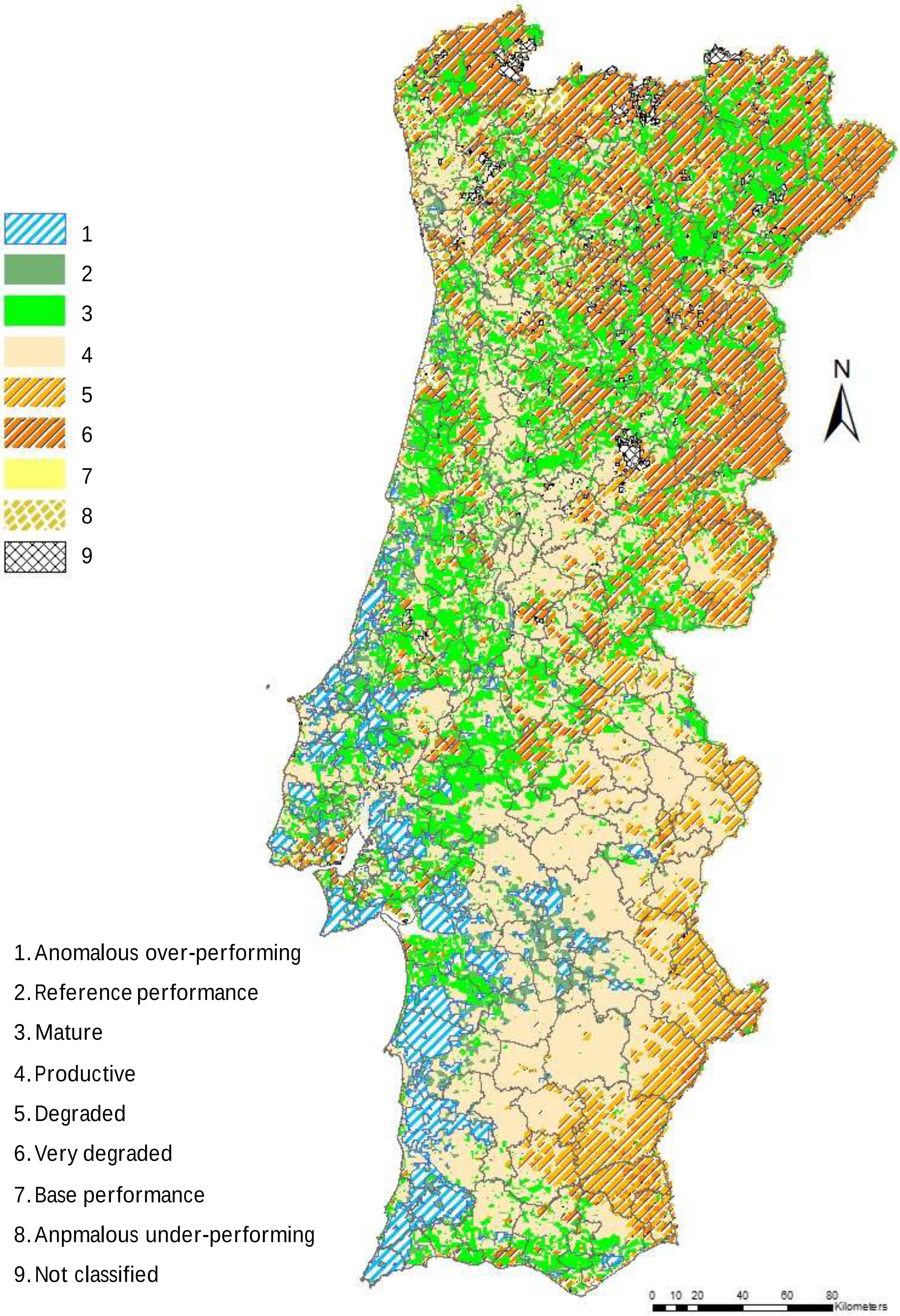

Fig. 5: Land Degradation Index (LDI) – Land quality status, for 2000/2010, in Continental Portugal, following Sanjuan et al., 2011 (synthesys by Rosário, L. do, National Focal Point of the UNCCD).

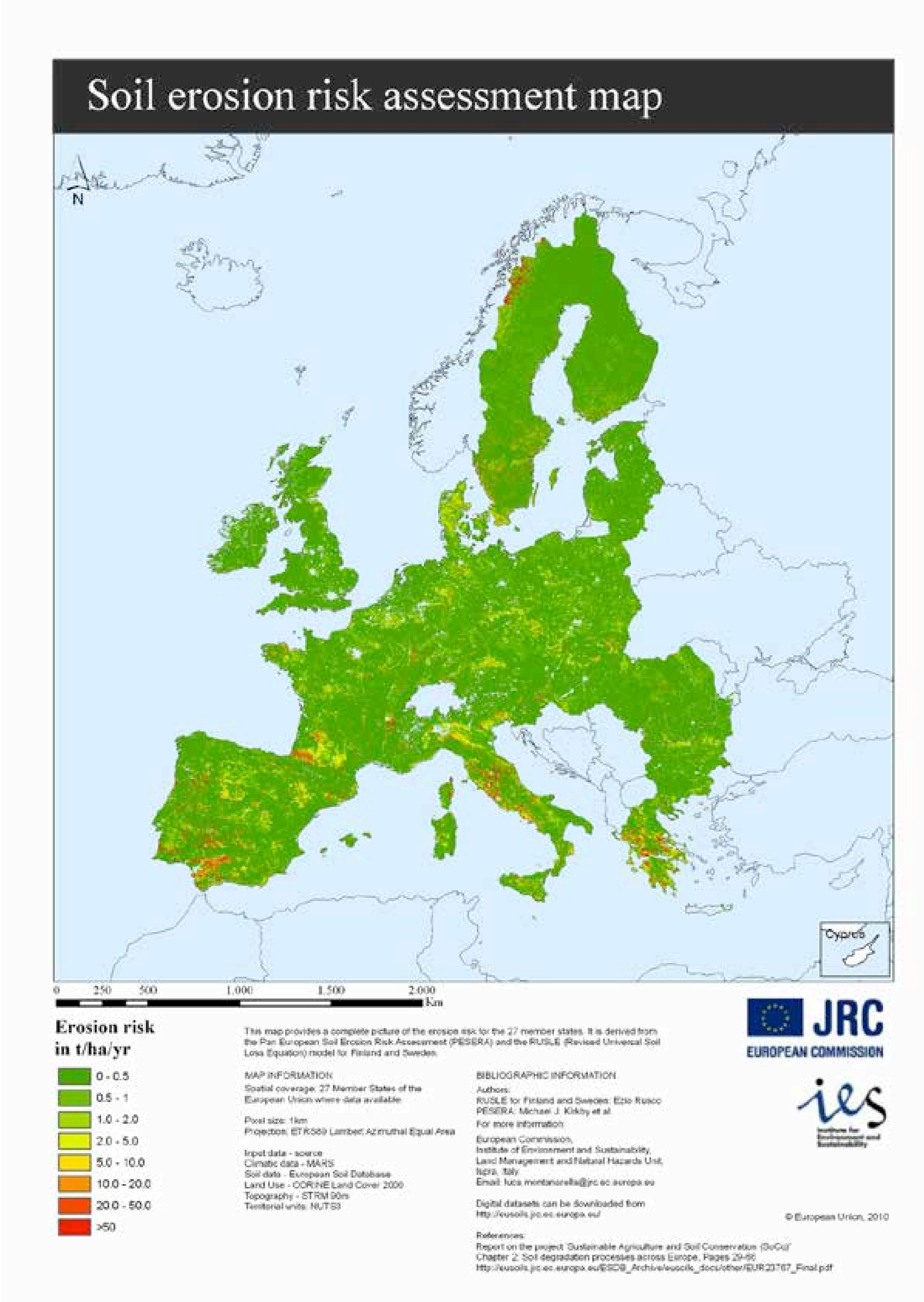

Fig. 6: The PESERA map of erosion risk for Europe (Kirkby et al., 2004).

4. FINAL REMARK

The Portuguese continental territory is highly diverse in what concerns physiography, climate and soils but, may generally be considered prone to high runoff erosion risk under the prevailing natural conditions. Land use and management, again quite variable along the country, and besides the good examples of soil protection in practiced in certain areas and cropping systems, do not help controlling that risk. Although trends of change towards climatic aridity eventually foreseen, and trends towards a new land use paradigm present scenarios that might enlarge the areas under risk, increasing awareness by farmers, land users and the global community are expected to help improving land management practices and models for a sustainable resource use, namely soil and water. This awareness should be translated into policies and regulatory actions, in a comprehensive Soil Act at national level, within the framework of the much expected European Soil Directive, still lacking consensus.

5. THE SELECTED CASE STUDIES

In spite of the general approach followed in the book, namely a focus on runoff erosion, the Portuguese case studies selected also address to a larger or shorter extent management practices following soil conservation principles. In fact, it is essential that, once identified soil degradation, solutions issued from well grounded research contribute to tackle the problem and point towards sustainable resources management and conservation within the framework of actual productive activities carried out in a living territory. As so, rather than a simple focus on runoff erosion research outcomes, article presented hereafter include results of research oriented towards sustainable management of farm land and afforested areas, that may taken as examples to better ground recommendations for practical application or policy design.

The four case studies cover distinct geographical settings (two in the North and two in the South of Portugal), and cover agricultural (3) as well as forest (1) land uses. The Douro valley vineyards are an example of the risks associated to permanent row crops cultivated in the extremes of such regional setting, and of the way a deep knowledge of local conditions provides solutions for tackling those risks. The largely negative experience of of adopting inadequate practices and models in cereal cropping throughout the country but especially in the South, fully justifies the attention given to this type of land use in two case studies, where good management farming practices adopted in conservation agriculture, clearly show how wrongly managed were past but still prevailing farming systems. The last case study addresses the topic from the side of recovery of degraded land by intense cereal cropping, performed within an active farm, that is is a known case study of sustainable farming. Forest cover provides a most efficient way to control erosion but, under Mediterranean conditions, suitable site preparation is required to improve soil quality, therefore, ensuring success in installation and growth of new forest stands, a topic addressed to in the second case study selected, that not only highlights the importance this so widely represented land use type but also addresses a crucial moment of forest stands life – the installation phase. The set of selected case studies is intended to contribute to a reliable picture of research carried out in soil protection in Portugal.

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CASE STUDY 8: EXTRACTION OF BIOMASS FROM FOREST SOILS - THE MAIN ASPECTS TO TAKE INTO ACCOUNT TO PREVENT SOIL DEGRADATION

Agustín Merino

Abstract

The use of logging residues after the clear cutting of a forest plantation to provide energy is being considered in different regions where forestry prevails. This article provides a brief summary of the most important aspects that should be taken into account to avoid soil degradation in forest plantations where intensive cuttings are carried out. The different processes such as compaction, erosion, loss of nutrients or changes in organic matter are illustrated with results obtained by a research group in northern Spain. The overall aim of these studies is to aid in the sustainable exploitation of this potential resource.

Key words: biomass; soil organic matter; erosion; nutrients.

1. INRODUCTION

The use of low grade timber as a source of energy or raw material for the timber transformation industry is currently being considered in different regions of Spain and other parts of Europe. In addition to the establishment of energy plantations, another possibility is exploitation of the logging residues that remain in plantations after final cutting. This practice could increase the profitability of forest land, and given that in many cases the presence of logging residues increases the risk of fire and spread of disease, it may also help in management of forest stands.

The increasing interest in bioenergy, along with other environmental aspects (accumulation of C, nutrient cycling) has led to the development of allometric equations for the quantification of arboreal biomass (for example; BALBOA, 2005; MONTERO et al., 2006; BRAVO et al., 2007). Such equations, which relate the weight of each arboreal fraction to tree variables that are easy to measure in the field, have enabled quantification of the arboreal biomass in some regions and thus estimation of the potential of this resource as a source of energy.

However, before logging residues are removed, their function in protecting the soil should be considered. Many tree stands are established on steep slopes, on nutrient-poor soil, and therefore the removal of biomass may favour erosion and deterioration of the nutritional status of the forest systems. Indiscriminate exploitation may affect the soil organic matter content. Information about the availability of the resource and data that will enable the elaboration of criteria to ensure sustainable exploitation of this resource are clearly required. Furthermore, with appropriate planning this may also provide an opportunity to obtain some environmental benefits, such as C capture.

According to the *Instituto para la Diversificación y el Ahorro de* Energía, *IDEA*, (Institute for the Diversification and Storage of Energy), the conditions in the north of the Iberian Peninsula, in particular in Galicia, are optimal for the generation of energy from forest biomass. However, some limitations, such as steep slopes and nutritional deficiencies make good planning necessary to ensure sustainable exploitation of this resource. Some measures that may contribute to this objective are discussed, in light of the results of different studies carried out in the region.

Soil compaction

Logging residues together with the humus layer provide an effective buffer against the weight of heavy machinery used in felling operations and in the preparation of the land for the next rotation. Removal of the remains therefore exposes the soil directly to the weight of the machinery. Under such conditions the soils, especially fine-texture soils are easily compacted. In addition, the impact of raindrops on the unprotected soil favours the development of surface crusts on the soil (MWENDERA and REYEN, 1994).

The data obtained in plantations in northern Spain show increases in the apparent density of up to 1.5 g cm^{-3} in land where arboreal biomass has been extracted and heavy machinery used (Figure 1; MERINO et al., 1998). The reduction in the pore space produced in these soils is sufficient to prevent root elongation (FROEHLICH et al., 1986; SKINNER et al., 1989), thereby affecting growth of the next rotation and also delaying the establishment of scrub and herbaceous vegetation (Figure 2), in addition to affecting run-off and erosion, which will be discussed further below. It is important to emphasize that recovery of such soils is a slow process (Figure 1), due to the difficulty experienced by the scrub plants to become established in compacted soil (RAB, 1996; FROEHLICH et al., 1986; EDESO et al., 1999).

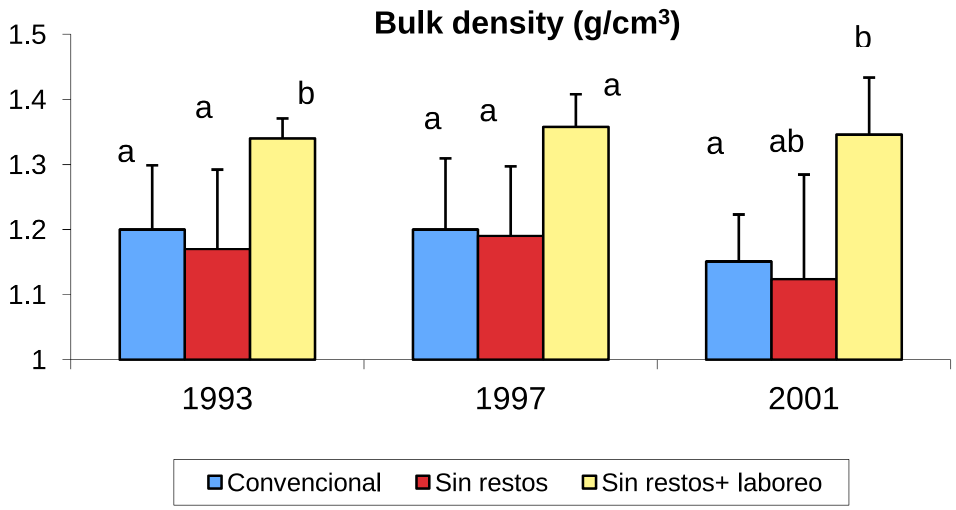

Fig. 1: In clay soils under moist conditions, the removal of all of the arboreal biomass and the introduction of heavy machinery may lead to significant compaction of the soil. Recovery of soil porosity is a slow process (Merino et al., 2004).

Fig. 2: Soil compaction resulting from the introduction of heavy machinery and the impact of raindrops limit growth of forest plantations.

Soil erosion and water quality

Soil erosion is of special interest in steeply sloping forest land, where shallow soils may be considered as a non-renewable resource. Erosion affects the forest system through loss of depth and fertility of the soil, which have repercussions on productivity. This process also affects aquatic systems linked to forest catchments, by reducing regulation of the water flow and increasing the turbidity and concentration of solutes in the water.

Although forest systems offer the best protection against soil erosion, in intensive forest exploitations, soil loss greatly depends on the practices carried out after felling. If the logging residues and the humus layer are left on site, the run-off will only increase slightly and the rate of erosion will be low. If on the contrary, large amounts of the remains are removed from steeply sloping areas, the degree of erosion may be significant (MILLER et al., 1988). In this respect, different studies carried out in Spain show that the total removal of forest biomass, as a result of fire (BENITO et al., 1991; FERNÁNDEZ et al., 2006) or intensive land preparation (OLARIETA et al., 1999; EDESO et al. 1999; FERNÁNDEZ et al., 2004) causes large increases in run-off and erosion. The effect of the indiscriminate extraction of biomass on erosion is particularly important in clay soils, in which low infiltration rates favour run-off (Figure 3).

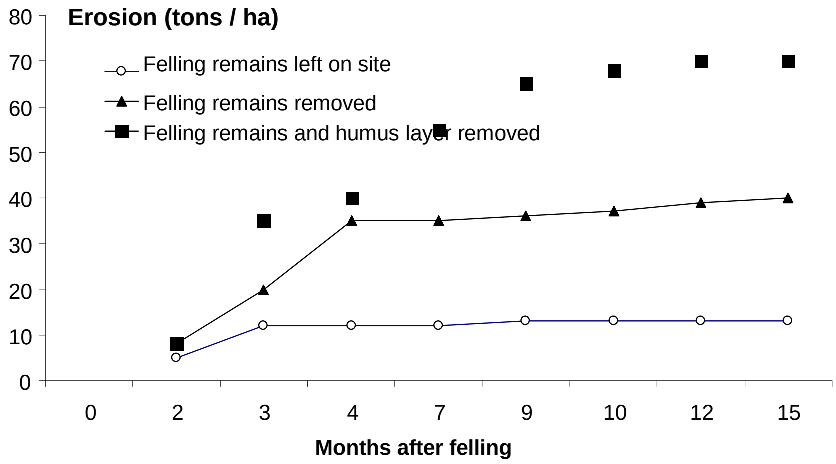

Fig. 3: If logging residues are left on site, low rates of erosion are maintained, even after clear cutting. Rates of erosion may be high in steeply sloping land where logging residues are removed (Edeso et al., 1999).

Clearly, run-off and erosion will increase with the amounts of logging residues removed. One way to avoid this in steeply sloping areas is to remove only the thickest branches. If the logging residues are left to dry on the land before being removed, a large part of the small branches and the leaves/needles will remain (Figure 4). In this type of exploitation, disturbance of the humus layer by heavy machinery will be minimal. A study in which selective removal of thick branches was carried out in an area of slope 35 % (BALBOA, 2005) showed that this practice slightly increased the generation of surface run-off (obviously the removal of the material reduced the water capture), without increasing erosion (Figure 5).

Fig. 4: If the biomass is removed some time later, when the material is dry, most of the leaves/needles remain on site and protect the soil from erosion.

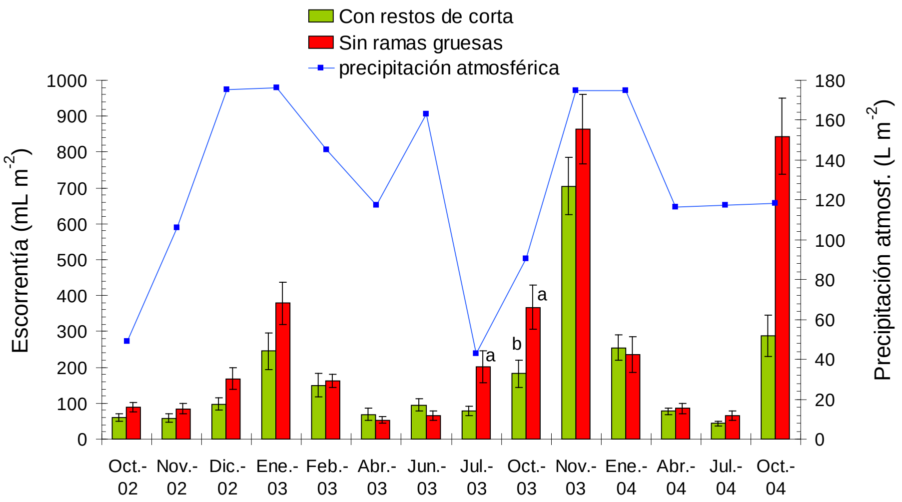

Fig. 5: In stands situated on moderate slopes, the selective removal of the thickest branches to leave the finest branches and leaves/needles leads to a relatively small increase in run-off. Note that the run-off is low in both plots, as the humus layer is retained in both (Balboa, 2005).

Soil fertility

The tree fractions that are richest in nutrients are the fine branches, leaves and in the case of eucalyptus, the bark. This means that, despite the lower weight with respect to the total biomass (in intensive plantations in northern Spain, between 15 and 35 %; BALBOA, 2005; MONTERO et al., 2006), the logging residues accumulate a large portion of the nutrients that the stand has assimilated during the rotation (FISHER and BINKLEY, 2000). If this material is deposited on the soil, the nutrients released during its decomposition may be available to the next rotation. Removal of the material, on the other hand, implies the extraction of a substantial amount of nutrients from the system.

In fact, exploitation of plantations may involve the extraction of P, K, Ca and Mg in quantities comparable to the soil reserves. The increase in the amounts of nutrients removed from the system in some cases may affect the nutrient status of the system (FÖLSTER and KHANNA, 1997; OLSSON et al., 2000; DAMBRINE et al., 2000; MERINO et al., 2005). This is especially important in northern Spain, where intensive forest plantations are often deficient in nutrients, particularly P, Mg and Ca (ROMANYÀ and VALLEJO, 1996; SÁNCHEZ-RODRÍGUEZ et al., 2002; MERINO et al. 2003).

Intensive exploitation of eucalyptus is clearly unfavourable in this respect. In plantations of *Eucalyptus globulus* in the north of Spain, conventional exploitation of these trees (for timber and bark) involves the extraction, every 18 years, of more than 80 % of the quantities available in the soil (MERINO et al., 2005). More Ca is extracted from these systems than the amounts supplied by natural processes (rainfall, mineral weathering) (Fig. 5). In the case of pine plantations, the situation is not as serious, although the amounts of some nutrients extracted are close to the amounts that the system can replace via natural processes (MERINO et al., 2005; RODRÍGUEZ-SOALLEIRO et al., 2007).

Phosphorus deserves special mention for several reasons. In acid and alkaline soils it is one of the most limiting elements for plant production given that the most readily available forms of P (HPO_4^{-2} and $H_2PO_4^-$) only occur within pH 4.8 to 6.2. Furthermore, soil systems have a low capacity to replace this element, given the extremely low quantities provided by mineral weathering and atmospheric precipitation. The main means of replacement is therefore internal, via the decomposition of plant remains. In this respect, most of the P extracted by the exploitation involves a net loss of the element.

However, the deficiencies caused by extraction of nutrients can be overcome by the application of a suitable fertilizer. This need is recognised in the two systems of Certification of Sustainable Forest Management (PEFC and FSC). Fertilization is easily carried out in plantations designed for mechanised labour, i.e. with low or moderate slopes and with pathways to allow movement of vehicles. In plantations not designed in this way, the application of fertilizers is more complicated because in many sites the topography does not allow the use of heavy machinery. Nonetheless it must be pointed out that the amount of fertilizers required to replace nutrients is much less than required for agricultural soils, and many forest managers opt to use light machinery or manual application for distributing mineral fertilizers.

One alternative to conventional fertilizers is the use of certain waste products that contain low levels of heavy metals or toxic substances. In stands developed on acid soils, ash generated from biomass plants may be used as this product is rich in K, Mg, Ca and P, supply of which favours growth of forest stands (Figure 6, SOLLA-GULLÓN et al., 2006; OMIL, 2007). An example of another waste product that can be used in these plantations is the sludge generated by the waste water treatment plants used in the dairy industry, and characterized by extremely low levels of heavy metals (OMIL et al., 2007). Input of organic materials of this type not only contributes to the replacement of nutrients, but also to maintaining the levels of organic matter, as will be discussed further below.

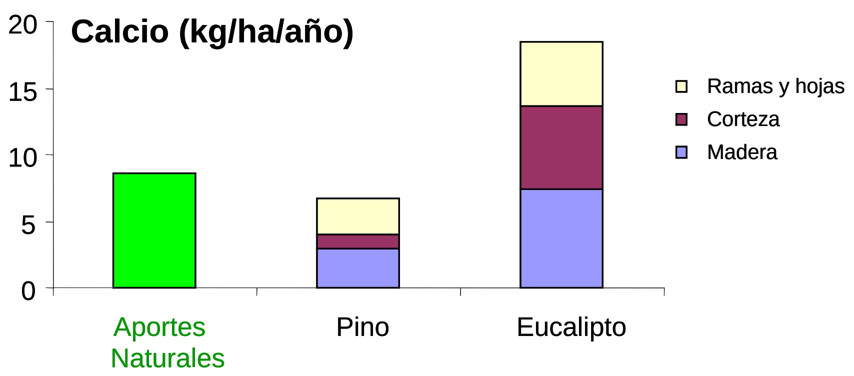

Fig. 6: The removal of nutrients by exploitation of rapidly growing species may exceed the inputs by natural means, such as weathering of minerals or atmospheric inputs. In the case of eucalyptus, the large accumulation of Ca in tree parts, especially the bark, should also be taken into account (Adapted from Merino et al., 2005). * Data from Dambrine et al. (2000).

Soil moisture content and temperature

The layer of logging residues and the humus layer together exert a mulching effect, thereby maintaining the soil moisture and buffering the temperature of the soil. This combined layer reduces losses by evaporation in two ways. On one hand it reduces the growth of accompanying vegetation and thus the loss of water by transpiration. On the other hand, it forms a highly porous layer with a low capacity for capillary rise. Land where logging residues are removed tends to be less moist and the drought period is prolonged (Figure 7; PÉREZ-BATALLÓN et al., 2001). In soils in zones with hydric limitations, the selective removal of logging residues may reduce these negative effects.

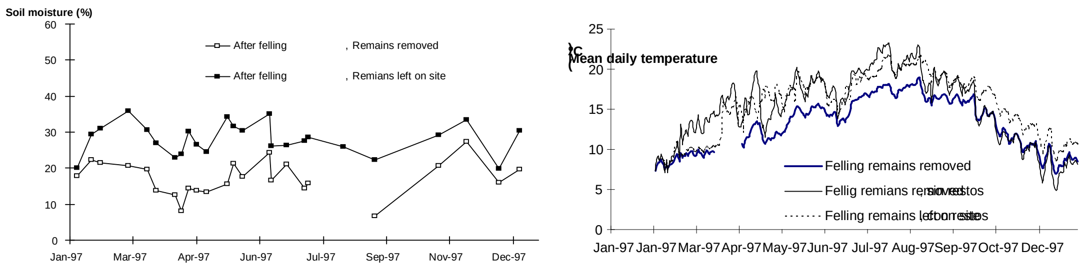

Fig. 7 The layer of logging residues forms a mulch that is effective at preventing evaporation and increased temperatures in the uppermost layers of the soil. When the logging residues are removed the drought period is prolonged and the soil temperature increases (Pérez-Batallón et al., 2001), with the consequent negative effects on survival of the next rotation.

Likewise, the daily temperature fluctuations in the soil increase considerably after felling (due to increased direct radiation reaching the soil and loss of heat during the night). However, the logging residues are effective buffers of thermal fluctuations (Figure 8) and increase the survival of the new plantation. Again, the selective removal of logging residues buffers these effects.

Organic matter content

Logging residues accumulate 20-35 % of the carbon content of a tree, and therefore they can contribute to maintaining the levels of organic matter in the soil.

This is especially significant because the soil organic matter is not only the most important short-term reserve of some nutrients, but it is also the soil component that is most involved in maintaining plant water reserves and in protection against soil erosion.

It must also be considered that increases in soil temperature following felling tend to favour soil microbial activity, although this effect also depends on soil moisture. In stands where the logging residues are removed, the lower inputs of plant remains along with the higher soil microbial activity will lead to a gradual loss of soil organic matter in the intermediate term. In fact, some studies have shown a reduction in the soil organic matter content in intensively managed plantations (TURNER and LAMBERT, 2000). The loss of soil organic matter as a result of the repeated removal of biomass is not consistent with the Kyoto Protocol, in which compensation by carbon capture in soils is proposed.

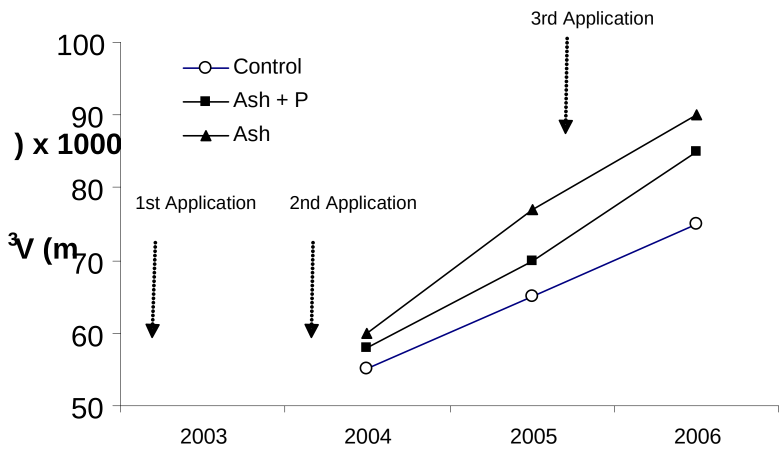

Fig. 8 Forest plantations destined for biomass production can be fertilized with ash generated in biomass plants, which is rich in nutrients. This practice not only improves the yield of the stand and its nutritional status, but also contributes to environmentally sound management of the waste product. The data in the graph correspond to a Pinus radiata plantation to which ash from bioenergy plants was added at repeated intervals, at doses of 4.5 t/ha (Omil, 2007).

This aspect may not only affect the organic matter content, but also the diversity of soil microorganisms, which may also affect certain inherent biological processes. Amongst these, forest soils play an important role as consumers of atmospheric CH_4 (MOSIER et al., 1991), whose potential is considerably reduced as organic matter is lost (STEUDLER et al., 1989).

The input of ash or other clean organic waste products may partly mitigate the loss of soil organic matter, not only through the direct input of organic C (which in the case of ash is in the form of carbon and therefore much more resistant to decomposition) but also through the positive effects on plant production, thereby increasing the amounts of leaf litter and fine roots (SANTALLA, thesis in prep.).

As regards C capture, one important aspect is the establishment of forest plantations destined for timber production, energy purposes or both, in marginal agricultural soils. In these situations, C is captured in both the biomass and in the soil, which may be particularly important in dense plantations. This leads to an improvement in the conditions of these soils, which usually contain low levels of organic matter. In this respect, the benefits have been estimated as approximately 0.1-0.5 tm C/ha/year (ROMANYÀ et al., 2000; PAUL et al., 2002; PÉREZ-CRUZADO et al., 2007). However, this may also contribute to climate change (LISKI et al., 2002; BRAVO et al., 2007). In this respect, recent data show that forest systems in Europe are currently storing between 9 and 12% of the anthropogenic emissions of CO_2, whereas storage of C in soils currently involves 3% of these emissions (LISKI et al., 2002). The increase in forested area also contributes to recovering the capacity of soils to consume atmospheric CH_4 (Figure 9).

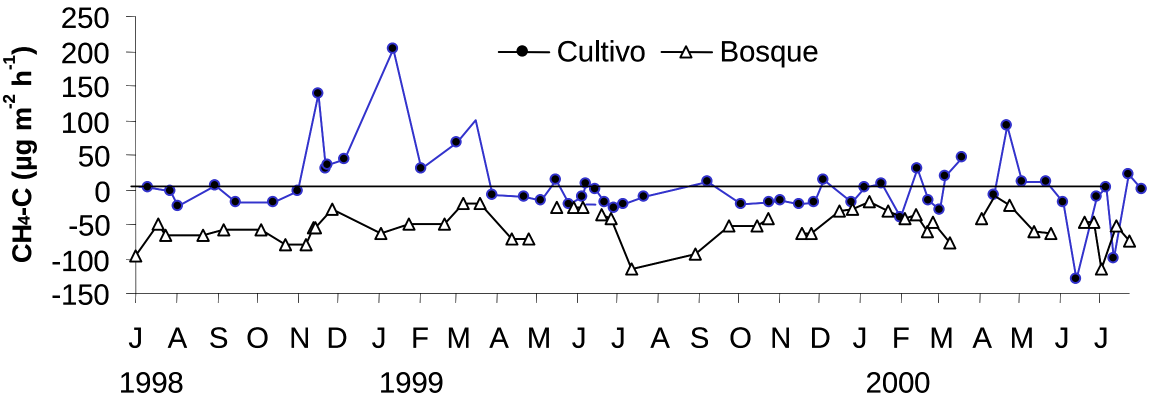

Fig.9. Agricultural soils have lost their ability to absorb CH4. Reforestation of agricultural land may result in soils recovering this capacity, which helps maintain atmospheric CH_4 in equilibrium. The figure shows the CH_4 flows in an agricultural soil compared with those in another adjacent soil reforested 40 years earlier (Merino et al., 2004).

Conclusions: The need to plan silvicultural practices to prevent damaging effects in soils

The continual extraction of biomass in forest systems may cause different types of soil degradation such as compaction, erosion, loss of nutrients and loss of organic matter. Most of these problems can be prevented or reduced by prior planning that identifies forest land that is suitable for this practice. In addition, prior assessment of the possible effects on soils and water are essential, along with analysis of the most appropriate silvicultural practices. If these aspects are considered carefully, biomass exploitation may also provide an opportunity to obtain a series of environmental benefits, such as the generation of renewable energy and the recapture of C in biomass and soils.

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