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# Soil Erosion Issues in Agriculture

*Edited by Danilo Godone and Silvia Stanchi*

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# Meet the editors

Danilo Godone holds a PhD in "Agriculture, Forest and Food Sciences", doctorate's topic was cryosphere's phenomena monitoring by geomatic methodologies. Currently he is a PostDoc grant holder, at Turin University, studying geomatic contribution in land management and analysis. His main research interests are, however, landslide, glacier and, more generally, natural disasters monitoring. During his activities he has developed skills in GIS, also by developing customised tools by Visual Basic programming, and land surveying with GPS or Laser Scanners. He is a member of NATRISK - Research Centre on Natural Risks in Mountain and Hilly Environments, in the same University. He acts as a freelance consultant, in the same topics, for other research bodies, training agencies and professionals, too.

Dr. Silvia Stanchi holds a PhD in "Agriculture, Forest and Food Sciences", with a focus on fractal theory approaches for the study of soil aggregation and structure. Currently, she is a PostDoc at Turin University working on soil hazards and vulnerability in mountain areas. Silvia Stanchi is a member of NATRISK - Research Centre on Natural Risks in Mountain and Hilly Environments, in the same University. She is actively involved in outreach and soil awareness raising activities targeting kids and the large public, and in international training activities focusing on mountain development and management.

# Contents

# Preface

Erosion is a natural process among soil dynamics concerning the movement of soil portions and their deposition in a different location. The intensity of particles removal is variable and leads to different environmental impacts. Soil formation rate is highly slower than any erosion phenomena, this fact suggest that erosion control and mitigation practices should be adopted in order to preserve soil as a crucial environmental resource.

Several natural events as water or atmospheric phenomena trigger erosion processes. Moreover anthropic activities, like inappropriate agricultural practices, deforestation, overgrazing, forest fires and construction activities, may exert a remarkable impact on erosion processes. The lack of appropriate knowledge of agricultural best practices or their disregarding, frequently due to the need of providing food in overpopulated areas, leads to the endangering of soil equilibrium. This aspect, also historically proven, is worsened in developing countries by critical socio-economical conditions and unfavourable climatic conditions. More generally, unsustainable land management policies, not only concerning agricultural sector, are key factors in soil endangering.

Moreover, in the current climate change scenario, weather related variables are increasing their impact on soil erosion; directly, by the amplification of their intensity (severe rainfalls, strong winds…), or indirectly, by worsening environmental conditions (droughts, heat waves…) thus leading to replace spoiled crops by the exploitation of marginal areas or forcing large human settlements to move towards more hospitable areas and available resources; in the worst case this migrations aim to settle in natural areas in order to acquire more space for moving population and for increasing cultivable land surface. This practice causes the reduction of natural areas such as forests, shrubland and consequently the simplification of landscape structure; moreover it influences water cycle and availability in the colonized area.

Soil erosion is a continuous process in Earth cycle and, if not properly faced, may conduct to extreme environmental consequences, like soil degradation o soil loss, threatening human activities and safety. In mountain and hillside areas soil erosion is an instability factor which may cause slope failures that can put in danger human settlements and infrastructures, from the agricultural point of view, extreme soil erosion may reduce nutrient availability thus reducing crop yield and causing land abandonment. Excessively impoverished soil, instead of evolving in other vegetation covers, could be involved in desertification processes.

This book, in its fourteen chapters, deals with several aspects of soil erosion, focusing on its connection with the agricultural world. Chapters' topics are various, ranging from irrigation practices to soil nutrient, land use changes or tillage methodologies.

The book is subdivided into four sections grouping different facets of the topic. In the first one several case studies are presented with the aim of introducing soil erosion issue in the world; in fact, chapters come from India, Spain and China. Each one present soil erosion features in a different geographical and climatic context, and various study approaches. The other three sections focus on a detail among the vast topic.

Section number two covers a typical cultivation, vineyard. An agricultural practice limited to a confined geographical milieu but characterized by a remarkable economic impact. The correct agronomical management of vineyards is a key factor in soil erosion reduction, in hilly environments, allowing, contemporarily, to obtain profitable yields from vines, as quality production is required by market, instead of mass production. The two chapters describe experimental approaches applied to vineyards located in Italy.

The third section theme is a geoclimatic one, since it concerns dry environments and their relationship with soil erosion theme management. The section includes chapters coming from different areas such as Africa (Uganda and Zimbabwe), South America (Mexico) and Europe (Italy) covering Arid, Semiarid and Mediterranean environments.

In the last section the erosion control matter is investigated. Chapters from various countries evaluate erosion control practices like the employment of afforestation to reduce grazing impact or the role of terracing, tillage and irrigation practice in soil erosion control, in cultivated areas.

In conclusion, this book approaches the soil erosion theme, concentrated on agriculture world. Certainly, due to the extent of the subject, the book is not a comprehensive collection of soil erosion studies, but it aims to supply a sound set of scientific works, concerning the topic. It analyzes different facets of the issue, with various methodologies, and offers a wide series of case studies, solutions, practices, or suggestions to properly face soil erosion and, moreover, may provide new ideas and starting points for future researches.

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# Part 1

## Case Studies

# 1

# Soil Degradation

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## 1. Introduction

The rise and fall of ancient civilizations were in direct proportion to the wise use or misuse respectively of the natural resources, in particular, land and water resources. Land is the finite resource which is devoted to the largest primary "Private Industry" namely, "agriculture". But unfortunately land is the least cared; most neglected and misused resource by almost everyone, either knowingly or unknowingly. The results are obvious and for everyone to see in the form of degradation and declining productivity of our resource base. The situation demands everyone's attention and immediate correction.

The pressure on our finite land resources is tremendous at present due to increasing population and competing demands of various land uses. The decreasing land-man ratio and continued dependence of a high proportion of the population on agriculture in developing nations is a matter of grave concern. That is why we are witnessing high rate of unemployment and under employment in the rural areas. Under such circumstances, the extent of unemployment can rise alarmingly unless measures are taken to either increase the intensity of land use or shift a significant proportion of the human resource out of agriculture to non-agricultural activities, or both, and both are not likely to happen that easily in the developing countries.

It is obvious that the population pressure, both human and cattle, and competing demands and needs of the society exerts tremendous pressure on the limited and shrinking land resources like soil, water, forests, vegetation, bio-diversity etc. Due to this pressure, there is severe degradation of the resources and large-scale change in the land use and land cover. Apart from this, particularly of late, there is a significant diversion of farm lands and water resources to non- agricultural purposes, exerting further strain on the already shrinking land and fast-depleting water resources.

Due to the population pressure in most developing nations, the existing forest areas are facing deforestation, cutting beyond the silviculturally permissible limit, unsustainable fuel wood and fodder extraction, shifting cultivation, encroachment into forest lands, forest fires and over grazing. These changes affect drastically the vegetation, rainfall and sedimentation levels in the lakes and reservoirs, hydrological cycle, bio-diversity and ultimately the land use. All such issues are now being widely discussed under the broad heading "global climate change". In spite of concerted efforts by the governments to check deforestation, large forest areas are already degraded and the remaining areas are in various stages of degradation. This has been conclusively proved through number of studies based on satellite imageries taken at regular intervals (Vasisht et al., 2003).

The land use dynamics in cultivated areas is unique and different than that is observed in the forest areas. Factors and processes like introduction of irrigation, topography, climate, floods and droughts, market fluctuations, input-output levels, price fluctuations, government policies, lifestyle changes and severity of degradation etc affect the types of crops grown, system of cultivation and input management in different regions of the globe (Lal & Stewart 1990).

Fig. 1. Forest fires have the devastating effect on all the elements of ecosystem including soil

Most of the times, the choice of the crops is not based on the suitability of the area, resulting in either over exploitation or under utilization of the soil, water and other resources. This leads to degradation of the land resources and ultimately to change in the land use pattern itself. A case in hand is the development of severe salinity in the Indo-Gangetic Plains and command areas in the Deccan Plateau of India due to uncontrolled and unscientific irrigation which has changed dramatically the land use of the area from multiple cropping to almost a single crop in about 11 m ha area (Suraj Bhan et al., 2001). If we consider the case of India, the latest estimate indicates that soil erosion, salinity and alkalinity, water logging and declining soil fertility has affected about 57 per cent (187.8 m ha) of the land resources in the country, threatening the sustainability of the resource base(Table 1).

Land resource includes soil, water, bio-diversity, climate etc. Soil is the most important component among all, as any effect on it, directly influences and changes other components also. Soil degradation is the loss of actual or potential productivity and utility of soils. It implies a decline in soils inherent capacity to produce economic goods and perform environmental regulating functions (Anthony Young 1998). Among the functions, agricultural productivity and environmental regulatory capacity depend on soil quality and relevant properties. Soil degradation is the temporary or permanent lowering of the productive capacity of land as a result of human actions or non-action. It covers the various forms of soil degradation, including erosion and fertility decline, adverse impacts on water resources, deforestation and forest degradation and lowering of the productive capacity of pastures etc. Loss of biodiversity and human-induced climatic change also has effects, direct and indirect, on productive potential of land resources (Pathak, 2010).

### 1.1 The global extent of soil degradation

Global Assessment of Soil Degradation (GLASOD) was the first attempt to estimate the severity and extent of soil degradation on a world basis (Oldman, 1988). A key feature of this study was that the degrees of severity were defined not in physical terms, as soil loss, nutrient decline, but on the basis of effects upon agricultural production. This was done because it allowed comparison between different types of degradation.

### 1.2 Degree of soil degradation

Four categories to express degree of soil degradation are recognized. The categorization was in terms of agricultural suitability, declined productivity and biotic functions. These are:

1. Slight: The terrain has somewhat reduced agricultural suitability, but is suitable for local farming systems: restoration to full productivity feasible: original biotic functions largely intact.
2. Moderate: The terrain has greatly reduced in agricultural productivity, but is still suitable for local farming systems: needs major improvements to restore productivity; original biotic functions partially destroyed.
3. Strong: the terrain is non-reclaimable (at farm level) and requires major engineering works for terrain restoration: original biotic functions largely destroyed.
4. Extreme: The terrain is irreclaimable and beyond restoration: original biotic functions fully destroyed.

| Degradation type | Area affected (m.ha) | Percent |
|---|---|---|
| Water Erosion | 148.9 | 45.3 |
| Wind Erosion | 13.5 | 4.1 |
| Chemical Deterioration (loss of nutrients, salinisation ) | 13.8 | 4.2 |
| Physical Deterioration (Water logging) | 11.6 | 3.5 |
| Total affected area | 187.7 | 57.0 |
| Land not fit for Agriculture | 18.2 | 5.5 |
| Total Geographical Area | 328.7 | 100.0 |

(Majhi et al 2010)

Table 1. Extent of Land Degradation in India

It is critically important whether a type of degradation is reversible, and if so over how long and at what cost. Earlier approaches to soil degradation, from productivity or agronomic point of view have now evolved to an environmental one in the last decades. The environmental point of view enables other degradation processes to be accepted, not only those processes affecting soils intrinsic characteristics (changes of the physical, chemical, biological soil properties, or agricultural use), but also those processes due to externalities. Universally known degradation processes are soil erosion, compaction, alkalinization, salinization, pollution, acidification, nutrient depletion and organic matter loss (USDA, 1998).

Out of a world land area of 13000 M ha, 4300 M ha are deserts, mountains, rock outcrops, or ice-covered, leaving a balance of 8700 M ha of usable land, meaning land with potential for cultivation, grazing, or forestry. For developing countries, about 1500 M ha or 25%of usable land are affected to some degree by degradation. The percentage degraded is highest in Africa and Asia, and lowest in South and Central America. About half the area of arable land and a quarter that of permanent pastures is degraded. Water erosion is given as the most widespread dominant type of degradation, with 836 M ha in developing countries, followed by wind erosion affecting 456 M ha, soil chemical and physical degradation 241 M ha, and salinization and water logging 836 M ha (Oldman, 1988).

More reliance can be placed on the estimates of strong and extreme degradation. The definitions imply that these refer to land that is largely destroyed, and probably abandoned from agricultural use. Moreover, since they refer to gullies, hillsides stripped off soil, salinized patches, and the like, such degradation is relatively easy to recognize and assess in semi-quantitative terms. The total world area of strongly degraded land is 305 M ha, of which 224 M ha is due to water erosion and 21 M ha to salinization. About 95%of this is in developing countries. The conservative estimates suggest that current loss due to degradation maybe more than 5 M ha per year. The 21 M ha of severely salinized land, probably representing saline patches that have been abandoned, is also largely in the tropics. It amounts to over 10 %of the irrigated area in developing countries and is steadily increasing as investments in soil conservation/reclamation programs are not forthcoming on expected lines.

| | Usable land M ha | All degrees of degradation | | Strong and extreme Degradation | |
|---|---|---|---|---|---|
| | | M ha | % | M ha | % |
| **Africa** | 663 | 494 | 30 | 129 | 8 |
| Asia | 779 | 748 | 27 | 109 | 4 |
| South/Central America | 714 | 306 | 18 | 48 | 3 |
| Developing countries | 656 | 548 | 25 | 286 | 5 |
| Developed countries | 555 | 417 | 16 | 43 | 2 |

(Source: Oldman, 1988).

Table 2. Global Assessment of Soil Degradation (GLASOD)

If it is assumed that most of this loss has taken place over the last 60 years, probably at an accelerating rate, the current loss becomes at least 5 M ha per year, or 0.3% of usable land of developing countries. An area of about 1500 M ha, or 25% of usable land in developing countries, has been affected by soil degradation of some kind, to a degree which appreciably (10 % of land) or greatly (15 % of land) reduces its productivity. About 300 M ha. or 5 % of usable land in developing countries have been so severely degraded, mainly by erosion, that for practical purposes they can be regarded as lost.

## 2. Causes of soil degradation

The causes of soil degradation are made up of natural hazards, direct causes, and underlying causes. Taking soil erosion by water as an example, the natural hazards include steep slopes, impermeable or poorly structured soils, and high intensities of rainfall. The direct causes are unsuitable management practices, such as cultivation without conservation measures, or overgrazing etc. The underlying causes are the reasons why such practices are adopted, such as the cultivation of slopes because the landless poor need food and non-adoption of conservation measures because farmers lack security of tenure (Hassan & Rao, 2001).

Water erosion was attributed more or less equally to deforestation, agricultural activities like the cultivation of land naturally at risk without adequate conservation measures and overgrazing. Wind erosion is primarily due to overgrazing and to a lesser degree, over cutting of vegetation. Soil chemical and physical degradation result primarily from faulty agricultural practices. The deterioration of soil physical properties occurs when farmers try to maintain crop yields by fertilizer use alone, without measures to maintain organic matter (Butterworth et al., 2003).

The direct causes of soil degradation like salinization are due to mismanagement of irrigation schemes and lowering of groundwater through extraction in excess of recharge (Singh et al., 1992). Adverse changes in river flow and sediment load are off-site consequence of forest clearance and erosion. Deforestation is resorted mostly for agricultural use than by felling for timber. Forest degradation is normally due to over cutting for fuel wood, domestic timber and fodder. Selective extraction of the best species in commercial logging is also another major cause for forest land degradation.

### 2.1 Economic and social reasons

Soil degradation need not be viewed as a consequence of failure by farmers to adopt conservation practices, or deforestation. It is only part of the picture and the root of the problem lies in economic and social circumstances. We need to view the situation from a socio-political stance, seeking for changes in the social structure and state polices, programs and developmental interventions, if measures to combat land degradation are to be successful (Greval & Dogra 2002, Srivastava et al., 2002).

Land tenure is rightly seen as a basic obstacle in sustainable management of land resources. It is natural that farmers are reluctant to invest in conservation measures if their future rights to use the land are not secure. Two kinds of property rights lead to this situation, insecure forms of tenancy and open access resources. Tenancy as such is not to blame, provided that there is legal security of tenure. In the 1980s, following a World Conference on Agrarian Reform and Rural Development (WCARRD), there was an impetus on reform of land tenure. Land reform programs were attempted in many countries, with limited success owing to opposition by strong vested interests (FAO, 1988).

Fig. 2. Cultivation on steep slopes without adequate conservation structures: cause for degradation of natural resources

Land shortage, brought about by population explosion has become a fundamental cause of degradation. Once farms are too small to support their children and all the good land is taken up for crop production activities, migration to sloping, semi-arid, or other areas with high natural hazards takes place. Frequently this will require clearance of forest. Soil conservation is normally applied on a participatory basis, through the approach of land husbandry. Forests, which serve the needs of local people for food, fodder and fuel wood, are more likely to be conserved, if responsibility for their management is given to the village or community

### 2.2 Vicious cycle of population, poverty and land degradation

A chain of cause and effect links direct and indirect causes of land degradation. The driving force is an increase in population dependent on limited land resources base. This produces land shortage leading to small farms, low production per person, increasing landlessness and in consequence, poverty. Land shortage and poverty together lead to non-sustainable land management practices, the direct causes of degradation. Poor or landless farmers are led to clear forest, cultivate steep slopes, overgraze village common lands like pastures or make short-term unbalanced fertilizer applications. These non-sustainable management practices lead to land degradation, causing lower productivity and lower responses to inputs. This has the effect of increasing the land shortage, thus completing the cycle.

Only the poor by no means cause land degradation. Irresponsible rich farmers sometimes exploit the land, but by and large farmers with secure tenure and capital are more likely to conserve natural resources. When natural disasters occur, rich farmers can turn to alternative sources of income, or borrow and repay in better years. These alternatives are not open to the poor (Srivastatva et al., 2002).

Fig. 3. Salinization of most fertile black soils due to excessive irrigation in India

In the past, rural populations had access to adequate land to meet their needs. When a disaster occurred, whether of natural origin or war, there were spare resources to fall back upon. They could take new land into cultivation, kill livestock, which fed upon natural pastures or go into forest and extract roots or hunt wildlife. Because of land shortage, these options are no longer available. Farmers are surrounded by other farmland, such common rangeland as exists is often degraded, and over large areas no forest remains. The options open are to work on the farms of others, non-agricultural occupations, enforced migration to the cities or ultimately dependence on famine relief. Many African nations face exactly the same situations even now (Young, 1998).

If we consider the case of India, the limited land area which is equal to only 2.5 per cent of the world's geographical area. It supports approximately 16 per cent of the world's human population and 20 per cent of the world's livestock population. The population of India has already crossed one billion mark and is still growing at the rate of about two per cent. This exponential growth of population (36.1 crores in 1951 to 102.7 crores in 2001) and dependence of more than 60 per cent of the population for their livelihood on agriculture and allied activities exerts tremendous pressure on the limited land resources of the country. At present, the per capita availability of land is only 0.15 ha, which will be further reduced to less than 0.07 ha in 2050 with an expected population of about two billion (Grewal & Dogra, 2001).

Hence the stress on limited land resources is going to increase day by day. Governments need to address the issues with all the seriousness. The link between population, poverty and soil degradation is now widely recognized. FAO reports 'A lack of control over resources, population growth and inequity are all contributing to the degradation of the region's resources. In turn, environmental degradation perpetuates poverty, as the poorest attempt to survive on a diminishing resource base'. Through force of circumstances, it is the poor who take the major role in the causal nexus between land shortage, population increase and land degradation. Thus rapid population growth can exacerbate the mutually reinforcing effects of poverty and environmental damage of which the poor are both victims and agents. Hence, in such nations population control needs to be taken on top priority to protect the natural resources base besides other socio-economic conflicts (FAO, 1988).

## 3. Processes and causes of soil degradation (Table 3)

Two processes lead to the loss of soil's capacity to perform its functions: those that change their physical, chemical and biological properties (intrinsic processes) and those that prevent their use by other causes (extrinsic processes)(Antony Young 1998).

| |
|---|
| **Intrinsic processes** |
| *Degradation of the Physical fertility* |
| Compaction |
| Crusting |
| Structural degradation |
| Soil loss: Water and Wind erosion |
| Mining |
| Urbanization of agricultural lands |
| Land movements by civil engineering for infrastructure projects |
| Excess water/waterlogging |
| *Degradation of Chemical fertility* |
| Loss of nutrient: Leaching |
| Extraction by plants (nutrient mining) |
| Run off loss of nutrients |
| Immobilization of nutrients |
| Acidification |
| Salinization |
| Sodification; alkalinization |
| Pollution |
| Degradation of biological fertility |
| Loss of organic matter |
| **Extrinsic processes** |
| Loss of accessibility: damage of roads etc. |
| Conversion to risk areas: Natural disasters etc. |
| Climate fluctuations |
| Inadequate agricultural policies |
| Illiteracy |
| Human induced degradations like degradation due to brick making, sand extractions etc. |

Table 3. Soil Degradation processes

### 3.1 Soil erosion

Soil erosion is the detachment or breaking away of soil particles from a land surface by some erosive agent, most commonly water or wind, and subsequent transportation of the detached particles to another location. Erosion is a natural process and is a critical factor in soil formation from rock parent material. Human activities are responsible for greatly


Fig. 4. Cultivation of crops not suited to the land (ginger in place of paddy): a cause for soil degradation


Fig. 5. Uncontrolled grazing by livestock: a cause for degradation

accelerating erosion rates, usually by reducing or eliminating plant and residue cover. However, once productive agricultural soils have been formed over periods of thousands or millions of years, erosion of the soil material is then usually very low or negligible because of the impacts of protective natural plant and residue cover. This exposes the soil to wind and water erosion forces, weakening the soil cohesive forces by tillage disturbance, and increasing the erosive agents, particularly by activities that increase surface runoff. Soil erosion is a serious problem and major cause for the declining productivity, particularly in the rainfed areas world over. If we consider the case of India, almost the entire rainfed area in the country, covering an area of about 70 m ha, is affected by severe sheet and rill erosion. Loss of topsoil is one of the major factors for the low and unstable crop yields obtained in the semi-arid and sub-humid subtropics of India (Sehgal & Abrol, 1994).

Fig. 6. Forest clearing for rubber cultivation: steep slopes and high rainfall in Kerala state (Southern India) makes the soil most vulnerable for severe erosion losses.

Gullies and ravines are also commonly seen in these areas. Wind erosion is dominant in the western regions of the country and to some extent in the coastal areas. It causes loss of topsoil, terrain deformation, over blowing and shifting of sand dunes. It is estimated that more than 45 per cent of India's geographical area is already affected by serious soil erosion and this proportion is increasing year by year.

It is estimated that the soil forming process needs hundreds of years for the formation of few inches of agriculturally productive soils. Under natural condition, undisturbed by man, equilibrium gets established between the climate of a place and the cover of vegetation that protects the soil layer. A certain amount of erosion does take place even under this natural cover, but it is slow and very limited in nature which is balanced by the soil that is formed by continuous weathering and other soil forming processes. When this balance is upset because of the cultural operations followed or any other reason, the removal of soil takes place at a faster rate than its renewal.

In sheet erosion, the movement of runoff water and eroded soil occurs in thin sheets continuously. When this moving sheet assumes sufficient velocity, its cutting action on the soil gets increased and results in the formation of rills, trenches or gullies. If the velocity of the runoff water is doubled, its energy increases fourfold and its erosive action on the soil is correspondingly increased and its capacity to carry soil particles is increased by 64 times (Government of Madras, 1954). The gullies tend to get deeper and wider with every succeeding rain and eventually cut up the agricultural lands into fragments and making it unfit for cultivation. Gully erosion is more evident and spectacular at the surface but sheet erosion is more dangerous as it is insidious and is seldom noticed before it is too late to remedy its destructive effects on heavy soils.

### 3.2 Erosion by water

Water erosion results from the removal of soil material by flowing water. The most common types of soil erosion by water are sheet and rill erosion on upland areas, channel and gully erosion in small watersheds and stream channel and bank erosion in larger catchments.

Sheet erosion is caused by the action of rain drops and shallow overland flows that remove a relatively uniform depth (or sheet) of soil. Because of uniform nature of the soil loss, it is often difficult to detect and gauge the extent of damage caused by sheet erosion.

Rill erosion occurs in well-defined and visible flow concentrations or rills. Soil detachment in rills is largely because of flow sheer stress forces acting on the wetted perimeter of the rill channel. Once detached, larger sediment particles move as bed load, rolling and bouncing down slope with the flow, and are almost always in contact with the soil surface. Smaller sediment particles (silts and clays) are much easier to transport and travel in the rill channels as suspended load. Rills are also the major pathways for trans- porting away sediment that is detached by sheet erosion (also known as interrill detachment). By definition, rill channels are small enough to be obliterated by tillage and will not reform in exactly the same location( Hallsworth, 1987).

Losses under dense natural vegetation are likely to be less than 1 t $ha^{-1}$ per year, under well-managed crops or with conservation works it, whilst, for crops such as maize or tobacco on moderate to steep slopes without conservation, rates of the order of 5o t $ha^{-1}$ per year are recorded in the savanna zone and upwards of 100 tin the humid tropics (Hassan & Rao, 2001).

Removal of soil by erosion and its renewal by rock weathering are natural geomorphological processes, so the question arises as to the rate of loss that is acceptable. The basis normally used is called the soil loss tolerance, defined as the maximum rate, which 'will permit a high level of crop productivity to be sustained economically and indefinitely'. Tolerances were established for different soils of the USA, mostly in the range 5-12 t $ha^{-1}$ per year. In the tropics, a value near the top end of this range, about 10 t $ha^{-1}$ per year, is commonly taken as a guideline, because on cropland it is difficult to achieve much below this rate in practice. This is equivalent to losing a soil thickness of o.8 mm per year, or 8 cm per century (USDA, 1988).

This estimate rests on dubious foundations, particularly as regards sustaining production 'indefinitely'. This word implies that it is the rate at which soil is renewed by rock weathering. The latter has not often been measured, but studies of rates of natural erosion show these to be more typically 1t $ha^{-1}$ per year, and it is reasonable to assume that weathering keeps pace with erosion. It may be that 'tolerable' erosion rates will sustain production for one or several generations, but, since they imply loss of nearly one meter of soil in 1000 years, they are not fully sustainable.

Fig. 7. Over grazing and neglect of village common lands leading to severe erosion

Fig. 8. Rill erosion in cultivated black soils

Because direct measurement has proved difficult, the use of modeling has been widespread. This is founded on the universal soil loss equation (USLE), which states that the predicted rate of erosion, in tons per hectare, is equal to the product of five Factors of erosion: rainfall energy, soil resistance, slope angle and length, crop cover, and conservation practices (Sehgal & Abrol, 1994). Based on experiment data, it is employed as a field guide to conservation; having obtained the predicted erosion for a site without conservation, erosion is reduced to the tolerable level by conservation practices as necessary.

It is not sufficient to know the rate of erosion in terms of soil loss. Before the considerable effort and expense of conservation works can be justified, it is necessary to know the effects on plant growth and crop production. An order-of-magnitude calculation illustrates the effect of the loss of plant nutrients. There is clear experimental evidence that concentrations of nutrients in eroded soil are over twice those in the soil from which they are derived, owing to selective removal of fine particles. Assuming that typical topsoil contains 0.2 % nitrogen, erosion of 20 t of soil will remove 80 kg of nitrogen, together with other nutrients. This is equivalent to carrying several bags of fertilizer away from each field every year. There is a further effect from loss of organic matter by erosion, causing degradation of soil physical conditions (Natrajan et al., 2010). There can be no doubt that erosion in excess of 50 t $ha^{-1}$ per year is common where steeply sloping land is farmed without conservation, and that erosion at such a rate has extremely serious consequences for the future.

Fig. 9. Most catastrophic form of erosion: Gully erosion

As one moves from smaller hill slopes to larger fields and watersheds, additional erosion processes come into play, because of the increasing amount of runoff water. Gullies are incised erosion channels that are larger than rills and form in regions of large runoff flow concentration. Ephemeral gullies are a common type of erosion feature in many fields. They are small enough to be tilled over, but re-form in the same location owing to the convergent topography in small catchments. Classical gullies are larger erosion features that cannot normally be tilled across.

Gullies and gully patterns vary widely, V-shaped gullies form in material that is equally or increasingly resistant to erosion with depth, U-shaped gullies form in material that is equally or decreasingly resistant to erosion with depth. As the substratum is washed away, the overlying material loses its support and falls into the gully to be washed away. The cost of restoring the areas affected with such kinds of severe erosion is at least 50 times more than the cost of preventing the events taking place (Young, 1998).

### 3.3 Erosion by wind

Erosion by wind occurs when wind speed exceeds a certain critical or threshold value. Soil particles can be detached and moved through suspension, saltation, or creep. Suspension usually lifts the smallest soil particles (clays, silts, organic matter) so high into the air mass that they are easily kept in motion and can travel for long distances. Soil particles that move by creep are larger sand grains and aggregates that stay in contact with the soil surface. Almost all times their motion is often through rolling and bouncing. Saltating soil particles are usually moderate in size, and once detached, move in trajectories up into the air and then back down to the soil surface (FAO, 1991).

Fig. 10. Neglected road side water drain forming into a huge gully

Fig. 11. Simple low cost bunding and bund planting can effectively prevent major forms of soil erosion

### 3.4 Classes of accelerated erosion

In cultivated fields 4 classes of accelerated erosion are identified based on degree of loss of surface layer and its spread over the fields

**Class 1:** Soils that have lost some, but on the average less than 25 per cent of the A horizon or the upper most 20 cm. Throughout most of the area, the thickness of the surface layer is within the normal range of variability of the uneroded soil.

**Class 2:** Soils that have lost, on the average, 25 to 75 per cent of the original A horizon or the upper most 20 cm. Throughout most cultivated areas the surface layer consists of a mixture of the original A horizon and material from below. Some areas may have intricate patterns, ranging from uneroded areas to severely eroded small areas.

**Class 3**: Soils that have lost, one the average, 75 per cent or more of the original A horizon or of the uppermost 20 cm. In most areas material below the original A horizon is exposed at the surface in cultivated areas; the plough layer consists entirely or largely of this material.

**Class 4**: Soils that have lost all of the original A horizon or the upper most 20 cm. In addition, some or all of the deeper horizons are lost throughout most of the area. The original soil can be identified only in small areas. Some areas may be smooth, but most have an intricate pattern of gullies.

Fig. 12. Growing hardy grasses (Pennisetum vahnikere) on farm bunds to conserve soil and water in semi-arid tropics

**3.5 Loss of soil structure**

Soil structure describes the arrangement of primary particles into aggregates of different sizes and shapes and the associated pore spaces between them. Therefore, a structured soil is heterogeneous; where a degraded, structure less soil is homogeneous. Soil structure significantly influences all processes that take place in the soil. It influences water infiltration (and hence runoff), the movement of water within the soil and the amount of water that can be stored in the soil. Soil structure also determines aeration levels in the soil, which are essential for the oxygen supply to roots, soil fauna and for aerobic microbial activity. Not only the soil structure but also the stability of the structure is of major importance. Structural stability determines the ability of a soil to withstand imposed stresses without changes in its geometric structure and functions. These stresses may be due to rapid wetting, raindrop impact, wheel traffic and excessive tillage( Lal & Stewart, 1990).

Soil physical degradation results when soil aggregates are destroyed by internal or external forces. Internal forces are applied when entrapped air breaks out of soil aggregates upon flooding. External forces appear in the form of rain impact or pressure and sheering for as exerted by animal trampling, wheel traffic and tillage implements.

Depending on the water content, this results either in pulverization or compaction of the soil. Soil physical degradation, however, depends not only on the degrading forces and stresses but also on the stability of a soil to withstand these stresses and its resilience to recover from different levels of short term and long term degradation. Mineral composition of soils also determines its structural stability. Salts like sodium in excess makes the soil vulnerable for destruction of its structure. Supplying adequate soil organic matter and adoption of proper irrigation technologies are essential for maintaining ideal soil structure that is required for the successful production of most crops (Lal & Stewart, 1990).

Fig. 13. Hardy local plant species having commercial value (Agave sps) are essential for successful soil conservation)

### 3.6 Compaction

Compaction describes the state of "compactness", i.e., bulk density of a soil. Compared with its undisturbed condition, a compacted soil exhibits reduced total pore space, especially because of a drastic reduction of the macrospores, and a pronounced discontinuity of the pore system within the profile. This affects the conductive properties of the soil and reduces its ability to retain air and water. Hence, plants growing under high evaporative demand suffer more from compaction than plants growing under low evaporative demand. Compaction also inhibits root penetration and development, thus affecting nutrient uptake and consequently, plant growth.

The most important cause of compaction is off-road wheel traffic and the use of heavy machinery in mechanized agriculture. Soils with high clay contents and well developed pore systems are generally more compressible than sandy soils. Two main types of soil compaction can be distinguished, namely, the surface layer compaction and subsoil compaction (Government of Madras, 1954).

*Surface layer compaction* describes the compaction in the upper part of the soil profile, i.e., in arable soils usually the plough layer, Compaction in the surface layer is dynamic and changes significantly over the cropping season, increasing with increasing machinery passes over the field and decreasing again with primary tillage for seed bed preparation for the following season. Adequate tillage effectively reduces soil compaction in the surface layer and its effects.

*Sub-soil compaction* affects soils beyond the surface layer at depths >30 cm. It is caused by heavy machinery. Swelling-shrinking, freezing-thawing and biological activities can alleviate compaction to a certain extent. Sub-soiling using specially designed equipment can in some cases alleviate sub-soil compaction, but is very energy demanding some times, soils become more dense than it was before sub-soiling because of the destabilization of the soil caused by the mechanical energy input from the sub-soiling operation. Compaction should be considered to be an irreversible, permanent form of degradation.

### 3.7 Sealing and crusting

Seals and crusts are consequences of rain and flooding on unprotected soil-surfaces. Under the impact of rain drops and the soaking effect of water, the bonds that hold the particles together become weak and the aggregates tend to fall apart. Individual particles become separated. These particles become rearranged and the finer particles tend to be washed into the cavities of the surface. There they form a very thin (1-5 mm) and dense layer that clogs the soil pores and seals the surface. These seals are usually very elastic. Typical characteristics of soil seals are that they do not crack and cannot be removed from the surface.

Fig. 14. Effective use of locally available raw materials like stones and rubbles can reduce the dependence on government aid for soil and water conservation

Soil crusts are formed by the same processes that form seals. They are much thicker than seals (usually 5-20mm) and can be separated easily from the soil surface, and they crack upon drying. Crusts are typically formed on soils with high contents of non swelling clay susceptible to dispersion. Seals that become hard upon drying are also termed crusts. Soils with a high content of fine or very fine sand, or silt are especially prone to sealing and crusting. The presence of exchangeable sodium in the soil can enhance clay dispersion and thus contribute to seal and crust formation.

Fig. 15. Engineering inputs wherever it is absolutely essential to make the projects successful

Fig. 16. Participation of local communities at every stage of soil and water conservation ensures greater success

### 3.8 Causes and effects of soil physical degradation

The main causes of soil physical degradation are inappropriate land use and soil management practices. All exploitative practices will ultimately lead to degradation and hence reduce soil productivity. In the developing world, land not suitable for cultivation, such as dry lands or steep terrain, is increasingly being cropped. The cultivation and husbandry practices associated with these land use systems are largely responsible for degradation. Ultimately, soil physical degradation leads to reduced plant growth, crop yields, and soil productivity. Soil and water are inseparable when we plan any conservation measure. Hence better husbandry practices to take care of soil, water and crop must go together in any conservation program (Hellin and Haigh, 2002).

## 4. Soil chemical degradation

Soil chemical degradation is the undesirable change in soil chemical properties such as pH, size and composition of cation exchange complex, contents of organic matter, mineral nutrients and soluble salts. Change in one or more of these properties often have direct or indirect adverse effects on the chemical fertility of soils, which can lead to a decrease in soil productivity(Suraj Bhan et al., 2001).

### 4.1 Soil pH and soil acidity

Chemically fertile soils have a pH range of 5.5-7.5. Soil pH is determined by the mineralogical make up (clay minerals, various metal oxides and hydroxides, lime etc), organic matter content of the soils and dissolved $CO_2$ in the aqueous phase. Any measure of pH below 7 is defined as the active acidity, whereas the ability of the soil to maintain a low pH level is referred to as the potential acidity. The active acidity represents the concentration of $H^+$ ions in the soil solution. Potential acidity includes exchange and titratable acidities where the former constitutes most of the latter in acidic soils. The exchange acidity includes the protons associated with the cation exchange sites on the clay mineral and organic fractions. The exchange acidity as a portion of total acidity varies with the nature of the soil and with the percentage base saturation. There is an equilibrium between active and exchange acidities and as the $H^+$ ions in soil solution is neutralized, the cation exchange phase brings new $H^+$ ions into solution. The source of soil acidity is humus, aluminosilicates, hydrous oxides and soluble salts.

Humic matter causes acidity through dissociation of $H^+$ ions in its carboxylic, phenolic, and similar $H^+$ ions yielding functional groups. The humic fraction is considered as the weak acid component of the acidity. Furthermore, the complexes of humus with iron and aluminum can produce $H^+$ ions upon hydrolysis. The charged sites associated with aluminosilicate clay minerals are occupied by various cations present in the solution phase. As the portion of basic cations such as Ca, K, Mg and Na are reduced through leaching or by plant uptake, the portion of the total charge occupied by $H^+$ ions increases. This process is accompanied by a reduction in pH as the dominating exchangeable $H^+$ ions controls the solution phase. When the soil pH falls below 6, Al in Octahedral sheets dissociates and is adsorbed in an exchangeable form by clays, thereby increasing the Al saturation. Exchangeable Al is the major cause of exchange acidity. When dissociated from the exchange complex as the $Al^{3+}$ ion, it produces $H^+$ ions.

Dissolution of soil minerals and application of ammonium fertilizers also can lead to soil acidity. Soil acidity limits the plant growth by toxicity and decreases the macronutrient base cation content. Furthermore, the solubility of Fe, Mn, and Al containing minerals are enhanced at low pH levels and the toxicity of these elements becomes a major problem. The activities of soil organisms, including nitrifying bacteria, are severely restricted at pH levels lower than 5.5. Regular liming of soils with suitable liming materials becomes essential in areas susceptible soil acidity. Regular testing for soil ph needs to be attended in such areas to monitor the acidity levels.

### 4.2 Salinity

Salinity is a common problem in arid and semi arid regions where evapo-transpiration exceeds rainfall. Under these conditions, there is not enough water to wash the soluble salts down the profile below the rooting zone. Thus, soluble salts originating from various sources accumulate in the soil profile at certain depths known as the salic horizon or at the soil surface, depending on the water regime. If not washed from the soil profile to a drainage system by the leaching fraction of water, the concentration of chloride, sulfate, carbonate and bicarbonate salts of Na may increase in the soil profile and cause salinity in a very short period. Salinization in its broad sense covers all types of degradation brought about by increase of salts in the soil. It thus includes both the build-up of free salts in the soil, salinization in its strict sense, and sodification, the replacement of cations in the clay complex by sodium. It is brought about through incorrect planning and management of canal-based irrigation schemes. Part of the water brought into the area is not used by crops but percolates down to groundwater. This leads to a progressive rise in the groundwater table, and, when this comes close to the surface, dissolved salts accumulate. Patches of salinized soil appear, as more or less circular areas of white, saline soil surrounded by a belt of stunted crop growth. A continued rise leads to water logging (Singh et al., 1992, Datta and Jhong, 2002).

This process happened extensively on the Indus plains of Pakistan; the water table began to reach critical levels in the 1940S, and salinization has since become widespread. A sequence of costly reclamation schemes was necessary to check the rate of land abandonment. It can be prevented by construction of deep drains. Reclamation is a more complex process, involving tube well construction; large-scale pumping to lower the groundwater table, followed by application *of water* much in excess of irrigation requirements in order to leach out salts, a wasteful and expensive procedure ((Datta & Joshi. 1991).

Because salinization is easy to identify', and also takes place on the 'managed' environments of irrigation schemes, estimates of its extent are somewhat less unreliable than those for other forms of degradation, meaning that their range of error is not much above plus or minus 100 %. Another cause of salinity is the absence of drainage system or poor drainage especially in lowlands. A raised water table as a result of an ineffective discharge system is still another cause of salinity. The upward capillary movement of water carrying the dissolved salts, previously present at depth in the rooting zone. The degree of salinity is measured as electrical conductivity of a soil saturation paste or extract and is reported as deci-Siemens per meter (dS/m)(Young, 1998).

The source of soluble salts, besides irrigation water, are mineral weathering, fertilizers, salts used on frozen roads, atmospherics transfer of sea spray, and lateral movement of ground water from salt containing areas. Salinity affects plant growth by affecting water and nutrient uptake and through specific toxicity of Na, Cl and B. The dissolved salts in water increase the osmotic potential, thereby creating the so called physiological drought. Toxicity develops when the ions take up from soil solution accumulate in leaves. As water is lost in transpiration, the concentration of toxic ions increases and causes damage to various degrees, depending on the sensitivity of plants.

Salinity affects the mineral nutrition of plants by reducing the availability and uptake of nutrients through the interaction of Na and Cl with nutrient cations and anions and by interfering with transport of elements within the plant. Leaching with good quality water, providing adequate drainage, adoption of scientific irrigation techniques and growing of salinity tolerant crops are the strategies needed in salinity affected soils. Studies in India have indicated that these areas can be successfully used for inland fisheries and also shrimp cultivation (ICAR, 2008).

### 4.3 Alkalinity

Addition of salt to soils increases the concentration of Na in the soil solution more than those of Ca and Mg and alters the composition of the exchange phase in favor of Na, because the Na salts are the most soluble salts in nature.

Fig. 17. Providing sub-surface drainage to manage soil salinity/alkalinity

This increase in exchangeable Na ($Na_x$) is called sodification and soils degraded in this manner are referred to as sodic soils. The measure of sodicity is exchangeable sodium percentage, which is the ratio of $Na_x$, to cation exchange capacity. This parameter is sometimes expressed as the exchangeable sodium ratio, which is the ratio of $Na_x$ to other exchangeable cations.

If the soil solution contains $CO3^{2-}$ and $HCO3^{-}$ in excess of $Ca^{2+}$ and $Mg^{2+,}$ highly soluble Na salts of these anions hydrolyse and the soil pH rises above 8.5. This process is termed as alkalinization. Sodic soils do not necessarily have high pH, but in well-aerated soils, alkalinization often follows sodicity. Sodicity although a chemical property, has adverse effects on soil structure. As $Na_x$ increases, the binding effects of divalent Ca and Mg ions on clay particles are overcome by the dispersing action of Na ions. Dispersed particles move with water and quickly clog the soil pores, causing drastic reductions in water and air permeability in sodic non saline soils (Sharma et al., 2004). Application of gypsum, leaching with good quality water and proving of adequate drainage and growing of suitable crops are the management techniques needed to tackle these soils (Singh et al., 1992).

### 4.4 Depletion of soil organic matter

Organic matter is very important to the functioning soil system for various reasons. It increases soil porosity, thereby, increasing infiltration and water holding capacity of the soil, providing more water availability for plants and less runoff that may potentially become contaminated. This may be specifically helpful at mine sites where runoff may become acidic and contain high concentrations of heavy metals. The increased porosity also aids in easing tillage of the soil.

Fig. 18. Success through participatory approach brings smiles and long lasting impact

The organic fraction of the soil accounts for 50 to 90% of the CEC of mineral surface soils. The CEC allows important macronutrient cations (K, Ca, Mg) to be held in exchangeable forms, where they can be easily used by plants. Nitrogen, phosphorous, sulfur, and micronutrients are stored as constituents of soil organic matter, from which they are slowly released by aiding in plant growth. In addition, humic acids (a form of organic matter) accelerate soil mineral decomposition releasing essential macro- and micronutrients as exchangeable cations. In addition, organic matter adds erosion resistance to soils (Anand Swarup, 20110). The establishment of cover on disturbed soil surfaces is a common reclamation strategy. Cover soils facilitate the establishment and growth of vegetation. Many times, finding enough cover soil to cover disturbed surfaces can be difficult and costly. When there is not enough soil on-site to satisfy the demand, surface soils may be hauled in from other designated sites. If surface soils are excavated to recover minerals beneath, the surface soils may be stored until reclamation of the area takes place(Hegde &Daniel, 1994).

In these cases, surface soils may be stored for long periods, during which time, the soils may show reduced biological activity, in part due to bacteria, and invertebrates. Stored surface soils also reveal a loss of organic matter and nutrients. Therefore, organic amendments and fertilization of surface soils that have been in storage for several years are necessary to ensure rapid buildup of microbial populations and initiate nutrient cycling (Rao, 2010).

Fig. 19. Training through field visits and interactions with stakeholders: spreads the soil and water conservation technologies at faster pace

The accumulation of soil organic matter, namely humus, in soils starts with the production of biomass and approaches equilibrium dependent on the effects of factors such as climate, type of vegetation, topography, soil texture, and drainage conditions. At equilibrium, when additions by biomass and removal by mineralization are in balance, organic matter contents range from less than 1% in arid regions to over 20% in organic deposits of cool humid climates. Any change in these factors may disturb the system and a new equilibrium towards depletion of soil organic matter results. A highly disturbing factor in this respect is cultivation. Less organic material is returned to soil at harvest in most cropping systems, and tillage accelerates decomposition of soil organic matter. This process is more rapid in tropical climates. Conversion of forests and grasslands to crop lands promotes rapid decomposition of the organic matter present in soils. Soil erosion is another factor that causes significant reductions in soil organic matter content, first, by decreasing the overall productivity and thus the production of biomass some of which is returned to soil and second, by carrying away the organic matter present in the lighter fraction of the surface soils.

Fig. 20. Planting of leguminous, multi-purpose hardy plant species like Sima rouba in tropical wastelands can restore the soil health at a quicker pace

There are several different types of organic amendments, added for different reasons (Suganya & Sivaswamy, 2006). Mulches are organic materials applied to the surface (not tilled into the soil) primarily to reduce erosion. The more common mulches include paper, wood residues, straw, and native grasses. Surface mulches reduce wind velocities at the soil surface, shield the soil from raindrop impact, reduce evaporation from the soil surface, trap small soil particles on the site, reduce surface soil temperatures, and help prevent soil crusting. Manure, compost, and sewage sludge are other organic amendments generally incorporated into the soil by plowing, chiseling, crimping, or rototilling. These organic amendments benefit the cover soil for the many reasons, such as increased microbial activity, cation exchange capacity (CEC), porosity and water-holding capacity (Anand Swarup 2010).

Adoption of conservation agriculture, crop rotation with legumes, mixed farming, green manuring, green leaf manuring and application of organic manures like oilcakes, FYM and composts are recommended to maintain adequate soil organic matter (Raj Gupta et al., 2010). In developing nations both the cow dung and the crop residues are extensively used as fuel in rural homes. These activities are also listed as the cause for increased accumulation of green house gases. It was noticed that wherever the program of green energy " Bio-gas technology" is adopted, the health of rural women and soil improved considerably

(Swaminathan, 2011). This simple technology can bring down the green house gas accumulation also to a great extent. Once this technology is adopted, farmers tend to stall feed their animals as they do not wish to lose the cow dung. This activity helps in restoration of vegetation in village common lands and naturally soil health improves and soil degradation decreases/halts. Farmers bring all kinds of farm wastes to their backyard and decompose the wastes using bio-gas slurry. Hence the quantity and quality of production of organic manure increases and its use in farm improves the soil health. When rural kitchens are freed from smoke due to the use of biogas, the health of women folk improves automatically. Hence there is an urgent need to promote this simple technology in very large scale.

### 4.5 Loss of plant nutrients

Plant nutrient elements are continuously lost from soils by crop removal, erosion, leaching, and volatilization at rates determined by the type of vegetative cover, cropping system and climatic conditions. In intensive agriculture, much larger amounts of nutrients are taken away from soil with little return in crop residues, in many cases exhausting the nutrient reserves in soils (Tiwari, 2010). Basic nutrient cations such as Ca, Mg and K may be leached from soils under acidic conditions. Nutrient depletion and declining fertility is commonly observed in both rainfed and irrigated areas. Highly weathered soils occurring in the high rainfall areas are more prone to loss of fertility and chemical deterioration. According to the Soil Resources Mapping data, in India, about 3.7 m ha of land area is deteriorated due to nutrient loss and/or depletion of organic matter (Sehgal & Abrol 1994). It has been established by many studies that in many regions there is a net negative balance of nutrients and a gradual depletion of organic matter content level in the soils. Since in future the required demand for food production will have to be met through increased intensity of cropping, the problems of maintaining nutrient balance and prevention of emerging nutrient deficiencies will be a major concern in most of the cultivated lands.

Lowering of soil organic matter is the main cause of physical degradation and also affects nutrient supply. Degradation of soil physical structure has substantial affects on plant yield independently of chemical properties. Maintenance of the soil organic matter content is a key feature of management, since this underlies many other properties: resistance to erosion, structure and therefore water-holding capacity, and ability to retain and progressively release nutrients. Recycling of organic material also helps to prevent the development of deficiencies in micronutrients (Rao, 2010). Erosion is itself a cause of fertility decline, through removal of organic matter and nutrients. Even with no erosion, however, fertility decline can be brought about by other processes, notably, nutrient removal in harvest exceeding replacements, by natural processes and fertilizers.

Evidence is accumulating that fertility decline is extremely widespread, particularly in areas that have long been under annual cropping. Indeed, although it is a reversible form of degradation, the total consequences on lowering current agricultural production may be greater than those of erosion (Tiwari, 2010). In the Indian subcontinent, where fertilizers have been in use for 20 years or more since the green revolution, reports of nutrient deficiencies are becoming common. The explanation is that farmers first added nitrogen fertilizer, and obtained a good crop response; after some years, the augmented growth led to exhaustion of soil phosphorus reserves, and phosphate had to be added also; now, the same process is happening with respect to secondary and micronutrients, such as sulphur and zinc. A result of fertility decline is that responses to added fertilizers are now less than formerly. In India, Pakistan, and Bangladesh, rates of increase in fertilizer use have not been matched by crop yields. There are also records from long-term experiments. A striking example is a 33-year experiment in Bihar, India; despite changes to improved varieties, wheat yields declined substantially with nitrogen, phosphorus, and potassium fertilization, whereas they rose with additional farmyard manure (Tiwari, 2008).

### 4.6 Soil pollution

Soil pollution can result from mining or industrial operations, and from agriculture. Soil and groundwater pollution from agricultural activities has been up to now mainly a problem of temperate countries. As fertilizer and pesticide use increase in the developing world, it will become an increasing problem, calling for technical appraisal and legislative control (Chonkar, 2001).

Ecosystems are threatened by such contaminants and their interactions in the environment. The impact of urbanization and industrialization has been a major factor against the need of preserving the quality of soil, by reducing it via chemical contaminants, use of polluted waters for irrigation, and deposition of harmful particulates to the atmosphere. Ecosystems are threatened by such contaminants and their interactions in the environment. The sources of pollution are: (1) the waste waters; (2) the agricultural wastes, (3) the airborne pollutants, (4) the pesticides, (5) the urban wastes: a) sewage sludge; (b) composts; (c) fly ash; and (6) the industrial wastes: (a) pesticides and fungicides; (b) fertilizers; (c) detergents; and (d) chlorinated solvents.

These sources may cause fatal effects and/or irreversible destructions for human health and the environment. The levels of contamination together with their ability and mobility in decomposition and accumulation of the chemicals in the soil have been scientifically proven to be harmful to animals, plants, and micro-organisms via destroying the natural structure of water, soil and air which is balanced by nature.

Inactivated enzymes are the measure of heavy metal (divalent) toxicity readily reacting with proteins, amines and sulfohydryl (-SH) groups as well as Hg and Cd replacing Zn in metallic enzymes. Metal toxicity is known to decrease cell membranes permeability and change genetic characteristics of cells increasing risks of cancer. Mercury accumulates in fatty tissues as methyl Hg, whereas cadmium replaces calcium in bones and kidneys destroying their excretory function.

Metals such as lead, arsenic, mercury, chromium, cadmium and copper are found at acceptable limits in nature. However, these limits might increase with contamination from agriculture, industrial and infrastructural wastes. For instance, concentration of lead might increase by the gaseous emissions of vehicles and use of garbage compost as a fertilizer together with pesticides. Arsenic is naturally found at a level of 10 ppm in soils and might also increase to 500 ppm with the use of industrial wastes for agricultural applications. Mercury is added to soils by rain and irrigation waters as well as garbage compost. Mercury compounds are highly poisonous for humans and animals. Excess Hg affects the central nervous systems and develops blindness via $CH_3$ HgCl.

The major source of cadmium contamination in agricultural soils is the excess use of fertilizers with varying amounts of cadmium contents (0.1-90 mg/Kg) depending on the phosphate rock materials utilized in production. Such contaminations in soils will cause a 0.1 mg/kg increase of Cd in 20-30 years. Another significant Cd contamination source is the sewage sludge used in cultivated soils. Cadmium adsorption by organic or inorganic soil

compounds (minerals) depends on the soil texture, pH, and Ca together with other elements.
Chromium, the essential element for human and animal diets is not considered essential for plant nutrition with a tolerable level of 100 mg/kg in soils. Chromium is irrevocable in its role in the induction of the effect of insulin and proteins as well as stabilizing nucleic acid structures and activating some enzymes in glucose metabolism. Since pollutants and heavy metals such as persistent chlorinated organic compounds and Hg could cause genetic side effects, great care and effort should be paid to prevent the transportation of the said substances to soil for sustaining biodiversity. Hence, through the sustainable ecosystem concept, the status of pollutants in soil should be monitored permanently.

### 4.7 Desertification

The term 'desertification' falsely evokes the image of advancing deserts. While a desert is a unique ecosystem, desertified areas are not: they are disrupted ecosystems. Desertification means land degradation, loss of soil fertility and structure as well as the erosion of biodiversity in drought prone areas (Tuboly, 2000). Desertification is a land degradation process and it deals with the gradual conversion of productive land into less productive or unproductive ones. Thus, the problem is a continuous one. The presence or absence of a nearby desert has no direct relation to desertification. It is the excessive abuse of land in any patch or land under arid ecosystem which can initiate desertification process. (Nagarajan, 2000). Land degradation is a more acute problem faced by the farmers in the dryland regions, especially the small and marginal farmers. Soil erosion by runoff is the principal cause of land degradation particularly in the rainfed agro-eco regions of India and Africa. Consequences of land degradation could be more disastrous in arid and semi-arid areas where the ecosystem is very fragile. Rainfall being the only source of sustaining the entire production system, these areas chronically suffers from low food and fodder productivity due to poor, erratic and unevenly distributed rainfall. The process of land degradation is highly dynamic and complex at times. Unfortunately, more often than not, it goes unnoticed by the very people dwelling in such less endowed areas which are already marginalized. The people are characterized by low literacy and awareness levels, poor socio-economic status and have low risk bearing ability (Subba Reddy et. al 2000).
A study desertification conducted using remote sensing and ground truthing in Bellary district in Karnataka situated in semi-arid region indicated that nearly 28 percent of total area (8.5 lakh ha) area faced severe desertification, major cause being vegetation degradation, salinization/alkalization and water resources degradation(NBSSLUP, 2005). Some of the other on-field indicators besides the scientific indicators, of the impending crisis are: Dying of older trees due to lack of enough capillary raise of water which indicates falling water tables in the area - for instance, if there is a need to irrigate mango trees and orchards, it is a sure indication of the alarming situation. In brief all the degradation processes together can be termed desertification and it is the final stage of decline of farming assets and consequently the food production potential.

## 5. Human induced accelerated soil degradation

The excessive demand for construction materials like bricks and sand for infrastructure projects in developing countries like India are causing huge soil degradation in peri-urban environment at an alarming rate. A study conducted at Bangalore, India revealed that sand supply from riverbeds to Bangalore is not able to meet the demand of booming construction sector. Enterprising farmers have taken up extraction of sand by washing surface soils of agricultural fields. Nearly 25 percent of sand supplied is from this source. Study revealed that significant employment and economic gains are realized at an ecological cost. Loss of surface soils, nutrient losses, crop yield losses, siltation of tanks, excessive ground water exploitation and soil erosion are taking place due to sand extraction(Table 4 and 5). Nearly 18000 ha of land which was usually used for growing the staple food of the region, i.e., Finger millet is going out of cultivation for few years to come(Rajendra Hegde et al., 2008).

Mining is another significant economic activity causing unrepairable damage to land resources (Vishwanath, 2002, Bhushan & Hazra, 2008). A case study conducted at Goa state in India indicated that large tracts of pristine forests of western ghats were lost to mining and the accumulation of mining wastes in the nearby paddy fields in the valleys have destroyed the highly productive soils. The economic and ecological damage is very long lasting. Such a development has lead to social conflicts between miners and farmers. Mining has both on site and off site ecological damage. Mining in Goa is done by open cast method which necessitates the removal of overburden overlying the iron ore formations. On an average about 2.5 to 3 tons of mining waste has to be excavated so as to produce a tone of iron ore. The average annual production of iron ore is about 15 to 16 million tones, in the process removal of which about 40 to 50 million tones of mining waste is generated. Such a huge quantity of mining waste creates a problem for its storage thereby causing severe environmental pollution.

Damage to the environment is mainly done by the reject dumps, pumping out of muddy waters from the working pits including those where the mining operations have gone below the water table, and slimes from the beneficiation plant. The damage is more evidenced during monsoon where the rain water carries the washed out material from the waste dumps to the adjoining low-lying agricultural fields and water streams. It is stated that the slimes and silts, which enter the agricultural field are of such character that they get hardened on drying. The washed out material from the dumps and the flow of slimes from the beneficial plants besides polluting the water causes siltation of water- ways, especially during monsoon. Such silting of water ways over the years may trigger years even flooding of the adjacent fields and inhabitated areas, especially during monsoon.

Desurfacing of farm lands for brick industry is another source of soil degradation. Thousand of ha of lands are losing their productive potential due to unscientific extraction of soils (Grewal & Kuhad , 2002). Recently technology of using fly ash for brick production has been evolved which may help in reducing the soil degradation to some extent (Kathuria, 2006).

Purposeful conversion of productive farming lands in to shrimp farming or urban development is taking place at a very large scale on coastal zones. In India, Goa a tiny state is a world famous tourist place. The state has low lying 18000 ha of vary productive paddy farming lands called Kazan lands. Slumping revenues from agriculture in Goa has led to breaching the bunds to allow saline water into the fields to raise fish, as this is far more profitable than cultivating paddy has become rampant. It is reported that khazan lands are extensively inundated for as many as 15 years and used for shrimp farming. The growing density of population poses another threat to the khazan lands. Goa's population is concentrated in the Mandovi-Zuari basin, which is also where the khazan lands are situated and almost all urban expansion has taken place at the expense of these lands. Threats to the khazan lands include those arising from general environmental degradation. Deforestation in the upper river catchment areas and mining activity have added to the silt load of the rivers. The sediment that gets deposited in the estuarine region have resulted in many acres of khazan lands now getting flooded during the monsoon.

The rivers have become heavily polluted near the towns and much of the waste material they carry flows into the khazan lands with the tides. And, this problem is compounded by the petroleum residues from barges, tankers and trawlers in the rivers."The problem is that any expansion that takes place in Goa has to be at the cost of the khazan lands. Good sign is that environmentalists attention is unquestionably focussed on the need to protect the khazan lands -- a valuable Goan heritage.

In every region such human induced degradation are taking place due to non adherence of environmental laws. Comprehensive policy is needed to make these enterprises ecologically tolerable.

| | Details | Per lorry load of sand | In one day (1000 lorry loads) | In one year | remarks |
|---|---|---|---|---|---|
| 1. | Soil used ($m^3$.) | 120 | 120000 | 438 lakhs | |
| 2. | Are of land used for soil excavation(ha) | 0.04 | 40 | 18600 | 0.3 m depth (normally) |
| 3. | Quantity of water used (litres) | 132000 | 1320 lakhs | 48180 million | |
| 4. | Quantity of silt-clay generated ($m^3$.) | 30 | 30000 | 11 million | |

Table 4. Soil degradation due to sand extraction in peri-urban and rural Bangalore

| Nutrient | Units | Surface soil | Sub surface soil | silt | Fertility Depletion/ ha | Fertility depletion /year* | Value of nutrient at present rates (Rs) |
|---|---|---|---|---|---|---|---|
| N | Kg/ha | 319.6 | 191.8 | 90.0 | 127.8 | 2.38 million | 25 million |
| $P_2O_5$ | Kg/ha | 83.7 | 25.2 | 11.7 | 58.5 | 1.09 million | 17.68 million |
| $K_2O$ | Kg/ha | 74.8 | 158.7 | 62.1 | 83.9* | * gained | - |
| Cu | ppm | 0.66 | 0.3 | 0.1 | 0.56 | - | - |
| Fe | ppm | 0.38 | 0.76 | 0.53 | 0.15 | - | - |
| Mn | ppm | 21.6 | 14.1 | 2.96 | 18.84 | - | - |
| Zn | ppm | 18.4 | 13.13 | 0.18 | 18.22 | - | - |
| B.D | g/c.c | 1.4 | 1.7 | - | - | - | - |

* 18600 ha of land being excavated to a depth of 30 cm in one year.

Table 5. Nutrients content and fertility depletion from surface soil of agricultural fields due to sand extraction

Fig. 21. Human induced accelerated land degradation: sand extraction from agricultural fields

## 6. Accelerated soil degradation due to climate change phenomenon

Due to climate change factors, various regions of the world are facing unprecedented aberrant weather situations like drought, floods, forest fires etc (Agrawal, 2008, Prasd & Radha, 2008). Role of soil is most significant in buffering the climate change phenomenon as soils and plants hold 3 times more carbon stock than atmosphere (Pathak, 2010).

The unprecedented rains leading to floods in 13 districts of northern Karnataka, India during 2009 was said to be one such event caused due to climate change phenomenon. Very high rainfall received over a short period of time in a region dominated by black soils, resulted in severe losses of crops, soils, soil organic matter and soil nutrients besides destruction of human and livestock lives and farming infrastructure.

The shallow depth and heavy texture of the soil made the situation to aggravate further. An estimate made after the calamity revealed that nearly 287 million tons of top soil, 8 lakh tons of soil nutrients, 39 lakh tons of soil organic matter were washed away. In monetary terms, about 853 crores worth of soil organic matter and 1625 crores worth of plant nutrients were lost from the region during this short period. In addition to this, nearly two-lakh hectares of cultivated fields in the flood affected area were deposited with sand along the river courses. These losses have severe long term implications on crop productivity and rural economy of this region (Natarajan et al., 2010).

## 7. Consequences of degradation

By definition, the direct consequence of land degradation is reduction in productivity. Where degradation is extreme, there is total loss of the resource. This is temporary in cases such as salinization and deforestation, and for practical purposes permanent in severe erosion. In soil degradation, the results may be lower crop yields, or a need for higher fertilizer inputs to maintain existing yield levels. Soil physical degradation and micronutrients deficiencies cause lower responses to fertilizers. Once the cutting of woodlands for fuel and domestic timber passes their rate of growth, the potential of the resource is reduced, and can only be restored by a radical reduction in cutting, which is economically and socially unpractical

Fig. 22. Large scale mining in India without adhering to environmental laws leading to land degradation(Mining of iron ores in Goa state)

There are three bases to assess the costs of land degradation in economic terms: lost production, replacement cost, and the cost of reclamation (Chinnappa and Nagraj 2007). In lost production, crop yields or other outputs are estimated for non-degraded and degraded land, and then priced. The two situations, with and without degradation, can then be compared. A weakness for the case of soils is the paucity of evidence for the physical reduction in output. A cleared forest has lost the capacity to produce timber for the 20, 30, or 50 years needed for its regrowth, besides which, continued use for agriculture will prevent any such restoration.

Replacement cost is based on estimating the costs of additional inputs, such as fertilizers, needed to maintain production at the same level as for non-degraded land. This is easier to assess than lost production, and also corresponds to what farmers seek to do. The clearest example of reclamation costs comes from salinization. For Pakistan, the cost of reclaiming 3.3 M ha of salinized land has been estimated at 9 billion (Young, 1998).

For deforestation and erosion, off-site costs resulting from reduction of river base flows and sedimentation of reservoirs must be added. The presently assessed life of eight Indian reservoirs was compared with that anticipated at the time of their design; in four cases, this was 30-40 %. In developed countries, off-site costs of erosion are often assessed as substantially higher than on-site loss of production, although in developing countries the opposite may be the case.

Fig. 23. Mining waste accumulation in paddy fields completely degrading the soil (Goa)

Fig. 24. Sand accumulation on agricultural fields in Nortern Karnataka, India due to floods

Economic analysis requires complex assessments of future production or input changes, and this introduces questions of the discounting of costs and benefits over time. It is possible to estimate the economic costs of degradation on an annual basis. This has been done for the South Asian region. The GLASOD estimates were taken as a starting-point, modified by additional fertility decline. Relative production loss for light, moderate, and strong degradation were assumed to be 5%, 20%, and 75%respectively. These reductions applied to average cereal yields. Fertility loss was also assessed on a nutrient replacement basis. The cost to South Asia of land degradation was estimated as:

**Cost, $ billion per year (Young, 1998).**

| | |
|---|---|
| Water erosion (on-site costs only) | 5.4 |
| Wind erosion | 1.8 |
| Soil fertility decline | 0.6-1.2 |
| Salinization | 1.5 |

Recent evidence would reduce the figure for erosion and considerably increase that for fertility decline. The total cost of land degradation to South Asia is about 10 billion a year, which is 7 % the agricultural production of the region. This estimate applies to soil-related forms of degradation, and refers only to on-site costs. The addition of water, forest, and rangeland degradation, together with off-site costs of erosion, would raise this figure substantially.

It is not unreasonable to say that the land degradation that has taken place up to the present is costing developing countries not less than 5%, and more probably nearer 10%, of their total agricultural sector production. Still more tentatively, this rate may be rising by 1% every 5-10 years.

The physical and economic consequences of land degradation are reflected in their effects upon the people. These include: lower and less reliable food supplies; lower incomes, resulting from loss of production or higher inputs; greater risk: degraded land is less resilient, and higher inputs mean that poor farmers are risking more capital; increased labor requirements, as when women walk large distances to collect fuel wood and water; in the case of farm abandonment, increased landlessness and social unrest.

In classical economic theory, 'land' was regarded as a fixed resource, to which factors of labor and capital were applied. With degradation occurring, it becomes a declining resource, and hence labor and capital become less efficient. If most farmers do not know about economic theory, they are well aware of it in practice. Land degradation means they must either accept lower production, or put in greater effort to maintain it at the same level (Chaturvedi , 2010).

The unprecedented rains leading to floods in 13 districts of northern Karnataka, India during 2009 was said to be one such event caused due to climate change phenomenon. Very high rainfall received over a short period of time in a region dominated by black soils, resulted in severe losses of crops, soils, soil organic matter and soil nutrients besides destruction of human and livestock lives and farming infrastructure.

All types of soil degradation can be caused very easily. However, a cure may be either slow and expensive or completely impracticable. Symptoms of degradation of the top soil include crusting, low fertility, etc. These problems can usually be alleviated by increasing the soil organic matter content and the adding calcium compounds such as gypsum or lime if the soil is acidic. Organic matter can be increased by changing crop rotations to include a pasture phase, by keeping plant cover, all the year round, by adding manures, and also by minimizing losses of soil organic matter, which can occur as a result of excessive or high intensity tillage. Organic matter increases soil aggregation and reduces clay dispersion in water. Calcium compounds flocculate the clay, thereby reducing dispersion and increasing soil stability. Clay floccules form the building blocks for larger compound particles such as aggregates. The use of such practices over a number of years can produce a soil structure that is better for agricultural production and is more stable. Compaction damage of top soils is usually not permanent because of the ameliorative effects of tillage, biological activity, and wetting and drying cycles.

Degradation of the sub-soil often occurs through compaction. Sub-soil tillage may make a temporary improvement, but often the soil will recompact to be as bad as or worse than it was before sub-soiling.

Soil salinity can be cured in principle by leaching. However, dry land salinity usually cannot be reversed but may be slowed or halted by planting of deep rooted plants (.e.g, salt tolerant trees) that can transpire enough water to stop further water table rise. Sodicity can be cured by displacing the sodium with calcium. However, this is difficult because sodic soils have an extremely low permeability to water. Displacement and leaching may be accelerated by not using pure water but using water with a high enough electrolyte concentration to keep the clay flocculated and hence the permeability high.

## 8. Land improvement

Not all changes to land resources are in the direction of degradation. Its converse is land improvement; all relatively permanent increases in productive capacity brought about by human action. Swamp rice cultivation is a special case in which the natural soil profile is radically altered by the formation of a pan, an impermeable horizon which checks loss of water by downward seepage.

Other examples of land improvement are drainage of swamps, terracing, systems of water harvesting, and reclamation of land from the sea. Were it not for land improvement, the productive capacity of Egypt and Pakistan would be a small fraction of its present level. The best-known ease of land reclamation from the sea is the polder system of Netherlands, the only country which, in international statistics, regularly increases its total land area. The leading example in the tropics is Bangladesh, where land productivity and security have been raised by a series of flood control works. More generally, land management which reverses soil fertility decline, for example by raising soil organic matter levels, is a less spectacular but potentially widespread form of land improvement.

Irrigation is the most widespread kind of land improvement; besides its direct action in improvement of water resources, it often leads to an increase in soil organic matter and productivity.

Land improvement also covers the reversal of degradation in its many forms, such as reclamation forestry on eroded hillsides, reclamation of salinized soils by leaching, and rehabilitation of degraded pastures. In resource inventories, it is the balance between degradation and land improvement that gives the net change. Current monitoring of land resource changes is insufficient to be able to say precisely which countries are improving and which degrading their land resources, but the available evidence, not least from field observation, is that the balance is frequently negative. A leading objective of resource management should be to reverse this situation (Srivastava et al., 2002).

## 9. Conclusions

The extent and severity of various forms of land degradation is alarming at present. This is caused by the excessive pressure on land to meet the competing demands of the growing population for food, fodder, fiber, and urban and industrial uses. If we are to meet the future increased demands and also maintain the productivity of the land, there is no alternative, except to manage and protect our scarce land resources more effectively than at present.

The challenge before us is not only to increase the productivity per unit area, which is steadily declining and showing a fatigue syndrome, but also to prevent or at least reduce the severity of the various forms of degradation, which has reached an alarming proportion. If the situation is not reversed at the earliest, then the sustainability of the already fragile ecosystem will be badly affected, threatening the livelihood security of not only the farmers but also every one. The situation needs immediate attention of all the stakeholders, from policy makers to farmers, involved in the management of the limited land resources of the earth.

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# 2

# Water Erosion from Agricultural Land Under Atlantic Climate

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## 1. Introduction

### 1.1 Background

Types and rates of water erosion depend on the following main factors: climate, soil, topography, land cover and use. Agricultural land use removes the vegetative cover resulting in accelerated wind and water erosion. Water flow and its paths are central to the study of water erosion (e.g. Flanagan, 2002). Erosion caused by water is best examined on the basis of the spatial context in which erosion takes place (e.g. Govers, 1987; Ludwig et al., 1996; Souchere et al., 1998). The smallest and simplest catchment can be defined by the area of overland flow adjacent to a single channel. Within a catchment, the major types of water erosion are: interril, rill, ephemeral gully and permanent, incised gully. Interril and rill erosion occur on hillslopes driven by overland flow (e.g. Toy et al., 2002). Rill erosion progresses to gully erosion when deeply incised channels are produced. Ephemeral gullies are periodic refilled by farming operations, whereas permanent incised gullies, which are wider and deeper, are not filled with normal framing operations (e.g. Toy et al., 2002; Flanagan, 2002).
In many areas of Northwest Europe, concentrated flow (rill and gully) erosion of agricultural land are particularly widespread, and this in spite of the low rainfall intensity characterizing Atlantic climate and a moderate topography. From 1980´s onwards, erosion studies have been a matter of interest in several European areas with loamy soils, frequently underlain by loess and/or calcareous parent material. This was the case in Pays the Caux and other regions in North and Northwest France (e.g. Boiffin et al., 1988; Auzet et al., 1993, 2006; Ludwig et al., 1996), South Downs in England (e.g. Fullen and Red, 1987; Boardman, 1990), Central Belgium (e.g. Govers, 1987, 1991; Poesen and Govers, 1990; Vandaele and Poesen, 1995) and the Province of Limburg in the Netherlands (e.g. Kwaad, 1991). Concentrated flow erosion was also described in other European regions with different climate, agricultural systems and soil types, for example in the Scandinavian countries (e.g. Uhlen, 1986; Oygarden, 1996; Hasholt et al., 1997) or in the Lake Leman area (e.g. Vansteelant et al., 1997).

There is also ample information showing soil erosion is a key factor in Mediterranean regions (Solé Benet, 2006; García-Ruíz, 2010). This environment, besides high rainfall intensity, slope gradient and low organic matter content, was traditionally characterised by a land use system (e.g. vineyards, olive and almond orchards) with scarce plant cover, which has been shown to be particularly prone to soil erosion (e.g. Martínez-Casasnovas et al., 2002; Solé Benet, 2006; García-Ruíz, 2010; Nunes et al., 2011). Permanent gullies are not an exception in these regions. Nowadays, rapid changes occurred in the agricultural system (i.e. abandonment of cultivated land, technological development, expansion of wine, almond and olive) might decrease or increase soil erosion rates (García-Ruíz, 2010). However, a recent study showed that erosion rates at the plot scale are generally much lower in the Mediterranean regions as compared to other areas in Europe (Cerdan et al., 2010). This was mainly attributed to high rock fragment content, which would reduce sheet and rill erosion rates. Also the fact that much of the arable land in Atlantic areas of some European regions where erosion has been most extensively studied is located on loess soil could help to explain these results.

Rill and ephemeral gully erosion, showing patterns remembering the loess Belt area, also was typified in Iberian regions located in the transition zone between Atlantic and Mediterranean climate, for example, Southern Navarra (e.g. Casalí et al., 1999; De Santisteban et al., 2006) and Northeast Portugal (e.g. De Figueiredo et al., 1998). Rainfall erosivity in these transitional regions is generally lower than in typical Mediterranean environments, where high intensity rains also are more frequent.

### 1.2 Geographycal context

Regions along the Atlantic coast in Northwest and Northern Spain (Galicia, Asturias, Santander, and Basque Country) are characterized by humid, temperate climate, opposite to Mediterranean regions. Rain intensities are moderate to low, like in other Atlantic areas in Western Europe, extending from Northern Portugal to the Scandinavian countries.

The surface area of Galicia is of about 27950 km$^2$. According to the UNESCO aridity index, Galicia is located at the humid region of Iberian Peninsula (P/ETP > 0.75). Mean yearly rainfall is within the range of 1400–1500 mm (Martínez Cortizas et al., 1999). Rainy months are mostly from October to May. Summers are often characterized by low total rainfall depths and dryness, even though thunderstorms with high-intensity rainfall are more frequent in this season (Font-Tullot, 1983). Thus, due water deficit in summer, the rain regime of Galicia presents, to some extent, transitional features between Atlantic and Mediterranean conditions. Rain erosion rates are expected to be moderate to high in a global perspective (Díaz-Fierros and Díaz de Bustamante, 1980).

In Galicia, traditional agricultural systems were characterized by the small size of fields and by a complex system of terraces and border features separating the fields. For several centuries, thousands of kilometres of stony walls acting as terraces have been constructed at the property boundaries and they have been an important element in erosion control. In recent years properties have been redistributed in some areas, increasing the average field size and facilitating more intensive farming practices. Therefore, nowadays both traditional and intensive management systems are found side by side.

Water erosion has been investigated since 1996 on medium textured (loamy to silty loam) soils developed on parent materials belonging to the Ordenes complex and to lesser extent also on loamy to sandy loam soils developed over granite (Figure 1), in A Coruña province, Galicia (e.g. Valcárcel, 1999; Valcárcel et al., 2003; Mirás Avalos et al., 2009). Results indicated that concentrated soil erosion (rill and ephemeral gullies) also was a widespread phenomenon, so that large parts of agricultural land are affected by soil losses. Similar findings have been reported in the neighbour region of Asturias, also characterized by Atlantic climatic conditions (Menéndez-Duarte et al., 2007).


Fig. 1. Location of fields where water erosion was surveyed.

In our studied area, both Hortonian and non-Hortonian runoff might occur (Valcárcel, 1999). In general, surface runoff is related to Hortonian flow occurrence, which in turn is more frequent on agricultural fields with a seedbed prone to crusting. In these conditions, overland flow is mainly due to low infiltration rates, lower than rainfall intensity. Saturated hydraulic conductivity is not a limiting factor for infiltration, after ploughing or seedbed preparation, but it was found to decrease with increasing cumulative rainfall and crust development so that values as low as 1- 3 $mm \cdot h^{-1}$ have been measured for sedimentary crusts (González García, 1999; Taboada Castro, 2001). Values of saturated conductivity reported for crusted surfaces in medium textured soils of Northern France are of the same order of magnitude (Boiffin et al., 1988). Therefore, low infiltration rates and runoff production are controlled, by two main factors: (1) the presence of a crusted soil surface and (2) a scarce water storage capacity in microrelief depressions. Seedbeds of spring and winter cereals and even those of reseeded grasslands change under the cumulative effect of rainfall becoming land surfaces characterized by poor soil infiltration capacity and poor surface water storage.

Surveys carried out in other Atlantic regions of Europe, as before mentioned, also show that in agricultural fields from temperate-humid Atlantic climate soil losses by rill and gully erosion are much more important than those caused by laminar erosion. Crust formation and surface degradation decrease the infiltration and produce runoff. Frequently, soil

management origins furrows, which favour soil incision, and therefore enhance the formation of rills and ephemeral gullies.
Notice also that during erosive events, apart from soil losses, manure and pesticide transports to surface water bodies can be produced. In fact, attention has been paid soil erosion in agricultural land not only because of concern for loss of soil fertility, but rather because of nutrients originated from the cultivated areas are an issue of concern, because they can threaten water quality.

## 2. Effect of soil surface changes and man-made agricultural features on runoff generation and soil erosion

The properties determining the capacity of an agricultural surface to produce runoff (i.e. soil infiltrability and surface storage) are strongly influenced by the structure of topsoil layers. Factors influencing soil surface structure and formation of soil crusts can be evaluated from field survey data (e.g. Ludwig et al., 1996; Souchere et al., 1998).
The primary mechanism leading to surface crusting was aggregate breakdown (Taboada Castro, 2001), which also produced small particles that are easy transportable by runoff water. Soil crusting consistently follows typical time and space sequences. Kinetics of soil crusting depends on factors such as soil composition (organic matter content, silt and clay content) and initial surface roughness.
To assess soil surface evolution as a function of cumulative rainfall in the soils of the Ordenes complex, field observations have been made after each important rainfall event, particularly when the soil surface was uncovered in late spring and autumn-early winter. Information recorded during this surveys included crusting stage, surface roughness, evidence of overland flow, sheet erosion, ponding and tillage erosion. Aggregate minimum diameter, i.e. the diameter of the smallest aggregates not integrated in the surface crust, also was directly assessed at the soil surface. As long as soil surface crust is developed, minimum diameter of soil aggregates outside this crust increases. Therefore this parameter can be used as a semiquantitative index of soil surface degradation by increased cumulative rainfall. Moreover, where significant rill erosion and/or sedimentation were observed, the site was surveyed for position of the channel and soil losses (Taboada Castro, 2001; Mirás Avalos et al., 2009).
Field identification of crust types and the associated state of degradation provided valuable information for predicting soil surface characteristics determining runoff, mainly infiltration and temporal storage capacity. Sedimentary crusts with a very low saturated conductivity (<5 mm/h) developed from freshly tilled surfaces after cumulative rainfall of about 150-200 mm (Mirás Avalos et al., 2009) or even after 50 mm (Taboada Castro, 2001). After a structural crust has been developed, infiltration capacity during heavy or moderate intensity rains can be about an order of magnitude lower than peak rain intensity. Therefore, field data are useful to evaluate the capacity of each land unit in the catchment to produce runoff and for modelling of concentrated erosion rates and risks (Figures 2, 3 and 4).
Runoff processes and runoff frequency at the catchment level have been found to show a wide complexity (e.g. Ludwig et al., 1996; Valcárcel et al., 2003) mainly depending on the interaction between soil properties influencing the structural state, agricultural practices and climate. For example, fields in long periods of rotation with corn showed high frequencies of runoff (Valcárcel, 1999).

On the other hand, runoff concentrates along features from topographical or agricultural origin. Man-made factors influencing runoff directions and runoff rate at the small catchment scale may be permanent (small roads and ditches) or temporary (ridges, dead furrows, etc.). In addition, size and geometrical configuration of farm fields also have been found to influence erosion rates. Therefore, field boundaries, headlands and dead furrows were mapped in field survey, independently from rill observations. Moreover land use data, including the nature of the crop, the rate of soil cover by growing vegetation and crop residues, the date, nature and direction of the farm operations have been also taken into account (Valcárcel, 1999). Actually, it is shown that rill lengths are determined by the route of the runoff and the location of the rill heads along the route. The route of the concentrated runoff is determined by topography and agricultural land use, which produce different types of linear depression features. These topographical and agricultural features form a runoff collector network which guides the flow to the catchment outlet (e.g. Ludwig et al., 1996).

Therefore, erosion rates have been explained taking into account the hydrographical structure of each catchment, which depends on both topography and lineal agricultural features. This is because concentrated flow erosion results from the hydrological connection between a runoff-contributing area where soil detachment does not necessarily occur and a collecting channel where flow discharge and velocity exceed the critical values for rill initiation and development. The hydrological structure of a catchment can be determined by identifying runoff collectors, runoff-contributing areas and the connection network between them (e.g. Auzet et al., 1993; Ludwig et al., 1996).

Both, analysis of concentrated soil erosion surveys and erosion modelling at the small catchment scale require information about soil surface stage and man-made features. Therefore, we focuses on several factors which depend on land use and land management and are thought to be most important for our study conditions: soil crusting, tillage direction, surface roughness, buffer strips and soil cover.

- *Soil crusting*. The stage of evolution of the soil surface has been shown to be associated with the hydraulic conductivity. A recent tilled soil is very permeable. Cumulative rainfall effects produce first a structural and finally a depositional or sedimentary crust. Sedimentary crusting affecting more than 80% of the soil surface has been observed mainly during two periods: in later spring after maize seedbed preparation and in autumn after grassland sowing.
- *Tillage* following the direction of maximum slope, on the one hand, increases flow velocity, causing a higher runoff peak and erosion rates and, on the other hand, reduces the surface storage capacity. Soil tillage perpendicular to the slope direction reduces flow velocity and increases surface storage capacity. Furthermore, which is most important, the drainage network can be fragmented, thus reducing the runoff contributing area. This is like dividing the total area, which originates runoff in smaller ones, producing less runoff and consequently less erosion. However, the link between tillage direction and runoff routing may be sometimes ambivalent: ridges and tracks created by tillage and seedbed preparation can be used as channels, thus promoting concentrated flow and increasing runoff flow velocity.
- *Surface roughness* has been found to change flow direction in catchments with gentle slope. After ploughing roughness is high, about 4 - 5 cm, but after seedbed preparation this figure is reduced to less than 1 cm with a high surface storage capacity, together a

high infiltration rate. Consequently, water depressional storage by microrelief can vary between 10-12 mm for a rough surface and less than 1 mm for a seedbed (Kamphorst et al., 2000).

- *Buffer strips* are a well-known conservation measure, thought not very much used in the area studied in this work. The effects of buffer strips are both, a reduction of the flow transport capacity and the increase of sedimentation. Hedges fences and ridges, left by certain cropping operations, such as digging up of potatoes have been observed to produce similar effects than border buffer strips at the field border.
- *Soil cover* by crop residues reduces the rainfall kinetic energy, diminishing soil detachment and crusting; on the other hand soil cover acts increasing roughness, thus reducing flow velocity.

## 3. Rates of soil erosion from field surveys

Several campaigns of concentrated erosion surveys have been conducted since 1996 in agricultural fields located at a 30-km radius from the town of A Coruña (Valcárcel et al., 2003; Mirás Avalos et al., 2009). Between 1996 and 2010 sedimentary crusts developed from freshly tilled surfaces even during the spring and autumn of 2004, which was the driest year of this time series. Concentrated flow erosion was observed during all the study period, except in autumn-early winter of 2004. Moreover, evidences of overland flow and more or less generalized interril erosion were observed during all the field survey campaigns of the studied time interval.

Table 1 list rainfall amounts from 1997 to 2004, whereas Table 2 shows average erosion rates during the same time period. For the sake of comparison, three subperiods were taken into account: 1997-2000, 2000-2001, which was the wettest year, and 2001-2004. Erosion rates during 1997-2000 were on average 3.29 Mg ha$^{-1}$ year$^{-1}$. These figures are of the same order of magnitude than the 2.68 Mg ha$^{-1}$ year$^{-1}$ averaged for the 2001-2004 timespan. The somewhat greater values of 1997-2000 when compared with the 2001-2004 period are in accordance with the rather higher rainfall of the former. In between, the extremely wet year 2000-2001 yielded soil loss rates by concentrated flow erosion of 36.81 Mg ha$^{-1}$ year$^{-1}$, thus about one order of magnitude greater than the average of the other years studied (Mirás Avalos et al., 2009).

| **Period** | **1997/98** | **1998/99** | **1999/2000** | **2000/01** | **2001/02** | **2002/03** | **2003/04** |
|---|---|---|---|---|---|---|---|
| **April 1 – June 30** | 329.0 | 435.4 | 249.0 | 268.0 | 159.0 | 242.0 | 201.0 |
| **July 1 – September 30** | 78.8 | 122.3 | 207.0 | 172.0 | 160.0 | 102.0 | 130.0 |
| **October 1 – December 31** | 475.5 | 214.1 | 580.0 | 747.0 | 196.0 | 731.0 | 524.0 |
| **January 1 – March 31** | 138.4 | 365.1 | 117.0 | 622.0 | 297.0 | 289.0 | 143.0 |
| **April 1 – March 31** | 1021.7 | 1136.9 | 1153.0 | 1809.0 | 812.0 | 1364.0 | 998.0 |

Table 1. Yearly and quarterly rainfall from 1997-1998 to 2003-2004 at the studied site (units in mm).

Summarizing, in our study area, the main situations of concentrated flow erosion that can be roughly distinguished are:

- No incision or limited rill incision, i.e., below 2 Mg ha$^{-1}$ year$^{-1}$ as, for example, in autumn-early winter and spring of 2004, respectively.

- Generalized rill and limited ephemeral gully incision in the class of mean values between 2.5 to 6.25 Mg ha$^{-1}$ year$^{-1}$. In this case, the contribution of each unit is very variable, ranging from about 1 Mg ha$^{-1}$ year$^{-1}$ to 31 Mg ha$^{-1}$ year$^{-1}$. This was the most common erosion pattern during the study period and was illustrated by observations in spring and autumn 1999, autumn 2002, and spring 2003 and 2004 (see Figures 2 and 3b as examples).
- Generalized ephemeral gully incision, which was observed during the extremely wet winter period, between October 2000 and February 2001. Again, the between site differences in erosion rates were large, ranging from 3.0 to 62.5 Mg ha$^{-1}$ year$^{-1}$. Figure 4 shows an example of heavy erosion observed in February 2001. Notice that erosion was so heavy during this period that not only the topsoil was removed but also the uppermost B horizon was affected.

| **Period** | **Surface** | **Rill + gully** | **Rills** | **Ephemeral Gullies** | **Gully/(rill + gully)** |
|---|---|---|---|---|---|
| | (ha) | (Mg ha$^{-1}$ year$^{-1}$) | (Mg ha$^{-1}$ year$^{-1}$) | (Mg ha$^{-1}$ year$^{-1}$) | (%) |
| **1997-2000** | 36.8 | 10.01 | 7.36 | 2.65 | 26.4 |
| | Average concentrated erosion rate= 3.29 Mg ha$^{-1}$ year$^{-1}$ | | | | |
| **2000-2001** | 10.8 | 397.6 | 46.04 | 351.56 | 88.4 |
| | Average concentrated erosion rate = 36.81 Mg ha$^{-1}$ year$^{-1}$ | | | | |
| **2001-2004** | 53.5 | 8.05 | 6.29 | 1.76 | 21.9 |
| | Average concentrated erosion rate = 2.68 Mg ha$^{-1}$ year$^{-1}$ | | | | |

Table 2. Average soil losses by concentrated erosion (rills and gullies) during subperiods of the 1997-2004 time span.

Fig. 2. Extensive rill erosion on a seedbed at a hillslope.

(a)

(b)

Fig. 3. a) Partial crusting at the soil surface and b) Crusting and rill initiation.

Fig. 4. Gully showing topsoil and subsoil erosion during a heavy erosion period in winter 2000-2001.

Erosion caused by ephemeral gullies supposed 26.4% of total soil losses during 1997-2000 and 21.9% during 2001-2004. Nevertheless, this ratio was much higher during the extremely rainy period of 2000-2001, accounting for 88.4% of total soil losses. Therefore, increased total concentrated flow erosion increases the proportion of ephemeral gully erosion.

The highest risks of concentrated erosion observed during the period of study were found for the following conditions: i) tilled surfaces prepared as seedbeds for in spring, and ii) surfaces also prepared as seedbeds for winter cereal or prairie renovation in autumn-early winter. This matched periods with a high proportion of sedimentary crusting (Valcárcel et al., 2003).

Survey results also showed important differences between ploughed soils and seedbeds, as no significant concentrated flow erosion was found in the former, even with high amounts of rainfall. The absence of runoff generation in the mouldboard ploughed surfaces can be attributed to the important temporal storage capacity in microrelief depressions, which are associated with the high roughness produced by ploughing. Depressional storage of rough surfaces can reach more than 10 mm $m^{-2}$ (Kamphorst et al., 2000). In opposite, because of the low surface roughness, spring and autumn-tilled surfaces, left bare produced high rates of

concentrated flow erosion as shown by measurements done at summer beginning or at the end of the winter, respectively.
Grassland is sowed in autumn on seedbeds with low roughness values, which also resulted in topsoil surface prone to crusting, where runoff was frequent. However, once a protecting soil cover developed, grassland prevented totally concentrated flow erosion. Because a temporal prairie protects the soil surface for a length of three or four years, concentrated flow erosion risk was highly reduced when rotations included temporal pastures.

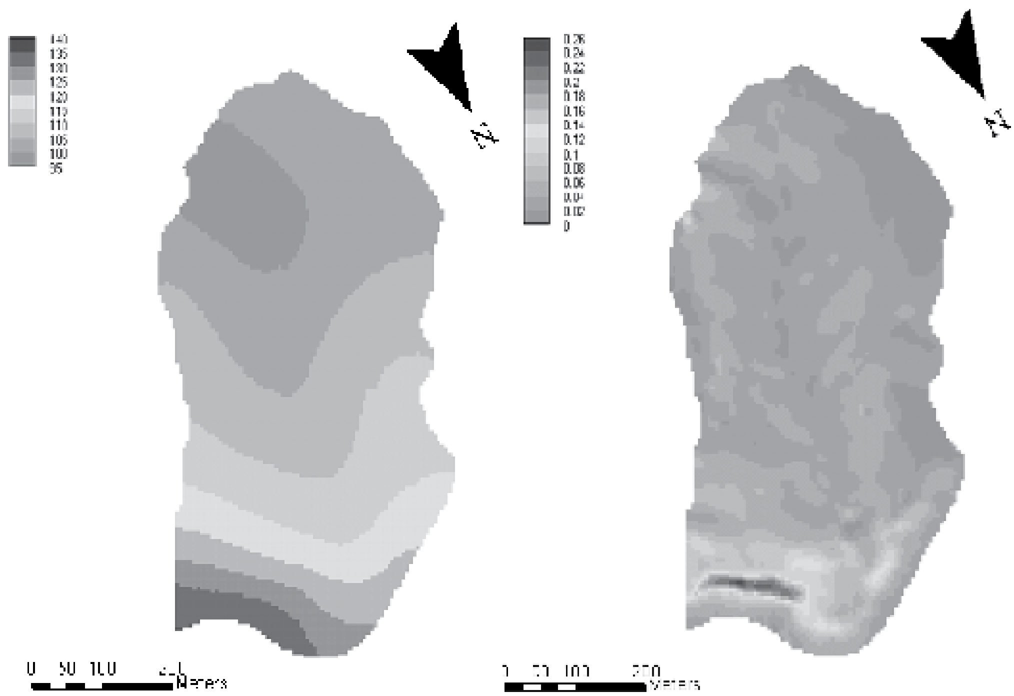
## 4. Modelling soil losses and runoff at the catchment scale

Erosion models consist of mathematical equations that compute estimates of soil loss, together with sediment yield, runoff and sometimes even water quality. These models require input values for climate, topography, soil and land use. Since more than half a century, many erosion models are available, each with particular strengths and limitations (e.g. Toy et al., 2002). Because rainfall, soil, topography and land use vary across a region or a catchment, soil losses and associated variables also show considerable spatial variability. Nowadays most erosion models can be applied in a spatially distributed way, which improves the accuracy of erosion estimations.
Several models have been developed for the purpose of estimating erosion and runoff at the scale of small agricultural catchments. Well-known examples are: CREAMS, ANSWERS, AGNPS, KINEROS, WEPP, EUROSEM and LISEM. Such models perform simulation of water losses and soil losses on the basis of meteorological information, crop phenology, agricultural practices and the physical characteristics of the watersheds. In the last decades they have played an outstanding role in hydrological planning and management at the catchment scale.
Distributed models discretize the physical medium in cells of required size, aiming recreation of the main processes of the hydrological cycle. The discretization process allows taking into account water balance and the transfer processes for each cell into which the catchment is divided. Furthermore, distributed models also are able to analyze the hydrological variables and parameters in such a way that the spatial variability found in agricultural catchments is reproduced. This is a very important issue, because parameters such as infiltration rates in cultivated fields change spatially and temporally, in relation with changes of the soil surface condition. However, accurate description of agricultural catchment where the main type of land use is arable cropping remains not easy, because of the large temporal and spatial variability.
LISEM (De Roo and Wesseling, 1996) is a physically based distributed model that estimates erosion at the catchment scale during a rainfall event. The main achievement of LISEM was that the model is fully integrated into a raster geographical information system (GIS) known as PCRaster (Van Deursen and Wesseling, 1992). This model allows assessing the effects of land use changes and to explore several soil conservation scenarios. Moreover, this distributed model pay particular attention to the influence of man-made factors, such as small roads, tillage direction and wheeltracks, and conservation measures, such as grass strips. Examples of tillage factors that can be responsible for the modification of runoff direction are tillage direction, dead furrows, dirt tracks and surface roughness.

### 4.1 Scenario building

Several scenarios were taken into account in our study. All of them used the topography of an agricultural catchment representative of the main conditions of the Ordenes complex area. This catchment is located at Mabegondo (Coruña province) and it is about 25 ha in surface with an average slope 4.17%. Moreover, scenarios were based on rotation schemes, agricultural operations and soil properties gathered during field surveys.

Topography measurements were made by means of an Abney level. Using PCRaster a digital elevation model (DEM) was elaborated, from which basic spatial catchment information was derived. Figure 5 shows DEM (a) and slope maps (b) of the Mabegondo catchment, with a grid size of 5 m x 5 m. The Mabegondo catchment was considered to be divided in 5 fields (Figure 6a). Land use in the field at the uppermost part (number one) was assumed to be grassland and the remaining fields (number two to five) were assumed to be cultivated with maize.



(a) (b)

Fig. 5. Mabegondo catchment: a) Digital Elevation Map (DEM) and b) Slope map.

In this work the LISEM 1.55 version for Windows (Jetten al., 1999) was utilized. The main processes incorporated by the model are: interception, surface storage, infiltration, runoff routing, splash detachment, flow detachment, channel flow, transport capacity of the flow and sediment routing (De Roo and Wesseling, 1996). Last versions of this model pay special attention to the influence of surface sealing, tillage direction and tillage features, like wheeltracks. Several methods may be chosen optionally to calculate infiltration: Green-Ampt (one or two layers), Richards and Holtan. In this work the Green-Ampt equation was used to assess infiltration into one layer soil. Surface storage in micro-depressions was estimated from the random roughness and the slope data.

An input dataset for LISEM consists of a series of raster maps, including:

- Maps based on topography: slope and local drainage direction.

- Land use maps: agricultural drainage network, surface cover, leaf area index, roads, etc.
- Maps with soil hydrological variables: saturated conductivity, initial moisture content, etc.
- Maps for describing soil surface: random roughness, hydraulic resistance, cohesion and aggregate stability.

The scenarios to be simulated roughly represent agricultural and soil conditions during a wet spring, after maize seedbed preparation. The soil surface is expected to have reached an important degree of evolution, similar to the first stage of a sedimentary crust. Information about values assumed for several input parameters at the five fields of the studied catchment are listed on Table 3. So, infiltration and random roughness were considered to be uniform within each field, even if natural variability was observed during our surveys. From the available experimental information, saturated conductivity was considered to be 5 mm $h^{-1}$ for maize seedbeds and 30 mm $h^{-1}$ for grassland (González García, 1999; Taboada Castro, 2001) and random roughness was fixed at 1.2 mm for grassland and 0.9 cm for maize seedbeds (Vidal Vázquez, 2002). In addition, the model requires data sets for initial moisture deficit, Manning n parameter, median diameter of particle size distribution (D50), aggregate stability, cohesion and soil cover, which also are shown in Table 3.

| Field N° | Area (ha) | Crop | Moisture Deficit | Ksat (mm/h) | Wheeltracks Distance (m) | Wheeltracks Width (m) | Wheeltracks Depth (cm) |
|---|---|---|---|---|---|---|---|
| **1** | 7.00 | Grassland | 0.03 | 30 | | | - |
| **2** | 1.88 | Maize | 0.03 | 5 | 12 | 0.4 | 2 |
| **3** | 6.65 | Maize | 0.03 | 5 | 12 | 0.4 | 2 |
| **4** | 1.96 | Maize | 0.03 | 5 | 12 | 0.4 | 2 |
| **5** | 7.49 | Maize | 0.03 | 5 | 36 | 0.4 | 2 |
| **Field N°** | **Manning n** | **D50 (µm)** | **RR (cm)** | **Agg.** | **Cohesion (kPa)** | **Soil Cover (%)** | |
| **1** | 0.2 | 65 | 1.2 | - | 3.25 | 90 | |
| **2** | 0.07 | 40 | 0.9 | 20 | 0.9 | 0 | |
| **3** | 0.07 | 40 | 0.9 | 20 | 0.9 | 0 | |
| **4** | 0.07 | 40 | 0.9 | 20 | 0.9 | 0 | |
| **5** | 0.07 | 40 | 0.9 | 20 | 0.9 | 0 | |

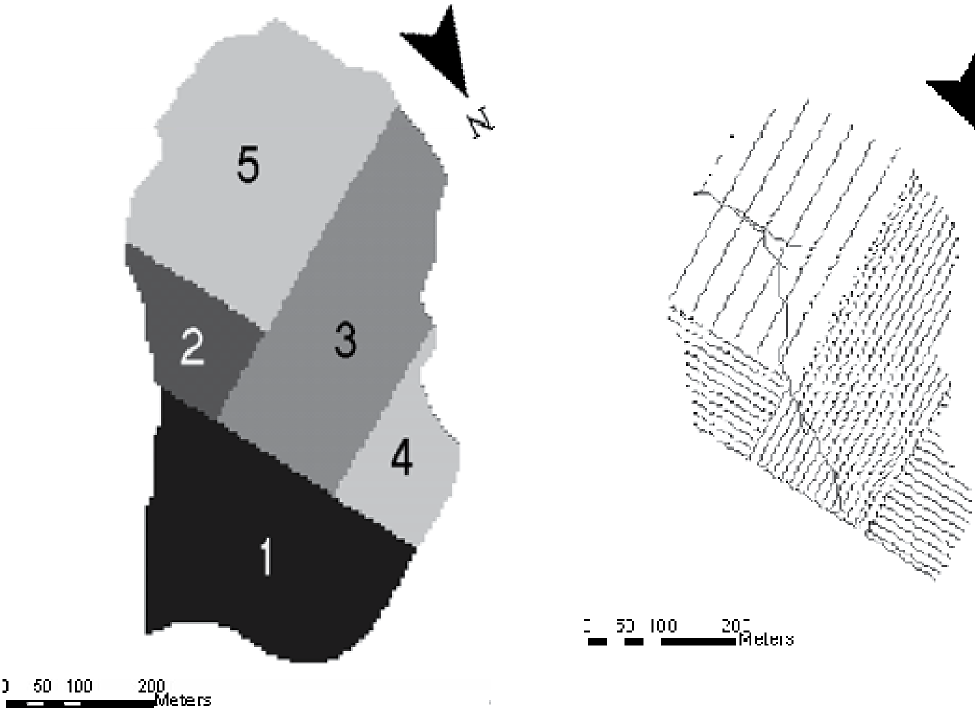
Table 3. Information about land use in the Mabegondo catchment and parameters used for each of its five fields in simulating runoff scenarios (Ksat = Hydraulic conductivity; RR = Random roughness; Agg. = Aggregate stability).

Simulations were carried out using synthetic storms for two different return periods, two and twenty-five years. These were built using the alternate block method, with intensity-duration-frequency data for A Coruña.

Main output of the model are erosion and sedimentation maps, a summary balance file with totals for different simulated terms (total rainfall, total discharge, peak discharge, total soil loss, etc.) and a time series file with information about discharge, solid discharge and sediment concentration in the catchment outlet. Optionally, also runoff maps at imposed intervals during the simulated event.

### 4.2 Predicting the effect of soil conservation measures and tillage features

As examples, next we show results for the simulated effect of grass strips at the borders of the cultivated fields and for taking into account wheeltracks as runoff channels. We assumed a 2 m width of the buffer strip, whereas wheeltracks of 0.4 m width and 2 cm depth were defined along the field largest side at 12 m intervals (Figure 6b, Table 3). Four scenarios were analyzed: 1) neither wheeltracks, nor buffer strips, 2) buffer strips and no wheeltracks, 3) wheeltracks and no buffer strips and 4) buffer strip plus wheeltracks.




(a) (b)

Fig. 6. a) Mabegondo catchment: a) Distribution of the five fields used for scenario building and b) Map showing the position of wheeltracks on the four tilled fields. Topographical channels are also depicted.

Results are summarized in Table 4. Total runoff, peak runoff and erosion rates were very higher for scenarios with 25 year return period compared to those with a 2 year return period, which is an obviously expected result. Grass strips at the field borders decreased the discharge at the outlet of the catchment from 435.46 to 327.32 $m^3$ and from 3360.14 to 3174.60 $m^3$ for a two year and a twenty-five return period, respectively. In terms of discharge/rainfall ratios the reduction was from 9.29% to 6.99% and from 34.18% to 32.30%, respectively. Peak discharge also decreased for the two studied rainfall scenarios. For rainfall intensities with a two year return period erosion rates were less than 1 Mg $ha^{-1}$, whereas for a twenty-five years return period erosion rates were between 4 and 5 Mg $ha^{-1}$. Erosion rates when grass buffers are used would be reduced by 45.6% and 16.5% for rainfall intensities with two and twenty-five years return periods, respectively.

| Scenario | Without grass strips and without channels | | Grass strips | | Wheeltracks as channels | | Grass strips + Wheeltracks as channels | |
|---|---|---|---|---|---|---|---|---|
| Return period (years) | 2 | 25 | 2 | 25 | 2 | 25 | 2 | 25 |
| Total Rainfall (mm) | 18.76 | 39.35 | 18.76 | 39.35 | 18.76 | 39.35 | 18.76 | 39.35 |
| Total Infiltration (mm) | 16.64 | 25.38 | 17.08 | 26.06 | 15.62 | 24.12 | 15.89 | 24.49 |
| Total Discharge ($m^3$) | 435.46 | 3360.14 | 327.32 | 3174.60 | 690.08 | 3.673.92 | 622.43 | 3581.18 |
| Peak Discharge ($l\ s^{-1}$) | 235.43 | 1965.85 | 142.67 | 1373.09 | 518.79 | 2553.71 | 418.22 | 2183.77 |
| Discharge/ Rainfall (%) | 9.29 | 34.18 | 6.99 | 32.30 | 14.73 | 37.32 | 13.28 | 36.43 |
| Average Soil Loss (kg $ha^{-1}$) | 685.59 | 5174.36 | 470.79 | 4441.32 | 1497.60 | 13491.42 | 1224.59 | 13255.09 |

Table 4. Summary of simulated results for scenarios taken into account the effect of grass strips or/and wheeltracks.

When wheeltracks are modelled as channels allowing runoff routing, both total and peak discharge considerably increase. The discharge/rainfall ratio rises from 9.29% to 14.73% for a two year return period and from 34.18% to 37.32% for a twenty-five year return period, indicating that in relative terms channelling effects are more important in the former than in the later scenario. Peak discharge also rises comparatively more for the two year return period than for the twenty-five return period. Also average soil losses are more than two times higher when wheeltracks are taken into account, increasing from 0.69 to 1.50 Mg $ha^{-1}$ for a storm with a two year return period and from 5.17 to 13.49 Mg $ha^{-1}$ for a twenty-five year return period storm. Therefore, according with the simulation results wheeltracks effects are greater on soil losses than on water losses at the catchments scale. Taken into account tillage features for scenario building and assuming parameters (infiltration, surface roughness) corresponding to a crusted soil surface, soil loss results are of the same order of magnitude than those based on field surveys.

The use of grass strips in the scenario with wheeltracks somewhat reduces runoff and erosion rates, as expected, but the effect of this conservation measure is rather limited. So discharge/rainfall ratio decreases from 14.73% to 13.28% under the two-year return period scenario and from 37.32 to 36.4% under the twenty-five years scenario.

The above results show that erosion simulation using a distributed model is able to take into account the hydrological structure of the studied unit (i.e. a hillslope or a catchment), which should provide further insight to analyze the variability of erosion by concentrated flow. This way allows an adequate assessment of the total erosion rate. Moreover, using some simplifying hypothesis distributed models can help in identifying runoff collectors and runoff contributing areas.

Both, field observation and modelling provide complementary information, which allow overcoming the scarce information on concentrated flow erosion in the regions of Atlantic

Spain and should be useful for a sustainable management of agricultural land with the aim of reducing water erosion risks.

## 5. Summary and concluding remarks

Regions along the Atlantic coast in Northern and Northwest Spain are characterized by humid, temperate climate. Rain intensities are moderate to low, lower than in Mediterranean regions. In our study area, traditional agricultural systems were characterized by the small size of fields and by a wide system of terraces with stone walls and various border features, separating the individual fields. For several centuries thousands of kilometres of walls acting as terraces have been constructed at the field boundaries and they have been an important element in erosion control. In the last decades, properties have been redistributed increasing the average field size and facilitating intensive farming practices. Attention has been paid to agricultural soil erosion not because of concern for loss of soil fertility, but rather because of nutrients losses. In spite of the relatively low erosivity, concentrated flow erosion is widespread on agricultural fields and/or small catchments. Increasing surveys on soil erosion from agricultural land at the field and catchment scale that rill erosion is a common feature in most of the years, whereas in some years heavier gully erosion occurs. Thus, concentrated flow erosion has been demonstrated to be the most important water erosion type in Galicia and other regions of North Spain, which is in agreement with erosion features before described other areas of Atlantic Europe.

Cropping systems are partly responsible for concentrated flow erosion, which may become a severe environmental problem. Interactions between farm operations, climate and soil texture induce complex and rapid changes in topsoil structure and its hydraulic properties. For example, it has been shown that properties determining the capacity of the land to produce runoff, such as soil infiltration and surface storage are strongly dependent on the crusting of the soil surface layer. Concentrated flow erosion most frequently takes place on seedbeds and recently tilled soils in late spring and autumn or early winter, but heavier erosion episodes may occur in every season when the soil surface is left bare. In most of the studied cases ephemeral gully erosion may cause significant soil loses, ranging from 2 to 5 Mg $ha^{-1}$ for a single season; however locally erosion rates may reach between 25 and 50 Mg $ha^{-1}$.

A modelling approach was used to predict erosion at the catchments scale for a given event. This model is spatially distributed, so that it allows taking into account interaction within different fields in runoff production and soil losses. The input data are topography rainfall characteristics, soil surface state and other soil physical properties (infiltration, surface roughness, etc.). The influence of field geometry, agricultural features at the field border (for example headlands and dead furrows), tillage marks (for example wheeltracks) and conservation practices (for example buffer strips) that influence runoff and soil losses can also be analyzed by the model. Both, field observation and modelling provide complementary information, which allow overcoming the rather scarce knowledge on concentrated flow erosion in North Western Spain and are thought to be useful for a sustainable management of agricultural land with the aim of reducing water erosion risks.

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# 3

# Micro-Topographic Characteristics in Coordinate with Surface Erosion

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## 1. Introduction

In the process of water erosion, the transformation of surface morphology itself has an impact on the production of surface runoff, water flows, convergence, thereby affect the evolution of soil erosion types and erosion sediment yield (Burwell, *et al.,1968;* Johnson, *et al.,1979;* Onstad *et al.,1984*). From 1960s, many scholars had carried out a series of studies on the water erosion evolution process, including the research about surface morphology factors, measurement and design procedure, surface hydrological erosion processes and so on(Allmaras,*et al.*, 1972; Lehrsch, *et al.*, 1987; Wang, *et al.*, 2005). However, due to the randomness of surface topography in the process of erosion, there still had some limitation to people's understanding about the relationship between soil erosion and surface morphology(Huang, *et al.*, 1992, 2001, 2003) .

In this study, it simulated different surface erosion periods under different surface tillage micro-morphology. Based on the comparative analysis and quantitative study, the characteristic of micro-morphology with its response mechanism in coordinate with surface erosion was illustrated. The results would be helpful to promote understanding of soil erosion processes in depth.

## 2. Relative concepts

Soil surface morphology was the focus of this study, so it was necessary to have a specific definition about relative concepts.

### 2.1 Micro-topography

From the perspective of geography, geomorphology refers to the fluctuant forms in the Earth's surface, such as plains, basins, hills, plateaus, valleys, etc.. So the geomorphology was belong to a macroscopical term, it was also known as a large topography or terrain.

Relative to the megarelief, micro-topography means the undulating surface configuration with slight fluctuations of relative elevation (usually no more than 5-25 cm) in a relatively small area. It can be simplified to 3 kinds of terrains — the plane, slope and uneven terrain, which correspond to a row of probe points and their elevation values can be fitted into three kinds of two-dimensional geometric models — horizontal lines, diagonal lines and curves in the X-Z plane. The undulating terrain feature of micro-topography in loess tillage slope generated by human management is not only the direct result of slope soil erosion but also the important reason leading to the further development of slope erosion. It is a composite factor which can reflect various elements in dynamics of slope erosion as well as their interaction. Research on the topographical features and spatial variability of loess tillage slope has great significance in the formation and evolution of soil erosion and can provide data support for the construction of a prediction model which can reveal the transforming relationship between different erosion modes of the slope(Xue, *et al.*, 2008).

### 2.2 Micro-slope & micro-aspect

At the grid scale, micro-topography can be divided into uniform geographic grid cell. While the concept of micro-slope and micro-aspect mean the grid slope and grid aspect Micro-slope has the feature of relatively shallow depth, gentle terrain, and it could be extended to micro- topography.

It can be classified to 9 grades: <5 °, 5-10 °, 10-15°, 15-20°, 20-25°, 25-30°, 30-35°, 35-40°, >40°; while micro-aspect can be classified to 8 directions followed by true north, northeast, east, southeast, south, southwest, west, northwest.

## 3. Method

### 3.1 Experimental design

Experiments will be carried out in artificial simulated rainfall lobby of State Key Laboratory of Loess Plateau Soil Erosion and Dryland Farming (side-spray style automatic simulation rainfall system with nozzles height of 16m). is 2.0 m × 1.0 m × 0.5 m and the slope can be adjusted in the range of 0° ~ 30°.

Choose the sloping surface soil (0 ~ 20 cm) in Yangling District in Shaanxi Province for the experimental soil. Yangling District is located in the southern edge of the Loess Plateau with the longitude of 108.72° and the latitude of 34.36°, which belongs to the temperate semi-humid continental monsoon climate; its average annual rainfall is 637.6 mm. The soil type is the Loutu which is gray brown and loose with the granular or mass structure, and the soil particles are mainly silty sand. The main mechanical components are shown in Table 1.

| Particle diameter / mm | Mechanical components / % | Particle diameter / mm | Mechanical components / % |
|---|---|---|---|
| <0.001 | 36.28 | 0.05～0.25 | 2.70 |
| 0.001～<0.005 | 12.89 | >0.25 | 0.12 |
| 0.005～<0.01 | 6.88 | Physical cosmid | 56.05 |
| 0.01～<0.05 | 41.13 | | |

Table 1. Particle size distribution of the experimental soil in 0–20 cm depth

Tillage measures of artificial digging (AD), artificial backhoe (AB) and contour tillage (CT) are commonly used in loess slope farming, so these three kinds of tillage measures are arranged respectively in the corresponding etching tank to simulate different micro-topographic conditions. The main processes are as follows: 1) soil preparation: after the

soil samples are dried, they are seived (mesh 0.5 cm), and filled in the soil tank with 8 layers (soil bulk density of 1.30 g/cm$^3$ and moisture content of 10% ); 2) simulate different tillage measures including: ①contour tillage (CT): conduct the horizontal tillage on the slope surface in the direction which is perpendicular to the slope to form grooves and ridges with the ridge height of 7 ~ 10 cm and ridge distance of 30 cm; ②artificial digging (AD): use the pickaxe to dig on the surface with the depth of 5 ~ 8 cm and distance of 20 ~ 25 cm; ③artificial backhoe (AB): adopt backhoe to cultivate along the surface with the depth of 4 ~ 5 cm. Both AD and AB are gradually laid from the base of the hill to its top, forming undulating hills and depressions. Due to the impact of the slope and tillage modes, ridges and potholes formed by tillage have no spatial symmetry. In order to keep these tillage measures closer to natural conditions, farmers who have long been engaged in the same tillage are employed to set up these tillage measures. In order to make a comparison a linear slope is set as check (CK). Three rainfall intensities of 60 mm/h, 90 mm/h and 120 mm/h are selected and the slope is 15 °; two replications are set.
In the artificial rainfall experiment, sectional rainfall are carried out including 2 phases,: (1) before rain (BeR) stage: different micro-topographic forms has been prepared. (2) the phase when squama-like pits and small-scale overfall appear on the slope is called sheet erosion ($S_hE$).

### 3.2 Elevation data collect and represent

A laser range finder is used to measure the slope elevation informationin 2 sectional stages. The laser scanner has 3 parts: 1) XY table; 2) laser range finder (Leica Lai, the vertical error is less than 3 mm); 3) data acquisition and control system whose principle is similar to the gearing of dot-matrix printer. Each test sloping surface can obtain 3480 elevation points and each point represents the 0.02 m × 0.02 m range of the actual ground. These points have provided a high guarantee for the construction and application of high resolution data model as micro-topographic digital elevation model (M-DEM).
M-DEM is a continuous expressive method for the surface of micro-topography, which can reflect the undulating changes and trivial conditions. In recent years, with the development of geographic information system (GIS) technology(Tang, *et al.*, 2006), M-DEM has become the important data of regional soil erosion research. The setup procedure of M-DEM is as shown in Figure 1.

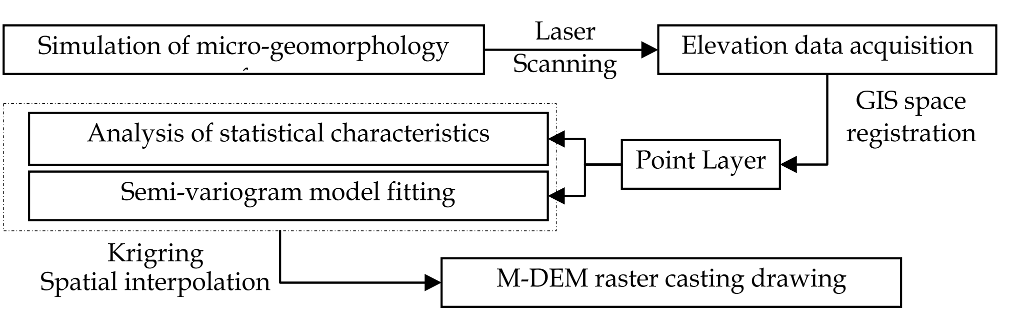


Fig. 1. The flow chart on elevation data collection and representation of the slope micro-topography

## 4. Results and analysis

### 4.1 Statistical characteristics of the relative elevation

#### 4.1.1 Normal distribution test

According to the classical statistical methods, normal distribution test on the density distribution function of micro-topographic relative elevation probability of each tillage slope before rainfall is conducted using the single-sample KS method. Table 2 shows the statistical results of K-S test result on the significance level when $\alpha = 0.05$. Accordingly, the elevation data can be used to generate M-DEMs.

| Tillage slope | Mean /cm | Std. Dev. | Min /cm | Max /cm | Skewness | Kurtosis | Median /cm | K-S Z | Distribution pattern |
|---|---|---|---|---|---|---|---|---|---|
| AB | 20.864 | 1.0622 | 17.8 | 24.1 | -0.25078 | 2.5702 | 21.0 | 5.015 | normal distribution |
| AD | 22.170 | 1.2511 | 18.3 | 25.3 | -0.2783 | 2.5735 | 22.3 | 4.744 | normal distribution |
| CT | 21.046 | 1.6965 | 17.1 | 24.3 | 0.0099255 | 1.8528 | 21.0 | 5.753 | normal distribution |

Table 2. The K-S test of density distribution function of micro-topographic relative elevation probability

#### 4.1.2 Analysis of spatial variability

Semi-variogram fitting curve model is adopted to analyze and describe the structure characteristics of relative elevation spatial variability of micro-topography, as shown in Table 3.

| Rainfall stage | Tillage measures | $C_0$ cm$^2$ | $C_0$+C cm$^2$ | C/($C_0$+C) % | semi-variance distance m | $R^2$ | RSS | Model |
|---|---|---|---|---|---|---|---|---|
| BeR | AB | 0.608 | 1.637 | 62.9 | 2.931 | 0.815 | 0.124 | Exponential |
| | AD | 0.409 | 1.677 | 75.6 | 0.492 | 0.93 | 0.097 | Exponential |
| | CT | 0.088 | 2.832 | 96.9 | 0.091 | 0.425 | 1.430 | Spherical |
| | CK | 0.021 | 0.145 | 85.5 | 1.782 | 0.962 | 5.151E-04 | Exponential |
| $S_hE$ | AB | 0.434 | 0.959 | 54.7 | 1.683 | 0.759 | 0.074 | Exponential |
| | AD | 0.446 | 1.589 | 71.9 | 1.053 | 0.934 | 0.092 | Exponential |
| | CT | 0.006 | 1.54 | 99.6 | 0.101 | 0.534 | 0.432 | Spherical |
| | CK | 0.0384 | 0.2238 | 82.8 | 2.0386 | 0.936 | 8.654E-04 | Gaussian |

Table 3. Statistical attribute values of spatial variability characteristic parameters of loess slope micro-topographic relative elevation with the rainfall intensity 90 mm/h

Semi-variogram is a function chart of the distance, generally represented by the variation curve. In geostatistics, the fitting curve models, corresponding to semi-variogram, which are commonly used including: spherical model, exponential model, Gaussian model, pure nugget model, power function model, Di Weisheng model. Different model selection criteria are: the closer the determination coefficient $R^2$ is closer to 1, the reference value of related equation will be higher; the smaller RSS value is, the better the fitting degree of the fitting model using theoretical semivariogram will be.

The ratios of base effect $C/(C_0 + C) > 75\%$, 75% ~ 25%, and 25% respectively indicate that the spatial correlation of variables is high, medium, and weak (Zuo, *et al.*, 2010). From Table 3, we can know that: with the constant evolution of the erosion processes, the micro-topographic relative elevation of different tillage slopes are in line with the fitting curve model of semi-variogram, which can better reflect the spatial structure; the tillage slope is significantly affected by the anthropogenic factors and stochastic factors dominate the leading role. Generally, the relative elevation values of loess slope micro-topography show a medium or high correlation, whose spatial variability is affected by the combined structural and random factors.

The fitting curve models is the approximation expression of semi-variogram, so it is necessary to conduct cross-examination on model parameters. An advantage of the cross validation test method (a kind of indirect method that combines with ordinary kriging) is that in the examination process the selected model parameters will be modified constantly until they reach a certain degree of accuracy. The basic idea is: in turn assume each measured data point has not been determined and the selected semivariogram model is used to estimate the optimal as well as unbiased value of this point based on n-1 other measuring points applying ordinary kriging( *Yang, et al., 2010*). Then test the rationality of the model by analyzing the error. Krigring cross-examination characteristic value of loess slope micro-topographic relative elevational spatial variability is shown in Table 4.

| | Rainfall | Regression coefficient | SE | $r^2$ | Intercept | r |
|---|---|---|---|---|---|---|
| AB | BeR | 1.021 | 0.007 | 0.866 | -0.43 | 0.93 |
| | ShE | 1.023 | 0.007 | 0.874 | -0.47 | 0.93 |
| AD | BeR | 1.125 | 0.01 | 0.784 | -2.78 | 0.89 |
| | ShE | 1.098 | 0.008 | 0.828 | -2.09 | 0.91 |
| CT | BeR | 1.111 | 0.004 | 0.95 | -2.34 | 0.97 |
| | ShE | 1.065 | 0.005 | 0.929 | -1.34 | 0.96 |
| CK | BeR | 1.033 | 0.007 | 0.867 | -0.63 | 0.93 |
| | ShE | 1.029 | 0.008 | 0.842 | -0.56 | 0.92 |

Table 4. Krigring cross-examination parameters of loess slope micro-topographic relative elevational spatial variability with the rainfall intensity 90 mm/h

The determination coefficient $r^2$ is an important indicator to measure the goodness-of-fit of the linear regression model. We usually use its square root - the correlation coefficient r to describe its relevance. When $|r| \geq 0.8$, it shows a high degree of correlation; when $0.5 \leq |r| < 0.8$, it shows the medium correlation; when $0.3 \leq |r| < 0.5$, it shows the low correlation; when $|r| < 0.3$, it is regarded as non-correlation(Yu, *et al.*, 2003). Accordingly, it shows that the semi-variogram fitting curve model can better fit the spatial variability of micro-topography. Krigring cross-examination and Krigring interpolation results of CT slope in BeR is shown as figure 2.

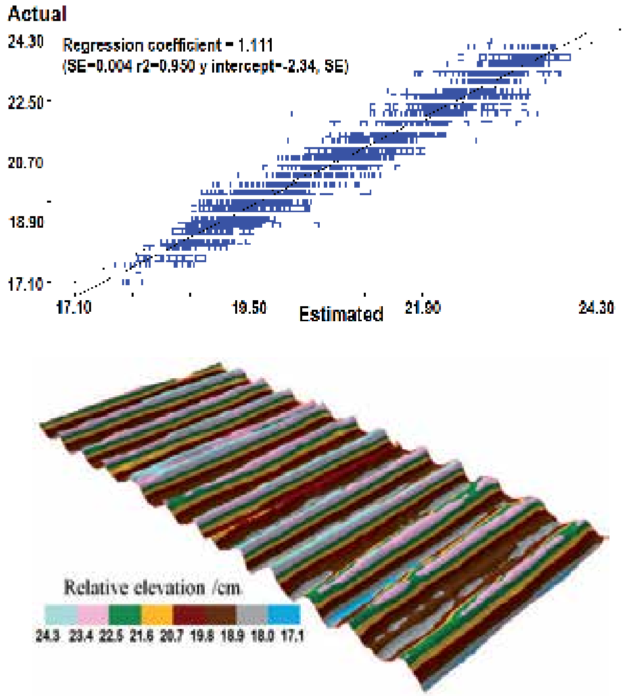


Fig. 2. Krigring cross-validation with its interpolation results of M-DEM of CT micro-topography in BeR.

#### 4.2 Micro-topographic statistical characteristics

Based on the findings above it can be said that the corresponding fitted semi-variogram model can better reflect the spatiotemporal variability of micro-topography after kriging. So, the method can be safety adopted at the application of the micro-topography with the rainfall intensity 60 mm/h and 120 mm/h. The statistical frequency of surface elevational points at different height classes under different tillage practice is shown at figure 3.It shows that the distribution of elevational points at different height classes under different tillage practice are significantly different.

On CK surface, elevational points values are more concentrated than other tillage, and 98% of the them are concentrated in the range of 0.22 ~ 0.24 mm. Correspondingly, its peak frequency is at 0.23 mm; On AH and AD surface, account for about 49.1% and 37.0% elevational points are concentrated in the range of 0.22 ~ 0.26 mm, and their peak frequency of the elevation are at 0.24 and 0.25 mm separately; On CT surface, accounts for 37.4% elevational points values are concentrated in the range of 0.20 ~ 0.24 mm, and its peak frequency is at 0.22 mm; Compared with CK, the statistical frequency of the surface elevation distribution of other micro-topographys are in weak variability and the curve approximately followed the normal distribution.

The variation of elevation at different rainfall process is different. Compared with the BeR, in the ShE stage, the rainfall influence the distribution characteristics of surface elevation in the order of AD> CT>AH, and the rainfall has nearly no influence to CK surface.

Statistical characteristics of slope elevation are calculated (see Table 5) by classical analysis methods (Hillel, 1980). It indicates that the tillage and rainfall have an important impact on the micro-topographic surface.

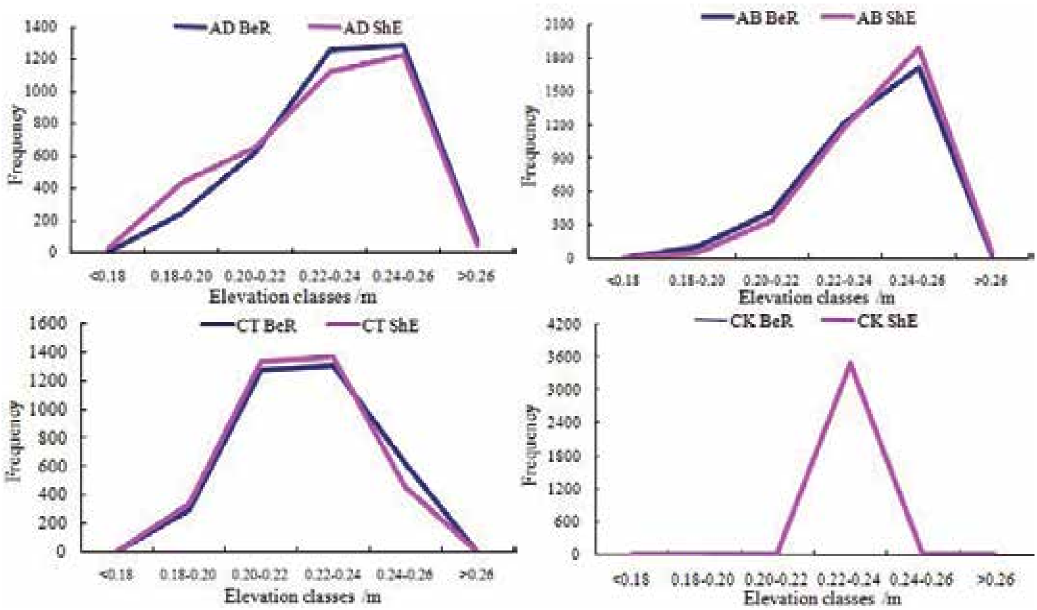


Fig. 3. Surface elevation distribution under different tillage

Due to smooth surface of CK, its $S_{\bar{d}}$ value is relatively small (0.005). While $S_{\bar{d}}$ value of CT, AD, AH surface decreased little separately after rainfall. Furth more, $S_{\bar{d}}$ value of CT surface reduced 22%. It shows the elevation distribution of the CT micro-topography tends to flattened overall in the ShE stage. Mean while, Cv value of CT surface is relatively high, and it shows that CT surface has a larger surface fluctuations than other tillage.

| Statistical parameters | tillage | stage | |
|---|---|---|---|
| | | BeR | ShE |
| Average $\bar{x}$ /m | CK | 0.229 | 0.229 |
| | CT | 0.220 | 0.220 |
| | AH | 0.228 | 0.230 |
| | AD | 0.251 | 0.240 |
| Standard deviation $S_{\bar{d}}$ /m | CK | 0.005 | 0.005 |
| | CT | 0.018 | 0.014 |
| | AH | 0.008 | 0.007 |
| | AD | 0.009 | 0.008 |
| Variation coefficient Cv | CK | 0.022 | 0.022 |
| | CT | 0.082 | 0.063 |
| | AH | 0.035 | 0.030 |
| | AD | 0.036 | 0.033 |

Table 5. Statistical characteristics of slope elevation

### 4.3 Micro-slope characteristics

Grid number to different micro-slope under rainfall intensity of 60 mm/h and 120 mm/h is shown as Figure 4. It is known that grid number is mainly concentrated in the range of: (1) AD: micro-slope of 0° ~ 5° and 10° ~ 20° in the stage of BeR, micro-slope of 0° ~ 5° and 10° ~ 15° in the stage of ShE; (2) AH: micro-slope of less than 25°in the stage of BeR, and no more than 20°micro-slope of AH surface in the stage of ShE; (3) CT: micro-slope of 0° ~ 5°, 15° ~ 30°, greater than 40° in the stage of ShE; (4) CK: micro-slope of no more than 10° in the whole rainfall process.

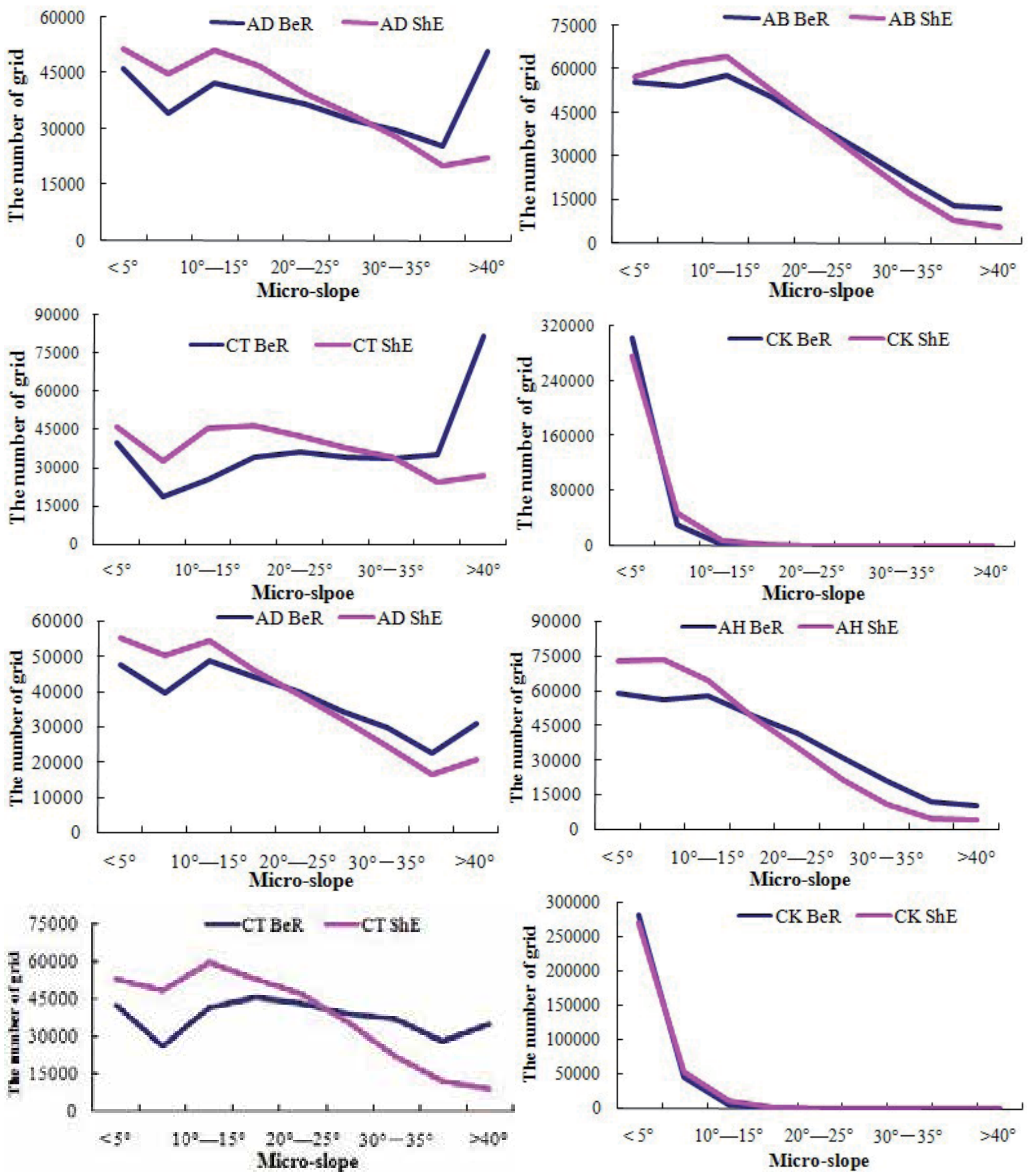


Fig. 4. Grid number to different micro-slope under rainfall intensity of 60 mm/h (A) and 120 mm/h (B)

Compared the feature in stage of BeR with which of ShE, it is known that the rainfall affect little to micro-slope on CK surface. The micro-slope grid number curve of AD, AH and CT surface present 'X-type', and they intersects at micro-slope 15° ~ 35° respectively. At the same time, with the increase of micro-slope, the statistical grid numbers shows decline trend generally. In addition, the intersection under 60 mm/h rainfall intensity is smaller than which under 120 mm/h rainfall intensity.

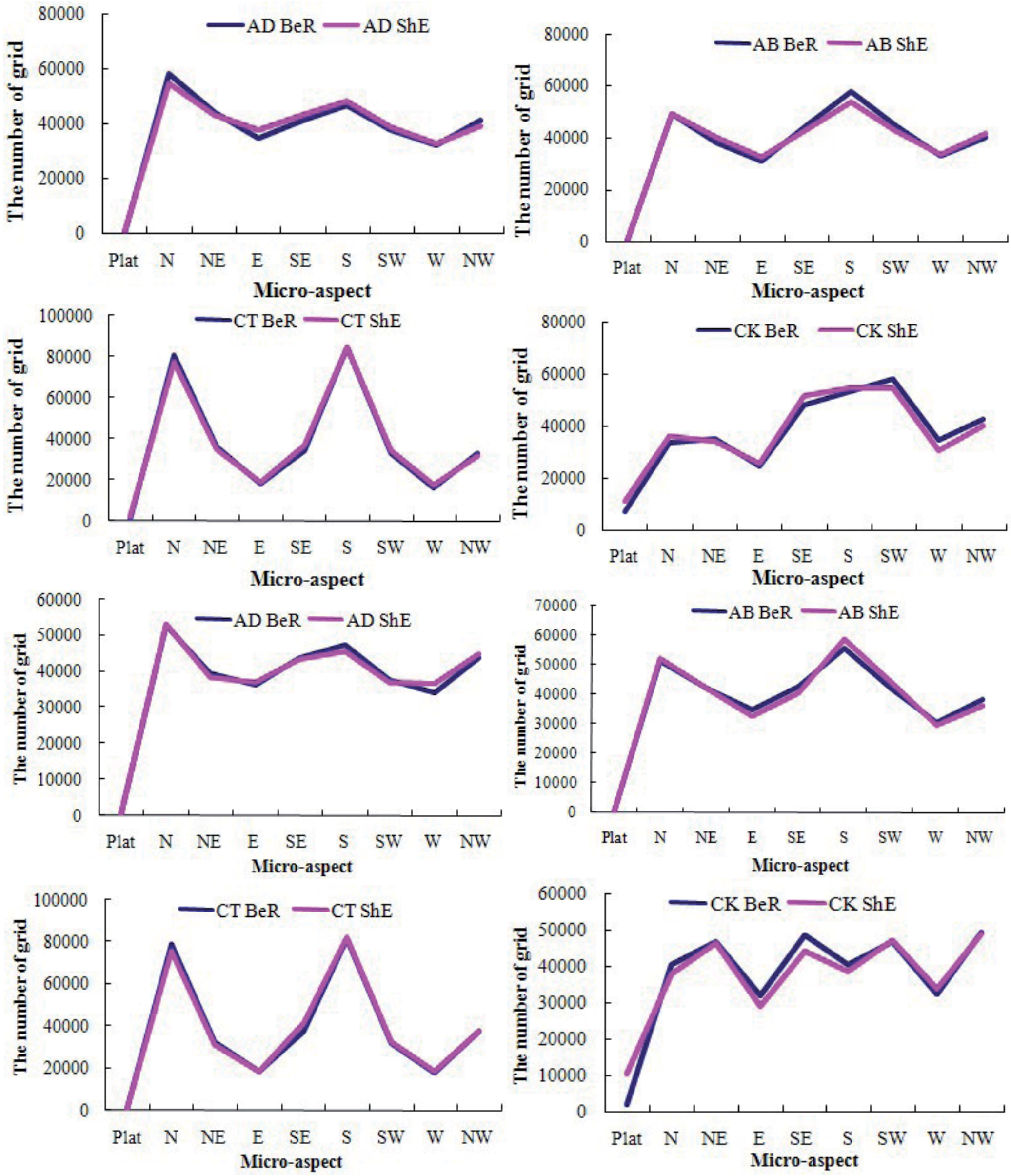


Fig. 5. Statistical grid number of surface micro-aspect under different tillage and rainfall intensity of 60 mm/h (A) and 120 mm/h (B)

### 4.4 Micro-aspect characteristics

Statistical grid number of surface micro-aspect under different tillage and rainfall intensity is shown as Figure 5. Compared with CK, the micro-aspect spatial distribution of AD, AH, and CT surface is significantly different. CT surface micro-aspect is mainly the direction of south or north, while AH and AD surface micro-aspect are the phenomena of contagious distribution at all directions.

At the same time, the 2 grid number curves of surface micro-aspect before and after rainfall are almost overlapped. It shows that though the rainfall can lead to change of surface micro-topography, it has little effect on the micro-aspect. Surface micro-aspect has the same trend of grid distribution whether it is under the rainfall intensity of 120 mm/h or 60 mm/h.

## 5. Conclusions

This paper analyzes the relative elevation variability of micro-topography in 3 typical farming slopes under rainfall intensity of 90 mm/h. and uses the single sample K-S to determine whether the density distribution of relative elevation probability belongs to the normal distribution. On this basis, classical statistical methods are adopted to analyze and describe the spatial variability characteristics. The results have shown that the density distribution functions of relative elevation densities of different tillage slopes are subject to the normal distribution which also have the spatial autocorrelation with weak variation. Secondly, the semi-variogram function is used to analyze the structure characteristics of relative elevation spatial variability of various micro-topography and the fitting curve model to conduct cross-examination. Studies have shown that: With the constant evolution of erosion processes, the semi-variogram fitting curve model can better reflect the variability characteristics of spatial structure of micro-topographys; On the whole, the loess slope micro-topography shows a medium or high correlation, whose spatial variability is jointly affected by the structural and random factors. Thirdly, the law of spatial self-correlation of micro-topographic relative elevation with the evolution of erosion. The results have shown that: the spatial correlation of loess slope micro-topographic relative elevation is generally shown as moderate and high self-correlation, and the index model can highly simulate the relative elevation changes of these micro-topographies. The experimental study on the micro-spatial variability of micro-topographic relative elevation has provided a reference for the further research on the micro-topography change coordinated with soil erosion.

Simultaneously, the definition of micro-topography, micro-slope, micro-aspect are firstly proposed. Based on the laboratory artificial simulated rainfall experiments, combining Laser scanning with GIS analysis, quantitative analysis about the micro-geomorphology characteristics are carried out. The results showed that:

In the ShE stage, the elevation distribution of the micro-geomorphology tends to flattened overall, and the rainfall influence the distribution characteristics of surface elevation in the order of AD> CT>AH;

Rainfall has little influence to micro-slope on CK surface. The micro-slope grid number curve of AD, AH and CT surface present 'X- type', and they intersects at micro-slope 15°~35° respectively;

The rainfall can lead to the transformation of surface micro-topography, but it has little influence to the micro-aspect. Forthmore, surface micro-aspect has the same trend of grid distribution whether it is under the rainfall intensity of 120 mm/h or 60 mm/h. All of these could provide theoretical basis for the in-depth study of mechanism and process of soil erosion.

Data used in this project are collected at micro-topography level, and the corresponding M-DEMs are taken as the basic research subjects. Also these M-DEMs are built at the stage of pre- and after-rain. In this study, it mainly reveal the characteristics of micro-topography in coordinate with surface erosion. In future, the project will expand to gull soil erosion experiments to further reveal the mechanism of soil erosion.

## 6. Acknowledgment

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4

# Losses of Soil and Nutrients from a Purplish Soil on Slopping Lands as Affected by Rain Intensity and Farming Practices

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## 1. Introduction

Increasingly aggravated nonpoint source pollution resulting from intensified farming activities has drawn wide concerns from the society (Quan et al., 2002; Zhu et al., 2005). Among the farming activities, overdose of chemical fertilizers is blamed the most. Sichuan Basin, home of the so-called purplish soil, is one of major grain production basis in China where farming on sloping lands is prevailing and intensive. The purplish soil, a unique soil type developed from an array of easily-weathering purplish parent materials, widely spreads in the basin and dominantly forms the sloping lands. The soil is usually characterized with low contents of organic matter and plant nutrients with poor structure, course texture, high saturated infiltration rate, low conserving ability of soil moisture and nutrients and thus, is very erosive, resulting in shallow top soil layers of less than 20-30 cm lying directly on the underneath parent materials. During rainstorms, the soil is easily saturated with rain water and then quickly forms runoffs discharging into ground waters with rich nutrients, nitrogen (N) and phosphorus (P) in particularly (Zhang, 1992). This has put great pressure on the widespread water eutrophication.

Previous studies documented in literature have well addressed the patterns of soil erosion and surface runoff with which how and how much soil nutrients are lost from the sloping lands. Most of the research was focused on N loss in surface runoff from farmlands (Hamsen and Djurhuus, 1996; Cookson et al., 2000; Havis and Alberts, 1993; Torstensson and Aronsson, 2000; Bergström and Kirchmann, 1999), in subsurface runoff from uplands induced by rains and/or irrigation in north China (Wang et al., 2005; Wang et al., 2006), from paddy fields (Yu et al., 1999; Wang et al., 1996; Wang et al., 1999) and from uplands of the red soil regions (Sun et al., 2003; Ji et al., 2006) in southern China. The factors considered in the related literature included climate (rainfalls, rain intensity, and so on), soil property, land use, etc., under artificial rains impacting on bare soils to study the patterns of nutrient losses (Fu et al., 2003; Kang et al., 2003; Ma et al., 2002). Nevertheless, the workers seldom considered the total nutrient losses from both surface runoff and subsurface runoff in one study but more often in a separate way. There is little, if any, information available on cultivation practices, fertilizer techniques such as fertilizer source, rate, timing and placement affecting nutrient losses from farmlands.

Therefore, the objectives of the field studies were to investigate effects of different rain intensity, fertilizer rates and placement, cultivation practices, mulch materials and its interaction with cultivation practices on amounts of soil erosion, amounts and pathways of nutrient losses during corn growing season from a purple soil on a sloping land in the Sichuan basin.

## 2. Materials and methods

The ongoing field experiments, initiated in 2007, are located in Songtao town, Ziyang city, Sichuan province of China, the upper reaches of Tuo River, one of major branches of Yangtze River, lying between E104°34′12″-104°35′19″ and N30°05′12″-30°06′44″. The area is hilly with an elevation of 395 m above sea level, enjoying subtropical climate with an annual precipitation of 965.8 mm of which 70% is distributed from June to September. The annual average temperature was 16.8 $^0$C ranging from the low of -3.6 $^0$C to the high of 36.5 $^0$C. The soil was a light textured purplish sandy soil and contained 5.1 g/kg organic matter, 0.56 g/kg of total N, 54.5 mg/kg of hydrolysable N, 0.94 g/kg of total P, 6.26 mg/kg of Olsen P, 13.8 g/kg of total potassium (K), and 97.3 mg/kg of ammonium acetate extractable K.

The experiment consisted of nine treatments and three replications, and was conducted in summer by growing corn and followed by winter wheat without fertilizers to homogenize the soil fertility for the next summer experiment. From 2007 to 2010, effects of different cultivation practices, rain intensities, fertilizer placement and interaction of cultivation practices and mulch materials on soil erosion, water runoff and nutrient losses from the sloping lands were studied. The first three experiments were carried out under artificial rainfalls with different intensities, and the fourth one was completely rain-fed. According to the local weather data, the maximum rain intensity in the region was 93.0 mm/h and 30.3 mm/10 min, an incidence of 3.7 rainfalls at 50-100 mm/h annually. Thus, the artificial rain intensities in this study were selected as 0.972, 1.741 and 2.255 mm/min up to 60 mm per rain event and lasted for 61.7, 34.5 and 26.6 min, respectively. During each rain event, no matter where the rain came from, artificial or natural, the surface runoff water, the eroded sediment, the subsurface runoff or seepage water were collected separately. The dry weight of sediment, and nutrients including N, P and K that were contained in the sediment, the surface runoff and the subsurface runoff water were analyzed.

The cultivation methods used in study were selected as down slope ridge cultivation, flat cultivation, contour ridge cultivation and strip cultivation. The first three were the recommended soil conservation practices for the local farmers. The strip cultivation, also a soil conservation technique, was in trial phase. Crop straw and plastic film are two commonly used mulch materials in the region, but the latter is more often used to keep soil moisture from surface evaporation under dry conditions and to raise soil temperatures in early growing seasons of crops. In the experiment examining interactive effects of mulch materials and cultivation methods on soil conservation and nutrient losses, they were combined into five treatments including down slope ridge cultivation + straw, flat cultivation + straw, contour ridge cultivation + straw, down slope ridge cultivation + plastic film and contour ridge cultivation + plastic film.

There were two N rates (300 and 450 kg N/ha), two K rates (0 and 150 kg $K_2O$/ha) and one P rate (150 kg $P_2O_5$/ha), consisting of three fertilizer treatments designated as $N_1K_0$ (farmers' practice), $N_1K_1$ (balanced fertilization), and $N_2K_1$(high N). Since P rate was the same in all the treatments, it did not appear in the treatment symbols. The N fertilizer used as urea (N 46%), P fertilizer as single superphosphate ($P_2O_5$ 12%) and K fertilizer as KCl

($K_2O$ 60%). All P fertilizer was used once at seeding as basal application; K fertilizer was split into 50% at seeding as basal application and 50% before corn earing stage as top dressings; N fertilizer was split into four times, 10% as basal application at seeding, two 20% at seedling stage and 50% before earing stage as top dressings. In the fertilizer placement experiment, there was water irrigation after each top dressing in one fertilizer splitting treatment and without water irrigation in the other to compare effect of irrigation on corn yield and on soil erosion and nutrient losses.
The corn (*Zea mays*) cultivar in the study was Chengdan 18, obtained from the Crop Institute, Sichuan Academy of Agricultural Sciences. The corn, with a plant population of 42000 plants/ha, was usually seeded in early April and harvested in later July.

## 3. Results and discussion

### 3.1 Effect of rain intensity on soil, water and nutrient losses from the purplish soil

Rain intensity, empowering raindrops erosive forces, determines quantity or severity of soil erosion (Foster et al., 1985). This is clearly shown in Fig.1 that the average amounts of water runoff and sediment yield under a heavy rain (2.255 mm/min) were 2.2 and 1.3 folds of those under a medium rain (1.741 mm/min), and 74 and 72 folds of those under a small rain (0.972 mm/min), respectively. The subsurface runoff generated by heavy rain events was measured as 6.0 mm, 90% of medium rain events and 86% of small rains events, reflecting a negative correlation between subsurface runoff and rain intensity. This negative correlation can be attributed to the fact that it takes a longer time for rain water to infiltrate into the soil depth during a small rain event than during a large rain event. Also, heavy rains usually produce much greater surface runoff than small rains. As the rain intensity increased, the total runoff coefficient increased from 0.15 to greater than 0.5 at which surface runoff became dominated. The maximum amount of subsurface runoff accounted for only about one-third of the surface runoff.

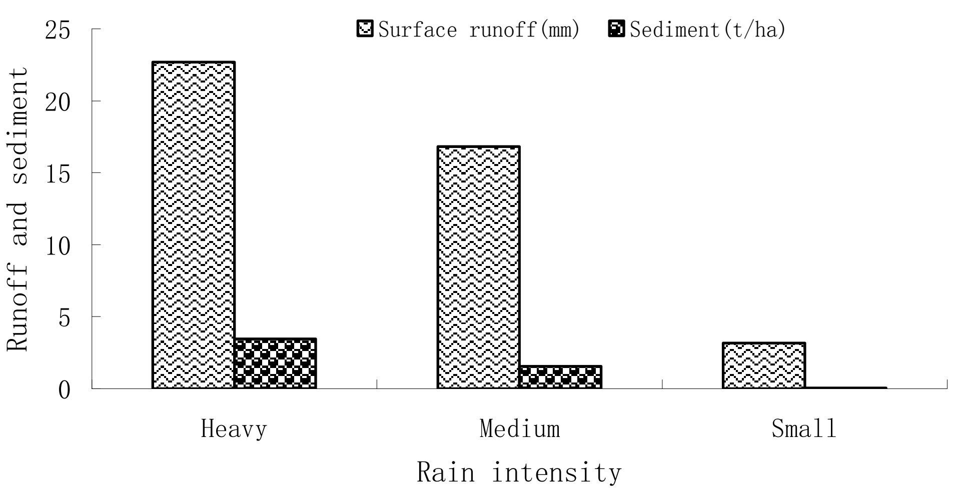


Fig. 1. Influence of rain intensity on amounts of runoff and sediment yield

The total losses of soil N and P from the soil were closely related to rain intensity, while their soluble portions (readily available forms) varied and highly nutrients specific (Table 1). Though P removal from the soil through runoff water tended to be rain intensity dependent, this portion was minor compared to its total loss. The majority of lost P was removed with sediment (often >90%) due to its nature in soil that it is always tightly adsorbed/fixed by the soil particles/colloids. Thus, the extent of P loss paralleled with the yield of sediment generated. The principal pathway of N lost from soil was just opposite to P, i.e., mainly

| Rain intensity | Surface runoff | | Subsurface runoff | | Sediment | | Total | |
|---|---|---|---|---|---|---|---|---|
| | N | P | N | P | N | P | N | P |
| Heavy | 0.81 | 0.03 | 2.68 | 0.01 | 1.80 | 3.18 | 5.29 | 3.21 |
| Medium | 0.32 | 0.01 | 2.87 | 0.01 | 0.90 | 1.48 | 4.09 | 1.50 |
| Small | 0.04 | 0.00 | 3.66 | 0.00 | 0.03 | 0.04 | 3.73 | 0.04 |

Table 1. Amounts (kg/ha) and pathways of N and P losses as affected by rain intensity

through runoff rather than through the sediment. Amounts of N lost were increased with a decrease in rain intensity and found its way into the subsurface runoff. It is suggested that the best method to control soil N loss by rains is to control water runoff, especially for subsurface runoff, while for P it is best to control soil erosion.

### 3.2 Different cultivation practices on soil, water and nutrient losses from the purplish soil

Among three types of commonly used cultivation practices (flat, down slope ridge and contour ridge cultivation) on sloping lands in the southwest China, the flat cultivation caused the most serious soil erosion and water losses (Table 2). Runoff water from the flat cultivation was 1.8 folds of that produced by the down slope cultivation and 25 folds of that by the contour ridge cultivation. The amount of soil eroded from the flat cultivation was 1.9 folds of that by the down slope cultivation in contrast to nil from the contour ridge cultivation during a small rain. The susceptibility of the flat cultivation to form surface runoff may be resulted from the soil characteristics of easy encrustation under direct raindrops striking. The encrustation on soil surface restricted rain water infiltration into soil depths and eventually generated surface runoff. Compared to the flat cultivation, the contour ridge cultivation performed best in control of soil erosion under any rain intensities, but excellently controlled water runoff only under small to medium rain intensities. Under heavy rains, however, the contour ridge cultivation behaved slightly better than the down slope cultivation in reducing water runoff by 2.5%.

Since runoff and sediment are actually the carriers responsible for nutrients migrating out from soil in a rain event, they both determine the amounts and pathways of nutrient losses (Table 2). Among three cultivation practices, the flat cultivation resulted in the highest amount of P loss through the increased sediment loss, while the contour cultivation most effectively lowered P loss via reduction of sediment loss. Amount of N loss from the soil, however, was the least from the down slope cultivation because it facilitated surface runoff and impeded water infiltration to leach soil N from the profile. The results imply that the contour ridge cultivation was highly effective in controlling soil erosion but less effective in reducing N leaching from the purplish soil like playing seesaw.

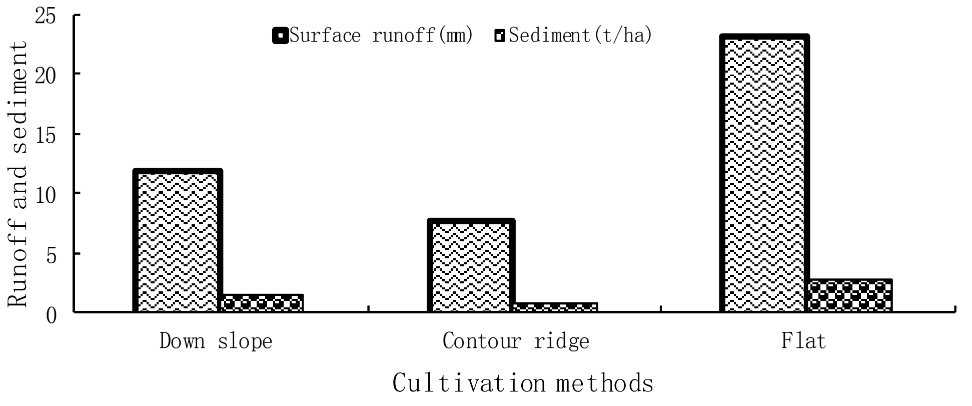


Fig. 2. Amounts of runoff and sediment yield as affected by cultivation methods

| Cultivation | Surface runoff | | Subsurface runoff | | Sediment | | Total | |
|---|---|---|---|---|---|---|---|---|
| Down slope | 0.24 | 0.01 | 2.89 | 0.00 | 0.84 | 1.46 | 3.97 | 1.47 |
| Contour | 0.26 | 0.01 | 3.63 | 0.01 | 0.45 | 0.80 | 4.34 | 0.81 |
| Flat | 0.66 | 0.02 | 2.70 | 0.00 | 1.43 | 2.43 | 4.80 | 2.45 |

Table 2. Amounts (kg/ha) of N and P losses through different pathways as affected by cultivation practices

### 3.3 Effect of balanced fertilization on soil, water and nutrient losses from the purplish soil

Different combinations of N and K fertilizers based on the same P rate had remarkable effect on soil, water and nutrient losses from the purplish soil (Table 3). If K was omitted from the fertilizer treatment, water runoffs and soil erosion, when measurable, from the minus K treatment were always significantly higher than those from the other two treatments under small to medium rain intensities. During a heavy rain event, the $N_1K_0$ treatment produced equivalent amounts of water runoffs and soil sediment to $N_1K_1$ treatment, which were significantly higher in the surface runoff and the sediment and lesser in the subsurface runoff than the $N_2K_1$ treatment. Averaging the data generated from the three rain intensities, the overall runoff coefficient was o.41 for $N_1K_0$, 0.30 for $N_1K_1$ and 0.28 for $N_2K_1$ treatment. The results proved that balanced fertilization with addition of K could considerably reduce water runoff and soil erosion due to the fact that addition of K improved corn growth and land cover which in turn reduced the raindrops impact on soil surface, impeding soil encrustation, increasing water percolation and eventually minimizing total runoff and soil erosion.

| Rain intensity | Fertilizer[1] | Surface runoff | Subsurface runoff | Total runoff | Sediment |
|---|---|---|---|---|---|
| | | ------------------------mm----------------------------- | | | t/ha |
| Small | $N_1K_0$ | 0.00 | 15.00 | 15.00 | 0.00 |
| | $N_1K_1$ | 0.00 | 10.04 | 10.04 | 0.00 |
| | $N_2K_1$ | 0.00 | 10.69 | 10.69 | 0.00 |
| Medium | $N_1K_0$ | 15.48 | 15.21 | 30.69 | 63.39 |
| | $N_1K_1$ | 2.89 | 13.59 | 16.48 | 12.32 |
| | $N_2K_1$ | 5.99 | 14.87 | 20.86 | 16.66 |
| Heavy | $N_1K_0$ | 19.8 | 7.76 | 27.55 | 72.09 |
| | $N_1K_1$ | 19.25 | 6.34 | 25.59 | 72.74 |
| | $N_2K_1$ | 10.66 | 10.58 | 21.24 | 22.54 |

[1] There were two rates of N applied, 300 and 450 kg N/ha as referred to $N_1$ and $N_2$; two rates of K fertilizer applied, 0 and 150 kg $K_2O$/ha as referred to $K_0$ and $K_1$; the rate of P was 150 kg $P_2O_5$/ha for all treatments. The same applies to Table 4 and Table 5.

Table 3. Amounts of runoff and soil erosion through different pathways as affected by rain intensity

| Rain intensity | Fertilizer | Runoff N (kg/ha) | | | Sediment N (kg/ha) | | Total loss |
|---|---|---|---|---|---|---|---|
| | | Surface | Subsurface | Subtotal | Subtotal | Available | |
| Small | $N_1K_0$ | 0.00 | 7.54 | 7.54 | 0.00 | 0.00 | 7.54 |
| | $N_1K_1$ | 0.00 | 7.29 | 7.29 | 0.00 | 0.00 | 7.29 |
| | $N_2K_1$ | 0.00 | 8.07 | 8.07 | 0.00 | 0.00 | 8.07 |
| Medium | $N_1K_0$ | 0.52 | 7.06 | 7.58 | 0.43 | 0.03 | 8.00 |
| | $N_1K_1$ | 0.08 | 5.82 | 5.90 | 0.09 | 0.01 | 5.99 |
| | $N_2K_1$ | 0.11 | 8.57 | 8.68 | 0.11 | 0.01 | 8.79 |
| Heavy | $N_1K_0$ | 0.40 | 7.68 | 8.08 | 0.54 | 0.03 | 8.62 |
| | $N_1K_1$ | 0.25 | 6.28 | 6.53 | 0.27 | 0.04 | 6.81 |
| | $N_2K_1$ | 0.61 | 8.16 | 8.77 | 0.26 | 0.02 | 9.03 |

Table 4. Amounts of N loss through different pathways as affected by rain intensity and fertilization

Different combinations of N and K fertilizers affected N loss from surface runoff, subsurface runoff, total runoff in an order of balanced fertilization ($N_1K_1$)< farmers' practice ($N_1K_0$) < high N rate ($N_2K_1$) (Table 4). The runoff N from the balanced fertilizer treatment was less or significantly less than from the other two treatments under any rain intensity when there was any runoff generated, illustrating that balanced fertilization was able to effectively reduce N loss through runoffs while high N rate intensified its loss. The total N carried away with the sediment in the balanced fertilizer treatment tended to be much lower than in the other two treatments in a medium or heavy rain event. As a whole, the N lost with the sediment was increased with an increase in rain intensity.

The average amount of N loss from different treatments and rain intensities was measured as 7.6 kg N/ha, equivalent to 2.5% of N applied to the corn field, among which the subsurface runoff N dominated, accounting for 88.3%-100% dependent on rain intensity. N loss through the rest pathways was relatively minor (<12%).

The main pathway of P loss in a rain event was somewhat different from N. Since P loss was mainly carried away by sediment, a small rain event that produced little or no sediment yield did not pose much risk to P loss (Table 5).

| Rain intensity | Fertilizer | Runoff P (kg/ha) | | | Sediment P (kg/ha) | | Total loss |
|---|---|---|---|---|---|---|---|
| | | Surface | Subsurface | Subtotal | Subtotal | Available | |
| Small | $N_1K_0$ | 0.000 | 0.005 | 0.005 | 0.00 | 0.000 | 0.005 |
| | $N_1K_1$ | 0.000 | 0.004 | 0.004 | 0.00 | 0.000 | 0.004 |
| | $N_2K_1$ | 0.000 | 0.010 | 0.010 | 0.00 | 0.000 | 0.010 |
| Medium | $N_1K_0$ | 0.013 | 0.010 | 0.023 | 0.68 | 0.005 | 0.699 |
| | $N_1K_1$ | 0.002 | 0.007 | 0.009 | 0.13 | 0.001 | 0.139 |
| | $N_2K_1$ | 0.004 | 0.011 | 0.014 | 0.17 | 0.001 | 0.184 |
| Heavy | $N_1K_0$ | 0.013 | 0.005 | 0.018 | 0.73 | 0.006 | 0.745 |
| | $N_1K_1$ | 0.005 | 0.007 | 0.013 | 0.44 | 0.002 | 0.456 |
| | $N_2K_1$ | 0.012 | 0.004 | 0.016 | 0.34 | 0.003 | 0.359 |

Table 5. Amounts of P loss through different pathways as affected by rain intensity and fertilization

In a medium rain event, amounts of surface runoff P and sediment P were in an order of farmers' practice > high N rate > balanced fertilization; while in a heavy rain event, it was shifted to farmers' practice > balanced fertilization > high N rate. Addition of K effectively reduced P loss.

As shown in Table 6, loss of K from soil was increased with an increase in rain intensity. In a small rain event, there was no surface runoff K because of no surface runoff, but all the K lost went through the subsurface leaching. The balanced fertilizer treatment ($N_1K_1$) significantly reduced amount of K leached out compared to the minus K treatment ($N_1K_0$), but further increased N rate did not help reduce K leaching. In a medium rain event, addition of K significantly reduced K loss from both surface runoff and sediment but only slightly reduced K loss from the subsurface leaching. The overall effect of K fertilization on preserving soil K pool against its loss was excellent in medium rains. In a heavy rain event, however, the balanced fertilization treatment only slightly lowered K loss compared to the minus K treatment while increased N rate behaved much better.

| Rain intensity | Fertilizer | Runoff K (kg/ha) | | | Sediment K (kg/ha) | | Total loss |
|---|---|---|---|---|---|---|---|
| | | Surface | Subsurface | Subtotal | Subtotal | Available | |
| Small | $N_1K_0$ | 0.00 | 0.777 | 0.78 | 0.00 | 0.00 | 0.78 |
| | $N_1K_1$ | 0.00 | 0.397 | 0.40 | 0.00 | 0.00 | 0.4 |
| | $N_2K_1$ | 0.00 | 0.717 | 0.72 | 0.00 | 0.00 | 0.72 |
| Medium | $N_1K_0$ | 0.79 | 0.82 | 1.61 | 12.56 | 0.09 | 14.17 |
| | $N_1K_1$ | 0.27 | 0.647 | 0.92 | 2.49 | 0.02 | 3.41 |
| | $N_2K_1$ | 0.33 | 0.79 | 1.12 | 3.51 | 0.03 | 4.63 |
| Heavy | $N_1K_0$ | 0.86 | 0.427 | 1.29 | 15.40 | 0.12 | 16.69 |
| | $N_1K_1$ | 0.60 | 0.487 | 1.08 | 14.03 | 0.05 | 15.11 |
| | $N_2K_1$ | 0.67 | 0.403 | 1.07 | 7.29 | 0.07 | 8.36 |

Table 6. Amounts of K loss through different pathways as affected by rain intensity and fertilization

Similar to N and P, the pathways of K lost from the soil were through surface runoff, subsurface leaching and eroded sediment. When there was no surface runoff such as in small rains, subsurface runoff or leaching was the only way for K removal from soil. As long as there was surface runoff generated, K could be removed through both runoff and leaching. The partitioning of K in the water phase of runoffs and the solid phase of eroded sediment was highly dependent on rain intensity, land cover, soil property, the size of soil readily available K pool serving both crop growth and leaching. In this study, the soil available K lost in medium to heavy rain events accounted for 7%-27% of total K loss, which was very comparable to what was observed in the common fields (Meng, 2007). This indicated that most of K was lost through soil sediment, accounting for 73% - 93%. Yet, the soil available K removed by the sediment was very low, only 0.02-0.12 kg/ha or 0.4%-1.0% of its total loss. This may be a result of extraction of available K from the sediment by rain water and then dissolving into runoff water which was evidenced by much higher K measured in the surface runoff (Table 6).

### 3.4 Effect of fertilizer placement on soil, water and nutrient losses from purplish soil

There were three methods of fertilizer placement employed in this study, one time application of all fertilizers at seeding, a splitting application that incorporated all P, K and 10% of total N in to soil at seeding, and top-dressing of remaining N twice (20% and 20%) at seedling development and once (50%) before earring stage, of which one treatment coupled with water irrigation after top-dressing and the other treatment without irrigation. From the field observation and measurement, the N contained in the surface runoff was affected by the fertilizer placement to a rather small extent, especially at later stage of corn growth (Fig. 3). The two types of splitting fertilizer applications were always better than one time application in reducing N loss though surface runoff at early corn growing stages. The splitting application coupled with water irrigation could further reduce N loss through surface runoff except one measurement that was caused by a heavy rain right after fertilization. Owing to high N loss (averaged as 3.68mg/kg in surface runoff and 31.74 mg/kg in the leachate) from the plot of one time application, the corn plant suffered from N deficiency at later growing stages and yield loss (data not shown). Thus, one time N application, despite its simplification, is a highly risky practice that sacrifices for environment, crop yield and economic returns.

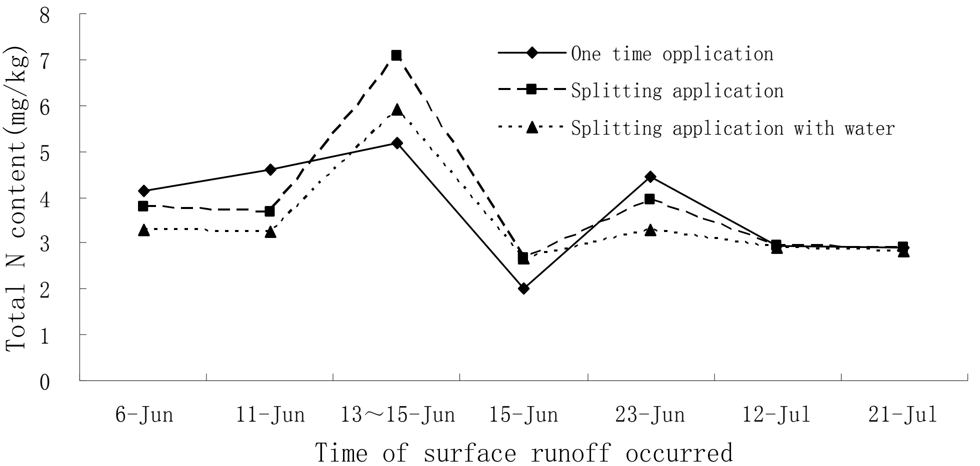


Fig. 3. Total N concentrations in surface runoff as affected by fertilizer placement and time

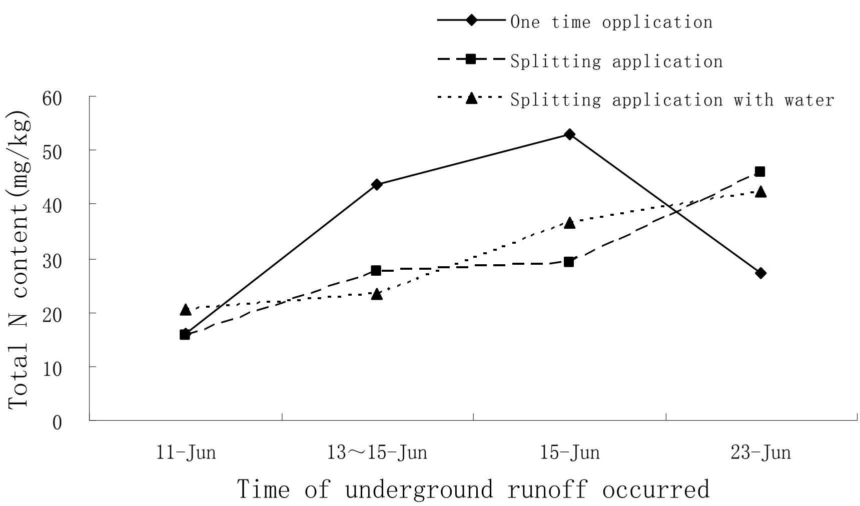


Fig. 4. Total N concentrations through leaching as affected by fertilizer placement and time

P loss from the soil showed an opposite pattern from N that their peaks and troughs were just reversely matched. This can be attributed to the fact that as described above, N loss was mainly though runoffs, especially though leaching, while P loss was mainly through sediment loss. Thus, if a rain event can produce high surface runoff, the subsurface runoff or leaching water from this rain event must be low. Most of P loss was observed in the early to middle growing stages of corn but fluctuated sharply as rain intensity changed (Fig. 5). In the later stage, the loss of P from the soil leveled off as no heavy rains occurred during that period of time. Unlike the sediment P, the P in the leaching water was diminishing with advances of growing season, a reflection of freshly added fertilizer P is more vulnerable to leaching loss than after it reacts with the soil (Fig. 5, Fig. 6). For this reason, the one-time fertilizer application resulted in less P loss through leaching than the splitting application which repeatedly activated the soil P from acidification effect of nitrifying ammonium released from urea after its hydrolysis reaction in the soil (Bouldin and Sample, 1958, 1959). The average total P concentration was measured as 0.097 mg/kg in the surface runoff water and 0.031 mg/kg in the leachate, less than one-third of the P concentration in the surface runoff. This further substantiated that P is always cohered to or adsorbed by the soil particles so that it is virtually less mobile with leaching water or resistant to leaching loss compared to N and K.

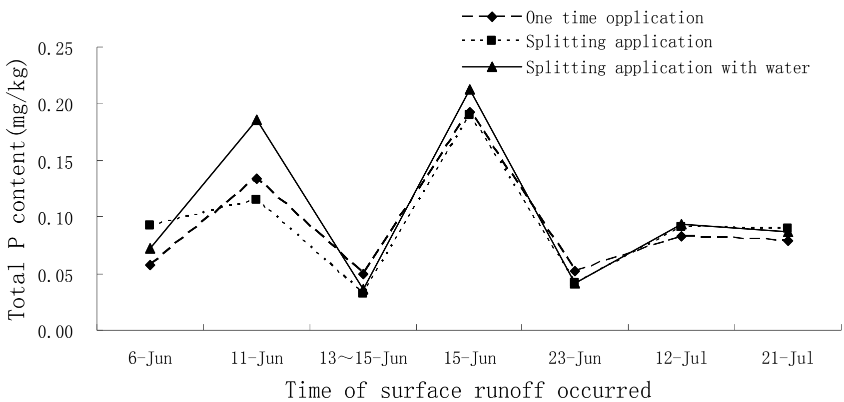


Fig. 5. Total P concentrations in surface runoff as affected by fertilizer placement and time

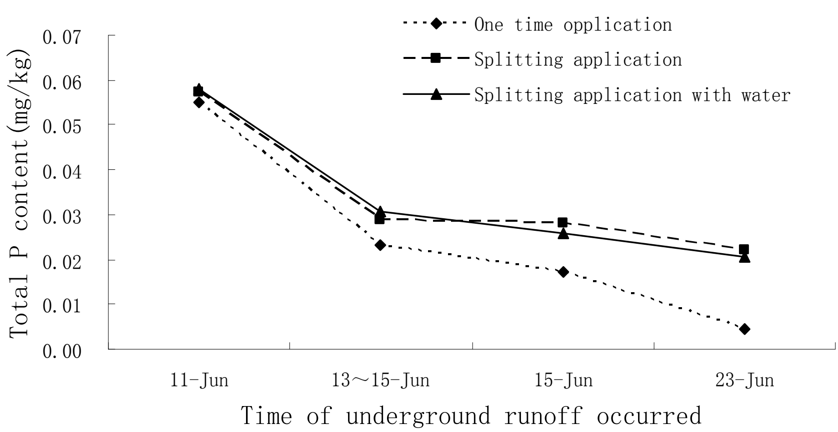


Fig. 6. Total P concentrations through leaching as affected by fertilizer placement and time

The patterns of K loss from the soil were similar to P loss through leaching water, that is, the amount of K removal with surface runoff or leaching water deceased over time (Fig.7-8). The mean concentration of K was measured as 14.9 mg/kg in the first surface runoff water and 6.8 mg/kg in the first leaching water. Thereafter, the K concentrations in the surface and subsurface runoff waters were dropped dramatically no matter what the rain intensities or the types of fertilizer placement were, reflecting that this soil was capable to adsorb K against leaching loss. In June 23, a weak K loss peak occurred, probably due to release of K from the soil K minerals weathering after one-week dry period.

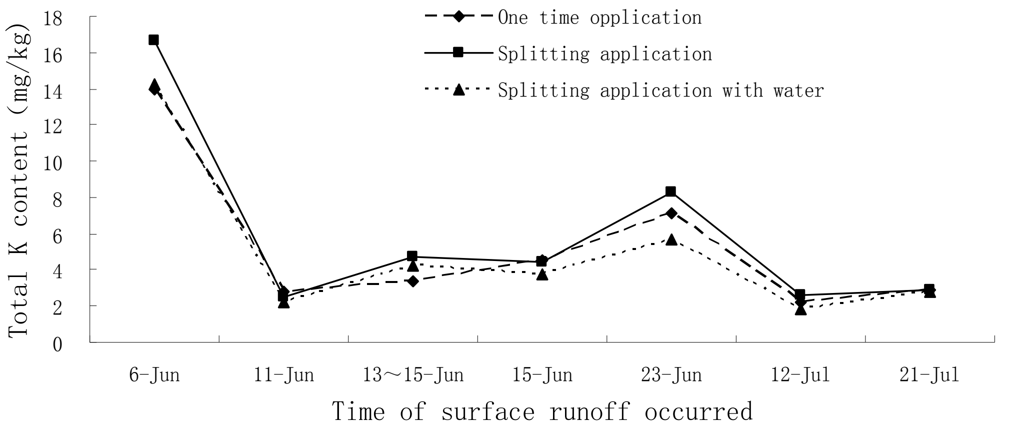


Fig. 7. Total K concentrations in surface runoff as affected by fertilizer placement and time

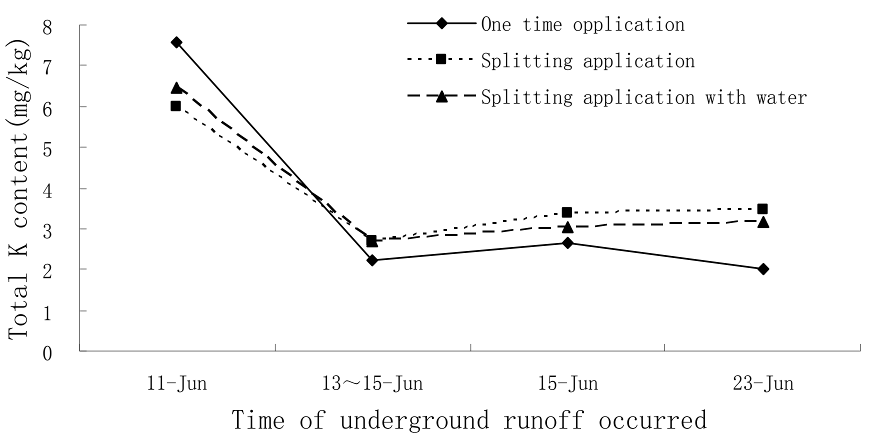


Fig. 8. Total K concentrations through leaching as affected by fertilizer placement and time

### 3.5 Effect of surface mulch on soil, water and nutrient losses from purplish soil

In order to examine the effect of surface mulch on soil erosion and nutrient losses, the experiment was carried out with paired design with no mulch vs surface mulch using wheat straw or plastic film, the mostly commonly used two mulch materials in the region, under the same cultivation practices. There were four types of cultivation practices employed, including down slope ridge, contour ridge, flat (no ridge) and strip cultivation (a soil erosion reducing technique). Compared to the no mulch plots, surface runoff was significantly reduced by 73.9%-86.2% in the plots mulched with straw and considerably reduced by 16.8%-19.8% in the plots mulched with plastic film (Fig. 8). Since the volume of surface runoff water and subsurface leachate from the soil induced by rainfalls were in a reverse relationship of ups and downs, the straw mulch treatments increased subsurface leachate by 15.4%-156.4%, while the plastic mulch treatments reduced subsurface leachate by 38.3%-65.5%. Even though, the total runoff water from the three straw mulch treatments was the lowest among all the treatments. Different combinations of surface mulch with cultivation practices significantly affected the soil erosion (Table 7). Compared to no mulch treatments, the straw mulch was extremely effective in reducing soil erosion.

| Treatment | Surface runoff (mm) | Subsurface runoff (mm) | Total runoff (mm) | Surface runoff/Total (%) | Sediment (t/ha) |
|---|---|---|---|---|---|
| Down slope ridge | 247.0 | 54.1 | 301.0 | 82.0 | 25.4 |
| Flat | 263.7 | 62.9 | 326.6 | 81.2 | 22.3 |
| Contour ridge | 195.4 | 96.4 | 291.8 | 68.3 | 4.3 |
| Contour strips | 197.4 | 33.8 | 231.2 | 85.4 | 9.2 |
| Down slope + straw | 64.5 | 138.7 | 203.2 | 31.7 | 0.5 |
| Flat + straw | 56.4 | 52.7 | 109.1 | 48.7 | 0.8 |
| Contour ridge + straw | 26.9 | 111.3 | 138.2 | 19.9 | 0.1 |
| Down slope + plastic film | 205.4 | 33.4 | 238.8 | 86.4 | 10.8 |
| Contour ridge + plastic film | 156.7 | 71.7 | 228.4 | 67.9 | 2.2 |

Table 7. Soil erosion and water losses as affected by cultivation practices and mulch materials

The sediment eroded from the three treatments ranged from 0.1 to 0.8 t/ha or 2.3% to 3.6% of its corresponding counterpart treatments without mulch. When the down slope cultivation that is usually considered as the most vulnerable practice was covered with straw, the soil erosion was reduced from 25.4 to 0.5 t/ha, making the vulnerable practice anti-erosion. When covered with plastic film, the soil erosion from this plot was cut down by 14.6 t/ha (57%). Contour ridge cultivation turned out to be the best in reducing soil erosion among all the no mulch treatments. After covered with plastic film, soil erosion further reduced by 2.1 t/ha (49%).

The results suggest that contour cultivation was the best in controlling soil erosion and strip cultivation was excellent in reducing water loss when there was no mulch material on the soil surface. Straw mulch could be considered as the best practice for soil and water conservation on sloping lands, which was able to convert the vulnerable down sloping cultivation to a conservative practice.

| Treatment | Runoff N | | | Sediment N | | Total loss |
|---|---|---|---|---|---|---|
| | Surface | Subsurface | Subtotal | Subtotal | Available | |
| Down slope | 12.4 | 17.5 | 29.9 | 22.8 | 1.6 | 52.7 |
| Flat | 16.1 | 23.2 | 39.3 | 19.0 | 1.5 | 58.3 |
| Contour ridge | 9.2 | 29.9 | 39.1 | 3.7 | 0.3 | 42.8 |
| Contour strips | 14.7 | 11.1 | 25.8 | 7.3 | 0.5 | 33.1 |
| Down slope + straw | 3.4 | 42.2 | 45.6 | 0.4 | 0.0 | 46.0 |
| Flat + straw | 3.6 | 16.2 | 19.7 | 0.6 | 0.0 | 20.3 |
| Contour ridge + straw | 1.7 | 25.5 | 27.2 | 0.1 | 0.0 | 27.3 |
| Down slope + plastic film | 11.9 | 8.0 | 19.9 | 9.1 | 0.6 | 29.0 |
| Contour ridge +plastic film | 8.4 | 16.9 | 25.3 | 1.9 | 0.1 | 27.2 |
| Average | 9.0 | 21.2 | 30.2 | 7.2 | 0.5 | 37.4 |

Table 8. Amounts (kg/ha) of N lost as affected by cultivation practices and mulch materials

As stated above, since N is lost mainly through runoff rather than with sediment, particularly through subsurface runoff or leaching process, any practice that can reduce runoff is able to reduce N loss (Table 8). Thus, the contour strip cultivation had the lowest total runoff and N loss (25.8 kg/ha of runoff N) as well among the no mulch treatments in which the runoff N ranged from 25.8-39.3 kg/ha; the flat cultivation + straw and the down slope cultivation + plastic film treatments were able to reduce N loss to its minimal levels (19.7-19.9 kg/ha of runoff N) and followed by the two treatments of contour ridge cultivation + mulching with straw and film (25.3-27.2 kg/ha of runoff N).

| Treatments | Runoff P | | | Sediment P | | Total loss |
|---|---|---|---|---|---|---|
| | Surface | Subsurface | Subtotal | Subtotal | Avail. | |
| Down slope ridge | 0.96 | 0.14 | 1.09 | 26.22 | 0.23 | 27.31 |
| Flat | 0.97 | 0.14 | 1.10 | 22.01 | 0.21 | 23.11 |
| Contour ridge | 0.61 | 0.31 | 0.92 | 4.70 | 0.04 | 5.62 |
| Contour strips | 0.56 | 0.09 | 0.65 | 9.49 | 0.08 | 10.14 |
| Down slope + straw | 0.22 | 0.36 | 0.58 | 0.52 | 0.00 | 1.10 |
| Flat + straw | 0.20 | 0.14 | 0.34 | 0.83 | 0.01 | 1.17 |
| Contour ridge + straw | 0.05 | 0.31 | 0.36 | 0.12 | 0.00 | 0.48 |
| Down slope + plastic film | 0.77 | 0.06 | 0.83 | 10.97 | 0.10 | 11.80 |
| Contour ridge + plastic film | 0.61 | 0.18 | 0.79 | 2.34 | 0.02 | 3.13 |
| Average | 0.55 | 0.19 | 0.74 | 8.58 | 0.08 | 9.32 |

Table 9. Amounts (kg/ha) of P lost as affected by cultivation practices and mulch materials

The overall P losses from the soil induced by rainfalls were relatively small with an average of 9.32 kg/ha (Table 9). Since P loss is mainly through sediment accounting for 24.0%-96.0% of total P loss (an average 92.1%), any farming practices that increase or decrease soil erosion will lead to an increase or a decrease in P loss. Therefore, the down slope cultivation and flat cultivation had the highest sediment yield and P loss, and the three straw mulch treatments produced little sediment and P loss only ranging from 0.12 to 0.83 kg/ha, or 2.5% to 3.8% of the no straw mulch counterparts. Besides, the treatment of down slope ridge cultivation + plastic film was also effective in controlling P loss and followed by the ones with straw mulch.

Amounts of K loss as affected by cultivation practices varied greatly, spanning from the low of 5.6 kg/ha to the high of 558.5 kg/ha, with much greater highs than those of N or P (Table 10). This provided a sharp contrast for the treatment effect in control of K loss. At the low extremes, the contour ridge + straw behaved excellent with a minor loss of K by 5.6 kg/ha in total, accounting for about 5.8% of K lost from the contour ridge treatment and only 1% from the down slope ridge treatment, and followed by the treatments of down slope ridge + straw and flat cultivation + straw.

| Treatments | Runoff P | | | Sediment P | | Total loss |
|---|---|---|---|---|---|---|
| | Surface | Subsurface | subtotal | subtotal | Available | |
| Down slope ridge | 7.5 | 1.0 | 8.5 | 550 | 4.4 | 558.5 |
| Flat | 6.7 | 1.1 | 7.9 | 455.8 | 3.7 | 463.6 |
| Contour ridge | 5.7 | 1.5 | 7.2 | 88.6 | 0.7 | 95.8 |
| Contour strips | 7.4 | 0.5 | 7.9 | 185.9 | 1.4 | 193.8 |
| Down slope + straw | 2.8 | 3.6 | 6.4 | 9.6 | 0.1 | 16.0 |
| Flat + straw | 2.3 | 0.9 | 3.3 | 16.2 | 0.1 | 19.5 |
| Contour ridge + straw | 1.9 | 1.3 | 3.2 | 2.3 | 0.0 | 5.6 |
| Down slope + plastic film | 6.6 | 0.5 | 7.1 | 238.2 | 1.8 | 245.3 |
| Contour ridge + plastic film | 5.4 | 0.7 | 6.2 | 45.8 | 0.4 | 51.9 |
| Average | 5.2 | 1.2 | 6.4 | 176.9 | 1.4 | 183.3 |

Table 10. Amounts (kg/ha) of K loss as affected by cultivation practices and mulch materials

## 4. Conclusion

A comprehensive study was conducted to investigate effects of different rain intensities, fertilizer rates and placement, cultivation practices, mulch materials and its interaction with cultivation practices on amounts of soil erosion, pathways and amounts of nutrient losses during corn growing season from a purple soil on a sloping land in the Sichuan basin. Amounts of surface runoff and sediment eroded from the soil were increased with an increase in rain intensity while the subsurface runoff was inversely proportionate to rain intensity. Effects of straw mulch on reducing soil and water losses were superior to use of plastic film. Straw mulch significantly reduced surface runoff by 73.9% - 86.2% in spite of increased subsurface runoff by 15.4% - 156.4%, resulting in reduction of total runoff by 32.5% - 66.6% and soil erosion by 96.4% - 98.1%. Though use of plastic film alleviated the subsurface runoff and total runoff to some extent, the difference was not significant compared to the traditional cultivation. Amount of N lost from the soil was measured as 37.4 kg/ha, of which N loss through subsurface runoff accounted for 70.1%. The straw mulch decreased total N loss by 12.8% - 65.1% despite some increase in N loss through subsurface runoff. Amount of P lost from the soil was relatively small (9.32 kg/ha) but it was mainly removed out of the field with the sediment, accounting for 92.1% of total P loss. Amount of K lost from the soil reached 183.3 kg/ha of which the loss through the sediment accounted for 96.5%. Mulching soil surface with either straw or plastic film effectively controlled P and K loss. Compared to the traditional, down slope ridge cultivation, the contour ridge cultivation produced higher corn yields while it lowered losses of soil, water and nutrients. The integration of flat cultivation and straw mulch can be a better practice as it does realize straw recycling, improve crop yields and reduce losses of soil, water and plant nutrients.

It was further revealed that the quantity of P lost from the sloping farmland was mainly through the sediment and influenced by rain intensity. Thus, to control P loss, soil erosion must be minimized first. Contour ridge cultivation proved to be highly effective in control of both soil erosion and P loss. Loss of soil N was mainly through runoff, especially through subsurface runoff during a small rain. Thus, minimizing runoff is the best way to control N loss from soil. The conventional contour cultivation tended to increase both subsurface runoff and N loss. In order to control N and P losses in the purple soil area, the integrated agronomic methods such as cultivation practices against soil erosion and water loss, soil depth improvement and organic matter enrichment can be adopted. The high N rate or one-time basal application susceptible to N loss should be always avoided and balanced fertilization that counteracted N loss should be promoted wherever possible. The total amounts of P and K lost from the farmer practice were the worst, indicating the crucial role of proper application methods in reducing P and K losses in the purple soil area.

## 5. Acknowledgement

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# Part 2

## Vineyards

# 5

# Soil Erosion Aspects in Agricultural Ecosystem

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## 1. Introduction

Agricultural activities have an important role in the primary sector of Italian economy, favorable climate characteristics have guaranteed the developing of numerous agricultural products, along the whole national land.
Within the present work, vineyard cultivations have been studied with the aim to analyze the effects caused by the intense cultivation practices, evaluating the annual soil loss and the management methodologies causing an increase or decrease of the erosion phenomenon.
Traditional hillside viticulture uses deep and surface tillage, this technique, also due to increased mechanization, causes deterioration of soil physical characteristics, surface erosion, transport of sediment, nutrient leaching. Controlled grass covering of the inter-rows has proved to improve the stability of soil aggregation, to mitigate soil water erosion by reducing run-off.
Runoff and sediment transport were partially controlled when the vineyards had this structure and when the work was undertaken in the traditional manner with weeding and digging to maintain the soil. The use of tractor cultivation since the beginning of the 1950s has favored the creation of furrows which are able to collect water and to generate channels. The intensive use of chemical weed killers in the 1960s and the absence of cultivation under the plants have enhanced the erosion processes which threaten the long term sustainability of the soil.
Our studies have been focused on the North-West Italy, in Monferrato area, that is characterized by hills landscape and vineyard cultivations. In presence of agricultural activities on sloping lands, the erosion phenomena could be very dangerous in terms of soil loss from organic matter, of great importance for plants growth and landscape quality. Erosive phenomena can be determined by atmospheric agents, but the most important, in terms of generated effects is rain. The erosion caused by rainfall events can be expressed in different ways, referring to the intensity (mm/h), or the water height (mm) and the kinetic energy. The rainfall erosion determines an effect, both in consequence of the rain drop impact, both for water volume flowing along the slope, in case of high intensity. The effect caused by rain impact (splash erosion) is also dependent to the drop dimensions, the erosion effects could be higher if drops increase in size and in

velocity. Velocity changes with the height rainfall, it increases and become constant in a distance of 20 m from the soil impact (terminal velocity), that is reached when the drop weight is balanced by the air resistance.

The rain impacts on soil, influences the soil erodibility that represents soil susceptibility to be eroded. The parameters involved in soil erodibility are soil aggregation, consistency and soil strength. Soil erodibility could be divided in two aspects, the first is the detachability and the second the transportability. In relation to the soil characteristics (i.e. texture), one aspects can be predominant on the others. The transportability of the soil is the most significant phenomenon, especially in presence of intense rainfall and considerable slope of the soil (hills cultivation). The effects of this common combination of factors is represented by channels/rills (that could reach deep dimension), representing the preferential roads causing soil removal. In consequence of the geometric characteristic of the soil (i.e. its slope), a first phase of transportability of the detached soil can be substituted by a deposit in a soil part with a minor slope degree.

The soil sediment transport phase, named runoff, is caused by numerous small channels that are known as rills and the phenomenon is the rills erosion. The soil parts between two or more rills are named interrill and also here erosion occurs, the interrill erosion. Rill and interrill erosion represent the overland flow and are considered diffuse erosion. In the interrill area the rain drop impact and superficial flow are the principle factors responsible of soil erosion, while in rills, soil detachability is caused by water flowing in channels. In presence of cultivated soil, rills have limited deep and length because the superficial morphology of the soil is frequently changed by the agricultural activities (mechanization), and soil micro-topography represents an important aspect influencing the rills characteristics. In Fig 1 has been possible to see as, in not cultivated soil, rills are more defined and deep and length can be interest the whole slope.

The soil loss and the rills formation represent a dangerous threat for vineyards, because determine the removal of a consistent part of organic matter useful for plant growth. Different methodologies are commonly used to reduce the erosion phenomena, as the vineyard arrangement along the contour line in order to break the water flow in the maximum slope lines. Another widespread alternative consists in the contribution provided by spontaneous vegetation growing in the vineyards inter-rows. Nevertheless in the first years (4-5 years) of the vineyard plantation, spontaneous vegetation is not present for the competition developed with vineyard plants, while is recommended after the start-up period. Vegetation provided a contribution against soil loss (De Baets et al. 2006) because limits the impact caused by water splash phenomenon on bared soil and the superficial runoff, increases the soil porosity and improves the water infiltration, reducing the soil compaction. In soil with high water content, vegetation can reduce the moisture level, avoiding also the roots asphyxia phenomenon. Vegetation represents also a source of organic matter for soil enrichment, but needs of management for growth control as the periodical cut or the weed-killer treatment around the plant trunks.

On the other side, vegetation can compromise the vineyard health in cases of water deficit or in areas characterized by limited water availability.

Numerous tests realized in situ on small soil plots (Arnaez et al., 2007) or bigger portion of vineyards (Tropeano, 1984; Cavallo et al. 2010), have demonstrated a reduction of soil loss during simulated or natural events, in presence of vegetated soil, because cover vegetation reduces the kinetic energy of drops impacting on soil and the consequence caused by rills formation.

Fig. 1. Rill erosion on bared soil

The natural erosion phenomenon has been reproduced by simulated events using an experimental equipment on a small soil portion and by the software modeling on slope scale adopting three models most widespread in literature, as showed in Fig.2.

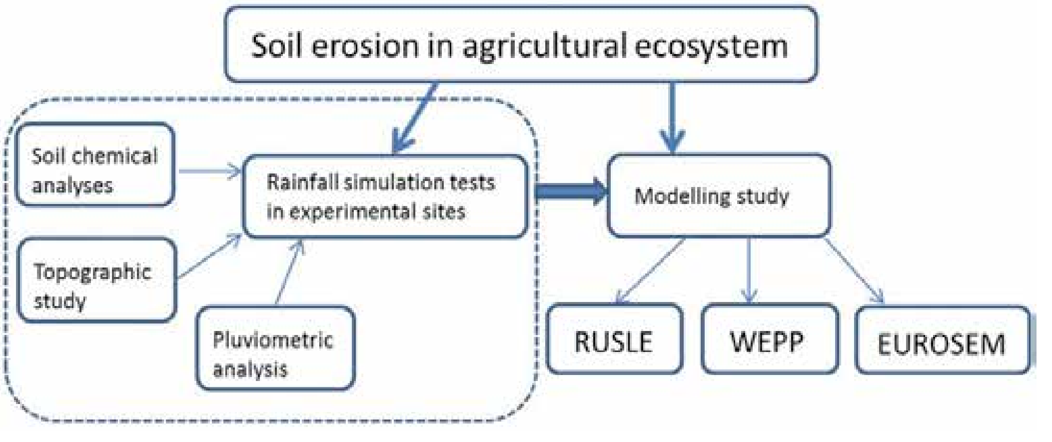



Fig. 2. Scheme of the experimental framework

## 2. Experimental device

The erosion effects caused by natural meteorological events have been studied through numerous tests simulating rainfall phenomena.
A simulator equipment has been built testing clods of 1 m$^2$ with different characteristics of moisture, soil topography, micro-topography and cover vegetation.
The simulator structure has been realized based on pre-existent models of previous research (Cerdà et al. 1997), Fig.3 represents the instrument used in situ for the rainfall tests. The simulator has a superior base of 0.40 x 0.40 cm where the nozzle (Spraying Systems 1/2HH-30WSQ and 1/4HH-14WSQ), was inserted (Fig.4). The rainfall simulator had four telescopic legs reaching a maximum height of 3.50 meters. In our tests the simulator height has been fixed to 2 m as suggested by previous research (Humphry et al. 2002; Arnaez et al. 2007).

Fig. 3. Experimental equipment

Fig. 4. Superior base and nozzle

Other important aspect is represented by the wind protection for simulated rainfall. A protection realized by plastic sheets, has been used as cover simulator to be sure that rain impacts on analysed soil. Through this system wind did not influence the rain direction and the general result of the tests (Fig.5).
The delimitation of the clod interested by the test has been realized by two thin metal sheets of 1m length for each of three sides and inserted within the soil to stop the water runoff in the tested clod (Fig.6).

Fig. 5. In situ rainfall test

Fig. 6. Wind protection

In the forth side, that is the downslope side, an iron gutter pipe has been positioned with the function to reach all the detached runoff from the tested clod. The gutter pipe is connected

by a tube to the sample tank for the subsequent laboratory analyses for the sediment estimation. The rain drops have been simulated using nozzles already proposed in previous case studies (Covert and Jordan 2009)., A pressure gauge has been introduced to measure the pressure of the water before the nozzle exit, between the water tube and the nozzle. Water was available through a tank and a pump positioned near the tested clod.
A calibration of the simulator has been realized before the in situ tests, in order to establish the rain intensity. The chosen intensities present similar characteristics to the real natural values, obtained by a rainfall analysis realized in the meteorological stations nearer to the experimental sites.
The three different nozzle have been used reproducing three different intensity: 40, 80 and 130 mm/h. Last intensity (130 mm/h) is typical of short but very strong summer events. The length of the rainfall events has been set to 20 min, while for the stronger events only 10 min, because more representative.
During the simulation numerous samples (water and sediment), have been collected in order to monitor the runoff during the test. It was established that the first information was the starting runoff. The subsequent sampling was realized each 5 minutes and its sampling time was 1 min.
The collected samples (4 or 5 in consequence of the starting runoff time), have been analysed through laboratory tests by which the samples have been dried in oven at 105° C for 24 hours. The dried matter permits to know the runoff amount and sediment soil loss distribution during the whole test length.

## 3. Experimental site

The experimental sites, choose for the realization of the rainfall simulated tests, involve vineyards with different characteristics. In presence of slope, the methodology for plant cultivation can be adapted to the soil characteristic and agricultural management. The most widespread techniques, concern the row plants disposal along the contour line or perpendicular to them, following the maximum slope lines. The experimental site analysed during the following tests present the contour line configuration of plantation, is in Piedmont region (North-West Italy), in a typical hill area named *Monferrato*. The experimental site is located in Castel Boglione town (near to Nizza Monferrato and Acqui Terme towns). Castel Boglione is extended on 12 km$^2$ at 260m a.s.l. *Monferrato* area involves the provinces of Asti and Alessandria and is almost exclusively characterized by hills. *Monferrato* area can be divided in three parts, *Low Monferrato*, including Alessandria province between Po and Tanaro rivers; *Up Monferrato* in South direction Between Bormida valley and Ligurian Appennino and the *Monferrato Astigiano* including the Asti province, delimited by Belbo and Versa torrents.
The experimental site is located at 200 m a.s.l. (4 km far from Nizza Monferrato), between the *Monferrato Astigiano* and *Up Monferrato* areas (Fig.7, 8).
The studied vineyard presents a cultivation of *Barbera* grapevine (typical of Piedmont region) and a disposal on row following the natural contour lines. Soil has been analyzed through chemical and physical analysis. The soil has a fine texture with a plasticity index > 40 and classified by the USDA classification as clay-loam, with the following percentage (Fig.9).
The chemical-physical analysis showed the soil as alkaline, with mean calcareous content and low content of organic matter.

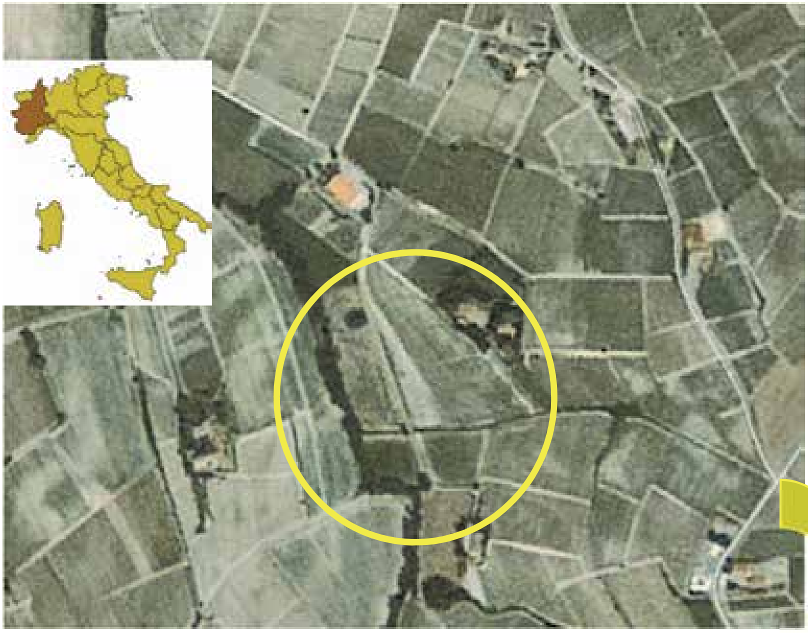


Fig. 7. Aerial image of experimental site

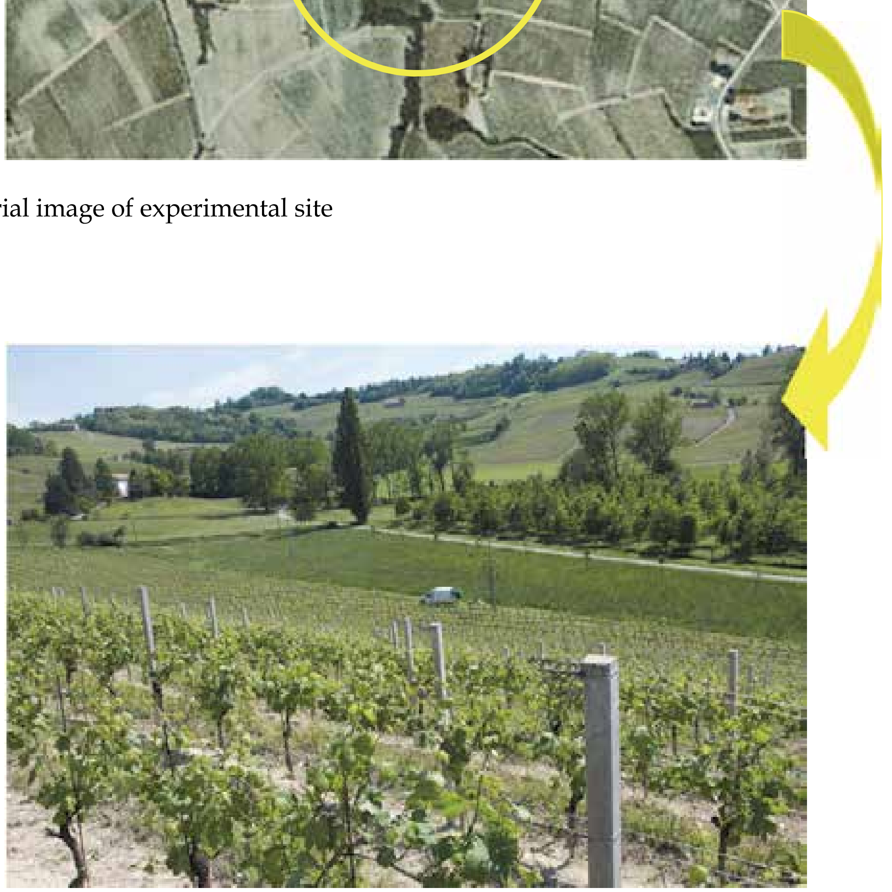


Fig. 8. Reference vineyard

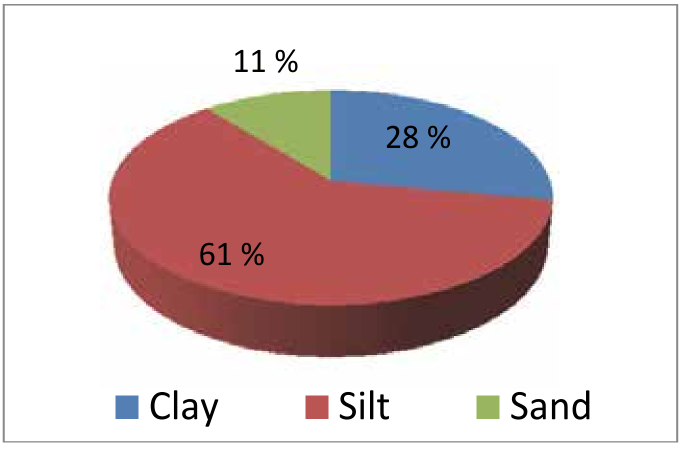



Fig. 9. Percentage of soil texture

| Main soil parameters | |
|---|---|
| *pH* | *8.4* |
| *Organic matter (%)* | *1.87* |
| *Carbonates (%)* | *23.08* |
| *Phosphorous (mg/Kg)* | *52.1* |
| *Potassium (mg/Kg)* | *185.4* |
| *Calcium (mg/Kg)* | *3054.62* |
| *Total Nitrogen (%)* | *0.37* |
| *Magnesium (mg/Kg)* | *18.4* |

Table 1. Main chemical parameters monitored in tested soil

Examined soil presents high cationic exchange with high Phosphorous, Potassium, Calcium and Nitrogen level, while content of Magnesium and Ammonium is low (Table 1).

## 4. Climatic and pluviometric analysis

Piedmont region is characterized by different climatic areas as consequence of co-existence of mountain, hill and plane areas. The North-West part of the region is surrounded by mountains reaching also 4000 m, characterized by cold climate and persistent snow. Hills and plane lands are not so far from mountain area, but present better climatic characteristics (temperate sub-continental), permitting numerous agricultural activities as vineyards cultivation. The rainfall events are more frequent in spring and autumn, especially in the mountain area, while are less consistent in the south plane part of the region. The rainfall events are also conditioned from the direction of the air masses: when they comes from South to North the presence of mountains create a limits and the rainfall events are more frequent on the hill and plane part of the region (Perosino and Zaccara, 2006).
The pluviometric analysis has been based on the data provided by the Hydrological Annals of the meteorological station of Nizza Monferrato and Acqui Terme in a period between 1915 and 2009. The monthly precipitation have been used for the definition of the mean

values; the maximum annual values at 1, 2, 6, 12 and 24 hours, have been used for the definition of the pluviometric probability curves for maximum length of 1 day and return time of 5, 10, 15, 20, 25, 30, 40 and 50 years. Maximum values at 10, 20, 30 and 60 minutes have been used for the reconstruction of significant erosive events in a time period between 1994 and 2004.

The absence of pluviometric data in correspondence of 1915-1930, 1940-1949 determined an analysis realized on shorter period (not referred to the whole period 1915-2009), and the mean values have been subsequently compared.

| MEAN MONTHLY AND ANNUAL PRECIPITATION (mm) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | 1915-1919 | 1920-1929 | 1930-1939 | 1950-1959 | 1960-1969 | 1970-1979 | 1980-1989 | 1990-1999 | 2000-2009 |
| **January** | 71 | 54 | 40 | 35 | 26 | 71 | 44 | 54 | 31 |
| **February** | 49 | 50 | 39 | 49 | 47 | 57 | 39 | 23 | 35 |
| **March** | 113 | 79 | 52 | 47 | 53 | 81 | 98 | 31 | 40 |
| **April** | 91 | 69 | 60 | 80 | 74 | 38 | 69 | 72 | 82 |
| **May** | 148 | 46 | 90 | 74 | 34 | 69 | 65 | 70 | 79 |
| **June** | 57 | 25 | 43 | 40 | 34 | 29 | 38 | 47 | 34 |
| **July** | 43 | 37 | 41 | 42 | 25 | 25 | 15 | 23 | 35 |
| **August** | 26 | 31 | 49 | 36 | 31 | 49 | 70 | 35 | 72 |
| **September** | 140 | 78 | 76 | 51 | 57 | 46 | 34 | 106 | 70 |
| **October** | 110 | 69 | 57 | 80 | 84 | 129 | 100 | 87 | 73 |
| **November** | 78 | 101 | 114 | 107 | 90 | 51 | 65 | 71 | 102 |
| **December** | 79 | 58 | 56 | 75 | 43 | 60 | 29 | 40 | 54 |
| **Annual** | **1005.0** | **694.7** | **716.2** | **716.7** | **589.7** | **705.6** | **666.9** | **658.8** | **704.7** |

Table 2. Pluviometric analysis summary describing the monthly and annual mean precipitation

Table 2 represents the mean monthly and annual precipitation in the nearer meteorological stations to the experimental site. Referring to the annual precipitation values, it has been possible to see that except for the first value, the mean value for the subsequent decades is about 700 mm.

## 5. Topographic analysis

The experimental site present a vineyard cultivation following the contour lines. Topographic relief has been realized through a total station in order to obtain a planimetric image of the tested area and some transversal sections. The relief has been realized from the down slope point and represents a cultivated field with 22 rows of plant with a distance of 2.70m.

| SECTION | MEAN SLOPE |
|---|---|
| **1** | 28.9 % |
| **2** | 25.8 % |
| **3** | 23.4 % |
| **4** | 23.9 % |

Table 3. Main studied section for slope determination

Four transversal sections have been traced for the definition of the mean slope (Table 3). Fig. 10 represents the plan design of the whole experimental site while the Fig. 11 is the second section with a slope of 25.8%.

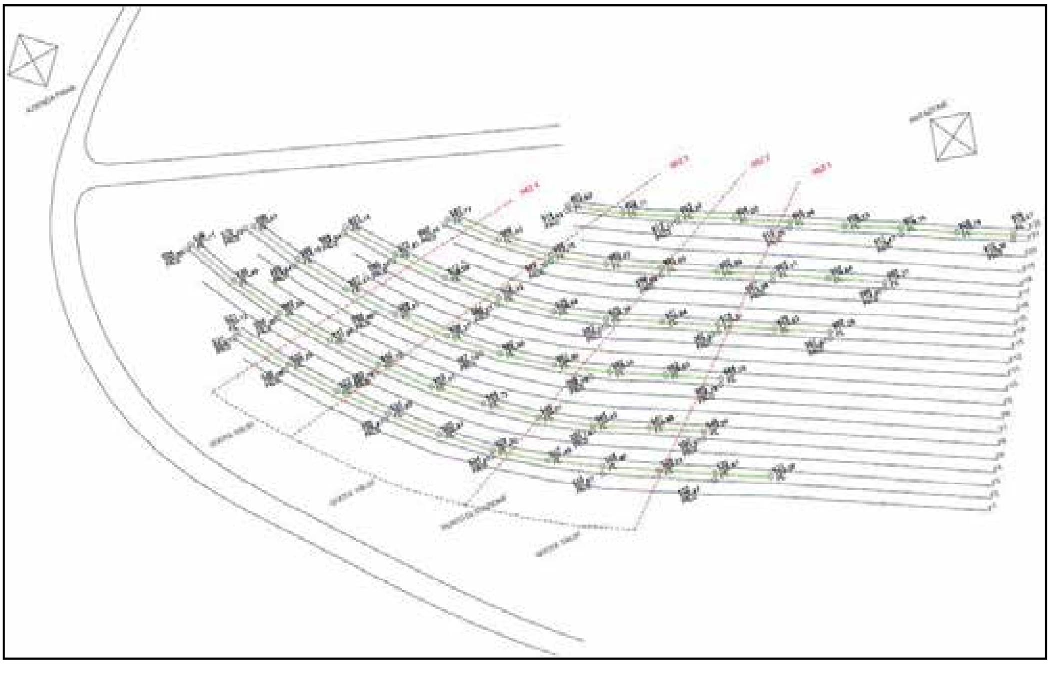
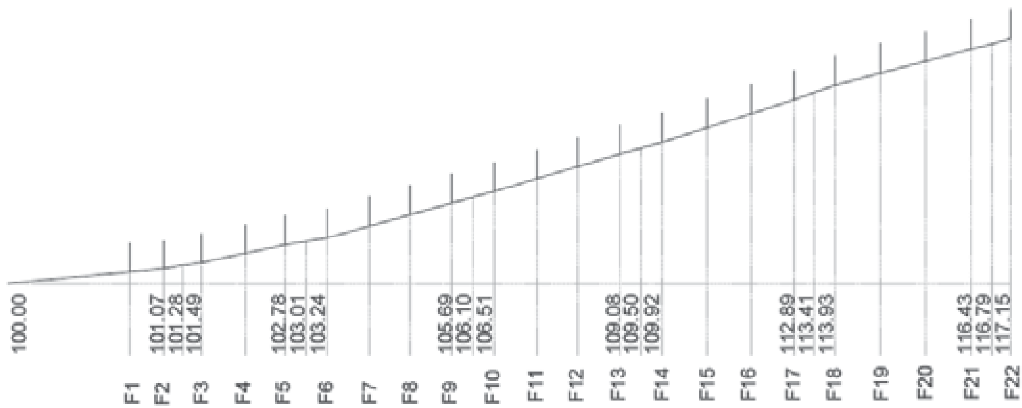


Fig. 10. Output of the topographic analysis on tested vineyard




Section 2

Fig. 11. Profile of section 2

## 6. Models description

Soil models tend to over-predict erosion for small measured values and under–predict erosion for large measured values. Risse at.al (1993) applied the empirically based USLE model, Universal Soil Loss Equation (Wischmeier and Smith, 1978) to simulate erosion from natural run-off (Nearing, 1998). Although the original USLE has been retained in RUSLE, the technology for factor evaluation has been altered and new data have been introduced with which to evaluate the terms for specified conditions.

The USDA - Water Erosion Prediction Project (WEPP) model represents a new erosion prediction technology based on fundamentals of stochastic weather generation, infiltration theory, hydrology, soil physics, plant science, hydraulics, and erosion mechanics. The hillslope or landscape profile application of the model provides major advantages over existing erosion prediction technology. The most notable advantages include capabilities for estimating spatial and temporal distributions of soil loss (net soil loss for an entire hillslope or for each point on a slope profile can be estimated on a daily, monthly, or average annual basis), and since the model is process-based it can be extrapolated to a broad range of conditions that may not be practical or economical to field test (Flanagan et al., 1995).

The European Soil Erosion Model (EUROSEM) is the result of European Commission funded research involving scientists from Europe and the USA. The model simulates erosion on an event basis for fields and small catchments. It uses physical descriptions to describe the process of soil erosion and is fully dynamic.

### 6.1 RUSLE

The Revised Universal Loss Equation (RUSLE) is an empiric soil erosion model, based on a multiplicative equation that predicts the amount of soil lost per hectare per year due to water erosion (sheet and rill erosion only). The RUSLE equation has been developed by the NRCS (Natural Resources and Conservation Services, a branch of the U.S. Department of Agriculture) over the course of the last 40 years.

The Universal Soil Loss Equation (USLE) model was based on the first concept of the separation and transport of particles from rainfall by Wischmeier and Smith (1965) in order to calculate the amount of soil erosion in agricultural becoming widely used and accepted empirical soil erosion model developed for sheet and rill erosion based on a large set of experimental data from agricultural plots.
The USLE has been enhanced during the past 30 years by a number of researchers. Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975), Revised Universal Soil Loss Equation RUSLE (Renard et al., 1997), Areal Nonpoint Source Watershed Environmental Resources Simulation (ANSWERS) and Unit Stream Power Erosion Deposition (USPED) represent an improvement of the former USLE equation. In 1996, when the U.S. Department of Agriculture (USDA) developed a method for calculating the amount of soil erosion under soil conditions besides pilot sites such as pastures or forests, RUSLE was announced to add many factors such as the revision of the weather factor, the development of the soil erosion factor depending on seasonal changes, the development of a new calculation procedure to calculate the cover vegetation factor, and the revision of the length and gradient of slope. The equation (1) of the RUSLE model is formed by 5 factors involved in the water erosion phenomena.

$$A\ (t/ha/y) = R \times K \times LS \times C \times P \tag{1}$$

where:
**A** = the predicted average annual soil loss from interrill (sheet) and rill erosion from rainfall and associated overland flow. Units for factor values are usually selected so that "A" is expressed in tons per hectare per year.
**R** = Rainfall-Runoff Erosivity Factor. "R" is an indication of the two most important characteristics of storm
erosivity: (1) amount of rainfall and (2) peak intensity sustained over an extended period of time. Erosivity for a single storm is the product of the storm's energy **E** and its maximum 30 minute intensity **I30** for qualifying storms. There are many equations that estimate the R parameter.
**K** = Soil Erodibility Factor. "K" values represent the susceptibility of soil to erosion and the amount and rate of runoff, as measured under the standard unit plot condition.
**LS** = Slope Length and Steepness Factor. The slope length "L" and steepness "S" factors are combined into the "LS" factor in the RUSLE equation. A "LS" value represents the relationship of the actual field slope condition to the unit plot.
**C** = Cover-Management Factor. "C" represents the effect of plants, soil cover, soil biomass, and soil disturbing activities on soil erosion. RUSLE uses a subfactor method to compute soil loss ratios, which are the ratios of soil loss at any given time in a cover-management sequence to soil loss from the unit plot. Soil loss ratios vary with time as canopy, ground cover, soil biomass and consolidation change. A "C" factor value is an average soil loss ratio weighted according to the distribution of "R" during the year. The subfactors used to compute a soil loss ratio value are canopy, surface cover, surface roughness, and prior land use.
**P** = Support Practices Factor. "P" represents the impact of support practices on erosion rates. "P" is the ratio of soil loss from an area with supporting practices in place to that from an identical area without any supporting practices. Most support practices affect erosion by

redirecting runoff or reducing its transport capacity. Support practices include contour farming, cross-slope farming, buffer strips, stripcropping, and terraces.

### 6.2 EUROSEM model

EUROSEM model (European soil Erosion Model) has been created by a European group of researchers at the end of '90 and has been based on the KINEROS program developed by Wollhsier et al (1990). EUROSEM model provided an erosion estimation due to rainfall and superficial runoff.
EUROSEM considers different aspects of the erosive phenomenon as

- drop interception due to the vegetative cover,
- volume and the kinetic energy of the rain drops,
- stagnation of water on soil for the micro-topography,
- runoff and sediment deposit.

The hydrographic basin has been represented by skew plains and channels that are respectively slopes and the hydrographic network (Fig.11).

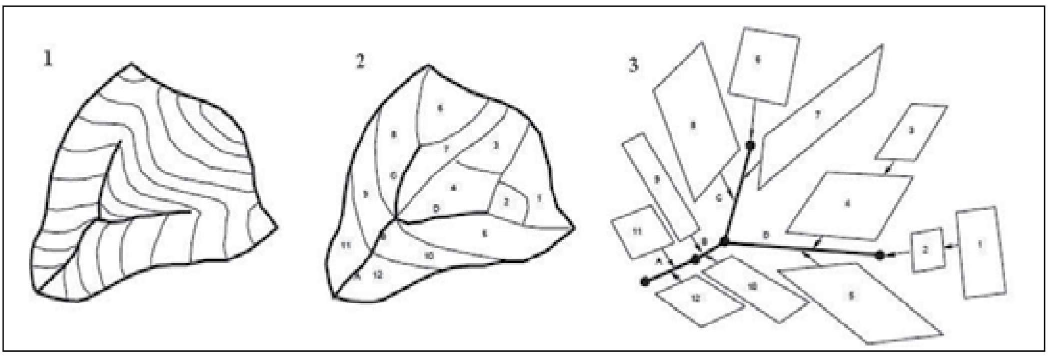


Fig. 12. EUROSEM plain and channels representation

EUROSEM model is based on the following mass transport equation (2):

$$\frac{\partial(AC)}{\partial t} + \frac{\partial(QC)}{\partial x} - e(t,x) = q_S(x,t) \tag{2}$$

Where:

- C ($m^3 * m^{-3}$) is the sediment concentration in the flow
- A ($m^2$) is the flow transversal area
- Q ($m^3 * s^{-1}$) is the water flood
- $q_s$ ($m^2 * s^{-1}$) is the sediment removal for unit length flow
- $e$ ($m^2 * s^{-1}$) is the superficial erosion
- $x$ (m) is the longitudinal coordinate
- $t$ (s) is the time

The slope can be also represented by a interril-rill scheme considering an overland flow running on soil surface.
EUROSEM can model the slope in two different versions, the first does not consider the presence of rills but only a superficial irregularity of soil, the second considers the rills as channels for the transport of the water flow coming from the interril. In the first case the soil

surface is considered as interril area and the flow direction is the maximum slope. In the second case the overland flow is directed to the rills channels with a slope (decided by EUROSEM), that is 1.4 time that of the plain element.

### 6.3 WEPP model

WEPP model (Water Erosion Prediction Project), has been created in the U.S.A. and represents one of the most advanced mechanistic model. WEPP could be applied to the temporal scale of the single event or to a multi-year events. The erosion estimation, by *profile* version, can be calculated on the slope scale or on a smaller surface (few square meters), while the *watershed* version permits the estimation for a small catchment. The *grid* model version guarantees a better results because the analyzed soil can be sub-divided and the mesh gives an higher precision in the results.

WEPP model has been based on seven different aspect concerning climate, water infiltration, hydric balance, vegetation, runoff, erosion and water transport in the hydrographic network; the model needs of numerous input data.

WEPP model permits to solve the lack of numerous detailed information. For example, inserting information about a short climatic period, the model compares them with other information present in the software libraries. In relation to the probability of a precipitation each day can be classified as wet (in presence of a precipitation), or dry (in absence of precipitation). The precipitation is considered water if the air temperature is higher than 0°C, otherwise is considered snow. Thanks to this approach is possible to obtain the hyetograph, knowing the total precipitation height. The water intensity is also important for the definition of the water infiltration percentage and the superficial runoff. The infiltration phenomenon is based on the Green and Ampt equation (1911), modified by Mein and Larson (1973) for constant intensity event and by Chu (1978), for variable intensity. The water partitioning between infiltration and runoff depends on hydraulic conductivity and saturation. If no detailed information are available the soil texture and cationic exchange are sufficient, and can be considered constant or variable i.e. for the presence of vegetation or soil management practices.

The water balance permits the estimation of the evapotranspiration rate, deep infiltration and interception by root systems. Vegetation is considered both in the alive part and in the decomposition part that can contribute to the runoff and the solid sediment transported.

WEPP model uses a geometric scheme based on a rill-interrill configuration dividing the slope in a sequences of homogenous areas (homogeneous in relation on the model parameters), in order to transfer all the results about the runoff and erosion values to the subsequent surface in the motion direction. The erosion component of the model calculates the soil detachment and the deposition along the profile that is subdivided in small parts. In the interrill the detachment is consequence of the rain impact. This portion of sediment is transported by the overland flow originated during the event reaching rills where can be transported within them or remain as deposit.

## 7. Results

The application of RUSLE model in the experimental vineyard gave as outputs the data set in table 4 and showed in Fig.13, that correspond to different percentage of cover vegetation.

| MEAN ANNUAL SOIL LOSS (t ha$^{-1}$ y$^{-1}$) | | | | |
|---|---|---|---|---|
| **Parameters** | **20%** | **40%** | **60%** | **80%** |
| **R** | 101,19 | 101,19 | 101,19 | 101,19 |
| **K** | 0,40 | 0,40 | 0,40 | 0,40 |
| **L** | 1,90 | 1,90 | 1,90 | 1,90 |
| **S** | 1,81 | 1,81 | 1,81 | 1,81 |
| **C** | 0,20 | 0,10 | 0,04 | 0,01 |
| **P** | 1,00 | 1,00 | 1,00 | 1,00 |
| **A** | **27,84** | **13,92** | **5,85** | **1,81** |

Table 4. RUSLE model results in terms of soil loss varying the cover vegetation

Increasing the cover vegetation from 20 to 80%, the soil loss values significantly decrease from 28 to 2 t/ha*y.

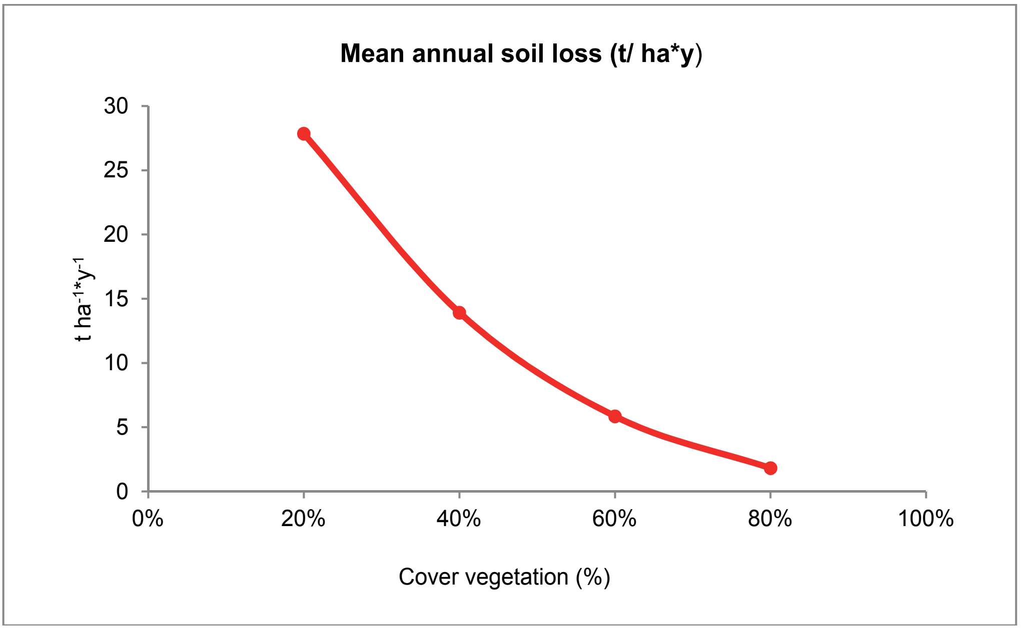



Fig. 13. Mean annual soil loss by WEPP simulation

Fig. 14, 15 represent the WEPP outputs for rain intensities of 40 and 80 mm/h in correspondence of two percentages of cover vegetation (70 and 10%). The rainfall events have variable duration from 10 to 40 minutes.

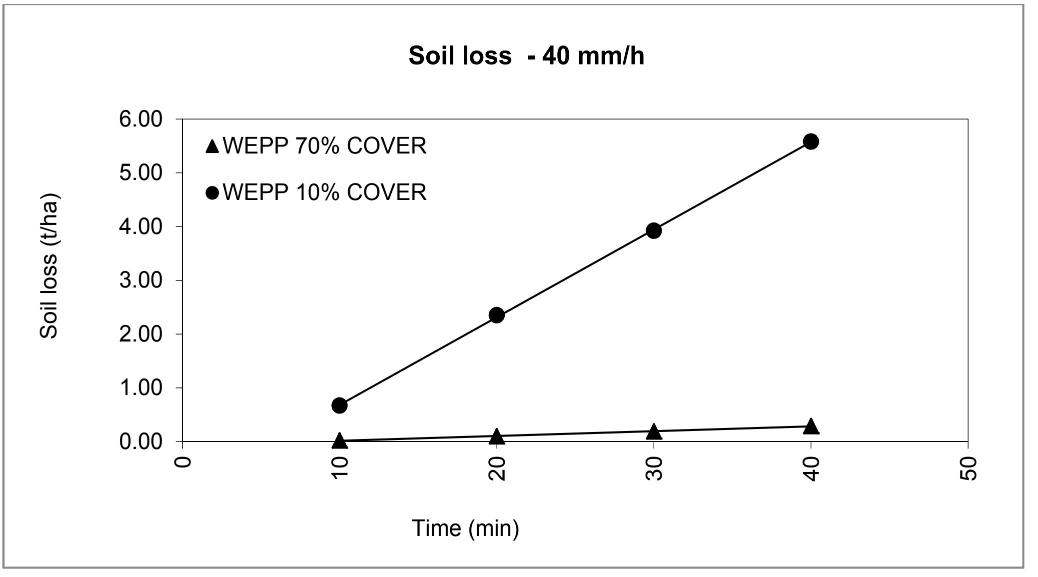



Fig. 14. WEPP soil loss simulation for different cover vegetation at 40mm/h

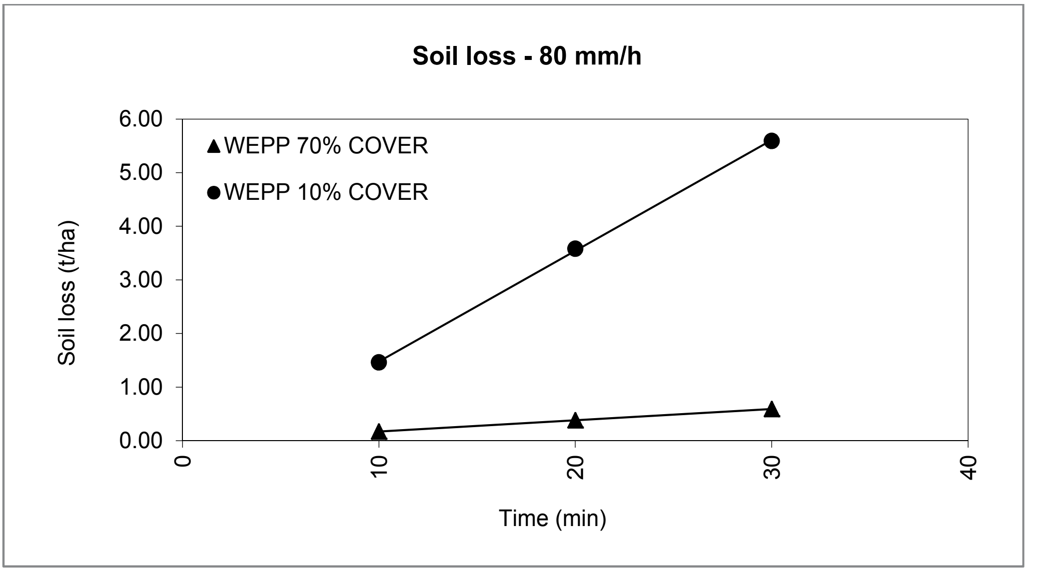



Fig. 15. WEPP soil loss simulation for different cover vegetation at 80mm/h

For both intensities (Fig.14, 15), soil with a low cover vegetation is much more threated by erosion phenomenon. This trend is more evident increasing the rainfall event duration.
Another sensible studied factor is the soil slope percentage. In the study case has been demonstrated that the increase of soil slope causes an higher soil loss. This trend is much more visible for stronger events characterized by an higher intensity and long duration (Fig.16, 17), where at 30 minutes the soil loss on 20% of soil slope is about 0.35 t/ha*y and on 29% of soil slope the soil loss is 0.6 t/ha*y.

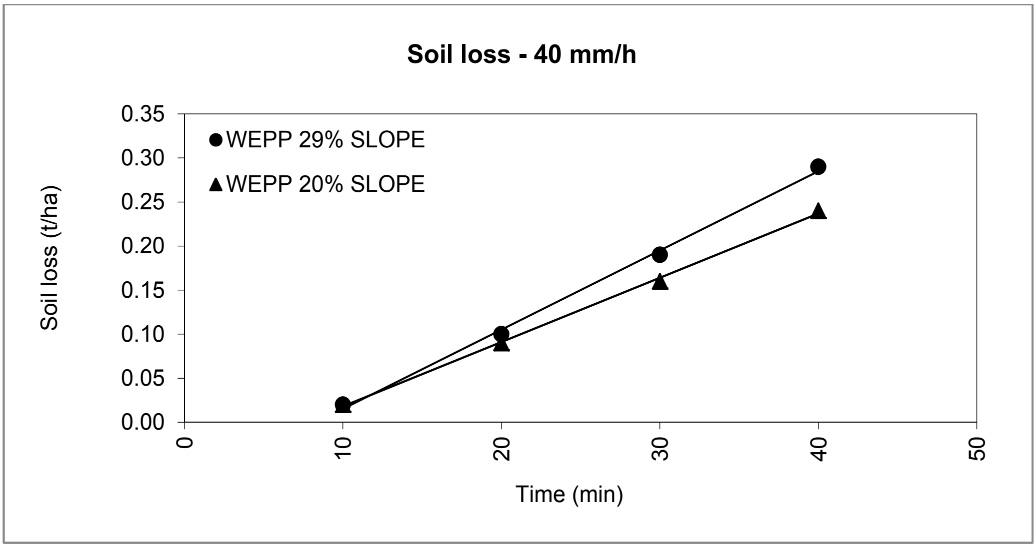



Fig. 16. WEPP soil loss simulation for different soil slope at 40 mm/h

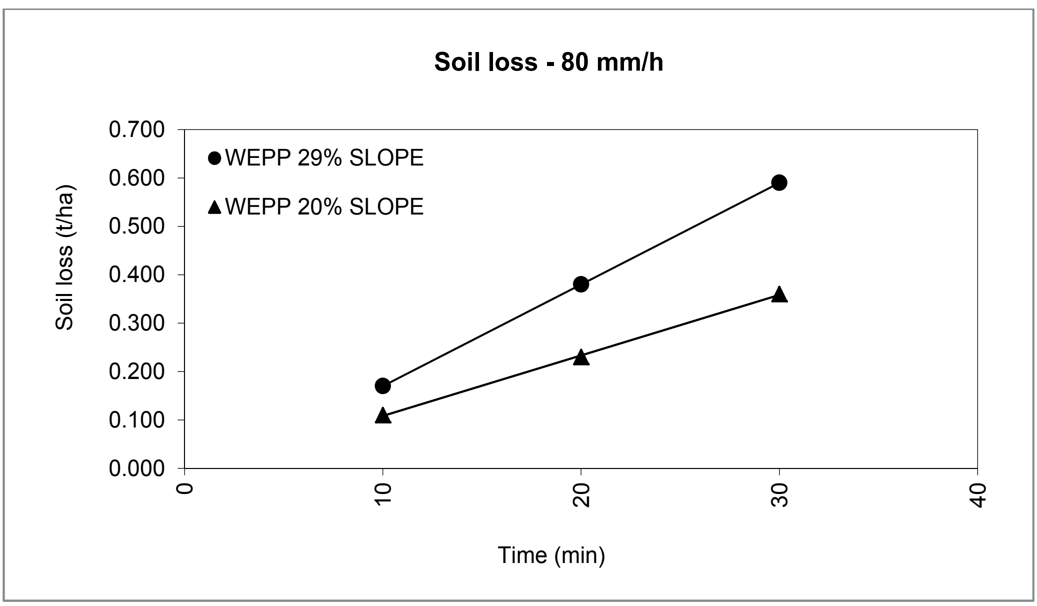



Fig. 17. WEPP soil loss simulation for different soil slope at 80 mm/h

EUROSEM model can be useful for the definition of the erosive effects caused by single events characterized by steps of intensity. Fig. 18, 19 below represent the estimation in terms of tons per year of soil loss for a single event with intensity characterized as follow.

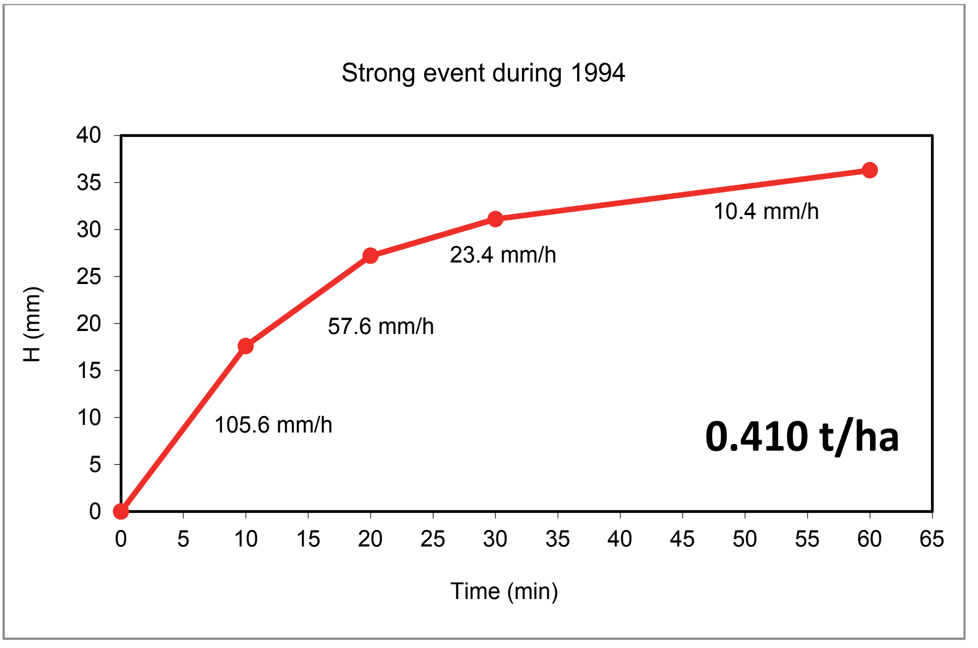



Fig. 18. EUROSEM simulation for event with variable intensities (1994)

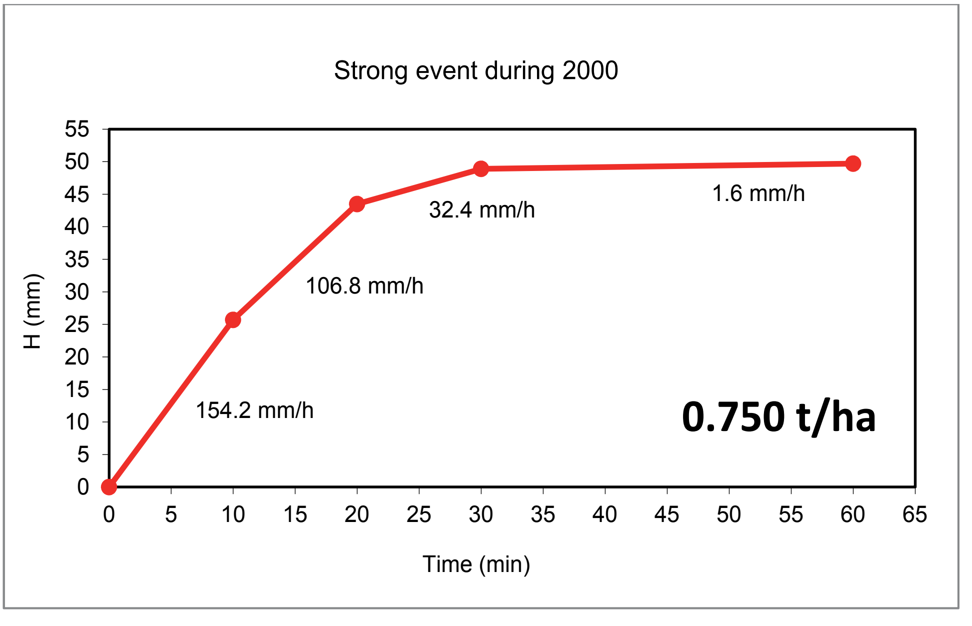



Fig. 19. EUROSEM simulation for event with variable intensities (2000)

The implementation of input parameters in WEPP (Flanagan anf Frankerberger, 2002) and EUROSEM (Morgan et al. 1998) models showed different trends of soil loss for different rainfall intensities and percentage of soil slope. In Fig.20, 21 the slope percentage is a mean value (29%) measured by the topographic analysis.

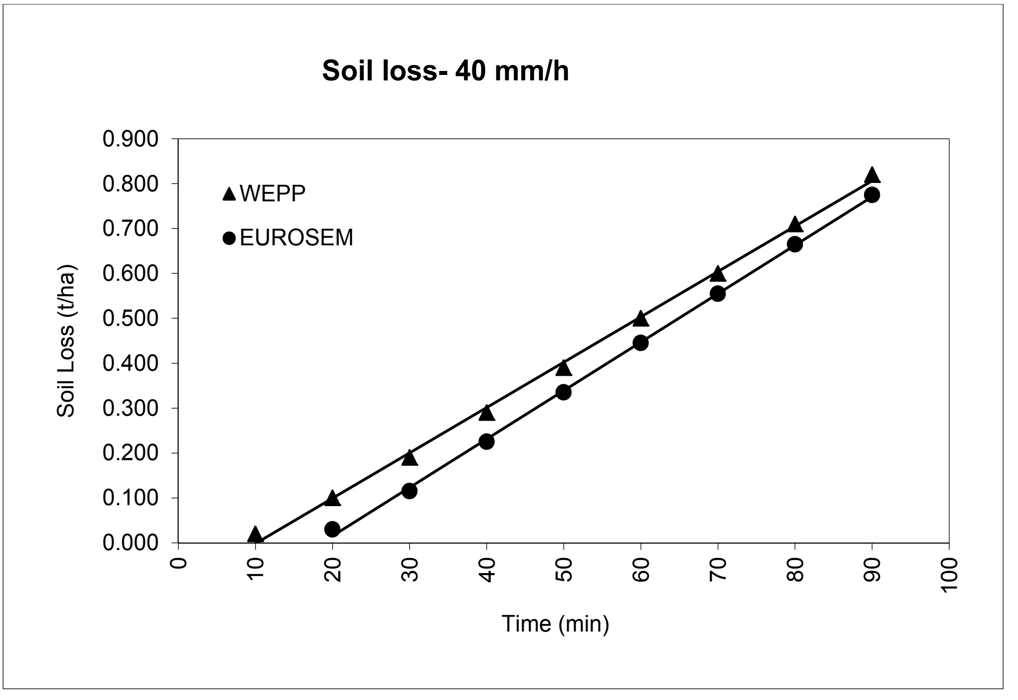



Fig. 20. WEPP and EUROSEM results for rainfall events of 40 mm/h

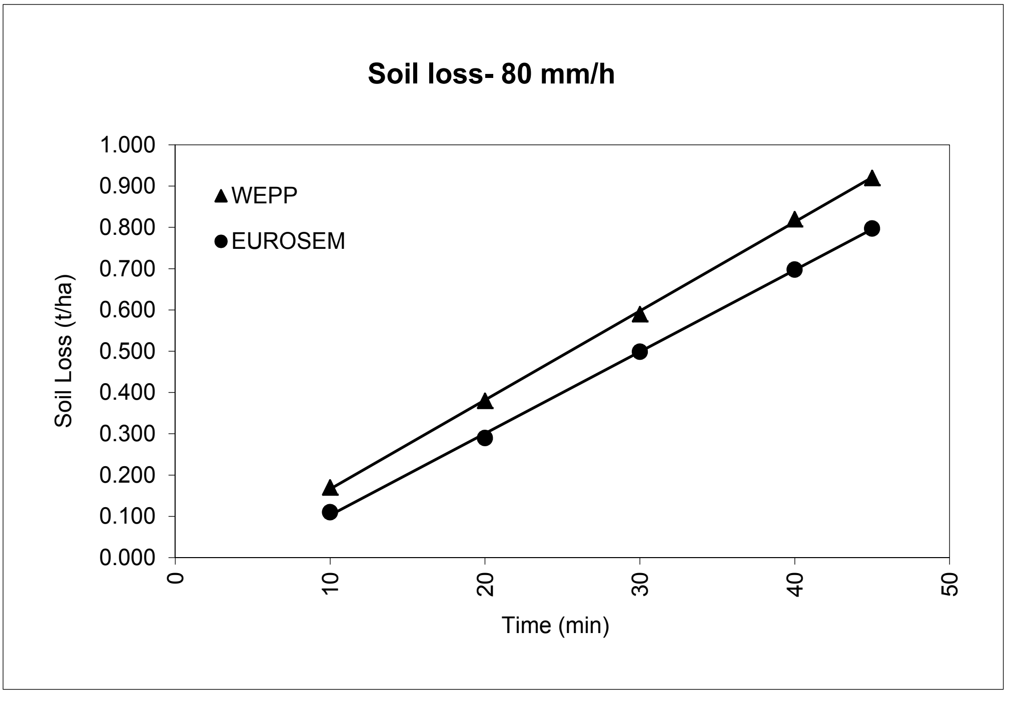



Fig. 21. WEPP and EUROSEM results for rainfall events of 80 mm/h

WEEP and EUROSEM could be applied to the vineyard scale for the annual erosion estimation but for their different characteristics the model outputs present some discrepancies. WEPP, in its *profile* version, is appropriate for the modeling of a single slope. EUROSEM model has been thinking for the erosion estimation at a basin scale and the output computational errors, if referred to a vineyard scale, is higher. The input data required for WEPP model are specific for the analyzed vineyard and concern the agricultural management practices. EUROSEM needs or more general parameters and has no differences between cultivated soil or hydrographic network basin, for this reason the same results could be associated to more than one configuration (slopes or hydrographic basin).

Soil loss results obtained for stronger events (Fig.22), have been compared with RUSLE model output. RUSLE output is equal to 3.83 t/ha*y considering a cover vegetation in the inter-row of 70%. Analyzing Fig.21 has been possible to see that in some cases (as for the stronger events occurred in 2000), a single event can be responsible of the major part of the soil loss during the whole year.

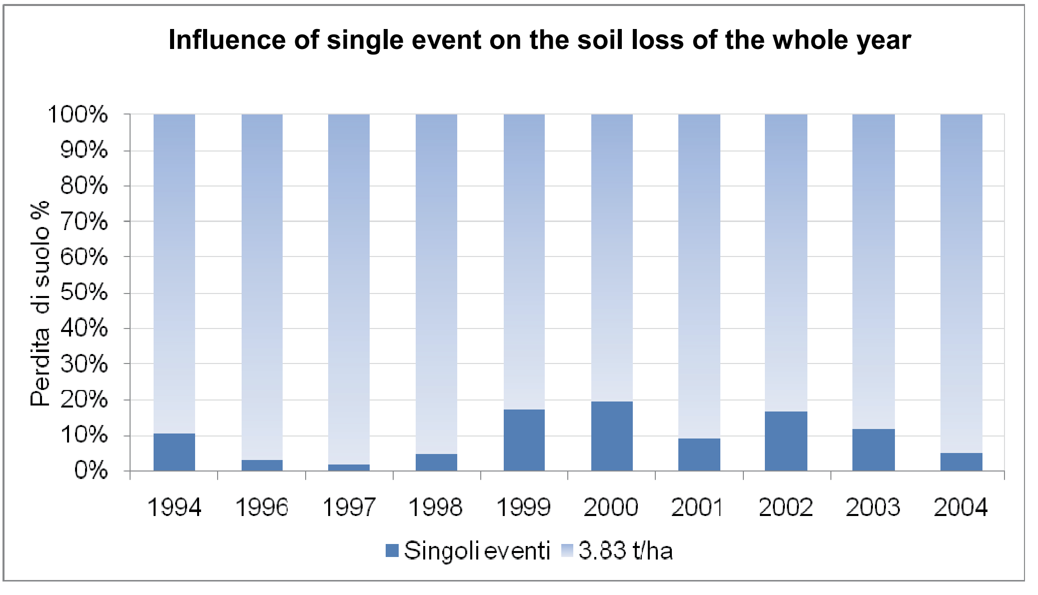



Fig. 22. Analysis of the influence of a single event on the total annual soil loss

Table presents some data concerning the experimental tests realized in situ describing main characteristics of the tested plots. In table 5 plots present variable slope with the same intensity, different percentage of vegetation cover and experimental soil loss. The soil loss is influenced by different parameters, especially by vegetation cover (where soil is bared the soil loss reaches higher value).

In table 6 have been compared experimental soil loss obtained by tests realized in situ and calculated soil loss obtained using WEPP model with the same input parameters. The compared results showed a difference in experimental and calculated results, in particular in tests 2 and 3, Fig.23 is the graphic representation of this difference measured in percentage.

| Tests | Slope (%) | Intensity (mm/h) | Vegetation cover (%) | Moisture (%) | Experimental Soil loss (g) |
|---|---|---|---|---|---|
| 1 | 16 | 85 | 69 (4) | 26.5 | 11.16 (3.66) |
| 2 | 25 | 85 | 0 | 31.6 | 42.84 (12.38) |
| 3 | 30 | 85 | 93 (5) | 31.0 | 5.61 (1.91) |

Table 5. Main results provided by experimental tests

| Tests | Experimental soil loss (t/ha) | Calculated soil loss (t/ha) |
|---|---|---|
| 1 | 0.112 | 0.210 |
| 2 | 0.428 | 3.850 |
| 3 | 0.056 | 0.270 |

Table 6. Comparison between experimental and calculated (by WEPP) soil loss

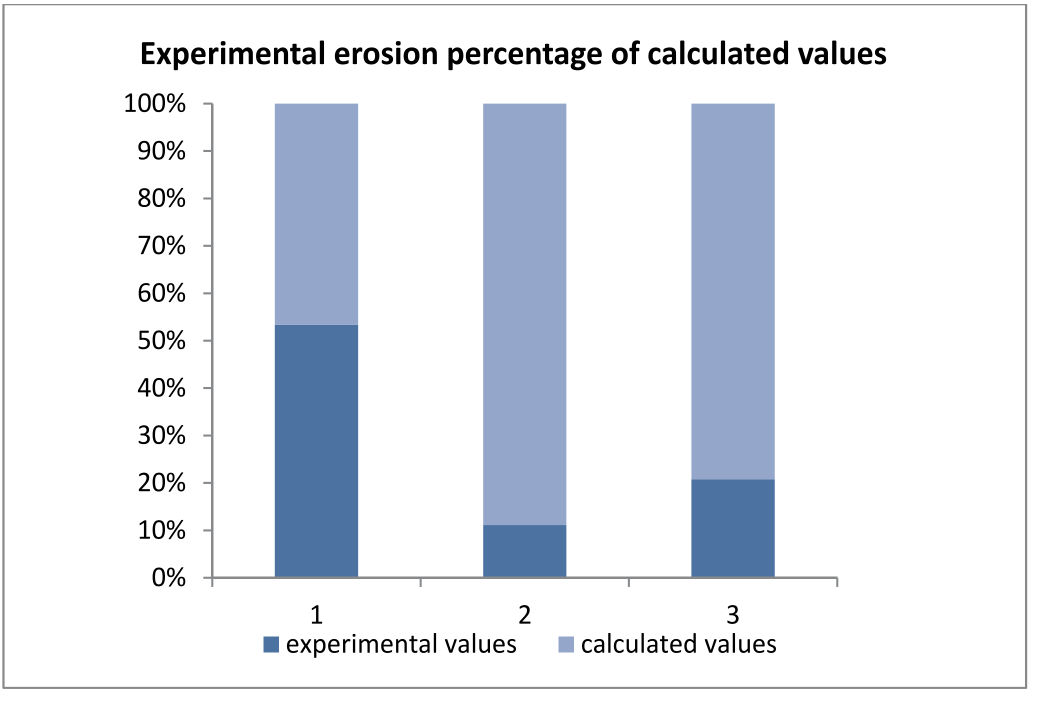



Fig. 23. Percentage of incidence of interrill erosion of the whole phenomenon

It is possible to assume that the event simulated by the experimental equipment, represents only the interill rate excluding the rill erosion phenomenon, in consequence of the small dimensions of the plot. Comparing the output provided by WEPP model (light blue in Fig.23), and the experimental results (dark blue in Fig. 23), it is possible to appreciate the incidence of the interrill rate on the whole phenomena.
The incidence of the interrill erosion, as showed in Fig.23, decreases with the increases of the slope gradient. The vegetated plot (test 3), characterized by higher slope, presents an interrill rate more evident if compared with the bared soil. The susceptibility of bared soil to the rills erosion is higher because soil is not protected by vegetation. In vegetated soil the predominant factor is represented by interrill erosion.

## 8. Conclusion

In the present research had been evaluated the applicability of three different models for the soil erosion estimation in Italian hills vineyards. The tested models, RUSLE, EUROSEM and WEPP are widespread in typical of different part of the world and have been proposed in numerous previous researches.
Models have been applied on an existent experimental site, located in the North-West Italy and validated by data obtained through experimental tests realized on near sites (Tropeano, 1984; Cavallo et al., 2010). RUSLE model gave results useful for a general estimation of the erosion phenomena, however the outputs are strictly dependent to the single parameters estimation and the model does not permit the simulation of the erosive rainfall events.
Concerning the two mechanistic models considered, WEPP has demonstrated the most reliable results in the erosion estimation, if compared with EUROSEM. Although WEPP requires a significant number of input parameters, generally not easily available. Another criticalities for both models is the non-automatically generation of rills, where the most important part of the erosion occurs. Both models need of the pre-definition of mean spatial definition of rills.
WEPP, even if showed some criticalities, can be considered a reliable instruments for soil erosion prevision in vineyard cultivation, useful in the agro-ecosystem management and the prevision of the best practices for the future agricultural activities.

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# 6

# Evaluation of Erosion Intensity and Some of Its Consequences in Vineyards from Two Hilly Environments Under a Mediterranean Type of Climate, Italy

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## 1. Introduction

Viticulture is one of the most diffused cultivations in the world. It is present in five of the six continents (except Antarctica) and in the first decade of XXI century vineyards have covered about 7.8 million hectares. In the same period, the global surface used for arable land and under permanent crops amounted to about 1.5 billion hectares (Bruinsma, 2003). Viticulture accounts only for about 0.5% of the whole agricultural area, and it can be inferred that the impact of this cultivation might be negligible at a global scale. In contrast, in the year 2009, the market of both wine and grapes moved around 300 billion of dollars in the world (AA.VV., 2010; FAO, 2011), putting the products of the viticulture at the top of the agricultural market. Unfortunately, also disadvantages induced by viticulture are considerable as most of the soils planted with vines have a fine texture and a moderate to steep slope (5-30°), and are submitted to climates where dry and rain seasons are rather alternated. Because of this, most of the vineyard soils are subjected to erosion.

In this chapter we briefly analyse the situation of viticulture in the world in terms of surface covered, parent materials and soils used, and types and degree of erosion usually found in the vineyards. Thus, we present the results of two managing experiences made in two different environments of Italy, where grass covered and harrowed soil vineyards were contrasted to assess the capability of the grass to reduce erosion. In the general conclusions, taking advantage of the results obtained, we give suggestions to preserve soil resilience and productivity.

### 1.1 Viticulture in the world

Viticulture is practised worldwide (Figure 1). The grape is cultivated on a commercial scale (for wine, fresh or dry grapes) in more than 40 Countries, mostly between 4° and 51° of latitude in the Northern Hemisphere and between 6° and 45° of latitude in the Southern Hemisphere, under a wide type of climates (Tonietto & Carbonneau, 2004). A small surface is cultivated with grape in some regions across the Equator line, in the desert and tropical climates, as well as vineyards go as far as the 55th parallel north and the 50th parallel south.

In some of the Countries belonging to the Old World viticulture has been practised for millennia, while many Countries of the New World, Asia and Oceania were engaged in viticulture very recently. The surface occupied by vineyards was about 8.8 million hectares in the 1950s and increased till more than 10 million hectares in the 1970s; then the surface considerably decreased till about 7.7 million hectares in 2009 (Figure 2). If one considers that, especially in the last three decades, the vineyard surface increased a lot in the Countries recently interested by viticulture, the result is that viticulture has decreased to a large extent in the Old World.

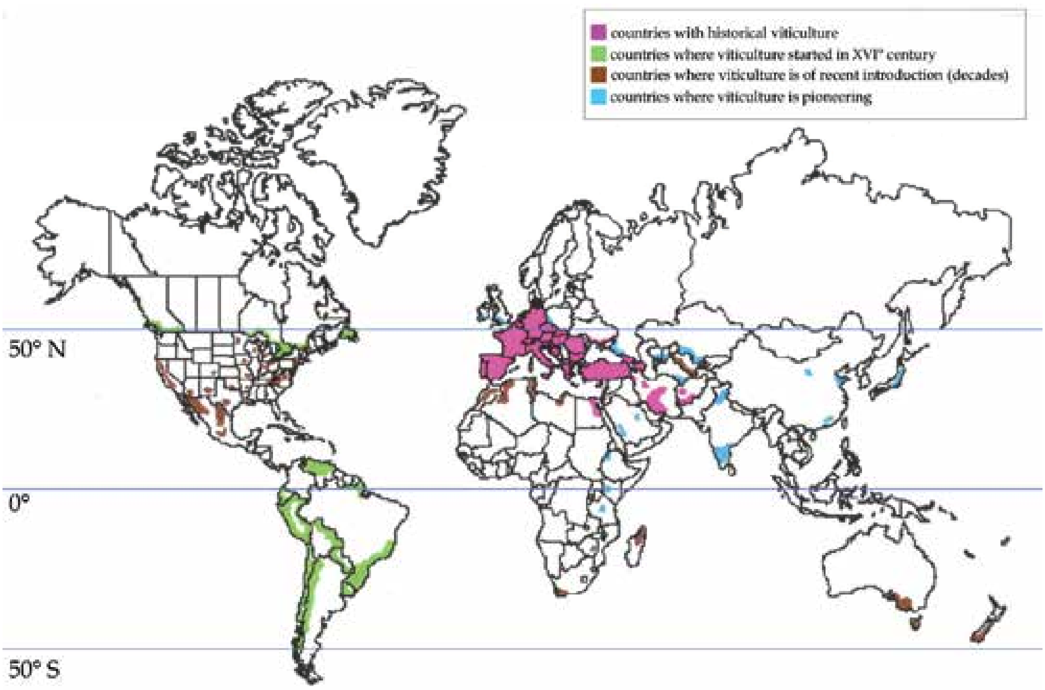


Fig. 1. Schematic representation of the World map with indicated the Countries where viticulture is practised.

Last statistics available from OIV, the World Viticulture Organisation (OIV, 2007), reveal that after a period of sustained growth that continued until the late 1970s, global vineyard surface started to decline as a result of European Union vine pull schemes and extensive vine pulls in the former Soviet Union. In 1998, the global vineyard surface had reached its lowest level since 1950: 7.6 million hectares. The change in the world vineyard hectarage is the result of different situations in Continents and Countries. For example, in North and

South America the surface under vines was 982,000 hectares; the USA area under vines levelled off at around 400,000 hectares, while Argentina and Brazil, as Chile to a lesser degree, maintained growth. In Africa, vineyards continue their expansion so to reach 394,000 hectares in 2007. In Asia, 1.649 million hectares are under vine. A large portion of the production is here destined for non-wine use, especially in Iran, Turkey and Syria. China's vineyards continue to increase in surface, although the Country's grape production is still dominated by table grapes. In Asia, China is not the only Country where vineyards are increasing surface, since Iran's area under vines (mainly destined to unfermented grape products) reached 330,000 hectares. Oceania continues to expand its vineyard surface, with a total of 204,000 hectares in Australia and New Zealand in the year 2007.

Europe has seen its area under vines steadily to decline since the year 2000. In 2007, the vineyard surface was 4.563 million hectares. Spain, France and Italy cover most of this large surface, with 1.169 million, 867,000 and 847,000 hectares, respectively (FAO, 2011). It should be noted that most of this decrease took place in France, which was the only European Union Country to undertake another extensive subsidised vine grubbing scheme. Continental trends should be kept in perspective by weighing the relative size of surface under vines on each continent: in the year 2007, Europe, the EU-15 (the European Union considering the early 15 member States) and the EU-25 (the European Union considering the 25 member States in 2007) accounted for 58.6%, 43.7% and 45.5% of the global vineyard, respectively.

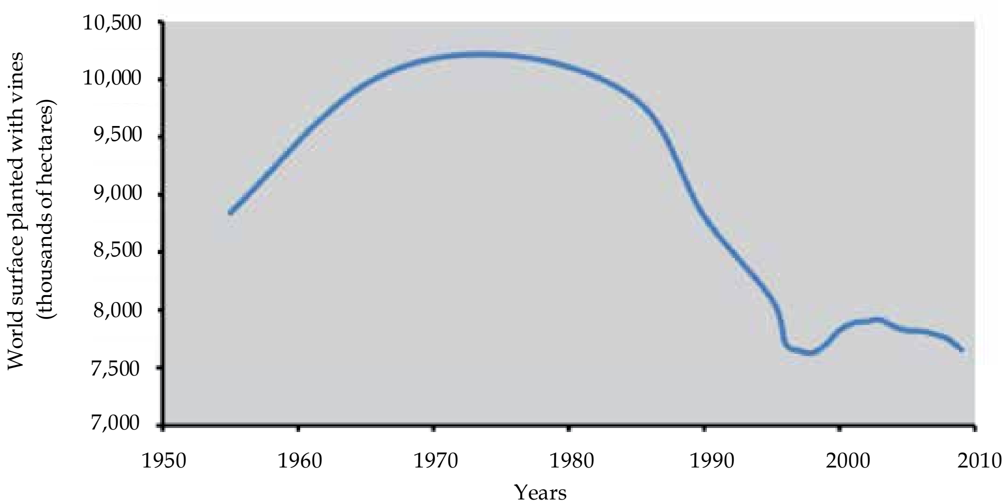


Fig. 2. Trend of the World's surface planted with vines from 1950 to 2010.

### 1.2 Viticulture in Italy

In Italy, viticulture dates back to 4000 years before present, when it was introduced in Sicily from Greece (Fregoni, 1998). A millennium after its introduction, viticulture had colonized so much of the southern Italy that the Greek people called this land Oinotria (*Οινωτρια*), namely the "land of the vines supported by stakes". During the successive millennium, the Etruscan first and the Romans then spread the viticulture in Italy and in the Roman Empire. Diffusion of vineyards became of so large scale that in the AD 92 the Roman Emperor Titus

Flavius Domitianus, preoccupied of the reduction of areas devoted to cereals cultivation, banned plantation of new vineyards and obliged to grub half of the vineyards (Suetonio Tranquillus, after AD 121). However, the reduction of vineyards persisted till the IV century, when the Roman Emperor Flavius Theodosius promulgated a law that contemplated the death penalty in case of vines grubbing. Nonetheless, the crisis of viticulture went on till the XIII-XIV century, surviving mainly in the proximity of monasteries, convents, churches and within the wall of the cities-states such as Florence and Siena. From the XIV-XV century, together with a certain economic and politic stability, viticulture regained surfaces and continued to increase till the middle of XIX century. Afterward, two plagues threatened the vine survival in Italy as in Europe: meldew (around 1850) and phylloxera (1880-1890). While the first illness was rather easily overcome by the use of sulphur, the second one exhausted vineyards till the first decades of the XX century. Only after the 1930, with the diffusion of the american rootstocks, viticulture recovered surfaces reaching a maximum of expansion in the 1950s with more than 3,700,000 hectares (Figure 3). Once again, after that period of glory, a surface reduction started and it persists till nowadays when a surface of about 800,200 hectares has been reached (Figure 3). Of

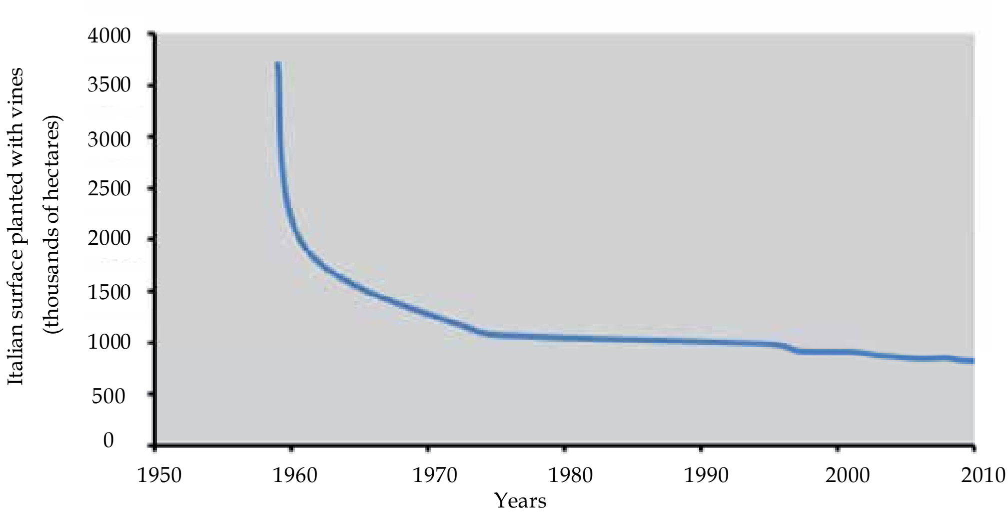


Fig. 3. Trend of the Italian surface planted with vines from 1960 to 2010.

these, about 730,000 hectares are devoted to wine production, while nearly 70,000 hectares are for human fresh grape consumption (ISTAT, 2009). Even though it is difficult to estimate the amount of surface devoted to vineyards during the Roman Empire or the Renaissance, it is assumed that the expansion reached in the 1950s should be the maximum reached in Italy in the story of viticulture. As a matter of fact, the 1950s and 1960s were two decades during which soil erosion was severe insomuch that the Italian soils lost about 30% of their water holding capacity (Pagliai, 2008). This was not only responsibility of viticulture, but at that time viticulture interested more than 12% of the whole Italian territory (which accounts to about 301,000 km$^2$). In those decades, vineyards were planted even in areas that never had hosted vines, including dormant landslides and hilly soils with a fine texture that were tilled topdown. Further, thanks to the availability of mechanical machines, the vines were planted after deep soil breaking up and levelling. Also because of this, many were those who put into relation expansion of viticulture and increase of hydraulic erosion processes (Tropeano, 1984; Costantini et al., 2001; Costantini, 1992; Kosmas et al., 2000; Pisante et al., 2005; Bazzoffi et al., 2006). Recently, to mitigate erosion, vineyards are usually planted with rows along isoipsae and the soil is grass covered, but erosion still persists. In fact, grassing was adopted as an universal cure whose efficiency in contrasting erosion should be assessed in the different pedo-climatic conditions.

### 1.3 Soil parent materials and soils devoted to viticulture

Many researchers with geologic background observed a connection between wine and geology, however pedologists believe that vine health and wine quality are conditioned by soil physical, chemical and mineralogical properties, which are only partly inherited from parent material. In fact, the nature of the parent material thoroughly influences soil characteristics but these latter are not exactly inherited from the substratum. For example, a soil can inherit a sandy texture if developed from granite or sandstone, while the presence of limestone in the parent material will slow the development of soil acidity (Brady & Weil, 2008). It is also true that availability of nutrients (P or N, for example), accumulation of organic matter or formation of soil aggregates depend on the type of pedogenesis and, hence, on the type of climate and vegetation that in time operated on a given parent material.

The grape quality and wine composition can be conditioned by soil texture, which determines water availability and related consequences such as drainage, evapotranspiration, nutrient availability. The reduced development of vine foliage caused by a mild water stress may result in a better bunch exposure to light and in a smaller fruit size, usually improving the grape quality too. A study of the soils of Burgundy (France) concluded that vines produce the best wine where the soil contains both clay and pebbles as the first improves water retention and cation exchange capacity, while the second favour the drainage (Huggett, 2006). However, most of the vineyards submitted to a Mediterranean climate occupy fine-textured soils and are often prone to erosion. In these territories, spatial and seasonal climatic variability may result in a wide and unpredictable rainfall fluctuation from year to year and could increase rates of erosion (Nunes & Seixas, 2003; Nearing et al., 2005). This rate is increased by other conditions such as the abandonment of protective agricultural practices and the (ab)use of soils derived from unconsolidated parent materials like marls, limestones, and fluvial and marine sediments (Poesen & Hooke, 1997; Martínez-Casasnovas, 1998). In fact, the nature of the underlying rock and the soil-rock relationships affect rate of water discharge, entity of the runoff and penetration of vine roots to varying degrees (Huggett, 2006). This means that the choice of the soil type where to plant vineyards or other orchards should be fundamental to ensure a long economical life to the plantations and to obtain high quality products.

It is hard to list all the parent materials and the related soil types on which vineyard are implanted because of the wide diffusion and the good adaptation of the vine, but a not exhaustive inventory includes the following lithologies from which the most used soils for vineyards originated. Granite is an igneous rock which has a hard and granular texture and an high content of quartz. Soils evolved on this parent material drain and dry out very quickly; for these reasons they frequently have an acid pH and store heat. Soils from granite occur in many wine-producing areas such as Beaujolais and the Rhône (France), Sardinia

(Italy), northern Portugal and Cape Cod (Massachusetts, USA). Igneous and metamorphic rocks generally result in stony, well-drained, and relatively unfertile soils. Basalt represents a common volcanic rock on which developed soils of many viticultural areas like Canary Islands (Spain), the Azores (Portugal), and Mount Etna (Sicily, Italy). Other regions that host vineyards planted in soils from basalt also are in western India, southern Australia, Oregon's Willamette Valley (USA), south-central France, northern Italy, and Hungary, but the largest one is the Columbia Valley Pacific Northwest, USA. Limestone and marls produce soils rich of plant-accessible calcium in French great wine regions like Champagne, Burgundy, Chablis, the Loire, southern Rhône valleys and Saint-Emilion in Bordeaux. Stony soils derived from limestone produce well drained soils with pockets of clayey or silty-clayey material that are rapidly colonized by vine roots. Chalk and similar soft, bedded limestones provide crumbly soils that are both well drained and have the capacity to retain water in the micropores of soil rock fragments or underlying parent rock. Soils from chalk have low fertility unless mixed with clayey or organic material coming from adjacent formations. Sandstones represent a combination of clay minerals and sand granules compacted together by pressure and time and may originate soils with a good drainage and nutrient status. Schist rocks based soils retain heat well and are rich in magnesium and potassium but they are usually poor in organic matter and nitrogen. In places, soils evolved on schist are preferred over those on granite because of their permeability, which favours penetration of rainfall and roots. Shale is a fine-grained sedimentary rock that gives moderately fertile soils that well retain heat, but have a very low porosity and a slow drainage that make them prone to erosion on impervious morphologies (Huggett, 2006). Alluvial deposits that are characterized by a combination of gravel, sand, silt and clay, formed over time from mineral deposits left by running water and that colonized geological terraces, frequently originate soils interested by viticulture.
Speaking about soil orders (according to the USDA classification), all of them are used for viticulture except Gelisols and Histosols. The reasons of this are obvious as Gelisols develop in very cold regions and are affected by cryoturbation and permafrost, while Histosols are organic soils that form in cold regions or/and in humid/perhumid conditions. These climatic conditions are extreme even for a so easy-fitting species like vine. The use of soils belonging to the orders of Vertisols and Mollisols are of particular interest for viticulture as the first ones are those that fracture during summer, while the second ones mostly develop on cold and humid areas. As Vertisols have been reported as harmful for the life of vines, the plantation of vineyard on this type of soils requires irrigation (to avoid fracturation) or the use of rootstock resistant to severe breakings of the roots. The use of Mollisols in viticulture requires protecting vines during winter, when temperature may go several degrees below zero.

### 1.4 Soil erosion in the vineyards

During the last decade the topic of soil water erosion in the vineyard has received much attention, due the great economic value of the viticulture products and the increase of problems caused by the erosion itself. Since 16% of world lands was estimated to be vulnerable to erosion hazard, water erosion is considered one of the major causes of soil degradation (FAO, 2000). In Europe, 12% of the land is estimated to be subject to soil erosion, which is considered to be one of the eight worst soil degradation threats (CEC, 2006a). Because of this, since 2006 the European Commission adopted the Thematic Strategy for Soil Protection with the aim to protect soils across the European Union. This strategy consists of *i*) a communication between the Commission and the other European Institutions (CEC, 2006a), *ii*) a proposal for a framework Directive (an European law), and *iii*) an Impact Assessment. The proposal for a framework Directive (CEC, 2006b) sets out common principles for protecting soils and, under this common framework's guidance, the EU Member States should decide how to use the soil in a sustainable way and how to effectively protect it on their own territory.

In Mediterranean regions, the concern of soil water erosion increased during the last decades, as a consequence of both natural and anthropogenic factors (Kosmas et al., 1997; CEC, 2006a). In recent years many efforts have been carried out by the research community in order to assess the soil erosion rates in agricultural or recently abandoned lands in Europe. According to data collected from several study cases by Cerdan et al. (2010), vineyards show the highest soil losses among cultivated lands, being lower only in comparison with bare soils. These authors observed that soil water erosion rates for bare soils, arable land and vineyards resulted unexpectedly lower in the Mediterranean zone than in other European regions. However, in the Mediterranean basin, in comparison with other typical crops, vineyards represent the form of agricultural land use that causes the highest soil loss (Raclot et al., 2009). In addition, in some parts of the Mediterranean area, vineyards represent the largest agro-ecosystem type and constitute the most important crop in terms of income, employment and environmental impact. As Spain, France and Italy are three of the world top ten grape-producing Countries and most of the territory of each one is in the Mediterranean area, many field studies on soil water erosion in vineyard have been carried out in these Countries. Some of them are cited as example here below.

In Spain, soil loss in vineyards was investigated at field scale in the wine region of Penedès (Catalonia) by Ramos & Martínez-Casasnovas (2007) and in Navarre by Casalí et al. (2009). In France, the intensity of soil erosion in the vineyards was studied in the Ardeche region by Augustinus & Nieuwenhuyse (1986) and, more recently and using different scales and methodologies, in Burgundy (Brenot et al., 2008) and in Languedoc-Roussilon (Blavet et al., 2009; Raclot et al., 2009; Paroissien et al., 2010). In Italy, Tropeano (1984) measured soil erosion for a two-years period in confined plots located in three vineyard areas in Piedmont (NW Italy).

Most of the investigations about soil losses in vineyards are based on experimental plot studies where erosion rates are measured under natural or simulated rainfalls. The runoff from each confined plot is usually collected with a Gerlach box type or discharge gauge and runoff samples are analyzed in order to measure their sediment concentration. Another approach to quantify soil erosion and deposition rates is based on stock unearthing measurements (Brenot et al., 2008). In addition, many authors estimated soil erosion rates using proper models such as the RUSLE, the Revised Universal Soil Loss Equation (Martínez-Casasnovas & Sánchez-Bosch, 2000). However, the considerable amount of studies on erosion is justified by the need to better quantify the risk and to know the role of the various factors that affect soil water erosion, in order to define sustainable management practices.

It is well known that the factors primarily influencing soil erosion are climate, topography, soil texture and soil management. In Europe, vineyards are often located on hilly areas subjected to a Mediterranean climate that makes optimum conditions for vine-growing, but also favour a strong erosion. Topography, especially slope gradient, is one of the factors that predisposes soil to water erosion. In the past, in order to make manual cultivation easier and also to intercept surface runoff, fields on sloping sides were generally terraced and cultivated with vine-rows oriented perpendicularly to slope. Nowadays, constitution (and maintaining) of a terrace system have often been replaced by land levelling and vine-rows running along the slope (Ramos & Martínez-Casasnovas, 2007).

In the Mediterranean basin, rainfall of high intensity and short duration that could occur in summer or autumn are responsible of most of the soil loss (Tropeano, 1984; Ramos & Martínez-Casasnovas, 2007; Raclot et al., 2009). The erosion rate differs in the various Countries, and was estimated to be 2.3 Mg ha$^{-1}$ year$^{-1}$ in Italy, 1.5 Mg ha$^{-1}$ year$^{-1}$ in France and 1.0 Mg ha$^{-1}$ year$^{-1}$ in Spain (Cerdan et al., 2010). Considering the mean value per land-use class, these Authors predicted a value of 17 Mg ha$^{-1}$ year$^{-1}$ for vineyards, that represents the highest value among all cultivations. Soil loss rates estimated in vineyard plots with different soil management techniques by Tropeano (1984) ranged from 0.2 to 47 Mg ha$^{-1}$ year$^{-1}$, with the highest values obtained for a vineyard that was deeply ploughed just before the period of observation. This annual rate was close to the 44 Mg ha$^{-1}$ year$^{-1}$ estimated by Lorenzo et al. (2002) in NE Spain. As a result of a single extreme rainfall event, soil losses up to 34 and 217 Mg ha$^{-1}$ were measured in SE France (Wainwright, 1996) and in NE Spain (Martínez-Casasnovas et al., 2002), respectively. The intensity of extreme rainfall events, which were increasing over the past 50 years, is projected to become more frequent (EEA-JRC-WHO, 2008) in consequence of global climate change and, consequently, soil erosion in vineyards is expected to increase again.

Besides topographic and rainfall characteristics, some soil management practices in vineyards could favour erosion. We have already mentioned here above that conservation measures such as the vine-rows orientation perpendicular to the slope and terracing have been abandoned in order to support field mechanization and increase the land productive potential. Blavet et al. (2009) reported the several changes pointed out by other authors in land use and farming techniques since 1970s in French Mediterranean wine-growing areas. These changes include the mechanization of the vineyard with straight rows of vines on trellises, chemical weeding to save manpower and the enlargement of vineyards to slopes so to obtain high quality wines. These anthropogenic modifications have made the soil more susceptible to water erosion as a high traffic of machinery on sloping vineyards increases bulk density, reduces water penetration and soil water holding capacity (Ferrero et al., 2005), that in turn produce effects even on soil permeability and, therefore, on runoff and soil erosion processes. Since factors like topography and soil texture are difficult to modify, different management techniques could be adopted as measures to reduce soil water erosion. Then, common farming methods like tillage and chemical or mechanical weeding between the rows have been compared with more conservative techniques such as mulching, grassing or rock fragments covering to evaluate the effects on runoff and soil loss (Blavet et al., 2009) and to obtain information about most suitable management practices for land protection.

Vineyards affected by degradation are generally subjected to sheet and rill erosion processes; in particular, during high-intensity rainfalls, the concentration of the overflow along preferential pathways could produce rills and ephemeral gullies (Martínez-Casasnovas et al., 2005), which form a provisional discharge network on the field. This causes negative on-site effects like soil and nutrient losses and, in a medium- to long-term, also fertility and organic carbon stock could decrease (Poesen & Hooke, 1997; Ramos &

Martínez-Casasnovas, 2004). Additional negative effects due to erosion require a direct economic engagement for the restoration of damages. The original soil surface can be restored by normal tillage operations, but the zones interested by ephemeral gullies will remain prone to be eroded during future runoff events. Environmental impact of soil erosion is also due to risk of pollution and inundations or muddy floods (Le Bissonais et al., 2002; Bechmann et al., 2009). Martínez-Casasnovas et al. (2005) evaluated on 5% of the income of the farm involved, the operation costs to redistribute the sediments over the field after an erosive rainfall event that had produced ephemeral gullies in vineyards of the Penedès region (NE Spain). The costs increase up to 7.8% of the income when the replacement of the nitrogen lost associated to soil erosion is taken into account (Martínez-Casasnovas et al., 2006).

In Europe, like in countries where wine-growing is emerging, the setting of sustainable management practices to handle land degradation is the primary aim of studies on soil water erosion in vineyards. Researches on that issue are nowadays more and more actual, considering the economic role of grape production and the effects of climate change on factors that control soil erosion processes.

## 2. Field experiences on the effect of soil managements in Italian vineyards

### 2.1 Background

In the last decades, many winegrowing areas of the Mediterranean region with soils developed from fine-textured parent materials have been subjected to an increase of water erosion that, mainly in hilly environments, has lead to soil degradation (Kosmas et al., 1997; Martínez Casasnovas & Sánchez Bosch, 2000; Dahlgren et al., 2001; Pla Sentís & Nacci Sulbarán, 2002; Martínez Casasnovas et al., 2005). As a matter of fact, nowadays the vineyard represents one of the most erosive land uses in the Mediterranean as well as in humid environments (Tropeano, 1984; Cerdan, 2010), with erosion rates so high to reach an average soil loss of about 1.4 Mg $ha^{-1}$ $yr^{-1}$ (Kosmas et al., 1997). This also causes a relevant loss of organic carbon and nutrients from the superficial soil horizons. In areas where the vineyard is the main land use, such a level of soil degradation is a serious agricultural and environmental problem as it causes loss of the fertile topsoil and pollution of the superficial water bodies where the eroded materials sediment (Pieri et al., 2007). In this context, soil productivity and other soil functions risk to decline irreversibly (Biot & Lu, 1995; Bruce et al., 1995) if correct practices to prevent erosion are not taken into account (Hudson, 1995; Morgan, 1995; Agassi, 1996). Because of this, many studies have been run to conceive managing practices able to reduce soil erosion and maintain or increase productivity in the vineyard. Most of these studies have focused on the adoption of permanent grassing (e.g. Schwing, 1978; Messer, 1980; Gril et al., 1989) or straw mulching (e.g. Carsoulle et al., 1986; Gril et al., 1989; Louw & Bennie, 1991).

In Italy, where vineyards are mostly implanted on hills, soil erosion has somewhere reached threatening levels (Federici & Rodolfi, 1994a, b; Sorriso-Valvo et al., 1995; Phillips, 1998a, b). In the Italian vineyards the erosion was also exacerbated by farming practices such as enlargement of vineyards, hedgerows removal, cutting of the elms (*Ulmus* spp.) and mulberries (*Morus* spp.) trees that supported the vines in the old vineyards (Italian term: *alberata*), mechanization of vineyards with constitution of straight rows with vines supported by wood or metal stakes, and chemical weeding of the inter-row. All these practices of inappropriate landscape and soil management have produced a rather even slope surface without redistribution of ground water circulation, so favouring activation of landslides and flow phenomena in addition to other forms of sheet and rill erosion. In the attempt to solve or reduce the problem, in the last decade many farmers have changed the previous soil management into permanent grassing of the inter-row, but the effects of this management on the soil erosion and properties have been scarcely studied, especially if one considers the variety of the Italian pedo-climatic environments.

### 2.2 Evaluation of erosion in tilled and grass covered soils in vineyards from north-western Italy

#### 2.2.1 Introduction

In Italy, Piedmont is a long established and specialized wine region and produces some of the best-known, top quality Italian wines (e.g. Asti Spumante, Barolo, Barbera) and it is the second largest (after Veneto) Italian exporting region. The region produces 11 DOCG (Denomination of Controlled and Guaranteed Origin) wines, over 38 in all Italy, and 45 DOC (Denomination of Controlled Origin), over 316 in all Italy, which account for almost 80% of the total regional production and represents about 15% of Italian production of appellation wines (Cusmano, 2010). Therefore, in this region the vines growing and oenological industry greatly contribute to the agricultural income. Vineyards in Piedmont cover more than 53,000 hectares, namely around to 7.3% of the Italian wine production area (730,000 hectares). In the Region, fresh grape production is negligible. According to the agricultural statistical database of the Piedmont Regional Administration, almost 90% of the vineyard surface of the region is on hilly area and near 2% on mountain area, and the vineyards are concentrated in the southern part of the region (Figure 4), in the Asti, Cuneo and Alessandria Provinces (Regione Piemonte, 2011a). Geology of the hilly sector of the southern Piedmont is made of Cenozoic deposits of the Tertiary Piedmont Basin, where three tectonic-sedimentary domains have been identified: Turin Hills, Langhe Basin and Monferrato Hills. The nature and structure of the bedrock strongly affect some morphological characteristics of this hilly area such as the asymmetry of the valleys and the intense soil erosion activity (Luino, 2005). More than the 50% of the hilly land of this region is characterized by soils that are considered to have high erodibility as they display values of the RUSLE K-factor (Wischmeier & Smith, 1978; Renard et al., 1997) higher than 0.047 Mg ha h $ha^{-1}$ $MJ^{-1}$ $mm^{-1}$ (Ipla, 2007). In fact, soils with high erodibility are characterized by K-factor values higher than 0.05 Mg ha h $ha^{-1}$ $MJ^{-1}$ $mm^{-1}$ (van der Knijff, 2000).

In this paragraph we report a 10-years (2000-2009) dataset of soil loss measurements made in vineyard plots differently managed in a Piedmont vineyard area, in order to analyze the impact of the management on runoff and erosion caused by natural rainfalls.

#### 2.2.2 Materials and methods

*Study site*

The research was carried out in a vineyard of the Experimental Vine and Wine Centre of Piedmont Regional Administration (Italy) called "Tenuta Cannona". The vineyard was in the hilly territory of Alto Monferrato belonging to Monferrato Hills geological domain (Figure 4), in the locality Carpeneto (44°40′ N, 8°37′ E), and covered an area with a mean elevation of 290 m, a SE aspect and a mean slope of about 15%. The climate is temperate, with a mean annual air temperature and a mean annual precipitation calculated on the 2000-2009 period of 13°C and 850 mm, respectively. Rainfalls are mainly concentrated in Autumn

(October and November) and Spring (April and May), while the driest months are June and July (Figure 5). Summer rainfall events are often of high intensity and short duration.
The soils derived from reworked Pleistocene alluvium, had a clay to clay-loam texture and were classified as *Typic Ustorthents, fine-loamy, mixed, calcareous, mesic* (Soil Survey Staff, 2010) or *Eutric Cambisols* (FAO/ISRIC/ISSS, 1998).


Fig. 4. Map of Italy with magnification of the Piedmont region and indication of the study site. Modified from Regione Piemonte (2011b).

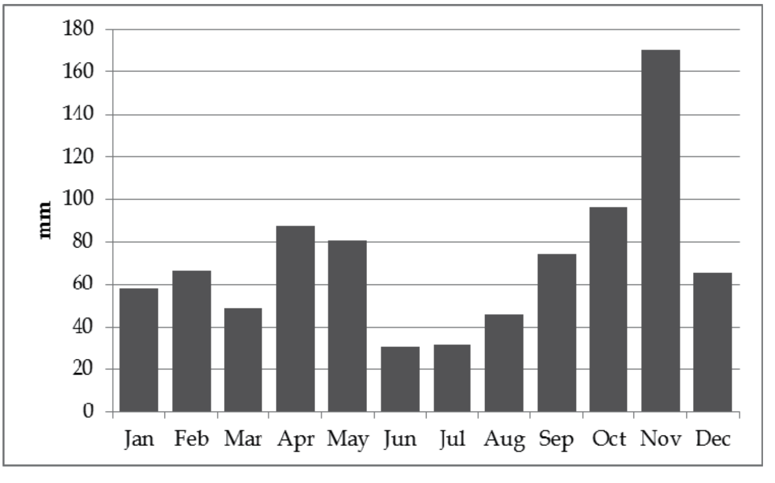


Fig. 5. Annual rainfall distribution in the study area.

*Experimental design*

The vineyard was planted in 1988 with Barbera vines. The vines were spaced 1.0 m on the row and 2.8 m between the rows. The rows were aligned along the slope. Investigation was carried out in three contiguous experimental plots (Figure 6). Each plot was 74.0 m long and

16.5 m width, for a total area of 1221 $m^2$ . Each plot included 7 vines rows. In each plot, the soil of the inter-row was managed with different techniques: conventional tillage (CT), reduced tillage (RT) and controlled natural grass covering (GC) with mulching. The inter-row of the CT plot was harrowed with a chisel and then with rotary cultivator at a maximum depth of 0.25 m. This soil management was the one traditionally adopted in the neighbourhood of the Experimental Centre. The inter-row of the RT plot was harrowed with a rotary cultivator at a depth of 0.15 m. Tillage operations in the CT and RT plots were carried out twice a year, usually in Spring and Summer, only occasionally in Autumn. The inter-row of the GC plot was mown twice a year, in Spring and Summer. Weeds under the rows of the three plots were controlled with a single herbicide application (Glyphosate®) in Spring, for a width of about 60 cm across the vine row. Chemical fertilizer was applied in the vineyard until 2004 by distributing once per year (in Spring or Autumn) the equivalent of 30 kg $ha^{-1}$ of N, 20 kg $ha^{-1}$ of P and 45 kg $ha^{-1}$ of K in form of complex fertilizers.
During the period of observation, once (in Spring) or twice a year (Spring and Autumn) soil samples were collected from each plot in three transects along the slope. Each transect consisted of three sampling places: the centre of the inter-row and the two tractor tracks; for each place, samples were collected at three depths (2-8, 12-18 and 22- 28 cm) by a soil corer with a known volume of 100 $cm^3$. On these samples, bulk density and soil moisture at field capacity were determined. In 2004, surface soil samples were also collected in duplicates to determine soil texture. The measured soil properties are shown in Table 1.

| | CT | RT | GC |
|---|---|---|---|
| Soil texture | [2] | [4] | [2] |
| Sand (60-2000 μm)(%) | 31.0±0.0 | 23.7±3.3 | 18.5±4.9 |
| Silt (6-60 μm) (%) | 33.5±0.7 | 26.0±0.9 | 25.5±2.1 |
| Clay (0-6 μm) (%) | 35.5±0.7 | 50.2±2.5 | 56.0±7.0 |
| Bulk density (g $cm^{-3}$) | 1.42±0.07 | 1.50±0.09 | 1.39±0.08 |
| Soil moisture at field capacity (%) | 24.5±2.2 | 21.0±3.3 | 26.7±3.2 |

Table 1. Soil properties of the three experimental plots (CT=conventional tillage; RT=reduced tillage; GC=controlled grass cover). Values are the average of *n* replicates, indicated within square brackets (± standard deviation).

Each plot was hydraulically bounded. Runoff and sediments coming from each plot were collected at the bottom by a drain. Each drain was connected to a sedimentation trap and then to a tipping bucket device for the runoff measurement (Figure 7). A portion of the runoff-sediment mixture was addressed to a sampler tank. After each rainfall event that was significant for the amount and the intensity of precipitation and during which sediments were transported in the sampler tank, a 1.5 L sample of runoff-sediment mixture was collected in order to measure the sediment concentration in the runoff. The samples were oven-dried and weighed to determine the sediment concentration. Sediments deposited along drains and in the sedimentation traps were collected and weighed. The total soil loss due to each event was obtained adding the two sediment values.
Hourly rainfall measurements was recorded by a rain-gauge station placed at about 200 m from the plots. Rainfall duration, total rainfall and hourly intensity were calculated from the data recorded by the rain-gauge.

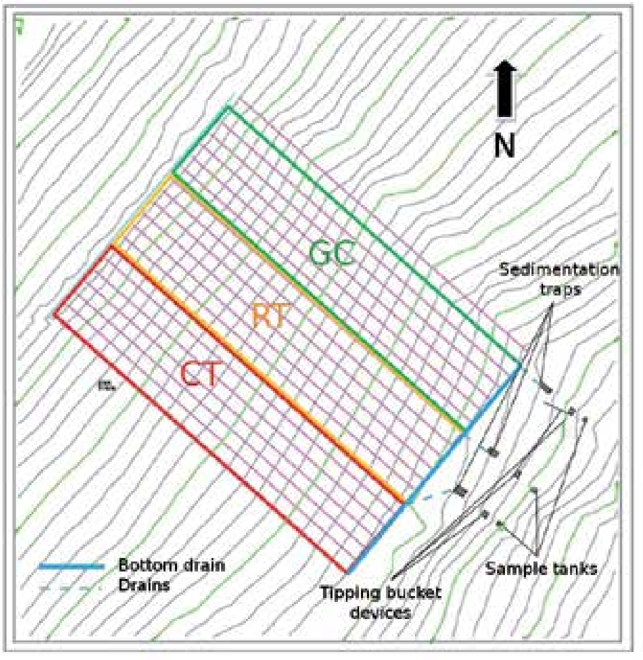


Fig. 6. Scheme of the experimental plots with indicated the position of the runoff collecting system.

Fig. 7. Sedimentation trap and tipping-bucket device for runoff discharge measurements.

#### 2.2.3 Treatment of the data collected

We took into account the rainfalls that produced runoff higher than 0.03 L $m^{-2}$ (equivalent to 0.03 mm) in every plot. During the ten years of observation, 156 rainfall events satisfied this

condition. The precipitation measured during each event was considered as the rainfall that caused runoff, and the runoff was associated to the precipitations that caused it.
In order to analyze the inflow-outflow collected data, rainy (rarely snowy) days were grouped on the basis of seasons and meteorological conditions, as it follows:

- for rainfalls of long duration and low intensity and snowfall, which occurred in Autumn and Winter and rarely in early Spring, the events were made of days with precipitation and the following days with runoff measurements due to drainage flow;
- for rainfalls of short duration, which generally occurred in Spring and Summer and rarely in Autumn, the events were separated by period of 12 (or more) hours without rainfall or with rainfall with hourly intensity below 0.2 mm $h^{-1}$;
- snowfalls were considered in case they were followed by rain, so to cause snow melting. In such occasions, snow and rain contributed to form runoff and generate soil erosion.

### 2.2.4 Results and discussion

*Runoff*

In the 10 years of observation, 8496.6 mm of precipitation were measured at the rain-gauge. The year with the highest precipitation was 2002, with 1257.0 mm of rain water, nearly twice of the amount measured in 2007, the driest year, with only 651.6 mm. In Table 2, the precipitations recorded during the observation period are listed and compared with the rainfalls that produced runoff higher than 0.03 L $m^{-2}$ and with surface flow.
The rainfall events that produced runoff were 156, namely 79% of the total precipitations. The annual distribution of rainfall producing runoff was consistent with the annual distribution of precipitation. In Autumn and Winter, about 80% of the total precipitation (2811.0 and 1609.2 mm, respectively) produced runoff. The proportion of water lost as surface flow was variable in the seasons: the runoff was minimum in Summer, when only 19% of the 760.6 mm of rain producing runoff flowed superficially, and maximum in Winter, with 10-years runoff that reached 70% of the total precipitation. Similar values were reported by Kosmas et al. (1997). As these authors pointed out, in Winter the soil usually has the lowest infiltration rate, due to high moisture content and to the high compaction of the plough layer, so causing high runoff.
Differences among runoff in the different plots are evident in each season. The total runoff was highest in the CT plot during Winter and Spring, while in Summer and Autumn the maximum values were registered in the RT plot. The 10-years amount of runoff was lowest in the GC plot in every season. Single-year runoff values recorded in Summer and Autumn have been in many cases highest in the RT plot and lowest in the GC plot. In contrast with this general observation, three times (in 2002, 2006 and 2007) Winter runoff was higher in the GC plot than in the tilled ones, and four times (in 2000, 2001, 2002 and 2004) the Spring runoff was lower in RT plot than in the CT one. The runoff values associated to rainfalls occurred from December to May have been more variable, probably because of the irrelevance of tillage and grass cover. In fact, the first tillage operation is generally made on May and the second during Summer, so the different soil management influenced differently the runoff production during Summer and Autumn.
The variability of data collected over 10 years confirmed the relevance of having long-term measurements of runoff and soil erosion under natural conditions. In fact, some of the studies that analyzed soil management effects on erosion in vineyard were based on data

obtained under rainfall simulations (Blavet et al., 2009), while others compared data of soil erosion related to natural rainfalls considering periods of observation of 2, 3 or 5 years, with a number of measured events from 9 to 20 (Tropeano, 1984; Kosmas et al., 1997; Ramos & Martínez-Casasnovas, 2007; Raclot et al., 2009).

*Soil loss*

On 156 rainfall events causing runoff, 63 have produced significant soil erosion. The selected events were the most erosive for each observed year. The concentration of sediments obtained from the sample collected after each rainfall event was considered constant during the entire rainfall event. The soil loss caused by the rainfall event was then calculated by the sum of the sediment deposited along channels and in the sedimentation trap plus the product of the sediment concentration per runoff. Runoff events were described with following variables: *i*) duration of each meteorological perturbation (considered on the basis of the described method), expressed in hours (h); *ii*) duration of rainfall within each event of precipitation, in hours (h); *iii*) amount of rain water (R, in mm) measured during the event; *iv*) maximum rainfall intensity over a 60-min period ($I_{60}$max, in mm $h^{-1}$); *v*) mean rainfall intensity (Im, mm $h^{-1}$) recorded during the rainfall event. For each plot, other variables were taken into consideration: *vi*) the measured runoff (RO, in mm); *vii*) the runoff coefficient (RC, in %), obtained by dividing RO for R; *viii*) the soil loss (SL, in kg $ha^{-1}$); *ix*) the soil loss per mm of rain (SLmm, kg $ha^{-1}$ $mm^{-1}$), obtained by dividing SL for R.

The selected events (Table 3) covered a wide range of weather conditions. The duration of the rainfall events ranged from 1 hour (8 August 2004) to 428 hours (from 23 November to 9 December 2002), in which the rainfall duration varied between 1 and 121.5 hours. The event with the longest precipitation occurred from 4 to 21 January 2008, when in 17 days (396 hours monitored) snow and rain fell during 136 hours. Rain water varied from 9 mm (14-16 April 2002) to 208.8 mm (12-30 November 2000). Maximum rainfall intensities over a 60-min period ($I_{60}$max) ranged from 1.2 mm $h^{-1}$ (17-21 April 2008 event) to 76.2 mm $h^{-1}$ (06 August 2002). Mean rainfall intensity (Im) varied between 0.5 mm $h^{-1}$ (17-21 April 2008) and 47.4 mm $h^{-1}$ (08 August 2004). The lowest amount of runoff was recorded from all plots during the 29-30 May 2003 event, when only 0.05, 0.07 and 0.1 mm flowed respectively from the CT, RT and GC plots. The highest runoff values were recorded during Autumn or Winter events for all plots: 132.99 mm in the CT plot from 23 November to 9 December 2002, 133.07 mm in the RT plot on 1-2 February 2009, and 97.15 mm in the GC plot on 12-30 November 2000.

The high variability of the runoff response is remarked by observing the runoff coefficients (RC). The lowest value was 0.3%, recorded in the CT plot during the event of 29-30 May 2003. After each rainfall event with RC lower than 1%, soil losses were measured. The highest RC values were unexpectedly greater than 100%. They were recorded for events occurred during the cold season (18-19 November 2002 and 2-10 February 2009). In these cases, an amount of runoff higher than the amount of rainfall was explained as the consequence of snow melting or the supply from ponds formed during previous precipitation events. Excluding cases with RC greater than 100%, the highest RC was 74.3%, obtained for the CT plot during the event that occurred from 23 November to 9 December 2002. Excluding Winter events, the highest RC (65.8%) was recorded in the RT plot in the period 26 April - 05 May 2009.

To best compare different events, the soil loss per mm of rain (SLmm) was considered. For this variable, the lowest value obtained was 0.006 kg $ha^{-1}$ $mm^{-1}$ in the CT plot after the 30-31 August 2000 event. In contrast, the highest value was 755.026 kg $ha^{-1}$ $mm^{-1}$, considering the soil loss measured during the 13-14 July 2008 event, in the RT plot.

| Year | Rainfall (mm) | | | | | Rainfall that produced runoff > 0.03 mm (mm) | | | | | Runoff CT (mm) | | | | | Runoff RT (mm) | | | | | Runoff GC (mm) | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Wi | Sp | Su | Au | Total | Wi | Sp | Su | Au | Total | Wi | Sp | Su | Au | Total | Wi | Sp | Su | Au | Total | Wi | Sp | Su | Au | Total |
| 2000 | 4.0 | 295.6 | 146.8 | 676.6 | 1123.0 | 0.0 | 129.0 | 96.8 | 575.4 | 801.2 | 0.0 | 2.5 | 1.0 | 193.7 | 197.2 | 0.0 | 1.7 | 2.0 | 157.2 | 160.9 | 0.0 | 9.2 | 1.7 | 183.7 | 194.6 |
| 2001 | 222.4 | 259.0 | 34.2 | 220.2 | 735.8 | 211.2 | 253.4 | 14.6 | 223.8 | 703.0 | 57.7 | 65.9 | 0.5 | 17.4 | 141.5 | 47.1 | 54.9 | 0.9 | 24.4 | 127.2 | 49.3 | 45.8 | 0.2 | 13.2 | 108.6 |
| 2002 | 163.2 | 350.0 | 182.6 | 561.2 | 1257.0 | 137.2 | 351.0 | 134.8 | 428.8 | 1051.8 | 2.8 | 74.0 | 11.7 | 146.5 | 235.0 | 2.5 | 54.8 | 11.0 | 144.1 | 212.4 | 7.2 | 43.8 | 9.2 | 129.2 | 189.4 |
| 2003 | 106.0 | 109.6 | 67.0 | 378.2 | 660.8 | 277.8 | 55.8 | 36.2 | 287.2 | 657.0 | 152.3 | 0.2 | 2.7 | 23.0 | 178.3 | 132.6 | 0.2 | 4.2 | 44.4 | 181.4 | 106.9 | 0.4 | 0.6 | 19.4 | 127.2 |
| 2004 | 249.6 | 172.4 | 78.4 | 269.0 | 769.4 | 200.6 | 205.2 | 47.4 | 224.0 | 677.2 | 74.6 | 52.2 | 8.4 | 10.4 | 145.7 | 43.0 | 36.9 | 8.4 | 30.4 | 118.7 | 37.7 | 36.2 | 5.3 | 3.4 | 82.6 |
| 2005 | 89.0 | 178.6 | 102.6 | 128.2 | 498.4 | 36.0 | 79.8 | 96.4 | 186.6 | 398.8 | 0.1 | 0.5 | 18.7 | 25.3 | 44.7 | 0.4 | 1.2 | 15.0 | 32.3 | 48.9 | 0.2 | 1.1 | 9.5 | 16.5 | 27.3 |
| 2006 | 279.6 | 77.0 | 51.8 | 331.8 | 740.2 | 158.0 | 19.8 | 28.2 | 241.2 | 447.2 | 15.9 | 0.1 | 0.4 | 16.5 | 33.0 | 14.1 | 0.1 | 2.5 | 53.9 | 70.6 | 17.5 | 0.1 | 0.5 | 22.9 | 41.0 |
| 2007 | 127.2 | 155.0 | 193.6 | 175.8 | 651.6 | 93.4 | 82.8 | 157.6 | 163.2 | 497.0 | 2.1 | 0.5 | 7.0 | 1.3 | 11.0 | 3.5 | 0.7 | 14.9 | 4.6 | 23.7 | 11.8 | 0.8 | 5.7 | 1.2 | 19.6 |
| 2008 | 230.4 | 275.2 | 114.4 | 293.2 | 913.2 | 186.4 | 182.4 | 65.0 | 204.8 | 638.6 | 9.2 | 7.3 | 10.9 | 27.3 | 54.7 | 6.7 | 13.2 | 12.3 | 28.2 | 60.3 | 5.7 | 4.2 | 2.1 | 9.6 | 21.6 |
| 2009 | 361.2 | 298.2 | 110.4 | 377.4 | 1147.2 | 308.6 | 270.8 | 83.6 | 276.0 | 939.0 | 293.0 | 75.5 | 10.2 | 21.7 | 400.4 | 171.4 | 86.8 | 21.9 | 15.4 | 295.4 | 116.5 | 26.5 | 2.0 | 16.0 | 160.9 |
| Total | 1832.6 | 2170.6 | 1081.8 | 3411.6 | 8496.6 | 1609.2 | 1630.0 | 760.6 | 2811.0 | 6810.8 | 607.9 | 278.8 | 71.6 | 483.1 | 1441.4 | 421.1 | 250.5 | 93.1 | 534.9 | 1299.6 | 352.8 | 168.3 | 36.8 | 414.9 | 972.9 |

Wi = Winter (December, January, February); Sp = Spring (March, April, May); Su = Summer (June, July, August); Au = Autumn (September, October, November).

Table 2. Total rainfall, amount of rainfall that produced runoff > 0.03 mm, and runoff measured from the three plots in each season during the period of observation (2000-2009).

Considering the soil loss of the three plots, the most erosive event in 10 years of observations was the storm occurred on 06 August 2002, when on the study area about 63.4 mm of rain fell in 5.3 hours, with a peak of rainfall intensity of about 75 mm $h^{-1}$. This event caused the highest soil loss in all the monitored plots, but it was particularly strong in the tilled plots as it occurred just 6 days after the execution of tillage operations; because of this, that event caused a soil loss higher than 21 and 23 Mg $ha^{-1}$ in the CT and RT plots, respectively, while in the GC plot the soil loss was about 1.1 Mg $ha^{-1}$. The lowest soil loss recorded was 0.100 kg $ha^{-1}$, measured on the CT plot after the 30-31 August 2000 event.

Considering the yearly erosion rate in the period of observation (Table 4), the soil loss produced in the tilled plots by the aforementioned rainfall event was exceptional as it was higher than the total sediment load measured on 6 of the 10 years of observations. Both runoff and soil erosion in 2002 were the highest of the period of observation for all plots (total: 147.8 Mg $ha^{-1}$). The yearly soil loss per mm of rain (YSLmm) was highest in 2005 for all plots, when every mm of fallen rain produced a mean soil erosion of 93, 158 and 15 kg $ha^{-1}$ for CT, RT and GC plots, respectively. The yearly soil loss rate and the yearly soil loss per mm of rain were generally highest for the RT plot and lowest for the GC one.

The soil loss measured during the whole period of observation was highest for the RT plot (Table 5) as the erosion rate was of 217 Mg $ha^{-1}$ during the 10-years. The CT and GC soil management allowed a considerable reduction of soil loss, accounting for about 47 and 88%, respectively.

The distribution of soil loss by season varied in each plot and generally was not directly related with the annual distribution of rainfall events that generated them. The total amount of rain fell in Summer in the years 2000-2009 was lower than that fell in other seasons, but it produced the highest soil loss in the tilled plots. In these plots, the Summer rainfall events have been more erosive than the others, with 45 and 38% of total soil loss for RT and CT, respectively. This fact has been confirmed by the soil loss per mm of rain (SLmm) index. The Summer value of SLmm for the RT plot was the highest (174.5 kg $ha^{-1}$ $mm^{-1}$), accounting for more than three times the annual average value (55.7 kg $ha^{-1}$ $mm^{-1}$). The SLmm was always the lowest in the GC plot and its seasonal variation was in the range 4.50-8.0 kg $ha^{-1}$ $mm^{-1}$. During the Summer events, the soil loss from the GC plot was similar to that of Spring and Winter (4.0, 3.6 and 4.3 Mg $ha^{-1}$, respectively in the three seasons); the most erosive season for the GC plot was Autumn, with 14.7 Mg $ha^{-1}$ and a soil loss of 8.0 kg $ha^{-1}$ for each mm of rain. During Autumn precipitations, the GC plot suffered 56% of total erosion.

The absolute highest sediment load was measured after Summer events in the tilled plots (98.0 and 43.4 Mg $ha^{-1}$ for RT and CT plots, respectively). The comparison of the soil loss per mm of rain measured for tilled plots versus the value obtained for the GC plot confirmed the protective action of the grass cover in Summer, when rainfalls generally have a more erosive power. The lowest rates of the soil loss per mm of rain were obtained for all the plots in Spring, when also the measured soil loss were very low as well.

### 2.2.5 Conclusions

The 10-years evaluation on the effect of soil management on erosion of Piedmont vineyards confirmed that, at least at slopes of about 15%, the grass cover plays an important role in reducing water and soil losses, as already reported by Tropeano (1984). In the period of observation, the formation of runoff occurred during 156 rainfall events, and this is an important assessment to consider in evaluating soil water erosion as this latter is a process consisting of two phases: detachment and transport of soil particles. In fact, after detachment, the soil particles may be transported by rain-splash, overland flow (sheet flow) or rills. As the runoff is the responsible for transport by overland flow and rill formation (Morgan, 1995), the measuring of the runoff events is per se an index of the potential soil erosion risk in a certain area.

The collected data showed that the adopted soil managements have strongly affected runoff, and that in the Mediterranean area the greatest amounts of runoff usually occur in Winter, as Kosmas et al. (1997) observed in France and Greece. During the observation period the highest runoff was measured in the CT plot, especially during Winter and Spring, while the lowest amount of water lost by surface flow was always in the GC plot. This study demonstrated that, other features (former soil, slope, topography) being equal, the soil management is responsible for the differences in runoff and soil loss and, probably, also for the variability of some soil properties such as soil moisture. In fact, soil moisture, compaction, crusting and evapotranspiration are strongly related to the adopted soil management, while vegetation cover plays an essential role in soil water balance and in determining soil loss (Vahabi & Nikkami, 2008).

During the period of observation, the highest soil loss was recorded in the tilled plots during Summer months, although the total rainfall and the related runoff have been lower in this season. Because of this, the tilled vineyards are highly subject to erosion in Summer. In fact, in the Mediterranean area, Summer rainfall events are often characterized by short durations and high intensities, so to induce high soil erosion rates. The relationships between rainfall characteristics (such as intensity, amount of rainfall and duration) and soil loss are not so obvious, considering that also many Autumn or Winter rainfalls may cause considerable soil losses. The most erosive storm recorded during the period of observation was in Summer and caused a soil loss of about 22 Mg ha$^{-1}$ in both of the two tilled plots, a value 20 times higher than that measured in the GC plot. Among the numerous variables that affect the soil erosion, the cited case demonstrated that tillage operations made few days before the rain precipitation may strongly affect the soil loss, provided that a high intensity rainfall event occurs (Raclot et al., 2009).

A comparison among the erodibility of seasonal rainfalls on each plot is possible referring to the soil loss for each mm of rain. The maximum value in the GC plot was 8.0 Mg ha$^{-1}$ mm$^{-1}$, related to a 10-years period autumnal rainfall. For the tilled plots, the highest values were 77.1 and 174.5 Mg ha$^{-1}$ mm$^{-1}$ (respectively for CT and RT plots), obtained considering Summer events. It is evident that the protective role of the grass is more effective in Summer, when the ground cover is highest and very erosive rainfall events usually occurs.

Assuming that the erosion has been homogeneous on the whole area of each plot, and considering the value of soil bulk density (Table 1), the thickness of soil lost in 10 years was computed to be 15.3 mm for the RT plot. The adoption of more conservative soil management techniques allowed a reduction of the removed soil thickness, which accounted for 8.2 and 1.8 mm in the CT and the GC plot, respectively.

| Event number | Date event | Sea-son | Duration (h) | | R | Imax | Im | RO GC | RO RT | RO GC | RC CT | RC RT | RC GC | SL CT | SL RT | SL GC | SLmm CT | SLmm RT | SLmm GC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Event | Rain-fall | (mm) | | | (mm) | (mm) | (mm) | (%) | (%) | (%) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) |
| 1 | 2-4/ 4/ 2000 | Sp | 44.0 | 20.0 | 61.0 | 8.8 | 3.1 | 0.5 | 0.5 | 3.5 | 0.7 | 0.9 | 5.8 | 0.58 | 4.24 | 24.62 | 0.01 | 0.07 | 0.40 |
| 2 | 8-11/ 7/ 2000 | Su | 72.5 | 5.5 | 29.6 | 8.2 | 5.4 | 0.7 | 0.7 | 0.6 | 2.4 | 2.3 | 2.1 | 12.66 | 11.03 | 7.62 | 0.43 | 0.37 | 0.26 |
| 3 | 30-31/ 8/ 2000 | Su | 26.0 | 3.0 | 22.0 | 20.6 | 7.3 | 0.1 | 1.0 | 0.8 | 0.5 | 4.5 | 3.8 | 0.10 | 181.79 | 92.98 | 0.00 | 8.26 | 4.23 |
| 4 | 20-21/ 9/ 2000 | Su | 8.0 | 8.0 | 11.0 | 11.0 | 1.4 | 0.4 | 0.8 | 0.6 | 3.6 | 7.4 | 5.5 | 91.73 | 182.17 | 97.97 | 8.34 | 16.56 | 8.91 |
| 5 | 11-13/ 10/ 2000 | Au | 42.0 | 22.0 | 91.4 | 22.0 | 4.2 | 11.7 | 19.0 | 15.4 | 12.8 | 20.7 | 16.9 | 464.74 | 1089.26 | 619.04 | 5.08 | 11.92 | 6.77 |
| 6 | 13-16/ 10/ 2000 | Au | 92.0 | 66.0 | 122.0 | 13.0 | 1.8 | 21.1 | 33.2 | 31.1 | 17.3 | 27.2 | 25.5 | 261.06 | 944.61 | 322.43 | 2.14 | 7.74 | 2.64 |
| 7 | 30-31/ 10/ 2000 | Au | 19.0 | 15.0 | 71.8 | 17.0 | 4.8 | 8.4 | 10.3 | 11.6 | 11.7 | 14.3 | 16.1 | 113.10 | 220.18 | 121.31 | 1.58 | 3.07 | 1.69 |
| 8 | 12-30/ 11/ 2000 | Au | 428.0 | 112.0 | 208.8 | 7.5 | 1.9 | 130.8 | 77.2 | 97.2 | 62.6 | 37.0 | 46.5 | 287.70 | 223.81 | 145.73 | 1.38 | 1.07 | 0.70 |
| 9 | 21-22/ 5/ 2001 | Sp | 34.3 | 18.3 | 45.6 | 20.8 | 2.5 | 0.4 | 0.3 | 1.1 | 0.8 | 0.7 | 2.5 | 1.52 | - | 2.26 | 0.03 | - | 0.05 |
| 10 | 1/ 9/ 2001 | Au | 8.0 | 8.0 | 46.2 | 11.5 | 5.8 | 7.3 | 10.4 | 3.8 | 15.8 | 22.5 | 8.3 | 1640.32 | 2211.49 | 192.59 | 35.50 | 47.87 | 4.17 |
| 11 | 14/ 9/ 2001 | Au | 2.0 | 2.0 | 49.0 | 32.0 | 24.5 | 7.0 | 7.6 | 6.2 | 14.3 | 15.6 | 12.6 | 2369.70 | 4989.83 | 1106.79 | 48.36 | 101.83 | 22.59 |
| 12 | 19-20/ 10/ 2001 | Au | 36.0 | 28.0 | 56.8 | 16.0 | 2.0 | 0.7 | 1.3 | 1.4 | 1.3 | 2.3 | 2.5 | 55.54 | 155.20 | 130.16 | 0.98 | 2.73 | 2.29 |
| 13 | 1-8/ 3/ 2002 | Sp | 156.0 | 42.0 | 58.0 | 4.5 | 1.4 | 1.9 | 1.3 | 2.9 | 3.3 | 2.2 | 5.0 | 53.35 | 34.54 | 17.67 | 0.92 | 0.60 | 0.30 |
| 14 | 14-16/ 4/ 2002 | Sp | 42.0 | 8.0 | 9.0 | 2.0 | 1.1 | 0.8 | 0.8 | 2.1 | 8.9 | 9.0 | 23.4 | 7.99 | 21.85 | 14.76 | 0.89 | 2.43 | 1.64 |
| 15 | 2-15/ 5/ 2002 | Sp | 308.0 | 90.5 | 166.6 | 11.0 | 1.8 | 69.9 | 51.1 | 34.3 | 42.0 | 30.7 | 20.6 | 7976.17 | 2518.50 | 2903.68 | 47.88 | 15.12 | 17.43 |
| 16 | 23/ 5/ 2002 | Sp | 13.0 | 4.5 | 31.2 | 12.4 | 6.9 | 0.6 | 0.9 | 2.5 | 2.0 | 3.0 | 7.9 | 170.66 | 67.83 | 376.07 | 5.47 | 2.17 | 12.05 |
| 17 | 13-17/ 7/ 2002 | Su | 68.0 | 20.0 | 59.2 | 16.0 | 3.0 | 0.2 | 1.1 | 0.3 | 0.4 | 1.9 | 0.4 | 11.38 | 130.89 | 12.19 | 0.19 | 2.21 | 0.21 |
| 18 | 6/ 8/ 2002 | Su | 5.3 | 5.3 | 63.4 | 76.2 | 12.0 | 10.7 | 8.5 | 8.8 | 16.9 | 13.4 | 13.8 | 21164.75 | 23487.52 | 1087.21 | 333.83 | 370.47 | 17.15 |
| 19 | 26-27/ 8/ 2002 | Su | 20.0 | 3.5 | 12.2 | 19.2 | 3.5 | 0.7 | 1.4 | 0.2 | 5.9 | 11.2 | 1.5 | 1268.14 | 1729.07 | 56.78 | 103.95 | 141.73 | 4.65 |
| 20 | 1-4/ 9/ 2002 | Au | 82.0 | 7.0 | 21.0 | 4.5 | 3.0 | 1.4 | 1.9 | 0.2 | 6.4 | 8.9 | 1.1 | 1395.43 | 1185.17 | 16.11 | 66.45 | 56.44 | 0.77 |
| 21 | 5/ 9/ 2002 | Au | 4.0 | 4.0 | 15.0 | 10.2 | 3.8 | 3.2 | 4.1 | 0.8 | 21.0 | 27.3 | 5.3 | 1224.88 | 2057.26 | 46.57 | 81.66 | 137.15 | 3.10 |

| Event number | Date event | Season | Duration (h) | | R | Imax | Im | RO GC | RO RT | RO GC | RC CT | RC RT | RC GC | SL CT | SL RT | SL GC | SLmm CT | SLmm RT | SLmm GC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Event | Rain-fall | (mm) | | | (mm) | (mm) | (mm) | (%) | (%) | (%) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) |
| 22 | 9-11/10/2002 | Au | 57.0 | 44.0 | 89.8 | 8.0 | 2.0 | 2.4 | 5.3 | 2.7 | 2.7 | 5.9 | 3.0 | 635.48 | 268.74 | 27.02 | 7.08 | 2.99 | 0.30 |
| 23 | 18-19/11/2002 | Au | 36.0 | 15.0 | 50.0 | 8.0 | 3.3 | 71.6 | 72.5 | 70.2 | 143.2 | 144.9 | 140.3 | 21385.76 | 16587.09 | 3028.72 | 427.72 | 331.74 | 60.57 |
| 24 | 23/11-9/12/2002 | Wi | 302.0 | 121.5 | 179.0 | 8.0 | 1.5 | 133.0 | 115.3 | 88.1 | 74.3 | 64.4 | 49.2 | 23155.84 | 11673.28 | 1965.47 | 129.36 | 65.21 | 10.98 |
| 25 | 20/5/2003 | Sp | 18.0 | 6.0 | 19.8 | 9.8 | 3.3 | 0.1 | 0.1 | 0.2 | 0.6 | 0.7 | 1.0 | 42.71 | 85.12 | 45.35 | 2.16 | 4.30 | 2.29 |
| 26 | 29-30/5/2003 | Sp | 17.0 | 8.0 | 16.8 | 9.6 | 2.1 | 0.1 | 0.1 | 0.1 | 0.3 | 0.4 | 0.6 | 0.65 | 1.19 | 1.30 | 0.04 | 0.07 | 0.08 |
| 27 | 28/6/2003 | Su | 4.0 | 4.0 | 21.2 | 10.2 | 5.3 | 1.1 | 1.5 | 0.3 | 5.1 | 6.9 | 1.3 | 198.44 | 557.09 | 9.71 | 9.36 | 26.28 | 0.46 |
| 28 | 18/8/2003 | Su | 7.0 | 4.0 | 15.0 | 14.0 | 3.8 | 1.6 | 2.8 | 0.3 | 10.8 | 18.3 | 2.0 | 642.28 | 1274.06 | 112.48 | 42.82 | 84.94 | 7.50 |
| 29 | 31/10-1/11/2003 | Au | 19.0 | 19.0 | 93.8 | 9.6 | 4.9 | 10.3 | 24.4 | 2.6 | 11.0 | 26.0 | 2.8 | 266.97 | 2296.14 | 21.10 | 2.85 | 24.48 | 0.22 |
| 30 | 7-8/11/2003 | Au | 32.0 | 20.0 | 23.8 | 4.2 | 1.2 | 0.2 | 0.3 | 0.5 | 0.8 | 1.4 | 2.0 | 1.71 | 3.30 | 3.36 | 0.07 | 0.14 | 0.14 |
| 31 | 21-28/11/2003 | Au | 181.0 | 106.0 | 153.2 | 10.0 | 1.0 | 11.5 | 19.5 | 16.2 | 7.5 | 12.8 | 10.6 | 320.62 | 781.76 | 177.94 | 2.09 | 5.10 | 1.16 |
| 32 | 30/11-5/12/2003 | Wi | 94.0 | 75.0 | 111.4 | 3.8 | 1.5 | 61.3 | 40.1 | 26.0 | 55.0 | 36.0 | 23.3 | 1850.65 | 1531.71 | 335.00 | 16.61 | 13.75 | 3.01 |
| 33 | 15-17/4/2004 | Sp | 43.0 | 38.0 | 28.4 | 3.6 | 0.7 | 1.0 | 0.8 | 0.2 | 3.6 | 2.9 | 0.7 | 9.08 | 8.29 | 1.26 | 0.32 | 0.29 | 0.04 |
| 34 | 3-6/5/2004 | Sp | 97.0 | 53.0 | 71.2 | 3.6 | 1.3 | 7.6 | 6.5 | 6.0 | 10.6 | 9.1 | 8.4 | 7.56 | 19.54 | 23.86 | 0.11 | 0.27 | 0.34 |
| 35 | 8/8/2004 | Su | 1.0 | 1.0 | 47.4 | 47.4 | 47.4 | 8.4 | 8.4 | 5.3 | 17.7 | 17.8 | 11.2 | 2582.09 | 5089.67 | 740.30 | 54.47 | 107.38 | 15.62 |
| 36 | 16/9/2004 | Su | 7.5 | 7.5 | 20.2 | 6.3 | 2.7 | 3.7 | 5.6 | 0.4 | 18.3 | 27.5 | 1.8 | 232.91 | 1053.64 | 20.14 | 11.53 | 52.16 | 1.00 |
| 37 | 31/10-2/11/2004 | Au | 62.0 | 42.5 | 85.0 | 6.5 | 2.0 | 5.3 | 16.6 | 2.2 | 6.2 | 19.6 | 2.6 | 26.38 | 714.08 | 10.99 | 0.31 | 8.40 | 0.13 |
| 38 | 8/7/2005 | Su | 3.0 | 3.0 | 25.6 | 25.0 | 8.5 | 3.8 | 3.5 | 1.1 | 14.8 | 13.8 | 4.3 | 5623.10 | 9151.38 | 277.01 | 219.65 | 357.48 | 10.82 |
| 39 | 23/7/2005 | Su | 2.0 | 2.0 | 26.2 | 23.6 | 13.1 | 4.1 | 2.5 | 3.0 | 15.6 | 9.4 | 11.4 | 3100.01 | 5262.27 | 407.96 | 118.32 | 200.85 | 15.57 |
| 40 | 27-28/8/2005 | Su | 17.0 | 14.0 | 35.8 | 19.4 | 2.6 | 10.5 | 8.3 | 5.5 | 29.4 | 23.2 | 15.2 | 4427.01 | 5789.88 | 599.88 | 123.66 | 161.73 | 16.76 |
| 41 | 8-9/8/2005 | Au | 27.0 | 19.0 | 92.8 | 21.6 | 4.9 | 22.4 | 25.7 | 13.9 | 24.2 | 27.7 | 15.0 | 3539.99 | 8245.93 | 1458.36 | 38.15 | 88.86 | 15.72 |
| 42 | 18/2-21/2/2006 | Wi | 59.0 | 43.0 | 75.4 | 10.6 | 1.8 | 13.0 | 10.1 | 16.2 | 17.2 | 13.4 | 21.5 | 64.75 | 30.34 | 113.52 | 0.86 | 0.40 | 1.51 |

| Event number | Date event | Sea-son | Duration (h) | | R | Imax | Im | RO GC | RO RT | RO GC | RC CT | RC RT | RC GC | SL CT | SL RT | SL GC | SLmm CT | SLmm RT | SLmm GC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Event | Rain-fall | (mm) | | | (mm) | (mm) | (mm) | (%) | (%) | (%) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) | (kg ha$^{-1}$ mm$^{-1}$) |
| 43 | 14-15/ 9/ 2006 | Au | 31.0 | 24.0 | 143.0 | 13.2 | 6.0 | 15.9 | 49.8 | 21.7 | 11.1 | 34.8 | 15.2 | 525.62 | 13120.18 | 3490.35 | 3.68 | 91.75 | 24.41 |
| 44 | 5-11/ 12/ 2006 | Wi | 94.0 | 46.0 | 56.6 | 12.4 | 1.2 | 0.8 | 1.5 | 5.7 | 1.4 | 2.6 | 10.0 | 18.61 | 21.88 | 62.43 | 0.33 | 0.39 | 1.10 |
| 45 | 22-23/ 1/ 2007 | Wi | 19.0 | 16.0 | 36.8 | 6.6 | 2.3 | 1.3 | 2.0 | 6.2 | 3.6 | 5.5 | 16.7 | 32.13 | 140.58 | 30.77 | 0.87 | 3.82 | 0.84 |
| 46 | 5-6/ 6/ 2007 | Su | 6.0 | 5.0 | 16.6 | 12.0 | 3.3 | 1.3 | 4.8 | 0.7 | 7.7 | 29.2 | 4.0 | 118.99 | 1900.09 | 93.11 | 7.17 | 114.46 | 5.61 |
| 47 | 30-31/ 8/ 2007 | Su | 13.0 | 13.0 | 64.8 | 25.2 | 5.0 | 5.1 | 7.3 | 4.5 | 7.8 | 11.3 | 6.9 | 2741.52 | 5961.59 | 126.86 | 42.31 | 92.00 | 1.96 |
| 48 | 21-26/ 11/ 2007 | Au | 91.0 | 79.0 | 67.0 | 5.4 | 0.8 | 0.9 | 3.1 | 0.7 | 1.3 | 4.6 | 1.0 | 5.95 | 151.78 | 1.38 | 0.09 | 2.27 | 0.02 |
| 49 | 4-21/ 1/ 2008 | Wi | 396.0 | 136.0 | 150.2 | 5.2 | 1.1 | 8.6 | 5.5 | 5.4 | 5.7 | 3.7 | 3.6 | 293.94 | 1096.90 | 255.94 | 1.96 | 7.30 | 1.70 |
| 50 | 17-21/ 4/ 2008 | Sp | 94.0 | 60.0 | 28.2 | 1.2 | 0.5 | 2.5 | 8.3 | 1.1 | 8.9 | 29.5 | 4.0 | 17.63 | 174.58 | 2.28 | 0.63 | 6.19 | 0.08 |
| 51 | 16-19/ 5/ 2008 | Sp | 49.0 | 27.0 | 56.6 | 9.8 | 2.1 | 0.5 | 0.6 | 0.6 | 0.9 | 1.1 | 1.1 | 3.12 | 12.19 | 3.00 | 0.06 | 0.22 | 0.05 |
| 52 | 13-14/ 7/ 2008 | Su | 6.0 | 3.0 | 30.0 | 28.2 | 10.0 | 4.8 | 4.0 | 1.1 | 15.8 | 13.2 | 3.5 | 592.91 | 22650.79 | 110.02 | 19.76 | 755.03 | 3.67 |
| 53 | 31/ 10-3/ 11/ 2008 | Au | 71.0 | 31.0 | 52.6 | 32.4 | 1.7 | 1.6 | 1.9 | 4.5 | 3.1 | 3.5 | 8.6 | 80.93 | 5273.35 | 380.58 | 1.54 | 100.25 | 7.24 |
| 54 | 3-7/ 11/ 2008 | Au | 89.0 | 50.0 | 99.6 | 8.6 | 2.0 | 25.2 | 25.2 | 4.7 | 25.3 | 25.3 | 4.7 | 277.08 | 11369.41 | 74.98 | 2.78 | 114.15 | 0.75 |
| 55 | 2-10/ 2/ 2009 | Wi | 132.0 | 70.0 | 89.6 | 5.0 | 1.3 | 124.5 | 133.1 | 68.1 | 139.0 | 148.5 | 76.0 | 871.58 | 15968.55 | 1498.59 | 9.73 | 178.22 | 16.73 |
| 56 | 28-30/ 3/ 2009 | Sp | 37.0 | 30.0 | 47.6 | 4.2 | 1.6 | 0.9 | 2.2 | 0.3 | 1.8 | 4.5 | 0.7 | 2.65 | 224.86 | 1.67 | 0.06 | 4.72 | 0.04 |
| 57 | 31/ 3-3/ 4/ 2009 | Sp | 66.0 | 31.0 | 44.8 | 9.0 | 1.4 | 14.2 | 8.5 | 2.8 | 31.8 | 19.0 | 6.2 | 455.39 | 9240.01 | 33.49 | 10.16 | 206.25 | 0.75 |
| 58 | 26/ 4-5/ 5/ 2009 | Sp | 207.0 | 50.0 | 108.0 | 7.4 | 2.2 | 54.4 | 71.0 | 22.5 | 50.4 | 65.8 | 20.8 | 380.72 | 2256.20 | 134.80 | 3.53 | 20.89 | 1.25 |
| 59 | 18/ 7/ 2009 | Su | 4.0 | 4.0 | 29.2 | 17.8 | 7.3 | 1.3 | 8.1 | 0.4 | 4.4 | 27.8 | 1.4 | 11.59 | 8704.21 | 3.30 | 0.40 | 298.09 | 0.11 |
| 60 | 08/ 08/ 2009 | Su | 4.0 | 4.0 | 32.6 | 23.0 | 8.2 | 8.4 | 12.0 | 1.4 | 25.8 | 36.7 | 4.2 | 533.46 | 4931.04 | 132.42 | 16.36 | 151.26 | 4.06 |
| 61 | 15-16/ 9/ 2009 | Au | 27.0 | 18.0 | 55.0 | 5.6 | 3.1 | 6.1 | 1.7 | 0.6 | 11.2 | 3.1 | 1.0 | 143.33 | 1247.67 | 11.40 | 2.61 | 22.68 | 0.21 |
| 62 | 7-9/ 11/ 2009 | Au | 37.0 | 37.0 | 61.8 | 5.6 | 1.7 | 0.9 | 0.9 | 0.7 | 1.5 | 1.4 | 1.1 | 0.94 | 26.24 | 0.70 | 0.02 | 0.42 | 0.01 |
| 63 | 21/ 11-1/ 12/ 2009 | Au | 223.0 | 62.0 | 83.2 | 6.8 | 1.3 | 13.1 | 11.0 | 14.2 | 15.7 | 13.2 | 17.1 | 26.10 | 263.94 | 3268.60 | 0.31 | 3.17 | 39.29 |

R: rainfall (mm); $I_{60}$max: maximum rainfall intensity over a 60-min period measured during the event (mm h$^{-1}$); Im: mean rainfall intensity (mm h$^{-1}$); RO: runoff (mm); RC: runoff coefficient (%); SL: soil loss (kg ha$^{-1}$); SLmm: soil loss per mm of rain (kg ha$^{-1}$ mm$^{-1}$). CT=conventional tilled plot; RT=reduced tilled plot GC=grass cover plot.

Table 3. Rainfall, runoff and soil loss characteristics of the 63 events measured in the period of observation (2000-2009) for the studied plots.

| | YR | YRO CT | YRO RT | YRO GC | YRO TOT | YRC CT | YRC RT | YRC GC | YSL CT | YSL RT | YSL GC | YSL TOT | YSLmm CT | YSLmm RT | YSLmm GC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | mm | | | | | % | | | $Mg\ ha^{-1}$ | | | | $kg\ ha^{-1}\ mm^{-1}$ | | |
| 2000 | 617.6 | 173.8 | 142.6 | 160.8 | 477.2 | 28.13 | 23.09 | 26.04 | 1.2 | 2.9 | 1.4 | 5.5 | 2 | 5 | 2 |
| 2001 | 197.6 | 15.4 | 19.7 | 12.6 | 47.6 | 7.79 | 9.95 | 6.37 | 4.1 | 7.4 | 1.4 | 12.9 | 21 | 37 | 7 |
| 2002 | 754.4 | 296.4 | 264.1 | 213.0 | 773.5 | 39.29 | 35.01 | 28.23 | 78.4 | 59.8 | 9.6 | 147.8 | 104 | 79 | 13 |
| 2003 | 435.2 | 86.0 | 88.7 | 45.9 | 220.6 | 19.75 | 20.37 | 10.56 | 3.3 | 6.4 | 0.7 | 10.4 | 8 | 15 | 2 |
| 2004 | 252.2 | 25.9 | 38.0 | 14.0 | 77.9 | 10.29 | 15.05 | 5.57 | 2.9 | 6.9 | 0.8 | 10.5 | 11 | 27 | 3 |
| 2005 | 180.4 | 40.8 | 40.0 | 23.5 | 104.3 | 22.63 | 22.20 | 13.01 | 16.7 | 28.4 | 2.7 | 47.9 | 93 | 158 | 15 |
| 2006 | 275.0 | 29.6 | 61.4 | 43.6 | 134.6 | 10.77 | 22.31 | 15.87 | 0.6 | 13.2 | 3.7 | 17.4 | 2 | 48 | 13 |
| 2007 | 185.2 | 8.5 | 17.3 | 12.0 | 37.8 | 4.61 | 9.31 | 6.48 | 2.9 | 8.2 | 0.3 | 11.3 | 16 | 44 | 1 |
| 2008 | 417.2 | 43.2 | 45.5 | 17.4 | 106.1 | 10.34 | 10.90 | 4.18 | 1.3 | 40.6 | 0.8 | 42.7 | 3 | 97 | 2 |
| 2009 | 551.8 | 223.8 | 248.4 | 111.0 | 583.2 | 40.57 | 45.01 | 20.11 | 2.4 | 42.9 | 5.1 | 50.4 | 4 | 78 | 9 |

YR: yearly rainfall that produced runoff; YRO: measured yearly runoff; YRC: yearly runoff coefficient; YSL: measured yearly soil loss; YSLmm: yearly soil loss per mm of rain, obtained dividing YSL by YR.

Table 4. Yearly rainfall, runoff and soil loss characteristics observed in the three plots (CT=conventional tilled, RT=reduced tilled and GC=grass cover) in the period of observation (2000-2009).

| | | | Winter | Spring | Summer | Autumn | Total |
|---|---|---|---|---|---|---|---|
| Number of events | | | 7 | 15 | 18 | 23 | 63 |
| | $R_{(RO>0.03)}$ | (mm) | 699.0 | 792.8 | 562.0 | 1831.6 | 3885.4 |
| CT | RO | (mm) | 342.4 | 155.4 | 66.9 | 378.9 | 943.6 |
| CT | SL | ($Mg\ ha^{-1}$) | 26.3 | 9.1 | 43.4 | 35.0 | 113.8 |
| CT | SLmm | ($kg\ ha^{-1}\ mm^{-1}$) | 37.6 | 11.5 | 77.1 | 19.1 | 29.3 |
| RT | RO | (mm) | 307.6 | 153.1 | 82.2 | 422.7 | 965.6 |
| RT | SL | ($Mg\ ha^{-1}$) | 30.5 | 14.7 | 98.0 | 73.4 | 216.6 |
| RT | SLmm | ($kg\ ha^{-1}\ mm^{-1}$) | 43.6 | 18.5 | 174.5 | 40.1 | 55.7 |
| GC | RO | (mm) | 215.7 | 80.2 | 35.0 | 323.1 | 654.0 |
| GC | SL | ($Mg\ ha^{-1}$) | 4.3 | 3.6 | 4.0 | 14.7 | 26.5 |
| GC | SLmm | ($kg\ ha^{-1}\ mm^{-1}$) | 6.1 | 4.5 | 7.1 | 8.0 | 6.8 |

$R_{(RO>0.03)}$: amount of rainfall producing runoff>0.03 mm; RO: runoff ; SL: soil loss; SLmm: soil loss per mm of rain.

Table 5. Total rainfall runoff and soil loss characteristics observed in the three plots (CT=conventional tilled, RT=reduced tilled and GC=grass cover) in the period of observation (2000-2010), subdivided by seasons.

### 2.3 Evaluation of erosion in tilled and grass covered soils in vineyards from central Italy

#### 2.3.1 Introduction

Among the natural factors that affect erosion, soil properties are considered as one of the most important. More specifically, erosion processes are influenced by soil characteristics such as particle-size distribution, stability of aggregates, organic matter content, soil chemistry and clay mineralogy (Lal, 1994). All these factors regulate rainfall acceptance and resistance of the soil to particle detachment, and their subsequent transport in form of suspension. Among the parent materials originating soils very susceptible to erosion, the plio-pleistocene pelitic (silt clay) marine sediments are the most diffused in Italy. Here, this type of soil substrate constitutes most of the hills and plains, and mantles the Apennines chain till altitudes that in some cases reach more than 1000 m. In the hill and mountain environments the presence of this parent material has given rise to zones where different types of erosion occurred also because, since the 1950s, many hectares of these areas were deforested and reclaimed to expand agriculture. As in many of these cases soils have not carefully managed, as often happened for vineyards (Pieri, 2007), a loss of organic matter and, consequently, structure and water holding capacity occurred (Pagliai, 2008). This has led to a poorly to moderate degree of aggregation with weak aggregates that make the soils evolved from fine-textured marine sediments susceptible to severe erosion processes (Philips & Robinson, 1998). In fact, in this type of soils aggregates are so weak that collapse under the impact of raindrops and soil develops surface seals and crusts that reduce permeability and, in turns, favour runoff with consequent formation of rills and gullies (Robinson & Phillips, 2001).

In Italy, Marche region is representative of the morpho-climatic context of the Adriatic sector (Gentili et al., 2006), and about 64% of the total surface (969,450 ha) is made of soils derived from fine-textured and alkaline marine sediments. These fine soils were reclaimed to agriculture centuries ago and, on the basis of the data published by ISTAT (2003), we estimate that nowadays about 60% of them are used for arable crops (cereals, legumes, vegetables), while about 3% are occupied by vineyards. In the last decades the surface occupied by vine has dropped after it had reached highest levels in the 1950-60s. However, till the 1950s the soil management of this vulnerable portion of the territory had a low impact on the soil organic matter content while the attention directed to the surface water circulation was so high that erosion was under control and soil fertility was maintained. In the following 50 years, most of the vineyard soils have become even more subjected to erosion, with the formation of rills and gullies during storm episodes occurring in the warm season (Gentili et al., 2006). Because of this, the control of erosion to preserve soil quality and fertility and to maintain agricultural productivity is one of the most pressing environmental objectives that is worthwhile to pursue from both ecological and economic points of view. In the last decade, in the attempt to control erosion, many vineyard soils have been grass covered, but this was often done with no consideration of the soil properties.

The objective of this study was to compare the effects of two different soil managements (harrowing and grass cover) on the water erosion of Inceptisols developed on hilly slopes from fine-textured plio-pleistocene marine sediments under a Mediterranean type of climate.

### 2.3.2 Material and methods

*Study site*

The study was conducted at the Experimental Station of the Agriculture Faculty of Ancona (central Italy), at about 11.4 km from the sea coast (Figure 8) and at an altitude of 203 m, in a vineyard that faces a SSE exposure and has a general slope of 5-6%. The mean annual air temperature is 13.3°C, with July and August as the warmest months and January as the coldest month. The mean annual precipitation is 780 mm, with the maximum rainfalls in April and September. The soil developed from fine-textured plio-pleistocene marine sediments containing carbonates.

The vines were implanted in 1993 after decades of cultivation with cereals. The soil was broken up at about 70 cm and the distance between rows was established at 2.80 m. The harrowed plot was ploughed till a depth of 25 cm for the first three years, afterwards only superficial harrowing operations (8-10 cm) were conducted twice per year, in Spring and Summer. The grass cover plot was ploughed till a depth of 25 cm for the first three years and, then, left to the spontaneous colonisation from herbaceous species, which have been mown two/three times per year. The equivalent of 30 kg ha$^{-1}$ of N in form of ammonium nitrate was distributed only for the first three years. On both harrowed and grass cover plots, for a width of about 60 cm under vine trunks, weeds were chemically eliminated (Glyphosate®).

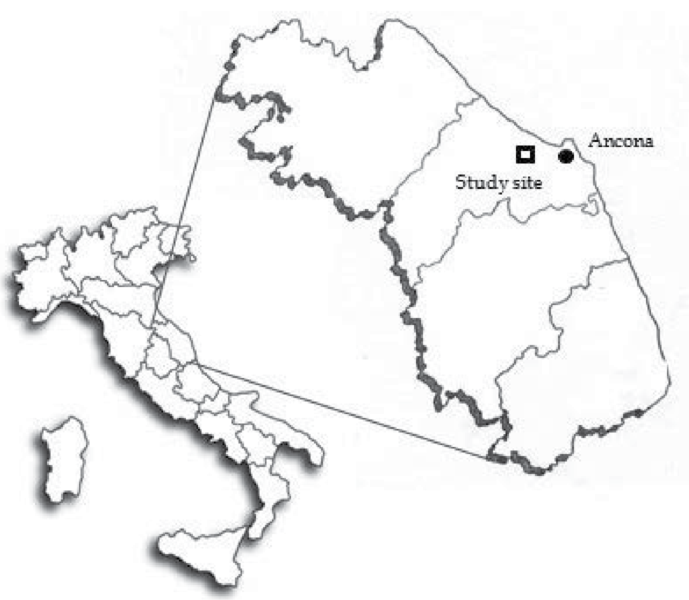


Fig. 8. Map of Italy with magnification of the Marche region and indication of the study site.

*Experimental design*

In 2008, in each plot, two soil profiles were dug in the inter-rows, excluding the surface treated with herbicide; because of this, the width of the trenches was around 2 m. The profiles were described and sampled by horizons. From the face of the profile, soil cores were collected by horizons to estimate their bulk density. The volume of the corer was

251.3 $cm^3$ and each horizon was sampled in double. The soil samples collected by mass were characterized for particle-size analysis after the samples were maintained a night under continuous gentle stirring in deionised water (solid:liquid ratio 1:5) and after dissolution of organic (Lavkulich & Wiens, 1970) and ferric (Mehra & Jackson, 1960) cements; coarse, medium and fine sands (2000-500 µm, 500-250 µm and 250-53 µm, respectively) were recovered by sieving while silt was separated from clay by sedimentation. The soil pH and the electrical conductivity were determined on a suspension with a soil:water ratio of 1:2.5. The available P content was evaluated according to the Olsen method, and the total C and N were determined by a Carlo Erba EA1110 dry combustion analyzer. The humic C content was estimated by the Walkley-Black method without application of heat (Nelson & Sommers, 1996), while the amount of total organic C was determined following Allison (1960). The exchangeable Ca, Mg, K and Na were displaced by a $BaCl_2$ solution and measured with an ICP-OES (Horiba Jobin Yvon Ultima 2). The mineralogical assemblage was determined on powdered specimens by x-ray diffraction with a Philips PW 1830 diffractometer. For the aggregates stability test, standard aggregates (diameter of 1-2 mm) were submitted to three treatments in parallel: *i*) submersion in water for 20 minutes in quiet; *ii*) submersion in water for 20 minutes in quiet plus vertical revolving shaker; *iii*) submersion in a 1M NaOH solution for 20 minutes in quiet plus vertical revolving shaker. For all treatments 20 g of standard aggregates were put in contact with 100 mL of liquid. For the *ii*) and *iii*) treatments, each suspension was put in a 500 mL plastic bottle and submitted to 25 vertical revolutions in 1 minute. On the suspensions obtained after the treatments, the particle-size analysis was carried on as described above.

One of the profiles in the harrowed soil and one in the grass covered soil were installed with plate lysimeters (Prenart soil disks with a diameter of 70 mm) to sample soil solutions after each storm. Three plate lysimeters were put at three different depths: 10, 20, and 40 cm. Soil solutions were collected the day after the end of the perturbation by applying a vacuum of -700 hPa. At the soil surface, in the inter-row of both harrowed and grass cover plots, three Gerlach systems were installed to collect runoff (solution plus sediments) the day after the end each perturbation. The suspensions collected were filtered through cellulose acetate filters (0.45 µm under $N_2$ pressure) to separate solution from suspended particles. Both filtered solutions coming from Gerlach and collected by lysimeters were analyzed for pH, electrical conductivity, concentration of cations (Ca, Mg, K and Na) by an ICP-OES, Horiba Jobin Yvon Ultima 2 and anions ($NO_3$ and $PO_4$) by an isocratic chromatograph Dionex CD20. The liquid phase of the runoff collected by Gerlach systems was characterised also for the concentration of $SO_4$, Cl, Br and F and for the DOC content, while the suspension of the runoff was analysed for total C and N and mineralogy.

Statistical analyses to compare differences among data-sets were performed by the one-way ANOVA with Tukey-Kramer post-test, paired t test, and the linear trend post-test to evaluate tendencies with soil depth (GraphPad Instat 3.1 software).

### 2.3.3 Results and discussion

*Soils*

The soil of the two vineyards showed a different thickness of the Ap horizons because of the different soil management (Table 6). In fact, the harrowed profiles showed an Ap horizon with an average thickness of 3-5 cm and a yellowish-brown colour, while the grass covered ones displayed Ap horizons 12-14 cm thick with an olive brown to brown colour. Both soils were classified as *Vertic Haplustepts, fine-loamy, mixed, mesic* (Soil Survey Staff, 2010) or *Vertic Cambisols* (FAO/ISRIC/ISSS, 1998). Soil structure was rather well developed in all the A and Bw horizons, but it was poorly developed in the BC horizons, those that still maintain morphologies of the parent material also because they were not interested by the breaking up. While in the harrowed soil there were nodules of Fe-Mn oxi-hydroxides at all depths, in the grass covered soil there were precipitations of gypsum below 20-22 cm of depth. In this *ustic* environment, the presence of gypsum was ascribed to the grass evapotranspiration.

The mineralogical assemblage was very similar in both soils, with the most abundant minerals represented by primary minerals such as calcite, plagioclases, and quartz (Table 7). Many phyllosilicates (micas, chlorites, kaolinite, 2:1 clay minerals) were present in small amounts and appeared to have been inherited by the parent material. The BC horizons of both soils showed the same mineralogical composition, indicating that both the soils developed from the same layers of the parent material. The similar mineralogy of the two soils and the absence of any meaningful trend with increasing depth was attributed to the fact that 12 years of distinct soil managements were not able to produce differences in terms of mineralogy. The presence of gypsum in the sub-superficial horizons of the grass covered soil was not detected by the x-ray probably because, even if the concretions were visible, the salt was quantitatively scarce.

The particle-size distribution after removal of cements (Table 8) was similar in both soils ($P>0.05$), confirming again these latter developed from the same parent material. In all the horizons, the silt and clay fraction were the most represented separates. The particle-size analysis without dissolution of cements (Table 8) showed a similar distribution of sand, silt and clay in the two soils ($P>0.05$). However, considering only the Ap and Bw1 horizons, the fine sand was higher in the harrowed than in the grass covered soil ($P<0.008$), while the contrary was true for the coarse sand ($P<0.002$). In the Bw3 horizon of the harrowed soil the high amount of coarse sand was ascribed to an enrichment of carbonates (see inorganic C in Table 10), which have probably improved structure stability (Gee & Bauder, 1986; Dimoyiannis et al., 2002). Observations at an optical microscope revealed that the coarse sand of the Bw3 horizon was partly made of carbonaceous remnants of marine bryozoa and secondary carbonates nodules. Hence, a help in improving the state of aggregation was probably due to the dissolution of the bryozoa remnants with successive re-precipitation of secondary carbonates. As the particle-size distribution after cements dissolution was similar in both soils, the differences obtained without cements dissolution were interpreted as the result of the soil management. In particular, the relatively high amount of coarse sand in the grass covered soils was attributed to a better state of aggregation of these aggregates. However, both soils manifested a relatively good state of aggregation as witnessed by a scarce amount of clay particles obtained by the particle-size analysis without dissolution of cements (Table 8), but also by the presence of a fairly good structure at the surface as well as in the sub-superficial horizons (Table 6). The aggregates stability test indicated that for a similar soil depth, the specimens of the grass covered soil always displayed the coarsest particle-size distribution (Table 9), indicating that the aggregates from this soil were more resistant to disaggregation than those of the harrowed soil. The relative resistance of the aggregates from the grass covered soil to the treatments with water and water plus vertical revolving shaker was ascribed to the presence of grasses. In fact, a better state of aggregation can be promoted by the soil shrinkage caused by roots (Materechera et al., 1992), and they were more abundant in the grass cover than in the harrowed soil. Further, according to Traoré et al. (2000), Whalley et al. (2005) and Fageria & Stone (2006), the presence of herbaceous vegetation increases the stability of aggregates thanks to the rhizodeposition of mucilaginous substances, but also to the humification of plant residues (mostly roots). Our observations were similar to those reported by Uren (1993) and Gregory (2006). In the case of the NaOH plus vertical revolving shaker treatment, the higher stability of the aggregates from the grass covered soil was still ascribed to the permanent grass cover, which may favour the formation of inorganic substances (secondary carbonates and/or Fe, Al, Mn oxi- hydroxides) that contribute to cement the particles.

The different soil management and state of aggregation did not affect the bulk density (Table 10), which, also in the superficial horizons, did not show any statistical difference ($P>0.05$). Evidently, for the soils derived from this parent material the bulk density depended more on the particle-size distribution than on the management and aggregation, at least after 15 years from the breaking up. Similar bulk densities and conditions were found in soils cultivated with cereals at about 25 km from our study site (Corti et al., 2006). Along the profile, variations of bulk density were probably inherited from the parent material rather than caused by pedogenesis. In fact, according to many authors (e.g. Bruckert & Bekkary, 1992; Attou & Bruand, 1998; Tuttle et al., 2005), this kind of parent material is often characterized by strata having different texture, fabric and variable density, which were originated by marine ingressions and seismic activity.

Both the soils showed similar values of electrical conductivity and similar decreasing trends with increasing depth (Table 10). The high electrical conductivity of the epipedons was ascribed to the arrival of marine salts blown by winds, a phenomenon rather frequent in coastal regions till several km from the seashore (Ruijgrok et al., 1995). In our case, the geographic position of the site, not far from the seacoast, was considered as responsible of these results. This fact acquires a relevant importance for these sloping soils as the amount of $Na^+$ arriving with aerosol can produce dispersion of clay minerals, so increasing vulnerability of soil to erosion (Shaikh et al., 1987; Schmittner & Giresse, 1999). The reaction of both soils was sub-alkaline (Table 10), with a tendency to increase with increasing depth ($P<0.0001$). At depth, pH values higher than 8.3 were attributed to the presence of Na-carbonate. The combination of high pH values and the presence of $Na^+$ was considered by Chiang et al. (1987) one of the most important factor in reducing aggregates stability and, consequently, increasing soil erosion. As observed in other agricultural soils (Doran, 1987; Janzen et al., 1992; Van Gestel et al., 1992), total N content was high in the superficial horizon of both soils even though no fertilization was applied for more than 10 years (Tables 10). Always at surface, the content of total N was higher in the grass covered than in the harrowed soil because of the higher organic C content. As expected, along both profiles the N tended to decrease with increasing depth ($P<0.0001$). In both soils, total C assumed different values from one horizons to the other (Table 10). If in the upper part of the profiles these differences are partly due to the decreasing with depth of organic C, in the sub-superficial horizons the differences were mainly due to the inorganic C (Table 10). In

| | Depth | Thickness | Colour[a] | Structure[b] | Consistency[c] | Plasticity[d] | Roots[e] | Boundary[f] | Other observations[g] |
|---|---|---|---|---|---|---|---|---|---|
| | cm | cm | | | | | | | |
| Harrowed | | | | | | | | | |
| Ap | 0-3 | 1-7 | 10YR 6/ 4 | 3f,m cr-abk-sbk | mfr, wss | wps-p | 2 mi, vf, f | ci | skl <1% |
| Bw1 | 3-30 | 18-28 | 2.5Y 4/ 3 | 2m,t pl → f,m,c sbk | mfr-fi, wss | wp | 2 mi, vf, f,m; 1co | cw | skl <1%; 1f Fe-Mn nodules |
| Bw2 | 30-60 | 26-31 | 2.5Y 5/ 4 | 2m,t pl → f abk | mfr-fi, wss | wp | 3 mi, vf, f, m, co | cw | skl <1%; 1f Fe-Mn nodules; erthw |
| Bw3 | 60-64 | 4-5 | 2.5Y 4/ 3 | 2m pl → f sbk | mfr, wss | wps | 2 mi, vf, f, m, co | cs | skl 1-3%; 1f Fe-Mn nodules; cutans |
| BC1 | 64-89 | 26-28 | 2.5Y 6/ 4 | 2th,m pl | mfr, wss | wps | 2 mi, vf, f, m, co | cs | skl 5-10%; 2f Fe-Mn nodules; 1 cutans |
| BC2 | 89-112+ | - | 2.5Y 4/ 3 | 2f pl | mfr, wss | wps | 1 mi, vf, f, m; $v_1$co | - | skl 5-10%; 1f Fe-Mn nodules; 1 cutans |
| Grass covered | | | | | | | | | |
| Ap1 | 0-1 | - | 2.5Y 4/ 4 | 3f,m cr | mfr, wss | wps | 2 mi, vf, f | cb | skl <1%; erthw |
| Ap2 | 1-4 | 3-4 | 2.5Y 4/ 3 | 3c,vc pr → f,m pl | mfr, wss | wps | 3 mi, vf, f | cw | skl <1%; erthw |
| Ap3 | 4-12 | 8-10 | 10YR 5/ 3 | 3c,vc pr → m abk | mfr, wss | wps | 3 mi, vf, m | cw | skl <1%; erthw |
| Bw1 | 12-21 | 9-10 | 10YR 5/ 4 | 2m,c pr | mfi, wss | wps | 2 mi, vf, f; 1 m, c | cw | skl <1%; erthw |
| Bw2 | 21-46 | - | 2.5Y 5/ 3 | 2m pl → f,m sbk | mfr, wss | wps | 3 mi, vf, f; 1 m, c | cb | skl <1%; 1 gypsum cncr; 1f Fe-Mn nodules; erthw |
| Bw3 | 46-64 | 18-40 | 2.5Y 4/ 3 | 2m pl → m sbk | mfr, wss | wps | 1 mi, vf, f, m, c | cw | skl <1%; 1 gypsum cncr |
| BC1 | 64-78 | - | 2.5Y 5/ 3 | 2m pl → f,m abk | mfr, wss | wp | 1 mi, vf, f, m, c | ab | skl 5%; 1 gypsum cncr; cutans |
| BC2 | 78-98 | 20-38 | 2.5Y 4/ 3 | 2th pl → f,m sbk | mfr, wss | wp | 1 mi, vf, f, m, c | cs | skl 5-10%; 1 gypsum cncr; 1f Fe-Mn nodules; cutans |
| BC3 | 98-113+ | - | 2.5Y 5/ 4 | 2th pl → f,m sbk | mfr, wss | wp | 1 mi, vf, f, m, c | - | skl 5-10%; 2 micro-mottles |

Table 6. Morphological description of the profiles dug in the harrowed and grass covered soils under vineyard.

[a] moist and crushed, according to the Munsell Soil Color Charts, 1992 Edition.
[b] 2=moderate, 3=strong; f=fine, m=medium, c=coarse, vc=very coarse, th=thin, t=thick; cr=crumb, abk=angular blocky, sbk=sub-angular blocky, pl=platy, pr=prismatic; → breaking into.
[c] m=moist, fr=friable, vfr=very friable, fi=firm; w=wet, ss=slightly sticky.
[d] w=wet, ps=slightly plastic, p=plastic.
[e] 0=absent, $v_1$=very few, 1=few, 2=plentiful, 3=abundant; mi=micro, vf=very fine, f=fine, m=medium, co=coarse.
[f] a=abrupt, c=clear; w=wavy, s=smooth; b=broken, i=irregular.
[g] skl=skeleton; 1=few, 2=plentiful; cncr=concretions; erthw=earthworms.

particular, where a sudden increase of inorganic C occurred, it was due to the presence of remnants of marine bryozoa. The total organic C and humic C contents, considering the whole profile, did not significantly differ ($P>0.05$) between the two soils (Tables 10). However, if only the upper 30 cm of soil are considered, the grass covered soil resulted more enriched of total organic C and humic C than the harrowed one ($P<0.03$, for both). This higher amount of organic matter in the grass covered soil was evidently obtained in the previous 12 years of grassing and was considered one of the factors that contributed to make the aggregates of the grass covered soil more stable than those of the harrowed soil. The higher amount of organic matter at surface was believed to be the reason of the higher amount of available P, even though its content on absolute value was low in both soils (Table 10).

| | Q | P | Ca | D | M | Ch | K | 2:1CM | A |
|---|---|---|---|---|---|---|---|---|---|
| Harrowed | | | | | | | | | |
| Ap | +(+) | ++ | +++ | (+) | (+) | + | (+) | + | - |
| Bw1 | +(+) | +(+) | +++ | (+) | (+) | + | (+) | +(+) | - |
| Bw2 | +(+) | +(+) | +++ | (+) | (+) | + | (+) | +(+) | - |
| Bw3 | + | ++ | +++ | tr | (+) | +(+) | (+) | + | tr |
| BC1 | +(+) | +(+) | +++ | + | + | + | (+) | (+) | tr |
| BC2 | +(+) | +(+) | +++ | + | + | + | (+) | (+) | tr |
| Grass covered | | | | | | | | | |
| Ap1 | +(+) | +(+) | +++(+) | (+) | (+) | (+) | (+) | +(+) | - |
| Ap2 | ++ | +(+) | +++(+) | (+) | (+) | (+) | (+) | + | - |
| Ap3 | + | +(+) | +++(+) | (+) | + | + | (+) | + | - |
| Bw1 | +(+) | + | +++++ | tr | (+) | tr | (+) | + | tr |
| Bw2 | +(+) | + | +++++ | tr | (+) | tr | (+) | + | tr |
| Bw3 | +(+) | +(+) | +++(+) | tr | (+) | (+) | (+) | +(+) | tr |
| BC1 | +(+) | +(+) | +++ | + | + | + | (+) | (+) | tr |
| BC2 | +(+) | +(+) | +++ | + | + | + | (+) | (+) | tr |
| BC3 | +(+) | + | +++(+) | (+) | + | +(+) | (+) | (+) | tr |

Q=quartz; P=plagioclases (mostly albitic); Ca=calcite; D=dolomite; M=micas;
Ch=primary chlorites; K=kaolinite; 2:1CM=2:1 clay minerals
(HIV and HIS at various degree of Al-OH polymerization, smectites, vermiculites, interlayered mica-2:1 clay mineral); A=amphiboles.
+=about 10%; (+)=about 5%; tr=about 1-3%.

Table 7. Semi-quantitative estimation of the mineralogical assemblage of the harrowed and grass covered soils under vineyard.

| | Depth | After NaClO + DCB | | | | | | Without dissolution of cements | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Sand | | | | Silt | Clay | Sand | | | | Silt | Clay |
| | | Coarse | Medium | Fine | Total | | | Coarse | Medium | Fine | Total | | |
| | cm | g kg$^{-1}$ | | | | | | g kg$^{-1}$ | | | | | |
| Harrowed | | | | | | | | | | | | | |
| Ap | 0-3 | 15 (2) | 14 (3) | 68 (5) | 97 (4) | 605 (15) | 298 (11) | 162 (9) | 227 (12) | 454 (13) | 843 (14) | 152 (12) | 5 (2) |
| Bw1 | 3-30 | 15 (3) | 11 (2) | 70 (8) | 96 (9) | 605 (17) | 299 (8) | 94 (12) | 160 (7) | 607 (18) | 861 (13) | 129 (10) | 10 (3) |
| Bw2 | 30-60 | 19 (3) | 10 (2) | 64 (8) | 93 (7) | 593 (6) | 314 (13) | 94 (10) | 236 (10) | 456 (17) | 786 (17) | 202 (14) | 12 (3) |
| Bw3 | 60-64 | 10 (3) | 7 (2) | 64 (4) | 81 (9) | 598 (1) | 321 (10) | 338 (16) | 212 (13) | 277 (11) | 827 (14) | 161 (16) | 12 (2) |
| BC1 | 64-89 | 15 (4) | 11 (2) | 51 (6) | 77 (8) | 624 (20) | 299 (12) | 34 (8) | 287 (11) | 440 (15) | 761 (12) | 221 (14) | 18 (2) |
| BC2 | 89-112+ | 16 (2) | 15 (3) | 45 (4) | 76 (5) | 622 (4) | 302 (9) | 83 (9) | 152 (7) | 472 (16) | 707 (18) | 281 (16) | 12 (2) |
| Grass covered | | | | | | | | | | | | | |
| Ap1 | 0-1 | 12 (2) | 11 (2) | 68 (4) | 91 (8) | 596 (22) | 313 (14) | 310 (18) | 146 (15) | 343 (22) | 799 (19) | 195 (18) | 6 (1) |
| Ap2 | 1-4 | 12 (3) | 13 (1) | 79 (6) | 104 (8) | 603 (7) | 293 (15) | 208 (14) | 172 (23) | 434 (28) | 815 (19) | 171 (22) | 15 (3) |
| Ap3 | 4-12 | 15 (2) | 12 (2) | 82 (5) | 109 (9) | 581 (9) | 310 (18) | 146 (16) | 183 (12) | 457 (21) | 786 (17) | 205 (15) | 9 (2) |
| Bw1 | 12-21 | 16 (3) | 14 (2) | 82 (6) | 112 (7) | 582 (19) | 306 (12) | 190 (15) | 243 (9) | 353 (27) | 786 (21) | 201 (18) | 13 (3) |
| Bw2 | 21-46 | 17 (2) | 12 (1) | 83 (8) | 112 (9) | 581 (26) | 307 (17) | 144 (18) | 235 (20) | 414 (25) | 793 (23) | 196 (24) | 11 (1) |
| Bw3 | 46-64 | 10 (2) | 9 (1) | 61 (4) | 80 (7) | 575 (26) | 345 (19) | 260 (21) | 237 (10) | 303 (26) | 800 (15) | 189 (14) | 11 (1) |
| BC1 | 64-78 | 8 (0) | 7 (1) | 53 (2) | 68 (3) | 586 (14) | 346 (11) | 265 (20) | 217 (13) | 274 (14) | 756 (21) | 236 (22) | 8 (1) |
| BC2 | 78-98 | 6 (0) | 5 (0) | 49 (2) | 60 (2) | 582 (13) | 358 (11) | 329 (13) | 193 (10) | 255 (13) | 777 (16) | 214 (16) | 9 (0) |
| BC3 | 98-113+ | 56 (4) | 11 (2) | 26 (1) | 93 (5) | 590 (4) | 317 (9) | 69 (8) | 184 (11) | 433 (17) | 686 (20) | 289 (18) | 25 (2) |

Coarse sand: 2-0.5 mm; Medium sand: 0.5-0.25 mm; Fine sand 0.25-0.05 mm.

Table 8. Particle-size distribution after cements dissolution and without cements dissolution for the harrowed and grass covered soils under vineyard. In parentheses the standard errors.

| | Sand | | | | Silt | Clay |
|---|---|---|---|---|---|---|
| | Coarse | Medium | Fine | Total | | |
| | $g\ kg^{-1}$ | | | | | |
| Harrowed | | | | | | |
| *$H_2O$* | | | | | | |
| Bw1 | 302 (13) | 227 (16) | 246 (13) | 775 (16) | 224 (16) | 1 (0) |
| Bw2 | 360 (17) | 207 (14) | 211 (13) | 778 (18) | 205 (16) | 17 (2) |
| *$H_2O$+shacker* | | | | | | |
| Bw1 | 167 (22) | 179 (15) | 277 (15) | 623 (22) | 334 (24) | 43 (2) |
| Bw2 | 176 (22) | 218 (18) | 246 (16) | 640 (24) | 327 (21) | 33 (3) |
| *NaOH+shacker* | | | | | | |
| Bw1 | 72 (13) | 36 (3) | 81 (7) | 189 (23) | 622 (33) | 189 (10) |
| Bw2 | 140 (15) | 48 (2) | 90 (5) | 278 (22) | 617 (30) | 105 (8) |
| Grass covered | | | | | | |
| *$H_2O$* | | | | | | |
| Ap3 | 492 (18) | 157 (12) | 160 (19) | 809 (13) | 190 (13) | 1 (0) |
| Bw1 | 559 (16) | 125 (13) | 167 (16) | 851 (13) | 140 (12) | 9 (1) |
| Bw2 | 420 (23) | 194 (15) | 210 (21) | 824 (13) | 175 (12) | 1 (1) |
| *$H_2O$+shacker* | | | | | | |
| Ap3 | 315 (20) | 190 (16) | 229 (20) | 734 (16) | 247 (18) | 19 (2) |
| Bw1 | 429 (15) | 164 (12) | 182 (14) | 775 (17) | 214 (15) | 11 (2) |
| Bw2 | 391 (14) | 166 (11) | 205 (15) | 762 (18) | 220 (16) | 18 (2) |
| *NaOH+shacker* | | | | | | |
| Ap3 | 122 (14) | 37 (5) | 123 (10) | 282 (29) | 601 (35) | 117 (6) |
| Bw1 | 265 (15) | 115 (13) | 96 (8) | 476 (36) | 479 (32) | 45 (4) |
| Bw2 | 229 (21) | 89 (7) | 106 (8) | 424 (36) | 484 (31) | 92 (5) |

Coarse sand: 2-0.5 mm; Medium sand: 0.5-0.25 mm; Fine sand 0.25-0.05 mm.

Table 9. Particle-size distribution of standard aggregates submitted to stability test. Aggregates came from horizons of harrowed and grass covered soils under vineyard. In parentheses the standard errors.

| | Bulk density | Electrical conductivity | pH ($H_2O$) | Total N | Total C | Inorganic C | TOC | Humic C | Available P |
|---|---|---|---|---|---|---|---|---|---|
| | g $cm^{-3}$ | dS $m^{-1}$ | | g $kg^{-1}$ | | | | | mg $kg^{-1}$ |
| Harrowed | | | | | | | | | |
| Ap | nd | 0.70 (0.03) | 7.95 (0.05) | 1.95 (0.02) | 48.08 (0.46) | 34.98 (0.47) | 13.10 (0.07) | 10.70 (0.18) | 0.6 (0.1) |
| Bw1 | 1.49 (0.12) | 0.30 (0.03) | 8.17 (0.03) | 1.52 (0.01) | 44.26 (0.18) | 35.08 (0.30) | 9.18 (0.19) | 8.53 (0.07) | 0.3 (0.1) |
| Bw2 | 1.60 (0.04) | 0.34 (0.03) | 8.13 (0.03) | 1.46 (0.00) | 41.49 (0.07) | 33.23 (0.18) | 8.26 (0.12) | 7.83 (0.58) | 0.2 (0.0) |
| Bw3 | 1.52 (0.01) | 0.32 (0.02) | 8.20 (0.03) | 1.07 (0.01) | 44.55 (0.41) | 36.65 (1.13) | 7.90 (0.81) | 7.84 (1.18) | 0.3 (0.3) |
| BC1 | 1.65 (0.02) | 0.24 (0.02) | 8.53 (0.04) | 1.04 (0.00) | 36.97 (0.16) | 31.97 (0.16) | 5.00 (0.00) | 4.45 (0.99) | 0.0 (0.0) |
| BC2 | 1.61 (0.02) | 0.29 (0.02) | 8.37 (0.00) | 2.03 (0.01) | 50.28 (0.36) | 45.12 (0.67) | 5.16 (0.43) | 2.67 (0.27) | 0.0 (0.0) |
| Grass covered | | | | | | | | | |
| Ap1 | nd | 0.63 (0.04) | 7.55 (0.02) | 2.46 (0.10) | 63.25 (2.24) | 34.53 (3.46) | 28.72 (2.03) | 28.24 (1.14) | 9.1 (6.9) |
| Ap2 | 1.60 (0.17) | 0.46 (0.03) | 7.93 (0.04) | 1.28 (0.02) | 49.35 (0.10) | 35.40 (0.56) | 13.95 (0.42) | 13.10 (0.52) | 1.6 (0.7) |
| Ap3 | 1.61 (0.10) | 0.26 (0.03) | 8.16 (0.02) | 1.27 (0.01) | 45.15 (0.42) | 33.41 (0.72) | 11.74 (0.45) | 9.66 (0.63) | 1.1 (0.7) |
| Bw1 | 1.61 (0.01) | 0.25 (0.03) | 8.22 (0.02) | 1.45 (0.86) | 64.31 (28.78) | 54.52 (28.78) | 9.79 (0.15) | 8.14 (0.37) | 0.5 (0.5) |
| Bw2 | 1.56 (0.04) | 0.27 (0.02) | 8.28 (0.00) | 0.62 (0.02) | 31.26 (0.08) | 22.46 (0.09) | 8.80 (0.01) | 7.26 (0.08) | 0.5 (0.0) |
| Bw3 | 1.59 (0.07) | 0.22 (0.02) | 8.44 (0.01) | 0.81 (0.00) | 31.49 (1.18) | 25.90 (1.18) | 5.59 (0.11) | 4.80 (0.24) | 0.9 (0.0) |
| BC1 | 1.60 (0.02) | 0.22 (0.02) | 8.44 (0.02) | 0.85 (0.02) | 25.53 (0.26) | 18.30 (0.40) | 7.23 (0.23) | 5.78 (0.25) | 0.6 (0.3) |
| BC2 | 1.58 (0.02) | 0.24 (0.03) | 8.42 (0.00) | 1.07 (0.01) | 25.94 (0.07) | 16.60 (0.07) | 9.34 (0.02) | 7.00 (0.49) | 0.1 (0.1) |
| BC3 | 1.61 (0.11) | 0.21 (0.01) | 8.59 (0.02) | 0.59 (0.01) | 47.08 (1.51) | 42.38 (1.60) | 4.70 (0.41) | 3.53 (0.04) | 0.0 (0.0) |

nd=not determined.

Table 10. Values of bulk density, electrical conductivity, pH in water, total N and C, inorganic C, total organic C (TOC), humic C and available P in the harrowed and grass covered soils under vineyard. In parentheses the standard errors.

*Soil solutions and runoff*

In Table 11 are shown the amount of the rain fallen during each perturbation and that of the solution collected by the lysimeters ranked into quantity classes. For most of the rainfalls (from March to December, with the exception of the event occurred on 3 June, 2009) water was collected from the harrowed soil only at the depth of 10 cm, while in the grass covered soil the lysimeters intercepted water mainly at 20 and 40 cm of depth. This was ascribed to the different soil management as, in the grass covered soil, the water infiltration is favoured by the presence of plant roots (Cerda, 1999) and by a better aggregate stability (also induced by plants), although in Summer the evapotranspiration tends to dry the topsoil. In contrast, in the harrowed soil water infiltration was low because of the weak structure that, under the impact of raindrops, favours the formation of surface sealing and crust (Robinson & Phillips, 2001), so to reduce permeability and promote water ponding. The different behaviour in terms of water infiltration due to the soil managements made possible to collect water during light rainfall events or after long periods of dryness only in the grass covered soil. A good penetration of water was observed in both soils only after intense events (those of 3 June 2009 and 25 March 2010, when it fell an amount of rain of 141 mm in 5 days and 51 mm in 2 days, respectively).
The chemical characteristics of rainfall and solutions collected by lysimeter are reported in Tables 12 and 13. The pH of the soil solutions, although it appeared to be rather dependent on that of the rainfall, did not show a clear trend with depth. It is reasonable that, with respect to the rainfall pH, the increases or decreases of the soil solution pH were controlled by the presence of carbonates in the rain water and by the chemical properties of the soil horizons. The electrical conductivity generally increased from rainfall to the soil solution collected superficially, and this was mostly due to the solubilisation of salts of anthropic origin and blown by wind. When the soil solutions were collected in the deepest lysimeters, the values of electrical conductivity slightly changed with increasing depth. However, both pH and electric conductibility did not significantly differ between the two soil managements. With the exception of some date, rain water contained relevant amounts of Ca, K and Mg, especially in Autumn. This fact indicated that the main source of these cations in the rain was the atmospheric dust coming from Sahara regions (e.g. Camarero et al., 1993; Rodà et al., 1993; Avila et al., 1997) or the mechanical works made in the surroundings (Moreno et al., 1996; Sanusi et al., 1996; Celle-Jeanton et al., 2009), but also the sea spray coming from the relatively close seashore (Lovett & Lindberg, 1984; Keene et al., 1986; Andrè, 2008). For the presence of Na and $PO_4$, the main source was considered the sea spray. The presence of $NO_3$ in the rainfall was rather occasional and probably due to local anthropogenic sources including agriculture (Praveen et al., 2007). When the rain water entered the soil and became soil solution, it started to equilibrate with minerals and plant absorption. Because of this, cations and anions acquired different concentration in function of the equilibria existing along the soil depth. The final result was that, in the grass covered soil solutions not only went deeper than 40 cm of depth more often than in the harrowed soil (Table 11), but on average they were also more concentrated in Ca, K, Mg, Na and $PO_4$. The behaviour of $NO_3$ differed from that of the other ions. In the harrowed soil, few times the solutions went deeper than 40 cm and for three times (1 May, 3 and 24 June 2009) they contained $NO_3$ in high concentration; in contrast, in the grass covered soil at least in five occasions the solution at 40 cm of depth contained $NO_3$ but in concentrations lower than in the corresponding solutions of the harrowed soil. The high

concentrations of $NO_3$ in the deep solutions of the harrowed soil was ascribed to the mechanical works made at the end of April to eliminate weeds. At the soil conditions of that period, the soil oxygenation induced by harrowing has probably favoured mineralization of organic matter, so producing unbalancing between availability of $NO_3$ and absence of plant absorption; as a result, $NO_3$ were leached. Similar results and conclusions were reported by Roggero et al. (2006) for arable soils close to those here studied. In the grass covered soil, $NO_3$-containing solutions arrived at depth only after mowing, which also induced unbalancing between availability and plant absorption, but the $NO_3$ concentration was low as plants may continue to absorb even after they had been mown. During the rest of the year, permanent grassing have ensured the soil solution may arrive at depth deprived of $NO_3$.

If we considered the amounts of runoff collected by the Gerlach systems (Table 14) no significant difference was detected in the amount of water or suspended soil particles between the harrowed and grass covered soils. This means that the grass covered, with respect to the harrowing, had no better effect on the loss of soil material due to erosion. The chemical characteristics of the runoff solutions (Table 15) did not show differences between the two soil managements with the exception of K and Mg, which were more concentrated in the runoff solutions of the grass covered than of the harrowed soil ($P<0.01$). These higher concentrations of K and Mg were attributed to the fact that the runoff solution, running through the grass cover, at the interface between below- and above-ground biomass, washed the vegetal tissues solubilising K- and Mg-containing compounds. The runoff solution was also characterized for its content of dissolved organic C, but also in this case the results did not significantly differ ($P>0.05$). Also the amount of total C and N comprising the suspended material of the runoff (Table 16) did not show significant differences between the two soil managements ($P>0.05$). However, the highest losses of N associated to the soil material transported by the runoff occurred in the two dates of 24 April and 4 June 2009 ($P<0.0003$), for both soil managements. The mineralogical analysis of the runoff suspended material (Table 17) indicated that the transported particles have a mineral composition similar to that of the topsoil, with a preference of 2:1 clay minerals and calcite, namely the easiest minerals to be floated.

### 2.3.4 Conclusions

The comparison of the response to erosion processes of harrowed and grass covered vineyard soils did not show substantial differences between the two managements, although the grass covered soil showed a better aggregate stability atop the profile. The collection of soil solution by lysimeters at different depths showed that the presence of a grass cover favoured water infiltration in depth, but the chemical quality of the soil solutions was similar in both soils for pH, electric conductivity, cations and phosphate content. The only significant difference occurred for the $NO_3$, which was found in higher concentration in the solutions collected from the harrowed soil after the Spring ploughing. Also the characteristics of the liquid and solid phase of the runoff collected by the Gerlach systems gave no statistically significant difference, with the exception of a higher amount of K and Mg in the solutions collected in the grass covered soil.

The results obtained by this study let us to conclude that the vineyard soil managements (at least the two compared in this experiment) in this fine-textured soils characterised by a

| | Rainfall | Harrowed 10 cm | Harrowed 20 cm | Harrowed 40 cm | Grass covered 10 cm | Grass covered 20 cm | Grass covered 40 cm |
|---|---|---|---|---|---|---|---|
| 12 March, 2009 (4) | 8.2 | | | | | | ++ |
| 2 April, 2009 (5) | 15.1 | + | | | | + | |
| 23 April, 2009 (3) | 33.8 | + | | | + | + | + |
| 1 May, 2009 (5) | 17.8 | + | | + | | + | + |
| 3 June, 2009 (5) | 141 | ++ | +++++ | +++++ | ++++ | +++++ | +++++ |
| 24 June, 2009 (4) | 52.7 | + | | + | | + | + |
| 13 October, 2009 (2) | 29.6 | | | | | + | |
| 19 October, 2009 (5) | 9.9 | | | | | + | |
| 25 October, 2009 (3) | 45.3 | | | | | + | + |
| 8 November, 2009 (6) | 42.5 | + | | | | + | + |
| 11 November, 2009 (2) | 30.9 | + | | | | + | ++++ |
| 12 December, 2009 (4) | 20.4 | + | | | | +++++ | +++ |
| 22 December, 2009 (8) | 11.3 | + | | | | + | ++ |
| 3 January, 2010 (6) | 32.6 | | + | + | | + | ++ |
| 7 January, 2010 (3) | 26 | + | + | +++ | | + | ++ |
| 15 January, 2010 (3) | 6.6 | + | | + | | + | + |
| 28 January, 2010 (5) | 18.9 | + | + | ++++ | | + | +++ |
| 5 March, 2010 (18) | 36.8 | + | ++ | ++++ | | + | +++ |
| 25 March, 2010 (2) | 51.5 | +++++ | +++++ | +++++ | | ++++ | ++ |

+=<10 mL, ++=10-40 mL, +++=40-100 mL, ++++=100-300 mL, +++++=>300 mL.

Table 11. Amount of rain-water fallen during each perturbation (in parentheses the number of days the perturbation lasted) and water collected by lysimeters at three depths in the harrowed and grass covered soils under vineyard.

| | | 12 March, 2009 | 2 April, 2009 | 23 April, 2009 | 1 May, 2009 | 3 June, 2009 | 24 June, 2009 | 13 October, 2009 | 19 October, 2009 | 25 October, 2009 | 8 November, 2009 | 11 November, 2009 | 12 December, 2009 | 22 December, 2009 | 3 January, 2010 | 7 January, 2010 | 15 January, 2010 | 28 January, 2010 | 5 March, 2010 | 25 March, 2010 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | Rainfall | 7.7 | 7.7 | 7.8 | 7.9 | 8.6 | 7.5 | 7.7 | 8.0 | 6.1 | 6.1 | 6.5 | 6.6 | 8.4 | 7.3 | 7.6 | 10.3 | 7.6 | 7.8 | 7.8 |
| | 10 cm | | 7.8 | 7.8 | 8.1 | 7.7 | 8.0 | | | | 7.0 | 7.7 | 6.5 | 7.2 | | 7.5 | 8.1 | 8.1 | 7.6 | 7.5 |
| | 20 cm | | | | | 7.8 | | | | | | | | | 7.3 | 7.3 | | | 7.7 | 7.6 |
| | 40 cm | | | | 7.8 | 7.8 | 7.8 | | | | | | | | 7.3 | 7.3 | 8.0 | 8.0 | 7.7 | 7.6 |
| EC | Rainfall | 0.0 | 0.0 | 0.2 | 0.3 | 0.0 | 0.0 | 0.6 | 0.4 | 2.0 | 1.5 | 0.9 | 0.7 | 0.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| (dS m-1) | 10 cm | | 0.4 | 1.1 | 1.1 | 5.7 | 0.4 | | | | 1.9 | 0.7 | 0.7 | 0.5 | | 0.3 | 0.4 | 0.3 | 0.2 | 0.3 |
| | 20 cm | | | | | 0.6 | | | | | | | | | 0.5 | 0.5 | | | 0.4 | 0.3 |
| | 40 cm | | | | 1.2 | 0.7 | 0.9 | | | | | | | | 0.4 | 0.4 | 0.4 | 0.4 | 0.6 | 0.2 |
| Ca | Rainfall | 1.6 | 21.4 | 144.2 | 285.0 | 237.0 | 42.0 | 87.1 | 37.1 | 187.1 | 37.1 | 102.1 | 92.0 | 108.9 | 0.0 | 0.0 | 2.2 | 2.2 | 0.0 | 0.0 |
| (mg kg-1) | 10 cm | | 181.0 | 252.5 | 45.5 | 50.5 | 92.0 | | | | 197.1 | 87.1 | 207.0 | 140.4 | | 29.9 | 7.2 | 8.9 | 47.8 | 2.8 |
| | 20 cm | | | | | 34.0 | | | | | | | | | 198.1 | 101.6 | | | 37.8 | 52.8 |
| | 40 cm | | | | 177.5 | 75.0 | 32.0 | | | | | | | | 91.6 | 21.6 | 37.2 | 32.2 | 57.8 | 37.8 |
| K | Rainfall | nd | 15.5 | 13.5 | 42.6 | 12.6 | 10.3 | 275.8 | 142.8 | 545.8 | 385.8 | 290.8 | 225.5 | 99.3 | 0.0 | 0.0 | 2.2 | 12.7 | 28.2 | 48.2 |
| (mg kg-1) | 10 cm | | 11.1 | 41.6 | 13.1 | 49.6 | 135.6 | | | | 480.8 | 215.8 | 180.5 | 83.8 | | 35.7 | 10.2 | 39.2 | 53.2 | 73.2 |
| | 20 cm | | | | | 12.1 | | | | | | | | | 35.7 | 5.3 | | | 68.2 | 43.2 |
| | 40 cm | | | | 62.1 | 14.1 | 8.1 | | | | | | | | 48.7 | 2.7 | 11.2 | 10.2 | 98.2 | 3.2 |
| Mg | Rainfall | nd | 0.3 | 2.4 | 3.7 | 2.2 | 11.7 | 11.3 | 0.0 | 21.8 | 0.3 | 0.0 | 0.0 | 0.0 | 13.0 | 5.4 | 14.6 | 16.1 | 0.2 | 1.7 |
| (mg kg-1) | 10 cm | | 9.2 | 13.2 | 8.7 | 14.7 | 76.7 | | | | 30.3 | 5.3 | 20.1 | 0.4 | | 0.0 | 10.6 | 10.1 | 2.2 | 6.2 |
| | 20 cm | | | | | 6.7 | | | | | | | | | 0.6 | 23.5 | | | 5.2 | 4.7 |
| | 40 cm | | | | 12.7 | 9.2 | 2.2 | | | | | | | | 0.2 | 13.0 | 11.6 | 9.6 | 10.2 | 7.2 |
| Na | Rainfall | nd | 5.4 | 4.5 | 42.7 | 15.2 | 0.7 | 13.8 | 13.3 | 13.8 | 0.3 | 5.3 | 0.0 | 29.1 | 12.5 | 7.5 | 14.1 | 5.6 | 0.2 | 4.7 |
| (mg kg-1) | 10 cm | | 9.7 | 20.7 | 9.2 | 5.7 | 226.7 | | | | 110.3 | 90.3 | 5.5 | 2.6 | | 0.1 | 13.6 | 7.1 | 0.2 | 2.7 |
| | 20 cm | | | | | 4.7 | | | | | | | | | 8.0 | 16.5 | | | 1.2 | 0.0 |
| | 40 cm | | | | 5.2 | 4.0 | 2.7 | | | | | | | | 13.0 | 7.0 | 10.1 | 5.1 | 15.2 | 0.7 |
| $NO_3$ | Rainfall | 0.0 | 0.0 | 1.1 | 11.6 | 5.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 47.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| (mg kg-1) | 10 cm | | 14.5 | 150.5 | 188.5 | 55.0 | 41.5 | | | | 0.0 | 16.3 | 0.0 | 0.0 | | 0.0 | 0.0 | 0.0 | 12.0 | 0.0 |
| | 20 cm | | | | | 65.0 | | | | | | | | | 0.0 | 0.0 | | | 0.0 | 10.0 |
| | 40 cm | | | | 49.5 | 27.1 | 38.8 | | | | | | | | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| $PO_4$ | Rainfall | 1.3 | 1.2 | 1.1 | 2.0 | 4.1 | 22.3 | 21.8 | 20.3 | 17.6 | 79.6 | 20.0 | 0.0 | 0.0 | 5.5 | 4.9 | 0.0 | 2.0 | 1.1 | 1.7 |
| (mg kg-1) | 10 cm | | 0.7 | 0.6 | 0.5 | 3.9 | 23.5 | | | | 25.0 | 158.1 | 0.0 | 0.0 | | 4.0 | 0.9 | 0.0 | 1.5 | 0.5 |
| | 20 cm | | | | | 2.4 | | | | | | | | | 0.0 | 5.4 | | | 1.6 | 2.6 |
| | 40 cm | | | | 0.2 | 1.3 | 5.6 | | | | | | | | 0.0 | 5.2 | 2.1 | 2.0 | 0.8 | 2.7 |

EC=electrical conductivity; nd=not determined.

Table 12. Principal characteristics of the rainfall precipitation and the soil solutions collected by lysimeters at three depths in the harrowed soil under vineyard. Standard errors are not shown to avoid overcrowding of the table. However, the highest proportion between standard error and mean amounted to 9-10% and occurred in few dates for $NO_3$.

| | | 12 March, 2009 | 2 April, 2009 | 23 April, 2009 | 1 May, 2009 | 3 June, 2009 | 24 June, 2009 | 13 October, 2009 | 19 October, 2009 | 25 October, 2009 | 8 November, 2009 | 11 November, 2009 | 12 December, 2009 | 22 December, 2009 | 3 January, 2010 | 7 January, 2010 | 15 January, 2010 | 28 January, 2010 | 5 March, 2010 | 25 March, 2010 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| pH | Rainfall | 7.7 | 7.7 | 7.8 | 7.9 | 8.6 | 7.5 | 7.7 | 8.0 | 6.1 | 6.1 | 6.5 | 6.6 | 8.4 | 7.3 | 7.6 | 10.3 | 7.6 | 7.8 | 7.8 |
| | 10 cm | | | 7.9 | | 7.8 | | | | | | | | | | | | | | |
| | 20 cm | | 7.8 | 7.8 | 7.9 | 7.8 | 8.0 | 7.6 | 7.8 | 7.7 | 7.3 | 7.8 | 6.6 | 7.1 | 7.3 | 7.5 | 8.1 | 8.1 | 7.6 | 7.6 |
| | 40 cm | 7.8 | | 7.8 | 7.9 | 7.8 | 8.1 | | | 7.8 | 7.0 | 8.6 | 6.5 | 7.1 | 7.2 | 7.5 | 8.0 | 8.1 | 7.6 | 7.6 |
| EC | Rainfall | 0.0 | 0.0 | 0.2 | 0.3 | 0.0 | 0.0 | 0.6 | 0.4 | 2.0 | 1.5 | 0.9 | 0.7 | 0.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| (dS $m^{-1}$) | 10 cm | | | 0.7 | | 0.3 | | | | | | | | | | | | | | |
| | 20 cm | | 0.9 | 0.6 | 0.3 | 0.4 | 0.3 | 1.0 | 1.1 | 1.0 | 2.1 | 0.8 | 0.6 | 0.5 | 0.5 | 0.6 | 0.4 | 0.4 | 0.3 | 0.5 |
| | 40 cm | 0.9 | | 0.7 | 0.7 | 0.4 | 0.4 | | | 1.2 | 1.8 | 0.9 | 0.7 | 0.5 | 0.9 | 0.9 | 0.3 | 0.3 | 0.4 | 0.2 |
| Ca | Rainfall | 1.6 | 21.4 | 144.2 | 285.0 | 237.0 | 42.0 | 87.1 | 37.1 | 187.1 | 37.1 | 102.1 | 92.0 | 108.9 | 0.0 | 0.0 | 2.2 | 2.2 | 0.0 | 0.0 |
| (mg $kg^{-1}$) | 10 cm | | | 211.0 | | 239.0 | | | | | | | | | | | | | | |
| | 20 cm | | 35.5 | 13.5 | 20.5 | 32.0 | 62.0 | 92.1 | 87.1 | 142.1 | 57.1 | 122.1 | 147.0 | 62.9 | 81.6 | 51.4 | 7.2 | 27.2 | 13.3 | 22.8 |
| | 40 cm | 104.0 | | 64.0 | 113.5 | 27.5 | 107.0 | | | 147.1 | 157.1 | 87.1 | 207.0 | 96.4 | 6.1 | 54.9 | 47.2 | 17.2 | 52.8 | 7.8 |
| K | Rainfall | nd | 15.5 | 13.5 | 42.6 | 12.6 | 10.3 | 275.8 | 142.8 | 545.8 | 385.8 | 290.8 | 225.5 | 99.3 | 0.0 | 0.0 | 2.2 | 12.7 | 28.2 | 48.2 |
| (mg $kg^{-1}$) | 10 cm | | | 35.6 | | 8.6 | | | | | | | | | | | | | | |
| | 20 cm | | 11.1 | 8.1 | 9.0 | 43.6 | 195.6 | 285.8 | 275.8 | 285.8 | 480.8 | 225.8 | 170.5 | 88.8 | 0.0 | 1.7 | 8.2 | 26.2 | 43.2 | 78.2 |
| | 40 cm | 13.6 | | 24.6 | 13.1 | 21.1 | 240.6 | | | 335.8 | 450.8 | 260.8 | 180.5 | 94.8 | 0.0 | 0.7 | 13.7 | 26.2 | 63.2 | 93.2 |
| Mg | Rainfall | nd | 0.3 | 2.4 | 3.7 | 2.2 | 11.7 | 11.3 | 0.0 | 21.8 | 0.3 | 0.0 | 0.0 | 0.0 | 13.0 | 5.4 | 14.6 | 16.1 | 0.2 | 1.7 |
| (mg $kg^{-1}$) | 10 cm | | | 7.2 | | 12.7 | | | | | | | | | | | | | | |
| | 20 cm | | 10.2 | 7.7 | 8.7 | 11.7 | 0.6 | 40.3 | 0.0 | 7.8 | 5.3 | 7.8 | 0.1 | 0.0 | 16.0 | 2.5 | 7.1 | 22.1 | 1.2 | 1.2 |
| | 40 cm | 10.2 | | 9.2 | 36.2 | 17.2 | 31.7 | | | 13.8 | 10.3 | 5.3 | 15.1 | 0.0 | 12.0 | 3.0 | 12.1 | 7.1 | 1.7 | 5.2 |
| Na | Rainfall | nd | 5.4 | 4.5 | 42.7 | 15.2 | 0.7 | 13.8 | 13.3 | 13.8 | 0.3 | 5.3 | 0.0 | 29.1 | 12.5 | 7.5 | 14.1 | 5.6 | 0.2 | 4.7 |
| (mg $kg^{-1}$) | 10 cm | | | 11.2 | | 3.2 | | | | | | | | | | | | | | |
| | 20 cm | | 8.7 | 1.7 | 0.7 | 36.7 | 3.7 | 165.3 | 10.3 | 15.3 | 12.8 | 20.3 | 0.0 | 0.4 | 15.5 | 3.6 | 5.6 | 9.1 | 0.2 | 3.2 |
| | 40 cm | 13.7 | | 13.2 | 14.2 | 119.2 | 38.7 | | | 35.3 | 12.8 | 15.3 | 0.5 | 0.4 | 15.5 | 2.6 | 4.6 | 2.6 | 0.2 | 1.7 |
| $NO_3$ | Rainfall | 0.0 | 0.0 | 1.1 | 11.6 | 5.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 47.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| (mg $kg^{-1}$) | 10 cm | | | 59.6 | | 10.4 | | | | | | | | | | | | | | |
| | 20 cm | | 44.5 | 26.5 | 5.4 | 17.5 | 0.0 | 7.8 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 10.0 | 0.0 | 0.0 | 0.0 |
| | 40 cm | 26.1 | | 20.2 | 25.2 | 14.3 | 0.0 | | | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| $PO_4$ | Rainfall | 1.3 | 1.2 | 1.1 | 2.0 | 4.1 | 22.3 | 21.8 | 20.3 | 17.6 | 79.6 | 20.0 | 0.0 | 0.0 | 5.5 | 4.9 | 0.0 | 2.0 | 1.1 | 1.7 |
| (mg $kg^{-1}$) | 10 cm | | | 1.5 | | 2.4 | | | | | | | | | | | | | | |
| | 20 cm | | 0.2 | 1.8 | 1.1 | 5.0 | 21.8 | 22.4 | 21.8 | 20.0 | 23.4 | 20.6 | 0.0 | 0.0 | 6.8 | 3.9 | 0.3 | 0.8 | 1.8 | 1.3 |
| | 40 cm | 0.8 | | 0.4 | 0.6 | 3.5 | 22.6 | | | 21.7 | 22.5 | 25.6 | 0.0 | 0.0 | 5.6 | 5.4 | 0.0 | 2.1 | 4.3 | 1.6 |

EC= electrical conductivity; nd=not determined.

Table 13. Principal characteristics of the rainfall precipitation and the soil solutions collected by lysimeters at three depths in the grass covered soil under vineyard. Standard errors are not shown to avoid overcrowding of the table. However, the highest proportion between standard error and mean amounted to 20-23% and occurred in few dates for Na.

| Date | Duration of the event | Amount of rainfall | Harrowed | | Grass covered | |
|---|---|---|---|---|---|---|
| | | | Solution | Suspended material | Solution | Suspended material |
| | days | mm | L $m^{-1}$ | g $L^{-1}$ | L $m^{-1}$ | g $L^{-1}$ |
| 4 February, 2009 | 3 | 14.4 | 4.010 (0.870) | 0.087 (0.012) | 2.575 (1.930) | 0.007 (0.001) |
| 12 March, 2009 | 4 | 10.4 | 0.875 (0.475) | 0.000 (-) | 0.930 (0.275) | 0.000 (-) |
| 2 April, 2009 | 5 | 19.8 | 0.310 (0.025) | 0.086 (0.009) | 1.775 (1.435) | 0.041 (0.033) |
| 23 April, 2009 | 3 | 40.2 | 0.200 (0.015) | 0.014 (0.003) | 0.965 (0.770) | 0.031 (0.023) |
| 1 May, 2009 | 5 | 17.8 | 0.055 (0.010) | 0.000 (-) | 0.323 (0.300) | 0.004 (0.001) |
| 3 June, 2009 | 5 | 141.3 | 1.380 (0.140) | 0.011 (0.003) | 0.990 (0.305) | 0.260 (0.231) |
| 22 December, 2009 | 8 | 11.3 | 0.240 (0.080) | 0.002 (0.001) | 4.720 (0.520) | 0.385 (0.096) |
| 7 January, 2010 | 3 | 26.4 | 3.830 (0.205) | 0.320 (0.029) | 3.100 (0.370) | 1.345 (0.745) |
| 5 March, 2010 | 18 | 36.8 | 4.970 (0.460) | 0.881 (0.241) | 3.290 (0.510) | 0.193 (0.155) |
| 25 March, 2010 | 2 | 51.5 | 4.780 (0.380) | 1.513 (0.250) | 4.250 (0.535) | 0.329 (0.152) |

Table 14. Volume of runoff and amount of suspended material collected by Gerlach from the surface of the harrowed and grass covered soils under vineyard. In parentheses the standard errors.

gentle slope did not seem to strongly affect the quantity and quality of the solutions and sediments running at the topsoil. In this type of soil, evidently, an inclination of about 5-6% appears unable to trigger off considerable erosion phenomena. Another possible reason of the absence of significant differences in terms of water erosion in the harrowed and grass covered soils has been attributed to the *ustic* soil moisture regime, which indicates a limited amount of water into the soil for great part of the year. In particular, precipitations were scarce and relatively well distributed during the considered study period. Further, the use of localised systems for measuring and collecting the runoff (Gerlach system), together with the spatial variability of the erosion processes, could have not allowed to figure out the real extent of the erosive phenomena in the whole parcels with different soil managements. The presence of $NO_3$ in the soil solution and its consequent leaching appeared associated with agronomic practices, such as ploughing in the harrowed soil and mowing in the grass covered soil, both able to produce unbalancing between availability of $NO_3$-N and plant absorption.

## 3. General conclusions

With the aim to assess the role of soil managements on soil erosion, we contrasted two soil managements (harrowing and natural grass cover) in vineyards of two areas, one in north-west and the other in central Italy, which represent two typical vine growing areas of the Mediterranean basin. Both study sites had similar features such as a hilly sloping aspect, fine-textured soils, and an *ustic* soil moisture regime; yet, they differed for the gradient, which was around 15% at the north-western site and 5-6% at the central site. For both sites, precipitations were characterized by an annual mean around to 800 mm concentrated in Autumn and Winter. Summer rainfall events are often for a short time but with high intensity.

| | | 4 December 2008 | 4 February 2009 | 12 March 2009 | 3 April 2009 | 24 April 2009 | 1 May 2009 | 4 June 2009 | 22 December 2009 | 07 January 2010 |
|---|---|---|---|---|---|---|---|---|---|---|
| pH | Harrowed | 7.89 (0.09) | 7.70 (0.02) | 7.74 (0.07) | 7.66 (0.89) | 7.37 (0.33) | 7.68 (0.21) | 8.14 (0.02) | 7.77 (0.05) | 7.72 (0.04) |
| | Grass covered | 8.08 (0.05) | 7.87 (0.03) | 7.87 (0.08) | 8.72 (0.25) | 7.96 (0.18) | 7.35 (0.22) | 7.96 (0.05) | 7.90 (0.03) | 7.61 (0.03) |
| EC (dS $m^{-1}$) | Harrowed | 0.14 (0.02) | 0.15 (0.02) | 0.14 (0.03) | 0.90 (0.67) | 0.22 (0.06) | 0.13 (0.06) | 0.41 (0.14) | 0.24 (0.05) | 0.21 (0.03) |
| | Grass covered | 0.21 (0.03) | 0.22 (0.03) | 0.22 (0.03) | 3.22 (0.20) | 0.48 (0.11) | 0.10 (0.03) | 0.31 (0.06) | 0.37 (0.07) | 0.14 (0.04) |
| Ca (mg $kg^{-1}$) | Harrowed | 19.89 (4.10) | 11.45 (0.85) | 10.80 (1.48) | 10.10 (3.2) | 14.90 (2.6) | 7.40 (1.76) | 30.95 (6.5) | 12.50 (3.75) | 13.35 (3.55) |
| | Grass covered | 31.34 (4.56) | 18.06 (4.12) | 15.50 (2.75) | 14.60 (3.23) | 33.00 (1.16) | 8.00 (2.88) | 20.40 (2.56) | 14.49 (2.81) | 10.11 (1.38) |
| K (mg $kg^{-1}$) | Harrowed | 3.13 (1.85) | 3.50 (1.52) | 1.10 (0.33) | 3.45 (1.75) | 3.35 (1.95) | 0.41 (0.11) | 3.50 (2.5) | 5.20 (2.16) | 4.10 (0.38) |
| | Grass covered | 5.49 (1.03) | 3.03 (0.43) | 2.70 (0.35) | 8.90 (1.55) | 7.50 (1.76) | 1.20 (0.26) | 15.10 (2.50) | 14.60 (3.55) | 6.90 (1,07) |
| Mg (mg $kg^{-1}$) | Harrowed | 2.36 (1.02) | 1.40 (0.30) | 1.10 (0.49) | 4.05 (2.35) | 1.65 (0.35) | 0.50 (0.28) | 3.45 (2.55) | 2.50 (0.88) | 3.10 (0.75) |
| | Grass covered | 2.51 (0.46) | 2.02 (0.33) | 1.80 (0.20) | 6.20 (1.09) | 3.10 (0.65) | 1.30 (0.45) | 3.60 (0.40) | 3.88 (0.66) | 3.10 (0.44) |
| Na (mg $kg^{-1}$) | Harrowed | 3.91 (0.35) | 3.55 (0.45) | 2.40 (0.64) | 11.50 (0.90) | 3.25 (0.05) | 0.60 (0.38) | 4.95 (4.05) | 9.50 (5.44) | 5.25 (0.05) |
| | Grass covered | 5.36 (0.63) | 3.50 (0.18) | 3.20 (0.15) | 12.90 (3.32) | 4.60 (1.02) | 1.10 (0.29) | 3.40 (0.31) | 8.10 (0.22) | 4.02 (0.38) |
| $NO_3$ (mg $kg^{-1}$) | Harrowed | 0.62 (0.01) | 3.44 (2.92) | 0.76 (0.37) | 28.45 (24.35) | 54.00 (41.06) | 3.36 (1.12) | 32.80 (27.5) | 0.00 (-) | 0.30 (0.29) |
| | Grass covered | 0.00 (-) | 0.62 (0.24) | 0.60 (0.25) | 2.60 (0.88) | 75.80 (21.73) | 4.08 (0.92) | 16.90 (5.93) | 1.70 (0.38) | 0.50 (0.16) |
| $PO_4$ (mg $kg^{-1}$) | Harrowed | 0.00 (-) | 0.27 (0.17) | 0.37 (0.13) | 1.15 (0.87) | 3.78 (3.37) | 0.16 (0.08) | 2.65 (1.40) | 0.36 (0.11) | 0.21 (0.14) |
| | Grass covered | 0.00 (-) | 0.10 (0.02) | 0.13 (0.04) | 4.68 (1.58) | 10.70 (3.55) | 0.21 (0.09) | 4.31 (1.86) | 8.09 (1.78) | 0.95 (0.24) |
| $SO_4$ (mg $kg^{-1}$) | Harrowed | 2.85 (0.95) | 2.82 (0.73) | 3.80 (1.21) | 12.35 (5.70) | 4.27 (3.27) | 2.85 (0.95) | 11.87 (7.12) | 19.94 (9.92) | 4.75 (2.85) |
| | Grass covered | 2.80 (0.64) | 2.85 (0.44) | 3.80 (0.60) | 27.54 (6.34) | 11.40 (3.47) | 2.85 (0.28) | 8.55 (2.52) | 5.70 (0.86) | 2.85 (1.22) |
| Cl (mg $kg^{-1}$) | Harrowed | 3.55 (0.35) | 5.37 (0.66) | 5.67 (0.81) | 50.52 (19.68) | 4.08 (2.16) | 5.14 (2.30) | 12.59 (10.46) | 38.29 (12.45) | 3.90 (1.77) |
| | Grass covered | 4.25 (0.36) | 7.45 (0.54) | 7.09 (0.69) | 87.92 (20.02) | 9.93 (1.83) | 7.45 (2.07) | 11.34 (3.48) | 28.36 (4.56) | 3.90 (0.82) |
| Br (mg $kg^{-1}$) | Harrowed | 0.00 (0.00) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) |
| | Grass covered | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) | 0.00 (-) |
| F (mg $kg^{-1}$) | Harrowed | 1.52 (0.76) | 0.87 (0.28) | 0.57 (0.15) | 0.85 (0.66) | 0.09 (0.02) | 0.28 (0.09) | 0.38 (0.19) | 0.00 (-) | 0.19 (0.00) |
| | Grass covered | 1.52 (0.42) | 0.76 (0.19) | 0.19 (0.04) | 4.94 (1.34) | 0.19 (0.08) | 0.19 (0.10) | 0.19 (0.11) | 0.18 (0.08) | 0.19 (0.10) |
| DOC (mg $L^{-1}$) | Harrowed | nd | nd | 1.37 (0.07) | 2.29 (0.25) | 5.67 (0.30) | nd | 6.56 (0.38) | nd | 9.50 (6.88) |
| | Grass covered | nd | nd | 2.79 (0.27) | 2.88 (0.38) | 6.36 (0.67) | nd | 7.12 (0.42) | nd | 10.15 (8.41) |

EC=electrical conductivity; nd=not determined.

Table 15. Principal characteristics of the runoff solution collected by Gerlach in the harrowed and grass covered soils under vineyard. In parentheses the standard errors.

| | 4 February, 2009 | 24 April, 2009 | 4 June, 2009 | 22 December, 2009 | 7 January, 2010 | 5 March, 2010 | 25 March, 2010 |
|---|---|---|---|---|---|---|---|
| | g kg$^{-1}$ | | | | | | |
| Total C | | | | | | | |
| Harrowed | 211.7 (65.4) | 138.3 (28.5) | 259.7 (44.6) | 597.3 (6.2) | 71.5 (0.1) | 35.5 (0.4) | 43.8 (0.2) |
| Grass covered | 213.7 (1.3) | 132.4 (52.5) | 206.5 (97.6) | 78.2 (0.8) | 61.8 (11.1) | 54.1 (6.7) | 44.75 (1.3) |
| Total N | | | | | | | |
| Harrowed | 11.2 (1.1) | 23.7 (3.3) | 44.3 (18.2) | 21.0 (2.0) | 3.9 (0.0) | 5.0 (0.0) | 4.6 (0.0) |
| Grass covered | 11.5 (0.2) | 18.2 (6.9) | 46.3 (24.1) | 5.9 (0.0) | 2.5 (0.2) | 6.7 (0.5) | 5.5 (0.9) |

Table 16. Amounts of total C and N comprising the suspended material of the runoff in the harrowed and grass covered soils under vineyard. In parentheses the standard errors.

| | Q | P | Ca | D | M | Ch | K | 2:1CM | A |
|---|---|---|---|---|---|---|---|---|---|
| 02 April, 2009 | | | | | | | | | |
| Harrowed | ++ | (+) | +++(+) | tr | tr | (+) | (+) | ++(+) | - |
| Grass covered | +(+) | + | ++++ | tr | (+) | + | (+) | +(+) | - |
| 22 December, 2009 | | | | | | | | | |
| Harrowed | ns[a] | ns | ns | ns | ns | ns | ns | ns | ns |
| Grass covered | ++ | + | ++++ | tr | tr | (+) | (+) | +(+) | |
| 07 January, 2010 | | | | | | | | | |
| Harrowed | ++ | (+) | ++++ | tr | tr | (+) | (+) | ++ | - |
| Grass covered | ++ | (+) | +++++ | tr | tr | (+) | (+) | + | - |
| 5 March, 2010 | | | | | | | | | |
| Harrowed | + | (+) | ++++ | tr | tr | + | (+) | ++(+) | - |
| Grass covered | ++(+) | + | ++++ | tr | (+) | tr | (+) | + | - |
| 11 March, 2010 | | | | | | | | | |
| Harrowed | + | (+) | +++++ | tr | tr | (+) | (+) | ++ | - |
| Grass covered | +(+) | (+) | +++++ | tr | (+) | (+) | + | + | - |

Q=quartz; P=plagioclases (mostly albitic); Ca=calcite; D=dolomite; M=micas; Ch=primary chlorites; K=kaolinite; 2:1CM=2:1 clay minerals (HIV and HIS at various degree of Al-OH polymerization, smectites, vermiculites, interlayered mica-2:1 clay mineral); A=amphiboles.
+=about 10%; (+)=about 5%; tr=about 1-3%.
[a]ns=not sufficient sample available

Table 17. Semi-quantitative estimation of the mineralogical assemblage of the suspended material forming the runoff of different dates in the harrowed and grass covered soils under vineyard.

Concerning with the relationship between grass cover and soil loss, the results obtained in the two sites conflicted. Where the slope was about 15%, the grass cover played an important role in reducing water runoff and soil losses. The protective role exerted by the grass cover was more effective in Summer, when very erosive rainfall events usually occur, producing the highest soil losses in the harrowed soil. In contrast, where the slope was about 5-6%, the grass cover reduced runoff, since water penetrated the soil more than in the harrowed soil, but it was ineffective in reducing soil erosion. Such different results might be partly attributed to the absence, for one site, of a long-term monitoring of precipitations, water runoff and soil loss, as soil erosion response can vary greatly from one year to another and is related with amount and pattern of rainfall. Further, part of the discrepancy could be also attributed to the different protocols and devices adopted in measuring the effect of rainfall: an automatic and large runoff sampler in one case, replicated manual Gerlach systems in the other case. However, just because the protocols were different, the results obtained are corroborated by different approaches as well as lab and data analyses. Following this consideration, on the basis of the results obtained we believe that the discrepancy in soil loss can be mostly ascribed to the different slope: at slopes of about 15% (or more), when erosion may be relevant, grass cover is able to reduce soil loss, while at slopes of about 5-6% (or less) the grass cover does not offer any advantage in reducing soil loss.

However, the grass cover of the inter-rows of vineyards is recommended for any slope even in a *ustic* environment for its positive effects. In this study we have shown that grass cover may improve the soil structure, increase the water infiltration, and stock organic N at surface after to have reduced the $NO_3$ concentration in the soil solution. The grass cover also increased the concentration of K and Mg in the runoff solution. However, since the grasses reduce runoff as the water is facilitated in penetrating the soil, this latter negative aspect may be considered as negligible with respect to the positive effects they exert.

The study of the influence of the soil management on the erosion processes and, to a larger extent, soil degradation should represent a high-priority issue within the "environmental agenda". In particular, since viticulture is a cultivation rather diffuse all over the World and, especially in Europe, vineyards are often implanted on soils vulnerable to erosion, the solution of this problem in vineyard soils could greatly contribute to reduce erosion at watershed or regional level.

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# Part 3

## Dry Environments

7

# Hydrological Effects of Different Soil Management Practices in Mediterranean Areas

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## 1. Introduction

In Mediterranean environment intensive agricultural activities are often practiced in steep slopes, where sometimes climatic, geomorphologic and land use factors (e.g. the high rainfall intensity, the scarce vegetal coverage, especially on the occasion of the early rainfalls, the low organic matter content of soils, etc.) worsen the impacts of soil erosion. In such contexts agriculture may play an important role both in terms of economic and social spin-offs (e.g. peopling of hilly marginal lands) as well as under the environmental aspect (e.g. control of erosion phenomena). This is the case of olive growing practiced in hilly lands with a low tree density (e.g. in Southern Italy), often subjected to torrential rainstorms. Therefore, soil degradation problems in such agricultural steep lands under semi-arid conditions must be accounted for through proper soil management systems with low environmental impacts (mainly on soil hydrology).

Until recently, the most common practice for soil conservation in many Mediterranean regions, as Andalusia (Spain, Gomez et al., 2003) and Sicily or Calabria (Italy) has been tillage: however, the tradition of frequent tillage, aimed at preventing competition from natural vegetation for water and nutrients with the olive tree and at facilitating olive harvesting, has exacerbated the problems of erosion and soil degradation (Gomez et al., 2009a). Alternative practices to tillage include: no-tillage with herbicides to maintain a bare and weed-free soil (which sometimes results in accelerated soil erosion due to an increase in water runoff) or the use of a cover crop to protect the soil during autumn and winter, either sown in early autumn or from the regeneration of the natural vegetation after the onset of rains (Gomez et al., 2009a, 2009c). The cover crop is controlled by mowing or by herbicide in spring to reduce the risk of competition for water with the trees, which represents the main limiting factor for plant growth in semi-arid lands, where the evapo-transpiration rate is very high and water resource is scarce.

Studies on soil erosion in orchards in Mediterranean environment have analyzed the hydrological effects of the traditional different managements systems (e.g. Dastgheib & Frampton, 2000; Gago et al., 2007; Gomez et al., 2003, 2009b, 2009c; Monteiro & Moreira, 2004); the important role for soil conservation played by the crop cover has been also highlighted, thanks to the rainfall interception and infiltrability increase (Kosmas et al., 1997; Gomez et al., 2003, 2009b, 2009c; Ramos & Martinez-Casasnovas, 2004).

In spite of the results achieved in these studies, the information about the impacts of different management practices on soil losses is still insufficient and does not allow a proper evaluation of the erosive risks in hilly olive groves across the different local conditions. This consideration is reflected in the contradictory results found in the literature. For example, Pastor et al. (1999) reported that, despite more rill erosion, no-tillage reduced soil losses as compared to conventional tillage, but Francia et al. (2000) measured the opposite effect in runoff plots. The few short term experiments mentioned can not capture the long-term effects of soil management and, to the present knowledge, no previous work has attempted to assess systematically the effects of all soil management practices on soil losses in olive orchards (Gomez et al., 2003). These latter Authors argued as well that "the scarcity of experimental results is the bottleneck for improving the estimation of management effects on the rate of soil losses in olive plantations. Until additional field experiments measuring actual soil loss rates, and field surveys estimating historical rates of soil loss, are carried out at different conditions and scales, erosion rates will remain highly uncertain. On the other hand, qualitative observations indicate that the magnitude of the erosion problem in olive groves on steep slopes is such that the role of alternative soil management in limiting soil loss should be urgently assessed".

However, because of the high variability that characterizes the Mediterranean environments, soil erosion varies considerably over space and time and in most cases it is inappropriate to extrapolate these measures to other spatial units, where different hydrological and erosive processes take place (Taguas et al., 2010). Thus further detailed investigations also at plot scale could integrate literature data, in order to estimate in different contexts the magnitude of the erosive risk: this latter, considering that monitoring activities of surface runoff and soil loss are time consuming and expensive tasks, can be assessed also through a modeling approach by mathematical simulation of water runoff and soil erosion processes.

As well known, prediction models are useful tools for monitoring and controlling the impacts of soil erosion (e.g. Engel et al., 1993; Licciardello et al., 2007; Zema et al., 2011). While the potential of process based models is greater in comparison to empirical ones, their complexity means larger data requirements, potentially greater problems of error propagation and increased difficulty in understanding the way the model simulates the erosion processes (Favis-Mortlock et al., 2001). Published comparisons between the two types show that the average error and model efficiency in predicting soil loss are similar (Morgan & Nearing, 2000; Tiwari et al., 2000). Thus, empirical models, mainly the Universal Soil Loss Equation (USLE; Wischmeier & Smith, 1978) or its derivatives (e.g. RUSLE; Renard et al. 1997), are still widely used (Gomez et al., 2003): in fact, the reduced data requirement and simplicity of USLE-type models (compared to process-based ones) make them useful tools for planning activities destined to soil conservation workers (e.g. Taguas et al., 2010).

Such considerations have stimulated research activities to evaluate and predict the erosion risks in hilly olive groves of Calabria region (Southern Italy), where olive growing represents a fundamental sector of local economy and the most important land use. Within such research activities, this paper aims at: (i) integrating the literature data on the hydrologic effects of three soil management practices (conventional tillage, no tillage and crop cover) typical of the Mediterranean olive groves; (ii) drawing indications on erosion prediction capability of the RUSLE model for the experimental conditions.

## 2. Materials and methods

### 2.1 The study area

The study area is located on the northern side of the torrent Menga valley near Gallina di Reggio Calabria in Southern Italy (Figure 1). The site lies at an altitude of approximately 250 m above sea level; predominant aspect is south. Soil has been classified as sandy-loam (USDA SCS, 1984). The climate of the area is typically Mediterranean, with a mean yearly precipitation of ca. 600 mm, most of which are concentrated in fall and winter periods. Mean monthly temperatures range from 11.5 °C in January, which is the coldest month in the year, to 26.5 °C in July (Bombino et al., 2004).

### 2.2 The experimental design

In 1991 a research group of the University of Reggio Calabria established nine experimental plots at the site (Figure 2), in order to monitor runoff and soil erosion under different slope and vegetation conditions (Bombino et al., 2002). The plots were characterized by different lengths and slope; the three longer (33 m) plots had a 9% slope, whereas three of the six shorter (22 m) plots had a 9% slope and three an 18% slope. A sheet metal cutoff wall, fixing 30 cm into the soil and protruding 20 cm above the ground surface, was installed around the upper and the two adjacent sides of each plot in order to hydrologically isolate the plots. On the lower side of each plot, a 1-m$^3$ tank was installed to collect runoff volumes and sediment loads. Rainfall has been recorded at the site since 1990, using a tipping bucket rain gauge, but measurements of runoff and sediment concentrations from the plots have been available since February 2002. Monthly and after each storm event, the sediment load collected in the tank was well mixed and several 1-liter suspended sediment samples were taken from different depths within the tank. The sediment concentration in each sample was determined by oven drying at 105 °C and the mean value of the samples was calculated. The sediment load from each plot was then calculated as the product of the mean sediment concentration and the water volume measured in the tank.

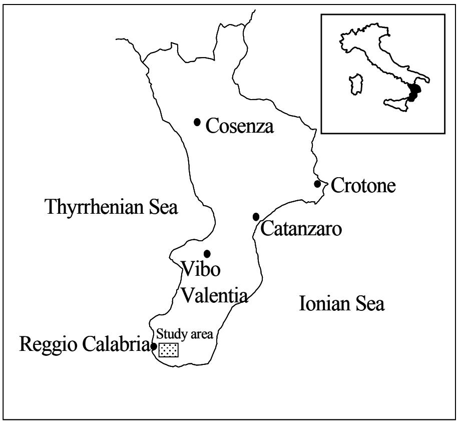



Fig. 1. Location of the study area.

In order to determine the plot vegetal coverage, monthly surveys have been performed in each plot since October 2001. The canopy cover of herbaceous and shrub layers (in %) was evaluated within 1 x 1-m$^2$ sample areas (at least 1 every 25 m$^2$).

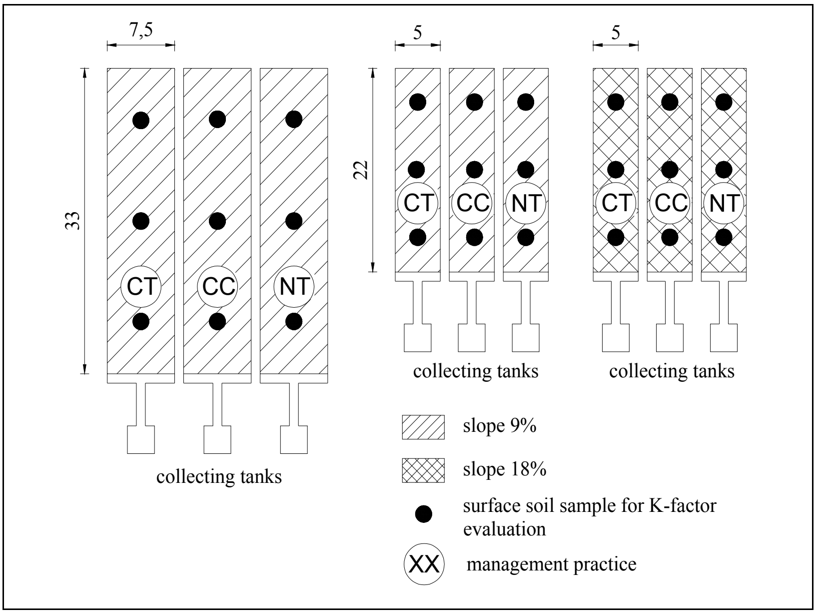



Fig. 2. Layout of the experimental plots (linear measures in metres) (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover).

### 2.3 Evaluation of the soil management practices

In the experimental plots three soil management practices commonly adopted in hilly olive groves of the Mediterranean areas were simulated and their hydrological effects were measured and compared. Conventional tillage (hereafter CT) and total weed killing through herbicide (hereafter no tillage, NT) were compared with a conservative practice (hereafter crop cover, CC) based on Low Dosage Herbicide Treatments (LDHT).

CT consisted of two to three passes, 10-15 cm deep, with a trough milling machines with subsequent soil compaction, generally starting after the first rain in October to control weeds in the whole plot. NT consisted of maintaining the soil weed-free and bare with 3 to 4 herbicide applications (acid glyphosate at a dose of 2.1 kg a.e. $ha^{-1}$ distributed manually or through a backpack sprayer) per year, mostly concentrated in late autumn (November-December). In CC practice the plots were subjected to two herbicide treatments in October and April (acid glyphosate at a dose of 0.23 kg a.e. $ha^{-1}$ distributed manually or through a backpack sprayer).

For all thesis runoff volumes and sediment concentrations were measured during a 7-year monitoring period (February 2002-December 2008); such values together with calculated soil losses were aggregated at monthly and yearly scales and then averaged among the plots subjected to the same experimental soil management practice (Figure 2).

### 2.4 Implementation of the RUSLE model

The RUSLE model was implemented at yearly scale in order to verify its prediction capability of soil erosion for the investigated soil management practices.

To calculate the R-factor, a simple equation correlating the erosivity index for the e-th event ($R_e$, MJ mm $ha^{-1}$ $h^{-1}$) and the corresponding rainfall height ($h_e$, mm) was utilized, due to the unavailability of rainfall records at sub-hourly scale in the meteorological database:

$$R_e = \alpha h_e^{\beta}, \tag{1}$$

$\alpha$ and $\beta$ are empirical coefficients for which the values of 0.18 and 1.59 respectively, calculated for the very close meteorological station of Messina (Bagarello & D'Asaro, 1994), were assumed (Table 1).

The K-factor (0.65 t $ha^{-1}$ per R-factor unit, Table 1) was averaged from K factors established for several soil samples collected within the investigated plots (Figure 2).

Topographic factor values $L_iS_i$ for the i-th plot were calculated by using the following relationship (McCool et al., 1989) (Table 1):

$$L_iS_i = \left(\frac{\lambda_i}{22.13}\right)^{m_i}\left(16.8 \ \sin\alpha_i - 0.5\right), \tag{2}$$

where $\lambda_i$ (m) is the slope length of the i-th plot, $\alpha_i$ is the slope angle (Figure 2); $m_i$ was calculated as follows:

$$m_i = \frac{f_i}{(1+f_i)} \tag{3}$$

being:

$$f_i = \frac{\sin\alpha_i}{0.0896 \ (3 \ \sin 0.8 \ \alpha_i + 0.56)}. \tag{4}$$

Because the C-factor changes continuously with cover and residue among cutting operations, the related values need to be established for different periods during the year, according the guidelines of Wischmeier & Smith (1978). Therefore, the monthly C-factors for the three soil management practices (Table 1) were calculated as a function of the plot vegetal coverage (reported for the investigated soil management practices in Table 2) through a regression equation ($r^2$ = 0.98; n = 6), correlating the C-values - in the range 0.0032-0.45, reported by Bazzoffi (2007) for vegetated or unvegetated olive orchards - to the corresponding per cent vegetal coverage.

| **RUSLE factor** | **Value or range** |
|---|---|
| *Max $R_e$ (MJ mm $ha^{-1}$ $h^{-1}$)* | 416.96 |
| *K (t $ha^{-1}$ h $t^{-1}$ $m^{-1}$ $mm^{-1}$ ha)* | 0.65 |
| *LS (-)* | 1.0 to 2.45 |
| *C (-)* | 0.01 to 0.40 |
| *P (-)* | 0.6 to 1.0 |

Table 1. Value or range of RUSLE factors for the experimental plots.

According to the guidelines of Wischmeier & Smith (1978), the P-factor was assumed equal to 0.6 (slope of 9%) or 0.8 (slope of 18%) in occurrence of tillage operations along contour lines for CT; otherwise a value of 1.0 was considered, because no erosion control practice was adopted (Table 1).

| Month | Plot vegetal coverage (%) | | |
|---|---|---|---|
| | *CT* | *NT* | *CC* |
| *January* | 25.9 | 16.7 | 58,1 |
| *February* | 36.8 | 21.8 | 71,3 |
| *March* | 40.9 | 29.5 | 82,2 |
| *April* | 52.8 | 33.0 | 22,4[1] |
| *May* | 57.1 | 34.2 | 43,4 |
| *June* | 54.5 | 32.8 | 45,6 |
| *July* | 48.3 | 25.6 | 41,1 |
| *August* | 32.1 | 19.8 | 36,9 |
| *September* | 36.5 | 21.2 | 44,3 |
| *October* | 49.6 | 26.7 | 19,4[1] |
| *November* | 2.3[1] | 9.8[1] | 33,9 |
| *December* | 10.8 | 11.2 | 47,1 |

[1] Treatment date

Table 2. Monthly values of plot vegetal coverage for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover).

### 2.5 Evaluation of the RUSLE model

Model performance was evaluated at yearly scale by qualitative and quantitative approaches. The qualitative procedure consisted of visually comparing observed and simulated values. For quantitative evaluation a range of both summary and difference measures were used.

The summary measures utilized were the mean and standard deviation of both observed and simulated values. Given that coefficient of determination ($r^2$) is an insufficient and often misleading evaluation criterion (Licciardello et al., 2007; Zema et al., 2011), the Nash & Sutcliffe (1970) coefficient of efficiency (E) was also used to assess model efficiency. As suggested by Krause et al. (2005) and Legates & McCabe (1999), E was integrated with the Root Mean Square Error (RMSE), which describes the difference between the observed values and the model predictions in the unit of the variable. The values considered to be optimal for these criteria were 1 for $r^2$ and E and 0 for RMSE. According to common practice, simulation results are considered good for values of E greater than or equal to 0.75, satisfactory for values of E between 0.75 and 0.36, and unsatisfactory for values below 0.36 (Van Liew & Garbrecht, 2003).

## 3. Results and discussion

### 3.1 Evaluation of the soil management practices

The results of the comparison among the hydrologic effects of the three investigated soil management practices (conventional tillage, no tillage and crop cover), typical of Mediterranean hilly olive groves, are reported in this section.

#### 3.1.1 Analysis at yearly scale

There was a clear difference in the runoff volumes yielded in the investigated management practices, with NT having the highest runoff coefficient and CC the lowest. The LDHT (CC practice) produced average surface runoff volumes lower by 28% than CT; conversely, complete removal of vegetal coverage through herbicide (NT) resulted in average runoff volume higher by 28% and 79% than CT or CC respectively. Consequently mean yearly values of the runoff coefficient for CC (10.7%) were appreciably lower than those recorded for CT (15.0%) and NT (19.4%) practices (Table 3).

The differences among the soil management practices in the average yearly runoff coefficients measured in this study are basically coherent with those measured in other investigations available in literature. Raglione et al. (1999) reported runoff coefficients of 3.5 and 12.8% for CC and CT respectively in Calabria (southern Italy). Bruggeman et al. (2005) measured average runoff of 184 and 66.5 mm year$^{-1}$ for orchards under CT and CC, respectively, in Syria, in an area with an average yearly precipitation of 400 mm year$^{-1}$. Francia et al. (2006) measured, in a loamy soil on a 30% slope, higher runoff coefficients in the treatment under NT (5.3%) and lower values for CT and CC (1.5 and 2.7%) respectively. Gomez & Giraldez (2007), in a sandy-loam soil on a 11% slope, measured runoff coefficients of 20 and 5.7% for CT and CC respectively. More recently, in Andalusia (Spain) Gomez et al. (2009b) in a 4-year experiment carried out in an olive tree farm on a sandy-loam soil found runoff coefficients of 6 and 16% for CC and CT practices respectively; in the same environment, Gomez et al. (2009c) recorded during a 7-year experiment in a young olive grove installed on a heavy clay soil the highest average yearly runoff coefficient (11.9%) for NT, which decreased to 1.2% for CC and to 3.1% for CT.

Sediment concentration in collected runoff samples was lower for plots subjected to LDHT (54% less than in CT plots) and higher for NT treatment (18% higher than CT) (Table 3).

The advantages induced by application of low doses of herbicide (CC) were particularly remarkably in terms of soil loss, decreased in this soil management practice by 57% and 71% with respect to CT and NT (Table 3). As well known, the soil loss depends not only on the runoff generation, but also on the sediment concentration of the water stream; both were greater under CT and NT treatments, which left for some periods along the year the soil unprotected and then exposed to the erosion risk. The records of the yearly soil losses for the three experimental soil management practices show a large inter-annual variability, with average values of 28.8 t ha$^{-1}$ year$^{-1}$ (with a standard deviation of 34.1 t ha$^{-1}$ year$^{-1}$) in the CT practice and 42.2 t ha$^{-1}$ year$^{-1}$ (± 50.0 t ha$^{-1}$ year$^{-1}$) under NT with the lowest average value recorded for CC (12.3 ± 14.7 t ha$^{-1}$ year$^{-1}$) (Table 3).

In all the observation years CC practice allowed to achieve soil losses very close to the tolerable value of 11-12 t ha$^{-1}$ year$^{-1}$ suggested by several Authors (e.g. Montgomery, 2007; Stone et al., 2000); conversely under CT and NT treatments such a threshold was always exceeded (Table 3).

A comparison between the yearly soil losses measured during the 7-year monitoring period of the present study and the values reported by other Authors in experimental runoff plots to evaluate soil erosion in olive groves has been carried out; the main results are reported in Table 4.

Kosmas et al. (1996) measured soil losses between 0 and 0.03 t ha$^{-1}$ year$^{-1}$in semi-natural olive groves in Greece with 90% of the soil covered by vegetation. Raglione et al. (1999) measured in Calabria total soil losses of 0.36 and 41 t ha$^{-1}$ year$^{-1}$ for CC and CT respectively in a 2-year plot experiment. In Syria Bruggeman et al. (2005) measured average soil losses of 11.2 and 41.4 t ha$^{-1}$ year$^{-1}$ in orchards under CC and CT respectively in an area with a slope of 24% for a 4-year period. Gomez et al. (2004) reported average soil losses of 4.0, 8.5 and

| Year | Rainfall (mm) | Cumulated surface runoff (mm) | | | Runoff coefficient (%) | | |
|---|---|---|---|---|---|---|---|
| | | CT | NT | CC | CT | NT | CC |
| *2002*[1] | 689.2 | 105.8 | 155.2 | 86.0 | 15.4 | 22.5 | 12.5 |
| *2003* | 843.3 | 136.5 | 180.2 | 103.2 | 16.2 | 21.4 | 12.2 |
| *2004* | 522.2 | 72.5 | 105.6 | 52.4 | 13.9 | 20.2 | 10.0 |
| *2005* | 690.4 | 113.4 | 120.6 | 76.4 | 16.4 | 17.5 | 11.1 |
| *2006* | 521.4 | 71.0 | 89.0 | 47.5 | 13.6 | 17.1 | 9.1 |
| *2007* | 690.4 | 113.4 | 120.6 | 76.4 | 16.4 | 17.5 | 11.1 |
| *2008* | 622.0 | 82.2 | 120.6 | 56.2 | 13.2 | 19.4 | 9.0 |
| *Cumulated* | *4578.9* | *694.7* | *891.8* | *498.1* | - | | |
| *Mean*[2] | *654.1* | *99.2*[a] | *127.4*[a] | *71.2*[b] | *15.0*[a] | *19.4*[b] | *10.7*[c] |

(a)

| Year | Rainfall (mm) | Mean sediment concentration (g $l^{-1}$) | | | Cumulated soil loss (t $ha^{-1}$) | | |
|---|---|---|---|---|---|---|---|
| | | CT | NT | CC | CT | NT | CC |
| *2002*[1] | 689.2 | 14.1 | 12.4 | 3.7 | 28.4 | 47.2 | 8.8 |
| *2003* | 843.3 | 6.0 | 5.6 | 3.6 | 40.9 | 52.9 | 17.3 |
| *2004* | 522.2 | 6.4 | 6.5 | 5.7 | 19.0 | 28.4 | 10.1 |
| *2005* | 690.4 | 6.7 | 8.1 | 4.4 | 32.2 | 44.6 | 14.8 |
| *2006* | 521.4 | 14.9 | 12.6 | 5.7 | 18.2 | 29.5 | 6.8 |
| *2007* | 690.4 | 6.3 | 8.4 | 4.7 | 32.2 | 44.6 | 14.8 |
| *2008* | 622.0 | 7.9 | 19.7 | 1.1 | 30.2 | 48.3 | 13.3 |
| *Cumulated* | *4578.9* | - | | | *201.3* | *295.4* | *86.0* |
| *Mean*[2] | *654.1* | *8.9*[a] | *10.5*[a] | *4.1*[b] | *28.8*[a] | *42.2*[b] | *12.3*[c] |

(b)

[1] February-December
[2] Values followed by the same letter are not significantly different at $P < 0.05$.

Table 3. a, b. Yearly values of the hydrological observations for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

1.2 t $ha^{-1}$ $year^{-1}$ from CT, NT and CC in a 3-year experiment on a heavy clay soil in Andalusia. In a 2-year study carried out n the same region, Francia et al. (2006) measured soil losses of 5.7, 25.6 and 2.1 t $ha^{-1}$ from CT, NT and CC respectively. Also in Andalusia, Gomez & Giraldez (2007) reported average soil losses of 21.5 and 0.4 t $ha^{-1}$ $year^{-1}$ for CT and CC in a different 4-year experiment. More recently, Gomez et al. (2009c) in a 7-year study reported soil losses of 2.9 t $ha^{-1}$ $year^{-1}$ for CT, 6.9 t $ha^{-1}$ $year^{-1}$ for NT and 0.8 t $ha^{-1}$ $year^{-1}$ for CC in a young olive grove installed on a heavy clay soil of Andalusia; in the same environment, Gomez et al. (2009b) in a 4-year experiment carried out in an olive tree farm on a sandy-loam soil recorded soil losses of 1.9 and 0.4 t $ha^{-1}$ $year^{-1}$ for CT and CC treatments respectively. Average soil losses measured in our experimental plots subjected to CT management practice (28.8 t $ha^{-1}$ $year^{-1}$) are coherent with the studies by Raglione et al. (1999), Bruggeman et al. (2005) and Gomez & Giraldez (2007), but generally higher than the observations reported in the other investigations (Francia et al., 2006; Gomez et al., 2004; Gomez et al., 2009b, 2009c). Also soil losses observed in the present study for NT and CC soil management practices (42.2 and 12.3 t ha$^{-1}$ year$^{-1}$ respectively) were generally higher than the observations found in the mentioned studies, except for data reported by Bruggeman et al. (2005) for CC, which are very close to the value achieved in the present study (Table 3). Even though the comparison of these values must be made with care due to relevant variability in the experimental climatic, morphological and management conditions among the examined studies and the limited duration of many of these databases (at most 4 years), the magnitude of the soil losses achieved in the present study highlighted the severity of the erosion phenomena in the experimental conditions and, as a consequence, the need of countermeasures to control and mitigate the erosive risks.

| **Study area** | **Authors** | **Soil losses (t ha$^{-1}$ year$^{-1}$)** | | |
|---|---|---|---|---|
| | | **CT** | **NT** | **CC** |
| Calabria, Italy | Present study | 28.8 | 42.2 | 12.3 |
| | Raglione et al. (1999) | 41.0 | - | 0.36 |
| Andalusia, Spain | Gomez et al. (2004) | 4.0 | 8.5 | 1.2 |
| | Francia et al. (2006) | 5.7 | 25.6 | 2.1 |
| | Gomez & Giraldez (2007) | 21.5 | - | 0.4 |
| | Gomez et al. (2009b) | 1.9 | - | 0.4 |
| | Gomez et al. (2009c) | 2.9 | 6.9 | 0.8 |
| Syria | Bruggeman et al. (2005) | 41.4 | - | 11.2 |
| Greece | Kosmas et al. (1996) | 0 to 0.03 | | |

Table 4. Soil losses in experimental plots to evaluate soil erosion in olive groves reported in the available literature.

### 3.1.2 Analysis at monthly scale

Figures 3 a, b and c illustrate the values (aggregated or averaged for 3-month periods) of surface runoff, sediment concentration and soil loss achieved in the experimental plots during the monitoring period. It is evident the remarkable reduction of all the hydrological variables recorded in the plots subjected to LDHT in comparison with the other soil management practices (and particularly with NT treatment). Gomez et al. (2009c) remarked a general reduction of runoff for all the hydrological variables along the monitoring period as the experiment progressed, contrary to what found in our experimental plots.

The analysis made at monthly scale highlighted that runoff was mainly concentrated from October to March, i.e. in the months characterized by the highest rainfalls and when the soil was moist after the dry season.

| Month | Rainfall (mm) | Surface runoff (mm) | | | Runoff coefficient (%) | | |
|---|---|---|---|---|---|---|---|
| | | CT | NT | CC | CT | CT | CT |
| *January*[1] | 47.3 | 8.2 | 10.8 | 6.7 | 17.3 | 22.9 | 14.3 |
| *February* | 47.5 | 7.4 | 9.8 | 6.0 | 15.7 | 20.7 | 12.7 |
| *March* | 60.7 | 8.8 | 11.7 | 6.9 | 14.5 | 19.2 | 11.4 |
| *April* | 46.7 | 6.0 | 7.7 | 5.0 | 12.9 | 16.5 | 10.8 |
| *May* | 31.5 | 3.7 | 4.5 | 2.7 | 11.8 | 14.4 | 8.7 |
| *June* | 22.6 | 1.3 | 1.7 | 0.9 | 5.8 | 7.6 | 3.9 |
| *July* | 20.7 | 1.3 | 2.6 | 1.2 | 6.4 | 12.4 | 5.7 |
| *August* | 15.3 | 0.2 | 0.6 | 0.1 | 1.1 | 4.0 | 0.7 |
| *September* | 52.2 | 5.3 | 8.4 | 3.6 | 10.1 | 16.2 | 6.9 |
| *October* | 77.7 | 13.9 | 16.6 | 9.1 | 17.9 | 21.3 | 11.7 |
| *November* | 69.0 | 12.6 | 15.3 | 9.5 | 18.3 | 22.2 | 13.7 |
| *December* | 128.1 | 26.4 | 35.2 | 18.3 | 20.6 | 27.5 | 14.2 |

(a)

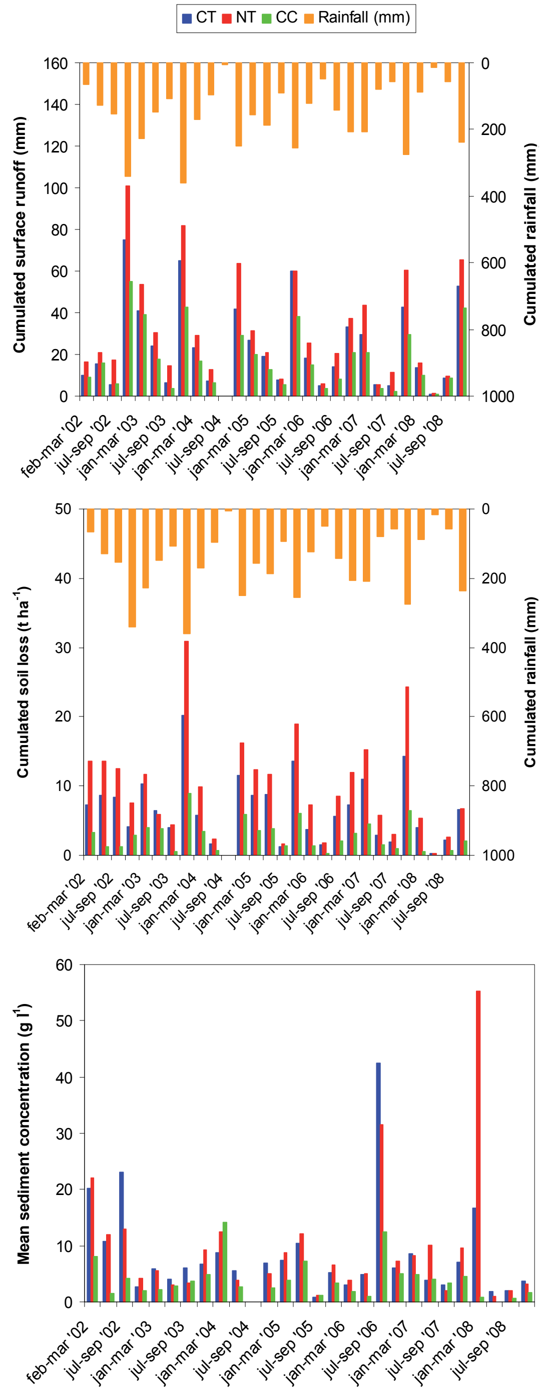
| Month | Rainfall (mm) | Sediment concentration (g $l^{-1}$) | | | Soil loss (t $ha^{-1}$) | | |
|---|---|---|---|---|---|---|---|
| | | CT | NT | CC | CT | NT | CC |
| *January*[1] | 47.3 | 9.3 | 28.5 | 2.3 | 2.1 | 3.3 | 1.0 |
| *February* | 47.5 | 11.1 | 14.5 | 8.3 | 2.7 | 4.5 | 1.2 |
| *March* | 60.7 | 8.2 | 7.9 | 2.5 | 2.7 | 3.4 | 1.0 |
| *April* | 46.7 | 7.8 | 8.8 | 3.7 | 2.1 | 2.8 | 0.9 |
| *May* | 31.5 | 7.8 | 9.2 | 2.2 | 1.6 | 2.2 | 0.4 |
| *June* | 22.6 | 2.1 | 2.3 | 1.5 | 0.7 | 0.9 | 0.3 |
| *July* | 20.7 | 9.61 | 5.0 | 1.3 | 1.5 | 1.9 | 0.2 |
| *August* | 15.3 | 16.4 | 11.5 | 4.3 | 0.2 | 0.3 | 0.1 |
| *September* | 52.2 | 7.2 | 6.2 | 4.2 | 1.7 | 2.5 | 0.7 |
| *October* | 77.7 | 5.6 | 5.9 | 2.6 | 3.3 | 4.6 | 1.5 |
| *November* | 69.0 | 5.5 | 6.7 | 3.3 | 2.4 | 3.8 | 1.1 |
| *December* | 128.1 | 5.3 | 6.6 | 3.1 | 5.5 | 8.3 | 2.5 |

(b)

[1] The mean values of January are calculated for the years 2003-2008

Table 5 a, b. Mean monthly values of the hydrological observations for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

Soil losses recorded under CC were systematically lower than under other soil management practices, particularly in the late autumn-winter-early spring (up to 60% and 72% less than CT and NT treatments respectively), when rainfall erosivity was higher; this is attributable to the reduction of both surface runoff and sediment concentration, linked to the higher vegetal coverage (in the range 33.9-82.2% of the plot area, Table 2), which helped to reduce soil erosion. In fact, the herbicide application at low doses assured the survival of some spontaneous species (represented mainly by *Crepis versicaria, Reichardia picroides, Inula viscosa, Salvia verbenacea, Oxalis pescapre, Arundo donax, Cynodon dactylon, Hedysarum coronarium, Foeniculum vulgare* and *Verbascum simatum*) and the presence of




(a)

(b)

(c)

Fig. 3. a, b, c. 3-month values of the hydrological observations for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

biomass residues (consisting of the depressed species laid on the soil) during the wettest months, shielding wide portions of soil from the erosive impact of rainfall. Conversely, total weed killing through herbicide (NT treatment), which destroyed crop residues, exposed the bare soil to the rainfall erosivity and thus to the erosion risks. In the summer months, characterized by low values of rainfall erosivity, the decay effects of weeds due to LDHT remarked since April helped to reduce competition for water between weeds and crop trees.

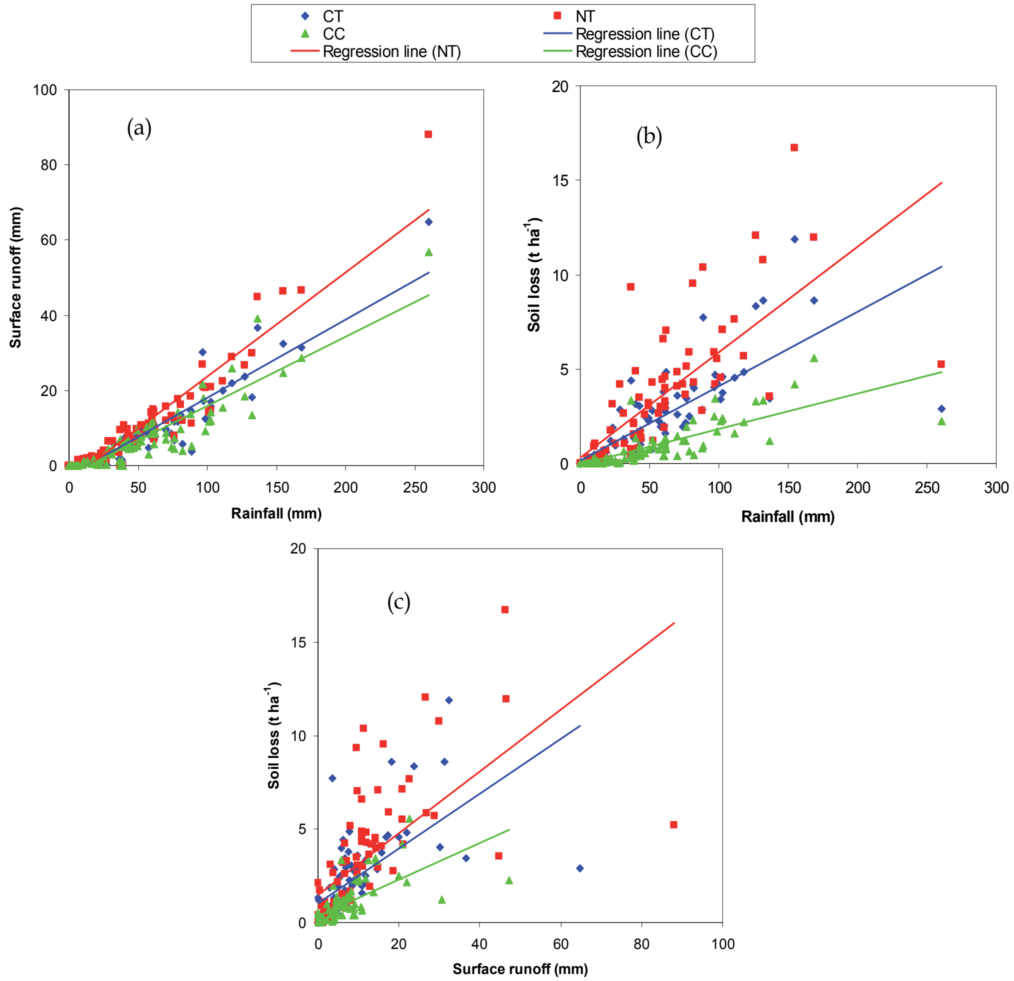



Fig. 4. a, b, c. Linear regressions among monthly hydrological observations for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

The highest reduction of soil erosion was recorded for CC in December (which is characterized by the highest mean rainfall amount), when a soil loss lower by over 55% than in the other soil management practices was achieved (Table 5).

As expected, monthly runoff volumes were well correlated with the corresponding rainfalls ($r^2$ always higher than 0.83, with the maximum value of 0.89 achieved for NT treatment).

Lower values ($r^2$ = 0.58-0.62) were found in the regression relationships between monthly rainfall and soil loss. Finally the latter was weakly correlated with the corresponding runoff ($r^2$ = 0.41-0.47, Table 6), highlighting that sediment losses generally did not follow the same patterns as runoff volumes (Figure 4).

On the whole, LDHT led to average soil losses lower by about 60-70% than in the other soil management practices investigated in the present study. Reduced soil losses depended not only on the lower runoff volumes (presumably due to the increased interception induced by the wider vegetal cover, to the higher soil infiltration capacity and to the greater flow resistance linked to the presence of vegetation stems, which helps to dissipate water stream energy), but also on the lower sediment concentration (Tables 2 and 5). These positive effects seem to influence erosion rates more efficiently than CT treatment, which in its turn increases the water retention within surface hollows left by tillage (due to the increased soil roughness) or infiltration capacity induced by the higher soil surface porosity in comparison with NT treatment.

| **Soil management practice** | **Runoff-rainfall** | **Soil loss-rainfall** | **Soil loss-runoff** |
|---|---|---|---|
| *CT* | 0.87 | 0.60 | 0.41 |
| *NT* | 0.89 | 0.58 | 0.43 |
| *CC* | 0.83 | 0.62 | 0.47 |

Table 6. Coefficients of linear regression among monthly hydrological observations for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

The results of the present study are consistent with the other similar experiences aiming at evaluating the effects of some management practices on soil erosion: such studies in general suggest to adopt CC practice in olive groves, which, establishing a proper vegetal coverage of the soil in olive grove lanes, thus reduces runoff volumes and, as a consequence, soil losses more efficiently than the most common CT treatment. No tillage should be avoided, due to the fact that keeping the soil bare by herbicide application just in the months characterized by the highest rainfall erosivity (i.e. in late autumn, early spring or during the winter) reduces soil infiltration capacity and roughness, increasing water runoff and stream velocity as well as yielding the maximum erosion rates.

### 3.2 Evaluation of the RUSLE model

The comparison among the soil losses measured in the experimental plots and the corresponding values predicted by the RUSLE model highlighted an unsatisfactory prediction capability at yearly scale. It is shown by the low coefficients of determination and efficiency as well as the high RMSE; also the differences between the measured and predicted standard deviations were high (Tables 7 and 8; Figure 5). The RUSLE model tended to overestimate soil losses for CT and, particularly, CC; on the contrary, soil losses measured for NT soil management practice were slightly underestimated (Figure 5).

For two (CT and NT) of the three simulated soil management practices the mean values of the predicted soil losses were close to the corresponding measured values with differences lower than 7%; also the differences between the measured and predicted cumulated soil losses, calculated for the entire 7-year monitoring period (201.3 versus 211.9 t $ha^{-1}$ $year^{-1}$ for CT treatment and 295.4 versus 273.8 t $ha^{-1}$ $year^{-1}$ for NT), were low. For CC soil management practice mean and total soil losses measured in the experimental plots and predicted by the RUSLE model differed instead by about 75-80% (Table 7). It means that, at least for the experimental conditions, estimations of soil losses performed by the RUSLE model must be considered with care, due to the fact that RUSLE is mainly meant to be used for long-term estimates of soil loss (Shrestha et al., 2006; Yoder et al., 2001).

| **Year** | **Soil loss (t $ha^{-1}$ $year^{-1}$)** | | | | | |
|---|---|---|---|---|---|---|
| | **Measured** | | | **Predicted** | | |
| | **CT** | **NT** | **CC** | **CT** | **NT** | **CC** |
| *2002* | 28.4 | 47.2 | 8.8 | 42.4 | 66.2 | 21.4 |
| *2003* | 40.9 | 52.9 | 17.3 | 56.1 | 61.0 | 37.3 |
| *2004* | 19.0 | 28.4 | 10.1 | 30.1 | 37.6 | 11.0 |
| *2005* | 32.2 | 44.6 | 14.8 | 24.7 | 29.9 | 25.2 |
| *2006* | 18.2 | 29.5 | 6.8 | 18.8 | 25.6 | 15.3 |
| *2007* | 32.2 | 44.6 | 14.8 | 18.2 | 27.5 | 20.8 |
| *2008* | 30.2 | 48.3 | 13.3 | 21.6 | 26.0 | 19.1 |
| *Cumulated* | *201.3* | *295.4* | *85.9* | *211.9* | *273.8* | *150.1* |

Table 7. Yearly and cumulated values of soil losses measured in the experimental plots and predicted by the RUSLE model for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover).

| Soil management practice | **Soil loss** | **Mean (t $ha^{-1}$ $year^{-1}$)** | **Std. Dev. (t $ha^{-1}$ $year^{-1}$)** | **$r^2$** | **E** | **RMSE (t $ha^{-1}$ $year^{-1}$)** |
|---|---|---|---|---|---|---|
| *CT* | *Measured* | 28.8 | 8.0 | 0.26 | -1.11 | 10.70 |
| | *Predicted* | 30.6 | 13.1 | | | |
| *NT* | *Measured* | 42.2 | 9.5 | 0.14 | -1.27 | 13.25 |
| | *Predicted* | 39.9 | 14.5 | | | |
| *CC* | *Measured* | 12.3 | 3.8 | 0.57 | -10.04 | 11.65 |
| | *Predicted* | 22.0 | 9.3 | | | |

Table 8. Statistics, efficiency and difference indexes of the RUSLE model at yearly scale for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) in the experimental plots.

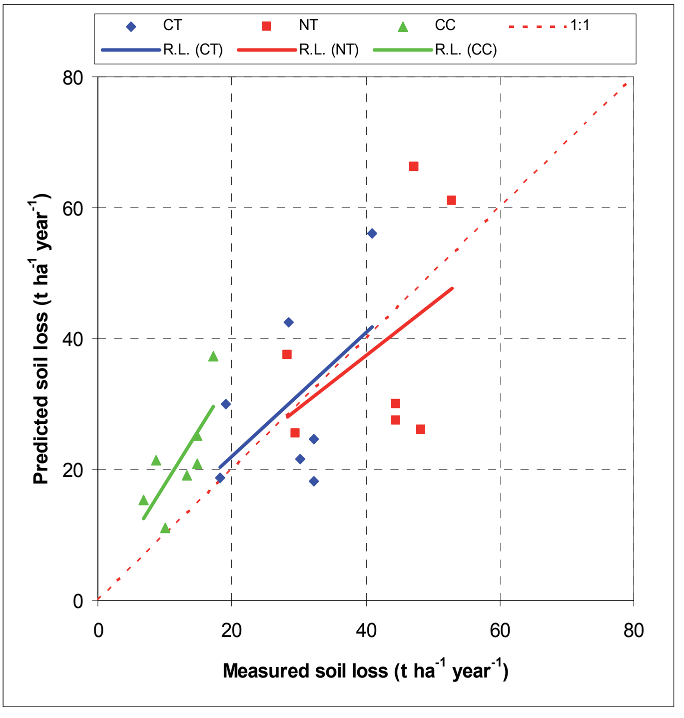



Fig. 5. Comparison between soil loss measured in the experimental plots and predicted at yearly scale by the RUSLE model for the investigated soil management practices (CT = Conventional Tillage; NT = No Tillage; CC = Crop Cover) (R.L. indicates the regression lines).

It can not be excluded that the unsatisfactory prediction capability of soil loss shown at yearly scale by the RUSLE model for the experimental plots can be attributable to:

- the unavailability of rainfall records at sub-hourly scale in the meteorological database, which, as mentioned above, forced the modeler to turn to an empiric regression equation for calculating RUSLE R-factors;
- the uncertainty of the calculated values of the C-factor (which is perhaps the most important USLE factor, because it represents conditions that can be managed most easily to reduce erosion, Ferreira et al., 1995; Renard et al., 1991; Renard & Ferreira, 1993; Yoder et al., 2001); it comes from the fact that the available soil management database lacked some important parameters (e.g. surface roughness and soil moisture) which can strongly influence soil loss estimation performed through the RUSLE model.

## 4. Conclusions

The present investigation has evaluated and simulated at plot scale the hydrological effects of three different soil management practices (conventional tillage, no tillage and crop cover through LDHT), commonly adopted in hilly olive groves of the Mediterranean environment. Although the monitoring of the erosion risk carried out in this paper is based on only 7 years of data and therefore results may change over a longer time period, the findings of this investigation highlight that, under the experimental conditions, the soil losses recorded for CT and NT practices are of high magnitude and thus unsustainable to avoid land degradation. Conversely, although the erosion rates achieved for CC practice in this study are generally higher than the observations reported by other Authors, LDHT, allowing to keep soil losses close to the tolerable value of 11-12 t $ha^{-1}$ $year^{-1}$ suggested by some Authors (e.g. Montgomery, 2007; Stone et al., 2000), results in a more efficient conservation practice in comparison to CT and NT and represents a valid alternative to these soil conservation practices. As a matter of fact, LDHT, assuring a suitable soil coverage during wet periods and a greater water availability to the olive trees in the dry seasons (thus reducing water competition with weeds), allows to mitigate the erosion risks and avoids negative impacts on crop productivity.

Unfortunately, also farmers operating in southern Italy, as remarked by other Authors (Gomez et al., 2009b; Helling & Haigh, 2002) in their respective countries, are in general reluctant to adopt soil management practices assuring a suitable crop cover and then high hydrological benefits during the wettest months (as LDHT), especially if they do not represent an immediate increase in the crop yield. Gomez (2005, 2009b) argued that the reasons for this reluctance is the need for a careful management of the cover crop to avoid competition for water with the olive tree (which is however basically limited) as well as the lower cost, for many farmers, of tillage (especially surface tillage) in comparison to cover crop soil management. This suggests the need of information activities by experts of soil conservation and farm advisers, purposing at illustrating the environmental benefits of cover crop soil management in olive groves, in particular: (i) immediately after olive planting; (ii) in young olive groves; or even (iii) in mature plantations with a very low tree density (especially in steep lands), where the canopy cover is low and the interception is rather limited. Thus, this kind of investigations may help to improve the countermeasures against soil erosion in Mediterranean slope zones, encouraging farmers to adopt soil conservation practices also through proper criteria of public financial support.

On a modeling approach, the present study has highlighted that the utilization of the RUSLE erosion model under the experimental conditions must be done with care, given that soil loss estimations have been reliable only for CT and NT treatments at a multi-year scale; presumably, a more complete hydrologic and geomorphologic database could improve model predictions.

Even though the outcomes of this study might contribute to soil conservation through sustainable management systems in agricultural lands characterized by high erosion risks, further research activities are finally needed not only to validate these results under different geomorphologic conditions, but also to assure a better understanding of runoff and erosion processes and to predict its effects with time.

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8

# Planting System on Permanent Beds; A Conservation Agriculture Alternative for Crop Production in the Mexican Plateau

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## 1. Introduction

Mexico is the world´s twelfth largest country. Almost one-quarter of the population depend on the farming sector for their livelihood. More than half the territory of Mexico is arid or semi-arid, and rainfall is the main factor limiting agricultural production (< 500 mm). Rainfed crops occupy 58 percent of the total sown area that is characterized by a large number of small-scale farmers. This shortage of farmland has resulted, mostly in the central part of the country (the high plateau or Mexican plateau), in an increment of sloping lands cultivated leading to severe soil degradation and erosion. Then, appropriate tillage/planting techniques for crop production have to be studied and promoted to be adopted to mitigate soil erosion and climatic constrains. This work should include the study of maize and small grain cereals like wheat, oat and barley as these are the most common crops in the Mexican plateau.

Indigenous farmers, on the other hand, have used raised-bed cultivation system for centuries for some row crops like corn, beans, and squash. In some states of central Mexico, raised field agriculture is a traditional system that dates back to as early as 300 B.C. (Crews & Gliessman, 1991). Currently, this planting system has been modified and widely adopted in the irrigated areas of northwest Mexico for other crops like wheat (Sayre et al., 2005). Similarly, due to the large potential to be fully adapted in dryland areas of the Mexican plateau, some farmers have started adopting the system. Regrettably, in both cases, considerable use of tillage operations is been applied. Even thought this system offers opportunities to grow crops more efficiently, the use of heavy tillage generally is promoting soil erosion, exacerbating effects of climate change, and increasing production costs. Indeed, Edward Faulkner (1943) with the publication of his classic book, "Plowman's Folly," challenged the conventional wisdom of the day by stating in the very first sentence of the very first page "Briefly, this book sets out to show that the moldboard plow which is in use on farms throughout the civilized world, is the least satisfactory implement for the preparation of crops." He went on to say, "The truth is that no one has ever advanced a scientific reason for plowing." These were revolutionary ideas at the time and met with ridicule and scorn (Triplett & Dick, 2008).

In addition to heavy tillage for seed bed preparation, another constraint for crop production is the climate change which nowadays is clearly evident. This change is most commonly expressed with higher temperatures and prolonged drought periods. Among the multiple consequences of those changes are the reduction of soil fertility and organic carbon (St. Clair & Lynch, 2010). Fortunately, the reversal can occur as increasing soil organic matter (OM) decreases atmosphere C pools. In this regard, some governments have been initiated efforts to develop markets for crop producers using C-sequestering cultural practices to sell C credits (Wilhelm et al., 2004). Thus, a technological proposal to ameliorate to some degree those ecological and economical constrains, is the application of conservation agriculture. This modern technique has evolved in different ways. One of them is known as the planting system in permanent beds.

This technology applied in the irrigated areas of northwest Mexico, with serious shortage of water in the reservoirs and even though most farmers still use conventional tillage, those that now grow wheat using the planting system on beds obtain 8% higher yield, use approximately 25% less irrigation water, and encounter at least 25% less operational costs compared to those still planting conventional tilled wheat on the flat using flood irrigation (Aquino, 1998).

On the other hand, most of the research work to develop the permanent beds technology has been carried out on wheat and maize crop but the planting system can be applied to many others such as oat, barley, beans, etc. The basic management for this technology consists mainly on leaving crop residues on soil surface as mulch, crop rotation, and bed reformation when needed (Verhulst et al., 2011). Research results have shown that if those practices are applied accordingly, grain yields can be even greater than those from a conventional planting system (Sayre, 2004; Govaerts et al., 2005), in addition to the improvement and conservation of soils (Govaerts et al., 2007). For example, diverse benefits on soil physical, chemical, and biological attributes have been identified from crop rotation and residue management (Torbert et al., 2007).

However, it is has been throughout reported that during the first years at the establishment of the permanent bed planting system, crop yields can be reduced as the net N immobilization is increased (Yadvinder-Singh et al., 2004) by microorganisms to undergo residue decomposition. This phenomenon reduces the N availability to plants to such extent that additional N fertilizer should be applied for several years until disequilibrium comes to an end (Gentile et al., 2010; Govaerts et al., 2006a) and crop yields become stable (Sayre & Hoobs, 2004). After this period, the bed planting increases the N use efficiency compared with conventional planting (Fahong et al., 2004) overall if the appropriate management practices are applied (Limon-Ortega et al., 2000). Nevertheless, there is some inconsistency between studies as other reports indicates that soil water or soil N is conserved when tillage is reduced in some environments, but not in others. The apparent inconsistency between studies suggests that complex interactions between climatic and edaphic factors affect the impact of tillage on soil water and soil N content. Therefore, more research is needed to elucidate why soil water and soil N status are not affected consistently across environments (Carr et al., 2003a; Schillinger, 2005).

This chapter aims on summarizing the dynamics of wheat and maize grain yields over years of research on permanent beds under the dry land conditions of the Mexican plateau. This includes the discussion on the effect of various management factors on the performance of those crops, as well as the advantages of this planting system and their effect on soil attributes.

## 2. Description of the planting system on permanent beds

Conservation tillage is broadly defined as any tillage system that leaves enough crop residues to adequately protect soil from erosion throughout the year (Reeder, 1992; Scillinger, 2005). Because erosion control is improved with increasing soil cover, systems with 30% or greater soil cover at planting time have been defined as conservation-tillage system (USDA-SCS, 1984). More recently the name of 'conservation agriculture' has been adopted to highlight this sustainable system from the narrowly defined 'conservation tillage' (Govaerts et al., 2009). Zero-tillage (no-till) is one of the primary categories of conservation agriculture, in which the soil is left undisturbed from the harvest of one crop to the seeding of the next one with only slight soil disturbance associated with creating a narrow slot to place the seed/fertilizer (Dickey, 1992).

Similar to zero-tillage, the permanent bed planting system is initiated using conventional tillage to allow the raising of well-formed beds and planting the initial crop. Subsequently, no additional tillage is used except to reshape the beds as required, by passing a winged-shovel in the furrow to maintain the shape of the bed (Morrison et al., 1990). This field operation is usually made before planting each crop but frequency may change depending on crop and/or soil type (Morrison & Gerik, 1983). No tillage is made on the top of the bed except that associated with planting. Since crop residues should be retained, a cutting-disc may need to be placed ahead of the reshaping shovel to allow its free passage through the furrow without clogging with residue during the simple bed-reshaping process. This depends upon the amount of crop residues left that generally vary according to the crop, yield and amount removed for fodder or other purposes. If the amount of crop residues left on the soil surface is the minimum required for a zero-tillage system to adequately protect the soil from erosion, it is likely that the cutting disk may not be needed. Actually this is the case in low- and moderate - rainfall environments of the Mexican plateau where the average grain yield of maize is less than 4 t/ha. The equipment used to reshape beds, with or without cutting-disk, can also be used for mechanical weed control in the furrow areas even in crops like wheat, oat and barley (Sayre, 1998).

The raised bed-planting technology for wheat-based cropping systems was developed in the irrigated areas of northwest Mexico by which a defined number of rows of wheat, or other crops, are planted on the top of beds with furrow irrigation between beds (Wang et al., 2009). Due to the benefits reported by farmers, the bed-planting technology needs to be applied in dryland production areas. The permanent bed planting system applied to irrigated or dryland conditions has various topological variants; bed width and number of seed-rows per bed are probably the most important. (Fig. 1a, b, c, and d). Bed width is mostly determined by user´s preferences to fit individual farm needs and grain yield potential. In general, for the case of small grain cereals, bed-width measured from furrow bottom –to- furrow bottom ranges from 0.75 to 1.6 m and the number of seed-rows per bed from 2 to 6, respectively. As the environment becomes drier and grain yield lowers, beds should be wider (Fig. 1d) and as the environment improves and grain yield increases, beds can be narrowed (Fig. 1b and c) (Sweeney & Sisson, 1988; Iragavarapu & Randall, 1997). In any case, bed width should match machinery to confine wheel traffic to the furrow bottom to maintain the cropping area (bed-top) free from soil compaction (Morrison & Gerik, 1988).

Most farmers that have adopted the bed planting system for dryland wheat, oat and barley production under conventional tillage in the Mexican plateau, plant two or three defined seed rows, regardless of the environment potential, on the top of narrow raised beds (about

80 cm wide) with modified conventional-drills. Analogously, the no-till equipment for small grain cereals to plant multiple rows per bed should be modified as it is not sold commercially. This implies that farmers have to modify no-till equipment so that it can plant through crop residues on beds. This modification requires not only money, but also time and creativity. In fact the machinery problem appears to be greater than any cultural or soil related problem (Morrison, 1985), and this will not be quite solved until machinery designers and agronomist interact to develop models with specific standards for the bed planting system. Those standards should be specific to each region and most common crops. For example, the type of planters should vary according to the amount of crop residues left which in turn depends upon farmer, crop type, and potential environment.


Fig. 1. Planting systems for dryland wheat production in a) conventional planting system on the flat; narrow-raised beds with b) two rows per bed and c) three rows per bed; and wide-raised beds with d) six rows per bed. Farmer´s fields in the Mexican plateau.

## 3. Benefits on soil attributes

The application of the permanent beds with residue retention as a form of conservation agriculture has several aims. One is to improve soil quality which is a concept based on the premise that management can deteriorate, stabilize, or improve soil ecosystem functions (Franzluebbers, 2002). The annual practice of crop rotation and rational crop residue management as minimum set is crucial to obtain the benefits on chemical, physical and biological soil attributes which are a function of OM (Chan et al., 2002). However, if those benefits are ultimately to be extended to improve grain yield, those practices should be accompanied with other factors as described in section 5 of this chapter.

### 3.1 Physical attributes

The primary soil physical characteristic influenced by OM is soil structure through aggregation and aggregate stability (Six et al., 1999). Organic matter improvement is in turn the result of crop residues left or incorporated into the soil. Soil aggregation is the process whereby primary soil particles are bound together into secondary units, usually by natural substances derived from root exudates and microbial activity (Soil Science Society of America, 1997). Reduction of soil crusting on the top of the beds as result of the improvement of soil aggregation (Egball et al., 1996; Fahong et al., 2004) allows a better crop establishment due to a rapid emergence (Guerif et al., 2001). Furthermore, top soil aggregate stability is considered as erodibility factor with strong influence on water run-off and erosion (Barthes et al., 2000). Nevertheless, there is no way universally accepted to measure this parameter (Diaz-Zorita et al., 2002). An approach that has provided an adequate description of aggregate stability is the fractal approach (Guerif et a., 2001). Examples applied to permanent beds can be found in Limon-Ortega et al. (2002) and Limon-Ortega et al. (2006).

For purposes of this chapter a rustic experiment to demonstrate the aggregate stability was carried out using two large clods from the same type of soil; one from a plot under conventional tillage and other from a plot under permanent beds for ten years (Fig. 2a and b, respectively). The experiment consisted on immersing both clods in tab water for about 30 sec. The effect of this immersion on the clod from conventional tillage was to disrupt the initial porosity impeding water infiltration after a rain. The opposite occurred on the clod from permanent beds; porosity was maintained allowing water to infiltrate. In the former case water stagnancy on the surface promotes soil erosion through run-off while in the latter soil erosion is greatly reduced.

Actually, soil erosion due to water run-off is the largest soil degradation process and it is associated to management practices which tend to leave the soil without protection when the rainy season starts. In Mexico, about of 85% of the territory is affected by this sort of degradation (Etchevers et al., 2006) and is frequently caused by the inopportune practice of extensive cattle grazing (Haulon et al., 2007). Regrettably, appropriate management of crop residues from the previous crop is a key to soil structural development and stability (Govaerts et al., 2009).

Relative to grain yield, it has been reported that differences in maize yield between tillage systems are attributed to a better water supply in no-till due to the maintenance of a larger mesopores and a greater hydraulic conductivity (Diaz-Zorita et al., 2004). According to Fig. 2, one might surmise that water supply is greater, and then grain yield, in permanent bed plots (2b) than in conventional tillage plots (2a).

Soil bulk density, another physical attribute, decreases as the soil organic C increases contributing to improve soil quality enhancing the performance of disk-type drills in seed placement in addition to the efficient C sequestration (Halvorson et al., 1999b). However, if a no-till system is practiced on the flat without beds there is a potential increase of bulk density, and thus soil compaction, as machinery circulates randomly (Jones et al., 1989). In the case of permanent beds, this constrain is confined to the furrow bottoms; bed tops remain intact without compaction.

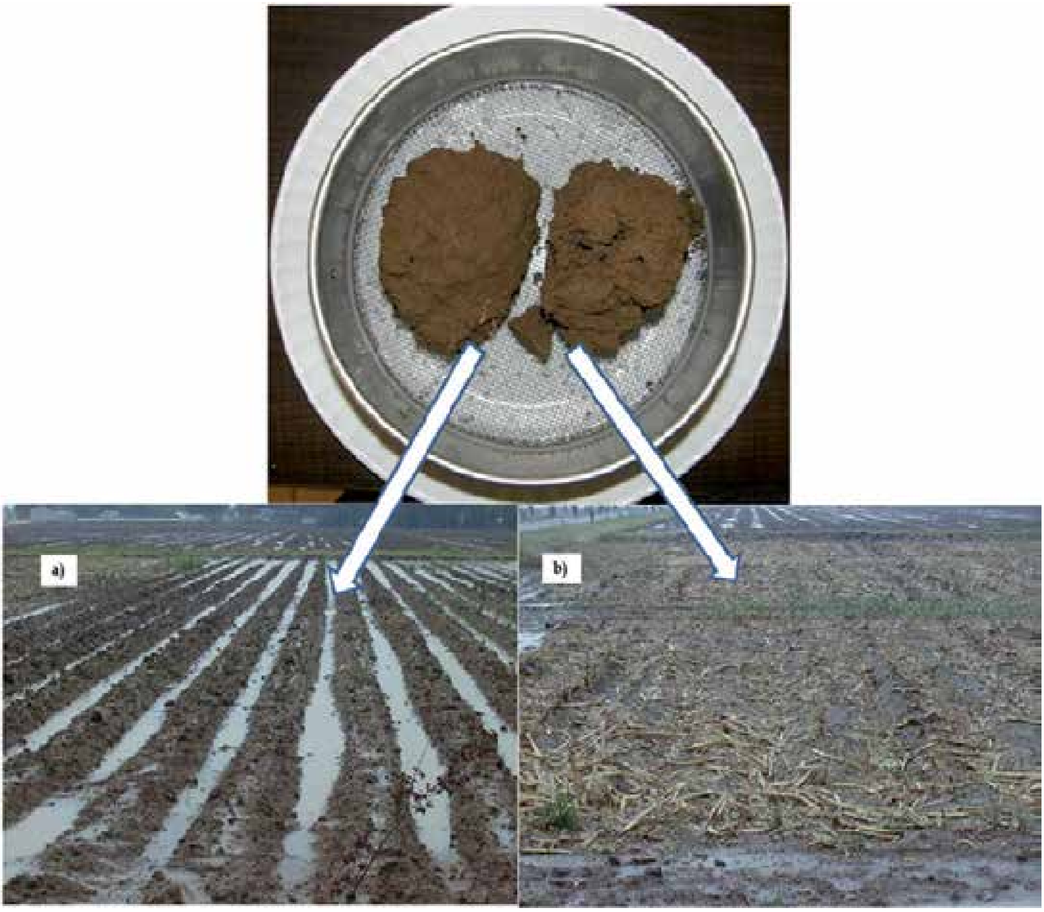


Fig. 2. Stability of two soil aggregates and its effect on water infiltration after a rain storm. Both aggregates from the same type of soil but different till system; a) conventional tillage and b) permanent beds. Arrows indicate the corresponding field plots. Chapingo, Mexico.

### 3.2 Chemical attributes

The work of Govaerts et al. (2007) in the Mexican plateau reported that sodicity as chemical parameter was highest for conventional-till raised-beds. However, sodium content in permanent beds varies according to the amount of residues left; as the amount is increased, sodicity decreases. The opposite was reported for pH; alkalinity slightly increases.

The decrease in soil salt concentration, on the other hand, resulting from the application of permanent beds may have a tremendous relevance for saline areas (Sayre et al., 2005). However, variations may occur depending upon crop residue management. For example permanent beds with residues burned increases electrical conductivity while the opposite occurs with residue retention (Verhulst et al., 2011). Fig. 3 from a field experiment at Chapingo, Mexico shows how salt concentration is reduced over time with the application of permanent beds with slight variations due to crop rotation.

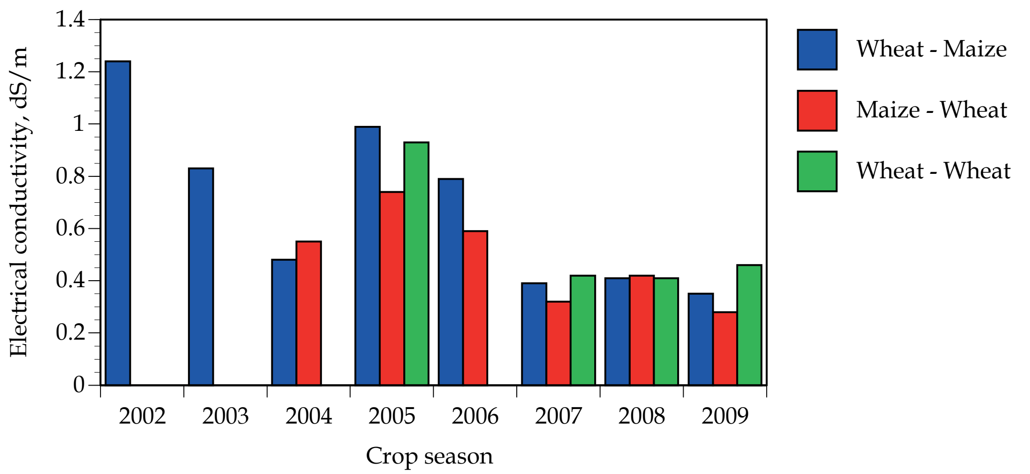


Fig. 3. Soil-salt concentration reduction as result of the use of permanent beds technology under three crop rotations; wheat – maize, maize – wheat, and wheat - wheat. Soil cores were taken from 0 to 30 cm depth. Chapingo, Mexico.

Even thought the initial soil salt concentration in this study was not high enough to be considered as saline soil, it is interesting to see how the electrical conductivity of the soil solution decreases over time from an initial of about 1.2 to 0.5 dS/m after a period of eight cropping cycles. According to this result, the electrical conductivity reduction for the wheat – wheat crop rotation is not as high as showed by the rotation with maize likely due to the higher quantity of crop residues left with this crop. This differential result is attributed to a poor aggregate stability (Verhulst et al., 2011) resulting from an inappropriate crop rotation (Limon-Ortega et al., 2009a).

### 3.3 Biological attributes

Biological erosion in México is the second largest soil degradation process after water erosion and it represents the rate of organic mater mineralization. Approximately 80% of the territory is affected by some degree of biological degradation (Etchevers et al., 2006). Optionally, the continuous return of crop residues increases the soil OM content. It is estimated that about 11% and 37% of C in corn residues and roots, respectively, ends up as OM indicating that the latter is the major contributor to maintenance of OM in the soil (Barber, 1979). This parameter plays a key role in the soil quality and it is interrelated to physical and chemical properties, far out of proportion to the small quantities present (Kubat et al., 2008). However, the increase in OM is not easily detected by chemical analyses in the short-term. Instead, measurement of soil microbial biomass (SMB) can be used as early indicator of the OM trend (Powlson & Brookes, 1987).

The amount of SMB is an important component of soil quality assessment because of its important roles in nutrient dynamics; decomposition of natural and synthetic organic amendments; and physical stabilization of aggregates. Crop residues, including roots, are a source of nutrients for SMB and plants. Soil microbial biomass is the living component of soil OM and although it comprises less than 5% of OM, it performs various critical functions for plant production in the ecosystem; is a labile source of C, N, P, and S; and is an agent of nutrient transformation (Dalal., 1998). Returning crop residues has a tendency to increase soil organic C and N (Malhi & Kutcher, 2007) and an example applied to permanent beds can be reviewed in Limon-Ortega et al. (2006).

## 4. Grain yield variations at the establishment of the permanent bed planting system

The adoption of the permanent bed planting system by farmers faces multiple restrictions, two of them are related to 1) issues of the N cycle and to 2) the visual perspective of wheat fields, i.e. the topological arrangement of plants on beds makes growers hesitate about the appropriateness of the system.

### 4.1 Variations due to N

Leaving crop residues on the soil surface, among other practices, is critical for the practice of zero tillage to get the benefits on grain yield. Likewise, it can take some time –roughly five years- before the benefits are evident (Govaerts et al., 2005) albeit other authors state that three years are enough (Triplett & Dick, 2008).Other reports indicate that'many' years are required to reach an equilibrium (Motta et al., 2000). This inconsistency leads to think that different effects actually appear due to local conditions, assigning priorities to specific factors and/or processes (Guerif et al., 2001), and offers a great challenge to discern and account for the impact of crop residue on nutrient availability (Schoenau & Campbell, 1996) for plant uptake.

As first instance, the soil N availability declines (Carr et al., 2003a) but the question still remains about how many years of good management it will take before the potential for greater N mineralization will be reflected in situ (Grant et al., 2002). The temporarily N declination is mainly due to organic residues added to the soil surface that should undergo decomposition through the SMB present in soil and/or residues (Cabrera et al., 2005). However, if the amount of N present in the residues is smaller than that required by SMB activity, additional inorganic N will be immobilized from the soil to complete the decomposition process (Corbeels et al., 1999). To offset this temporal deficiency, additional inputs of mineral fertilizer should be applied (Triplet & Dick, 2008) to improve the synchrony between nutrient availability and crop demands (Gentile et al., 2010). Nevertheless, in the long-term the cumulative effects of straw incorporation will play an active role in providing greater amounts of plant-available N from mineralization (Bird et al., 2001) and thus reducing the need of fertilizer inputs. This effect on N will depend upon crop residue quality (Gentile et al., 2010) evaluated as C and N as main parameters (Salinas-Garcia et al., 1997).

Yet, during some seasons and climates, the effect of N immobilization/mineralization at the establishment of a no-till system will inhibit biological activity which may be associated with either production or reduction of plant available nutrients (Schoenau & Campbell, 1996) which will be eventually reflected in grain yields. Fig. 4a and b is an example of what happened to maize and wheat yields, respectively, in a water-limited environment of the Mexican plateau.

Maize yield differences due to N application occurred during the first three years. Lowest yields were obtained with 0 kg N/ha and can be attributed to an initial N immobilization. However, after three years grain yields among N rates were similar (Fig. 4a). This indicates that effect of N immobilization on grain yield occurs only in the absence of N fertilizer suggesting that no additional rates are required to offset immobilization as other factors should be determining final yields.

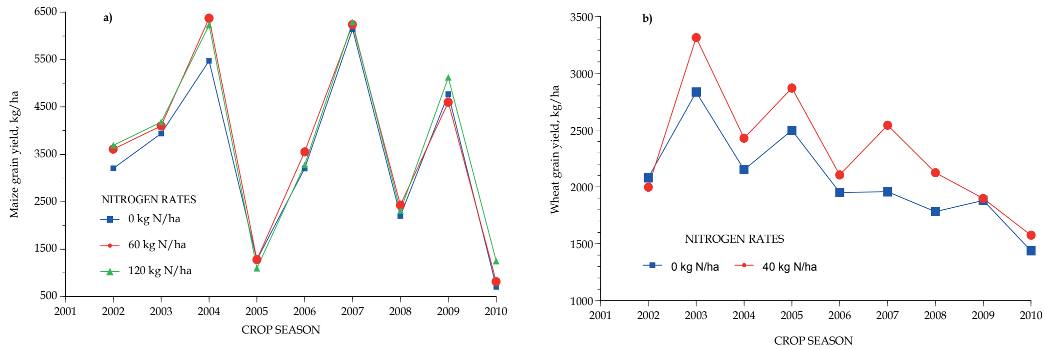


Fig. 4a and b. Maize and wheat yield response to N application and crop season from a trial on permanent beds with residue retention from a trial initiated in 2002 at Chapingo, Mexico.

In contrast, wheat grain yield differences among N rates (Fig. 4b) does not show a clear trend. In average, grain yield differences are clearly lesser than 500 kg/ha, except for the 2007 crop season when the difference was about 600 kg/ha. But from a practical point of view, these differences are minor if the economy of fertilizer acquisition and application is considered. Statistical analysis for this study suggests that wheat grain yields are mostly driven by rainfall amount and distribution albeit there was a slight reduction of soil $NO_3$ during the first three years.

It is clear that grain yield variations in Fig 4 can not be generalized to other locations as other limiting-factors may be determinants. Indeed, an additional fact in this study is that the application of P and K has been excluded in both crops. Statistical analysis (unpublished data) does not reveal these nutrients as limiting factors. Thus, one may surmise that P is bio-recycled from residues as water-soluble forms and moved with rain to the mineral soil below (Schoenau & Campbell, 1996; Motta el al., 2000). This is of major importance considering the world shortage of mineral P (Cordell, et al 2009).

### 4.2 Variations due to topological arrangement of small grain cereals

An additional farmers' constrains to adopt the bed planting system even using conventional tillage is the open space in furrows between beds without plants. Generally, they argue that this space is wasted and then grain yields reduced as the seeding rate is too light, or the space between beds too wide for maximum utilization of available resources. However, results from field studies have demonstrated the opposite; grain yields from planting on beds can be even higher than those from conventional planting on the flat. For example, Sweeney & Sisson (1988) reported 15 % wheat grain yield increases in a bed planting system compared to yield obtained in the 'flat' indicating that other factors such as rainfall patterns may be accounting for the differences between systems. The higher yield from beds can be attributed to greater resource utilization through the changes in row spacing and plant density configuration (Chen et al., 2008).

It is well known that wheat plants in narrow-raised beds can compensate, within certain limits, for the lost cropping areas of the open furrows between beds. The exclusion of wheat plants from the furrow bottoms between beds may stimulate tillering, or number of heads, and grain production in location/years with adequate precipitation (Gerik & Morrison, 1985). This increase in tillering can be attributed to more favorable soil moisture and nutritional conditions as well as greater light intensity (Siemens, 1963). Alternatively, if precipitation is relatively scarce and distribution inadequate, the number of rows per bed should be increased through a wider bed size to compensate for the low number of heads (Sweeney & Sisson, 1988; Iragavarapu & Randall, 1997). And under even harsher conditions, it is likely that a seed row in the furrow bottom may be needed to increase the number of heads but this option in this sort of environment has not been documented al least for the Mexican plateau.

## 5. Key components for a successful establishment of permanent beds

Most farmers in the Mexican plateau are small, near subsistence producers who practice extensive tillage and mono-cropping combined with direct removal of most crop residues mainly by baling for livestock fodder or by pasturing and/or burning. These traditional practices, which result in bare, exposed fields for most of the year, leads considerable run-off losses of rain water associated with the occasional heavy-afternoon thunder-showers, especially on the extensive sloping fields (Sayre et al., 2001).
Given this scenario, field research has been conducted to provide a sound basis technology to farmers of the Mexican plateau and other regions based on reduced/zero-till, crop rotation, and crop residue retention. One of those technologies is the planting system on permanent beds whose research results are based on long term trials carried out by diverse institutions. Some of the components identified for a successful establishment of this system are discussed below.

### 5.1 Seeding equipment

Conservation agriculture in its form of permanent beds has many advantages over conventional planting systems; however, crop residues on the soil surface complicate the planting activity (Torbert et al., 2007). Among the commercial equipment developed for direct seeding, there is no one designed specifically for the planting system on permanent beds. Under these circumstances, some agricultural research institutions in the world have developed their own prototypes with varying features among them (Hobbs et al., 2008). This surely has been made based upon how machinery designers conceptualize the planting system and on the particular situation of the farming areas including type of crops, amount of residues, and fertilizer requirements. Consequently, there should be a great variety of local designs ranging from equipments with a forward residue mover to planters with cutting disks. But a common factor is that those designs, including small-scale planters, should be able to penetrate untilled soils to both place and cover the seed, usually through variable amounts of crop residues on the surface (Sayre, 1998).
In general, appropriate field machineries are necessary for successful operation of this crop production system. Fertilizing and seeding machines must effectively interact with varying physical conditions of crop residues and surface soil. They must cut surface residues and soil and perform the desired functions with minimum disturbances of soil and surface residues (Morrison et al., 1990). This fact is critical as suboptimal plant stands may result when tillage is reduced, because crop residue maintained on the surface may interfere with penetration by seed delivery system and contribute to hair-pinning of residue into the bottom of seed furrows (Carr et al., 2003a; Torbert et al., 2007). When this is the case, a suboptimal population will result in lower final grain yield and grain quality (Geleta et al., 2002).

As additional rule of thumb, wheel traffic, not only for planters but cultivators, sprayers, and fertilizer spreaders, should be restricted to furrow bottoms; otherwise, the bed area can be compacted by the tires of the field equipment (Parsons et al., 1984). Similarly, tire width must be preferably minimized to avoid compaction on the edges of the bed area (Morrison, 1985). These principles are particularly important on coarse-textured soils, i.e. sandy loams and silt loams, since they tend to form traffic pans more readily than clayey soils (Mascagny et al., 1995).

The National Institute for Agricultural, Forestry and Livestock Research (INIFAP) of Mexico, developed a planter for this planting system at the 'Valle de Mexico research station' (CEVAMEX). This development has proved to be particularly successful on the basis of its ability to plant wheat and maize through relatively heavy crop residue (Fig. 5). This ability is based on the need to rotate crops as required for the permanent bed planting system.

Fig. 5. Prototype of disk-type planter for small-grain cereals and maize developed for the planting system on permanent beds.

This equipment is basically the assembly of commercial seed boxes and disk-type planters into a designed frame. One seed box is to drill small cereals and the other to plant maize. A gear mechanism in the frame permits to select the seed box to operate. The installation of the furrow opener allows planting and reshaping beds simultaneously. This planter has proved to be particularly successful on the basis of its ability to plant through the amount of wheat and maize residues resulting from the yields commonly obtained in the Mexican plateau. A feature of this drill is its paired-row configuration for wheat seeds, whereby rows are planted in pairs spaced 20 to 25 cm apart by means of two heavy-duty double-disk openers per bed. To plant maize only one planter per bed is needed and distance between them

should be adjusted accordingly. Since un-tilled soil does not flow, a small wheel in the back of the disk-type planters pushes the soil to close the slot and cover the seed.
Interestingly, this equipment does not include a hopper to band fertilizer at planting. This apparent 'lack' is partly explained in Fig. 4a and b where crop yield response to N application is negligible. However, it is important to emphasize that this only applies to about 400 mm dryland conditions of the Mexican plateau. Nevertheless, in areas with different conditions the need for fertilizer may be critical. Meanwhile, investment cost of this prototype is low which may promote its copy and then the adoption of the planting system on permanent beds.

### 5.2 Variety

Research results on bed-planting methods have shown that not all wheat varieties perform adequately on beds. One reason is that during the breeding process genotypes were generally selected in conventional planting systems (Freeman et al., 2007a). Thus a crucial first step in initiating research on bed-planting wheat is to test a wide spectrum of varieties with differing heights, tillering abilities, phonologies and canopy architectures (Sayre, 1998). Close cooperation between breeders and agronomists to jointly identify and understand the proper plant type needed for optimum performance on beds is highly recommended (Freeman et al 2007b).
Work in the Mexican plateau showed that only three out of eight Mexican wheat varieties recommended for rainfed areas performed acceptably on beds (Limon-Ortega et al., 2008). This differential response can be ascribed to plant height (Sweeney & Sisson, 1988), for example, a tall genotype may perform adequately on beds but not in a conventional planting system on the flat. This means that caution should be exercised when making general recommendations on the basis of studies in which only one variety was used (Siemens, 1963). Results from a study with six wheat varieties and seven locations are presented in Fig. 6 using a basic stability analysis.
This figure clearly shows that performance of wheat varieties changes with the environment. In low-grain yield-environments all varieties perform similar to each other, but as environment improves, grain yield differences become greater reaching up to 2 t/ha difference. This result is an indication on the importance to select the adequate variety before planting on beds. In this case, variety Nahuatl F2000 was the most stable across environments probably due to its tillering ability.
Yield components that determine wheat grain yield are heads per $m^2$, heads per plant, kernels per head and kernel weight and there are compensatory relations among them in response to the changes of environmental conditions and agronomic practices, such as row spacing and seeding rate (Chen et al., 2008). Research work has shown a consistent relationship between grain yield and number of heads; the former increases as the latter improves (Zhang et al., 2007; Chen et al., 2008). This suggests that factors constraining tiller survival should be considered to improve production under such planting systems (Pierce & Lizaso, 1993) regardless of the ability of wheat plants to adjust one yield component when another one is reduced due to environment or other factors (Carr et al., 2003b). In this scenario, if the environment is conductive, genotypes may have the ability to compensate under relatively lower seeding rates to establish good stands with many tillers, larger heads, or more kernels, resulting in higher grain yield (Geleta et al., 2002). According to Schillinger (2005), the number of heads is generally the most important yield component and is

primarily affected by management practices such as seeding rates and N inputs (Zhang et al., 2007). One way to optimize tillering and yield component formation is through the timing of N application (Weisz et al., 2001; Limon-Ortega & Villaseñor-Mir, 2006). A regression analysis with grain yield suggests that attaining 350 heads/m² is key to achieving about 3500 kg/ha of wheat in the Mexican Plateau (Limon_ortega, 2011). Assuming that spring wheat has a tiller survival rate of 70-75% (Zhang *et al*., 2007), it is then estimated that 500 – 466 tillers/m² should be targeted to attain an optimum grain yield in this region.

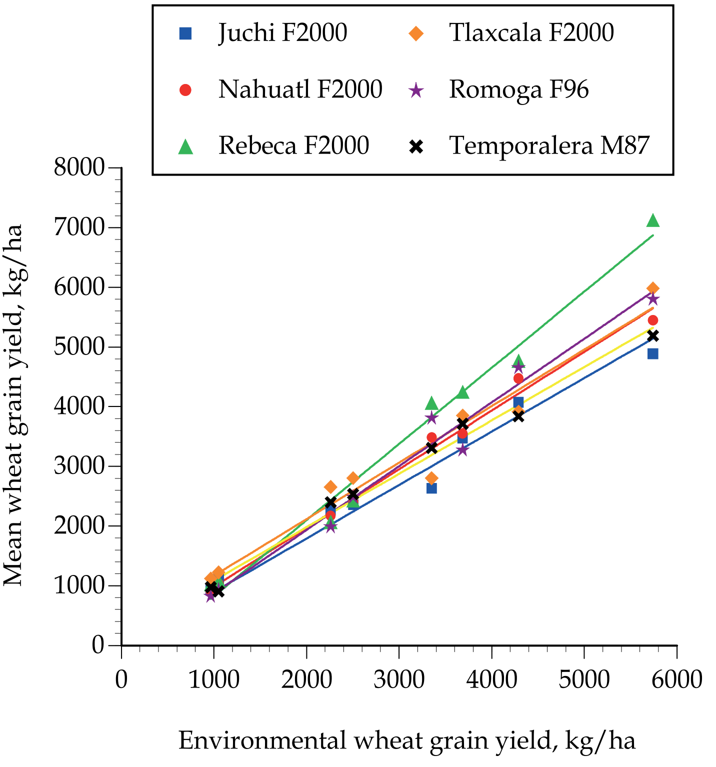


Fig. 6. Stability analysis of eight wheat varieties planted on narrow-raised beds in seven environments of the Mexican plateau (Courtesy, Dr. E. Villaseñor-Mir, 2005).

### 5.3 Crop rotation

Several researchers have suggested that phytotoxins from residues on the surface in a mono crop system may inhibit plant growth. The potential phytotoxic effect can not be reverted unless the period between successive crops provides sufficient time for residues to decompose and potential phytotoxic compounds be broken down or leached away (Carr et al., 2003b). Alternatively, diversifying crop rotations with different water use patterns and requirements can increase yield potential by influencing plant diseases and weeds (Campbell et al., 1990). The total crop yield increase and nutrient removal will depend upon the root depth of each crop (Grant et al., 2002). However, some crops in the rotation may cause reductions in subsequent wheat yields by decreasing the number of heads but they provide diversification and may prove beneficial when the yield and economics of the whole cropping system is considered (Norwood, 2000).

For the specific case of the Mexican plateau, it has been reported that maize-wheat as crop rotation is adequate. Otherwise, wheat grown in a mono-crop system tends to produce lower grain yields (Fig. 7). Data points in this figure are the average of four N rates (0, 40, 80 and 120 kg/ha).

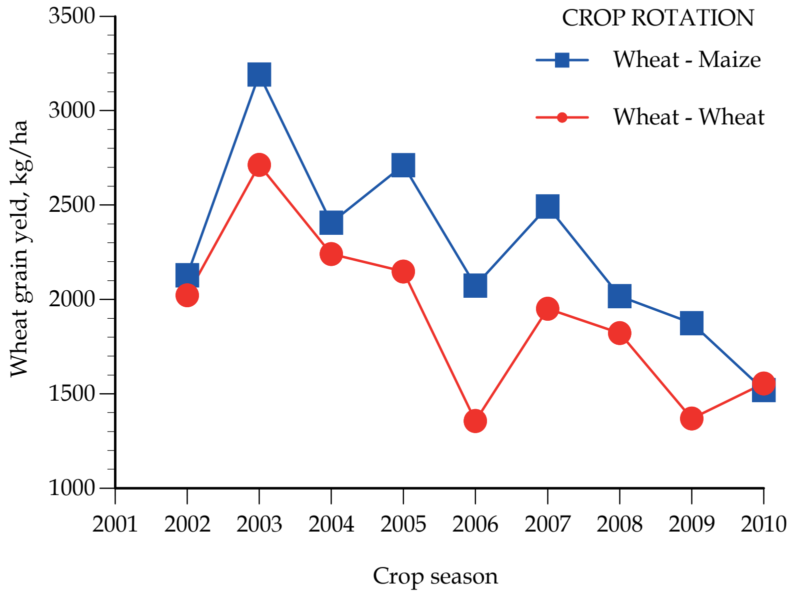


Fig. 7. Wheat grain yield variations when grown in rotation with maize and a mono crop. Chapingo, Mexico.

In general, wheat grain yield was greater for the rotation with maize compared to wheat – wheat rotation. One reason for this result is related to the development of root and foliar diseases as residues from the same crop serve as source of infection (section 6.1 of this chapter). Other relates to soil deterioration, mainly soil aggregate stability and its concomitant effects on numerous quality parameters. In general, wheat – wheat rotation produces less crop residue biomass compared to wheat – maize crop rotation. This results in a greater soil surface exposure to environment and thus deteriorating its aggregate stability.

### 5.4 Crop residues management

There are several constrains associated with crop residue management that may inhibit the farmer adoption of conservation agriculture. Currently many farmers in the Mexican plateau remove crop residues by baling to get an extra income. Farmers who convert to conservation agriculture, but continue to remove crop residues by any means, will usually produce lower yields than with conventional tillage practices. A good rule of thumb is that conservation agriculture should probably not implemented if it is not possible to maintain sufficient residues for adequate ground cover, especially under erosion-prone, low-but intense-rainfall dryland conditions (Sayre, 1998). When only certain amount of residue can be removed, the recommendation for removal rates must be based on regional yield, climatic conditions, and cultural practices. In this regard, agronomists are challenged to develop a procedure (tool) for recommending maximum permissible removal rates that ensure sustained soil productivity (Wilhelm et al., 2004). Since there is intense competition to use residue in many rainfed areas, especially by small- and medium- scale farmers, it is allowed to remove 50 -70 % of the residues as remaining portion will provide adequate benefit to the soil (Sayre et al., 2005). Similarly, more (or improved) knowledge about residue decomposition dynamics is essential for developing effective management strategies as no single residue management practice is superior under all conditions (Kumar & Goh, 2000).

The nature of crop residues and their management has a profound influence on soils over the short – and long -term (Schoenau & Campbell, 1996). Albeit chopping and incorporating crop residues is an acceptable practice for soil improvement, the planting system on permanent beds as conservation agriculture requires crop residues chopped and distributed uniformly over the soil surface, preferably during harvesting with the combine's chopper. But depending on the local conditions, sometimes is better to chop residues after harvesting when the season of winds is over. For example, winds of January and February in the Mexican plateau have the potential to blow off chopped crop residues. It is also important to point out that the initial location of crop residues at the soil surface, the clustering, and the spreading of fragments modify many soil physical factors (Guerif et al., 2001).

Residues, on the other hand, can enhance the loss of N fertilizer by volatilization from broadcast urea because urease enzyme present in the residues can increase the rate of $NH_3$ release (McInnes et al., 1986). Therefore, in a planting system like permanent beds with crop residues on soil surface is crucial the separation of fertilizer from the residues by placing fertilizers below the soil surface to increase fertilizer use efficiency. In irrigated conditions the incorporation can be accomplished by watering immediately after the fertilizer application (Limon-Ortega et al., 2000), while in rainfed areas this should necessary be done by mechanical means below the soil. Alternatively, fertilizer application can be scheduled to coincide with favorable conditions as predicted by short-term (48 – 72 h) weather forecast (Limon-Ortega, 2009b; Nielsen et al., 2005). Furthermore, guidelines routinely used for seed-banded N fertilization that only consider N application rate, should be modified to also consider N source and soil moisture (Mahler et al., 1989).

### 5.5 Bed formation and maintenance

Certain guidelines to begin the permanent beds should be followed (Griffith et al., 1990); 1) on nearly level soils with poor internal drainage, beds should not interfere with natural drainage-ways as ponded water in furrows may result, 2) on slopes of more than 6%, even with all crop residues left, beds should be oriented across the slope, 3) careful driving is needed to maintain proper furrow spacing between adjacent passes when raising beds the first time, any mistake made initially will be carried over from year to year, 4) a cultivator and planter with the same number of rows should be used as none drives straight all the time, 5) bed spacing from center –to- center must be exactly the same as spacing of planter rows, improperly adjusted winged-shovels may not move soil equally to both sides of the ridge, 6) to minimize potential planter centering and control problems, no attempt should be made rising beds too tall or 'peaked' (crowned) on top. Interestingly, harvesting individual rows on crowned beds, grain yield increases from furrow rows to center of the bed, while on flat beds, grain yield can be about the same on the top of the bed (Mascagni et al., 1995).

Furrow bottoms, on the other hand, should be periodically reformed to maintain the shape of the bed to provide surface drainage in situations of waterlogin (Morrison et al., 1990), and to restore the macroporosity to promote water infiltration into the soil and gas exchange. An additional benefit in certain production systems from re-shaping beds can be obtained in soils that tend to develop compaction constrains (Mascagni & Sabbe, 1990) that in other cropping systems occur from the wheel traffic of cultural practices applied at random over the field (Gerik et al., 1985).

## 6. Advantages and disadvantages of the permanent beds

The planting system on permanent beds as conservation agriculture for crop production offers many advantages and disadvantages to users. Both are mostly described by the key components for a successful establishment of the permanent beds in section 5 of this chapter. However, there is not a clear-cut to identify a component as absolutely advantageous as occasionally it might be beyond the farmer's control. Crop rotation, for example, is fundamental but not always feasible. This depends on the onset of the rainy season or expected market in a given year. If the onset of the season is delayed and the following crop in the rotation is maize, farmers should change the cropping plan to wheat or even oat depending upon the time the rain season is established. Regardless of this, other points to take into account about the planting system on beds are described below.

### 6.1 Foliar and root diseases

Direct seeding methods like permanent beds with crop residues left on surface have proved to promote the development of both foliar and root diseases and become yield-limiting factors if crops are grown without adequate rotation (Cook et al., 2000). For example, seedlings which encounter buried residues may be injured if the residues have not had enough time and moisture to lose their pathogenicity (Wuest et al., 2000). Thus, appropriate management techniques are needed to reduce the effects of these factors. Nitrogen fertilizer, on the other hand, has been identified as management factor affecting the incidence of diseases; in general, adequate N rates tend to reduce incidence but this varies according to moisture conditions (Halvorson et al., 1999a).

Foliar diseases, namely Yellow leaf spot (*Helminthosporium tritici-repentis*), and Septoria (*Septoria tritici*) and root diseases, namely take-all (*Gaeumannomyces graminis*), Rhizoctonia root rot (*Rhizoctonia solani*), and Pythium root rot (*Pythium spp*), become yield-limiting to wheat when grown without adequate rotation, especially in no-till plant systems. Nevertheless, the degree of incidence is influenced by the microenvironment formed by the configuration of the plant rows (Cook et al., 2000). For the case of planting on beds, there is a skipped furrow without plants that results in a more open space that affects disease incidence due to the microclimate from the row orientation (English et al., 1989). Foliar diseases that have been found to be suppressed by the planting system on beds include the sharp eye spot disease caused by *Pseudocercosporella herpotrichoides* and powdery mildew caused by *Erysiphe graminis* (Fahong et al., 2004; Sayre et al., 2005).

Visual scores taken from tillering to grain filling stage on the foliar diseases complex using a 0 to 10 scale, allowed to estimate the amount of initial disease and the exponential rate of disease increase (Fry, 1982) for each plot in a field study on permanent beds at Chapingo, Mex. Results showed that only the amount of initial disease was related to final wheat grain yield. Fig. 8 indicates that for every unit increase in the scale, grain yield will be reduced by 246 kg/ha (equation not showed). According to this figure, the wheat – wheat rotation has the largest visual scores and consequently the largest grain yield reductions. In contrast, the annual wheat – maize rotation showed less incidence of foliar diseases which resulted in greater grain yield.

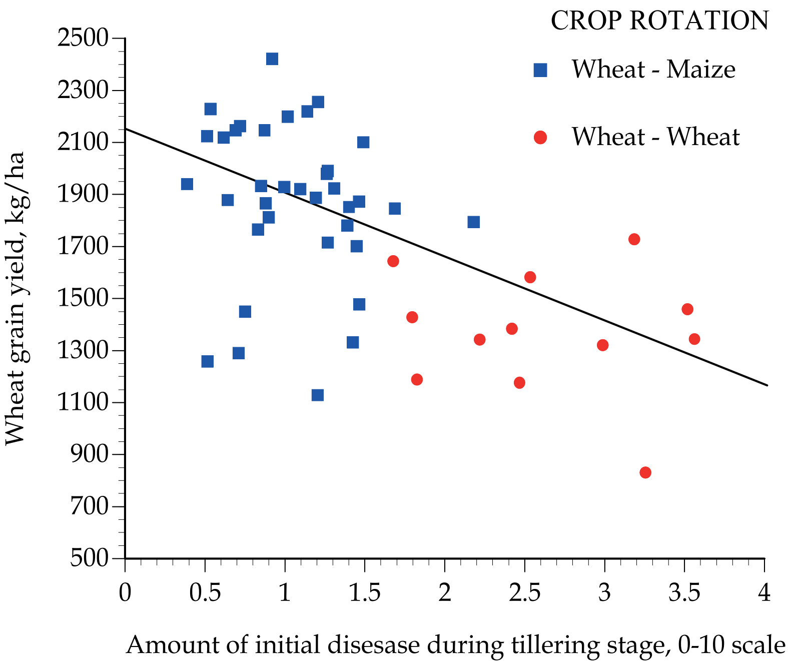


Fig. 8. Relationship foliar disease score - wheat grain yield in 2009 crop season. Chapingo, Mexico.

The work on root disease incidence on permanent beds in the Mexican plateau, on the other hand, has also shown the effect of crop rotation. For wheat – maize rotation with full residue retention, the root rot incidence in wheat was intermediate while for maize was even lower. However, in contrast to foliar diseases, the root rot incidence in both crops had minor influence on final grain yield as other factors such as water availability and nutrient status were more critical (Govaerts et al., 2006b).

### 6.2 Diking

Water is often the most limiting factor to dryland agricultural production. Practices that conserve water received as rainfall can greatly improve the potential for success of cropping systems (McFarland et al., 1991). One of those practices should include the bed planting system joined with furrow diking.

Furrow dikes are small dams formed periodically between the beds along the furrow bottoms. The furrow diking practice is known by many names, including tied-ridges; furrow damming; basin tillage; basin listing; and microbasin tillage. Furrow diking is a soil and water conservation practice that is very well adaptable to dryland crop production. It is most often used on gently sloping terrain in arid and semiarid areas where crops are grown under water deficit conditions (Jones & Baumhardt, 2003). Furrow diking in the Mexican plateau for wheat production on conventional-till raised-beds was first used in 2000 by Mr Emigdio Taboada, a wheat farmer at Nanacamilpa, Tlaxcala state. This farmer modified his conventional drill removing three planters and replacing them by three small furrow openers connected to an eccentric wheel which causes a trip movement to form small dikes. As immediate result wheat grain yields were improved, the amount of water runoff was substantially reduced, and water infiltration through the soil profile increased. Similar results have been reported for other places and crops (Jones & Baumhardt, 2003) indicating a strong correlation between grain yield and amount of rain during critical growth stages (Tewolde et al., 1993).

The application of furrow diking technology in a bed planting system is of particular importance in many semi-arid regions where rainfall is often of high intensity and short duration (Lyle & Dixon, 1997). This rainfall pattern is characteristic of many developing countries (Clair & Linch, 2010) including the Mexican plateau and will be surely extended to other areas of the world as the climate change will continue. For example, precipitation intensity in terms of the number of days with precipitation above 25 mm, shows a statistical significant increase in many areas of the globe (Porter & Semenov, 2005). Those changes in rainfall distribution can be parameterized by means of standard deviation (Monti & Venturi, 2007). The effect of furrow diking on water retention in conventional-till raised-beds appears to offset those climatic changes as shown in Fig. 9. Nevertheless, the implementation of tied ridges has no effect on soil parameters (Govaerts et al., 2007).

However, care should be taken in permanent beds as research has shown that furrow diking in every furrow may not be desirable as wheat grain yield can be slightly reduced (Saye et al., 2005). Alternatively, to improve yields furrow diking should be applied in alternate furrows (Limon-Ortega, 2011). This differential effects of furrow diking options on grain yield can be ascribed to an excessive amount of rain water accumulated in the soil profile due to the improvement of soil structure stability and suggests that there should be a balance between water conservation and drainage .

Fig. 9. Rainfall water retention with furrow diking in conventional-till raised-beds applied to wheat.

Apparently the added water through the sole use of conservation practices compared with more intensive conventional tillage, is enough to take full advantage of the often low and erratic growing-season precipitation (Grant et al., 2002). But care should be taken as contrasting results have been reported for crops like maize from wetter areas with rain amounts exceeding 900 mm where diking had little effect on grain yields (McFerland et al., 1991). The inconsistency of furrow diking in increasing grain yields can also be attributed to size of rain events –rainfall distribution. For example, small rain events (< 20 mm) can be lost to evaporation and then no-till with crop residues can be more effective than furrow dikes in improving water conservation in semiarid regions (Nielsen et al., 2005).

## 7. Conclusion

Given the large number of advantages of the planting system on permanent beds over the conventional planting for wheat and maize production, researchers have to joint efforts to accomplish two basic requirements. One is the work of agronomists with machinery designers to develop prototypes of planters that can be copied by small-scale farmers and be easily reproduced in local shops. Other is the joint work with breeders to identify and select the appropriate wheat and maize genotypes for the bed planting system. Furthermore, local governments should provide subsidies to allow those farmers to acquire planters and simultaneously provide some incentive to trigger the adoption of the system.

The stabilization period required to obtain the benefits of the permanent beds appears not to have a pronounced effect on wheat and maize yields under the rainfed (about 400 mm rainfall) conditions of the Mexican plateau. The adoption of this planting system as conservation agriculture and its effects on the improvement on soil attributes has the potential to reduce substantially the degree of soil erosion, as well as to improve the farmer's income by increasing grain yields and reducing production costs.

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# 9

# The Significance of Soil Erosion on Soil Fertility Under Different Tillage Systems and Granitic Sandy Soils in Semi-Arid Zimbabwe: A Comparison of Nutrient Losses Due to Sheet Erosion, Leaching and Plant Uptake

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## 1. Introduction

The plant nutrient status in the soils is dynamic and subject to a variety of transformations, gains and losses, depending on a number of factors including soil type, farming system and climate. Generally, soils lose plant nutrients in more ways than they gain them. The very common and obvious way is through the removal of crop harvests and residues. However, it is also known that erosion and leaching contribute immensely to the total nutrient loss from a field. In the case of nitrogen, denitrification/ volatilisation also contributes to the N loss (Stoorvogel and Smalling, 1990). For a sustainable management of nutrient systems some consideration has to be given to these losses and a balance struck between the losses and the possible gains that may occur in the field.

The components of a nutrient balance with regards to nutrient losses and possible nutrient gains are shown in Figure 1. As can be seen from this figure, the main losses of nutrients from the soil are through plant uptake, leaching and soil erosion. Volatilisation only involves nitrogen and is common under anaerobic conditions making it less important when compared to the first three forms of nutrient loss. Wind erosion in Zimbabwe is minimal and confined to a few areas to justify its exclusion from the components of nutrient balance. It is however, not clear as to what percentage of total nutrient loss is lost with each of the three major components, i.e. plant uptake, soil erosion and leaching.

Many investigations have shown that soil erosion results in loss of productivity, due to the modifications of the soil physical, chemical and biological composition. The degree to which these changes take place varies for the different soil types and from one agro-ecology to another (Kaihura, *et al.*, 1998). Given the very diverse agro-climatic conditions, results from one area cannot be successfully applied to the next. However, due to limited research findings in Sub-Saharan Africa, general estimations have been made and then extended to the whole region (van Reuler and Prins, 1992). In these countries soil is an essential input to farming as agricultural production is crucial to development and the livelihoods of the majority of the population depend on this sector (Barbier and Bishop, 1995). Yet inappropriate farming systems and overgrazing are rampant, nutrient budget is generally negative as soils are mined for nutrients resulting in negative environmental consequences of soil erosion and land degradation (van Reuler and Prins, 1992). Soil erosion - although considered as one of the many possible facets of land degradation (Biot, 1986) - constitutes a serious ecological and economic problem, which threatens agricultural production in Sub-Saharan Africa (Anderson and Thampapillai, 1990). Once initiated the process of erosion is self perpetuating in that fine soil particles that bind the soil particles together are lost and the soil becomes loose and more vulnerable to erosion, so erosion increases (Lowery and Larson, 1995).

In the case of nutrient uptake by the crop, many factors are involved, e.g. the light, temperature, air humidity, carbon dioxide, oxygen, water, macro- and micro-nutrients as well as the pH value of the soil solution and plant sap (Hekstra, 1996). According to Wrigley (1992), the nutrient uptake capacity is more closely related to root volume. What is important in nutrient loss studies however, is the availability of nutrients to the crop. This refers to the availability of a nutrient at the right time and in the right form and quantity. If the nutrients are not in the right form or quantity they may be fixed or leached. This reduces the amount of nutrients taken up by the crop, the rest being susceptible to leaching and fixation. Leaching of nutrients, therefore takes place at any time during crop growth when more soluble nutrients are found in the soil solution than can be taken up by the crop (Hekstra, 1996). These processes in turn deplete the soil of its nutrients and pollute the under-ground water. Fixation on the other hand is dependent on the pH, clay and organic matter content. Applied nutrients may be taken out of the soil solution and become immobilised or fixed on the soil's solids. This process counters leaching of applied fertilisers and thus adds to the soil's solid nutrient reserves (Singer and Munns, 1987).

The nutrient losses of concern for this study are the macro-nutrients, Nitrogen, Phosphorus and Potassium that are applied as fertilizers, even in low input agriculture. These are found in different forms in the soil. Some nutrients are more dynamic than the others, while some are more prone to fixation, leaching and/or washing away (Fitzpatrick, 1986). The macro-nutrients are also found in different concentrations in the soil solution. However, since soil erosion does not only render the soil of its available forms of nutrients, but also of its fixed and organic forms, it is important that total nutrients be considered when dealing with nutrient losses due to erosion, as this has a bearing on the productivity of the soil in the long term.

Nitrogen (N) in soils, unlike other nutrients does not originate from the soil mineralogy but a substantial amount of it originates from the air - through atmospheric deposition and symbiotic fixation (Singer and Munn, 1987). Fertilizer application is very important too as nitrogen is one of the most yield limiting nutrients (Brady, 1984; Stevenson, 1985). Nitrogen is a highly dynamic nutrient in the soil thus its status is very variable. It undergoes a wide variety of transformations in the soil, most of which involve the organic fraction. This makes the interpretation of available nitrogen content in the soil highly inconclusive as it changes within a short period. The common forms of nitrogen in the soil are (i) organic N, (ii) available N and (iii) fixed N (Stevenson, 1985). Only a small fraction of nitrogen in soils, generally between 1 - 2%, exists in available mineral compounds at any one time, i.e. as ammonium ($NH_4^+$) and nitrate ($NO_3^-$) forms (Brady, 1984). The main losses of N are as a result of (i) leaching, (ii) fixation, (iii) volatilization and (iv) erosion (Brady, 1984; Stevenson, 1985 Singer and Munns, 1987). While $NH_4^+$ ions are prone to fixation within the clay layers as well as volatilization as $NH_3$, $NO_3^-$ ions are prone to denitrification.

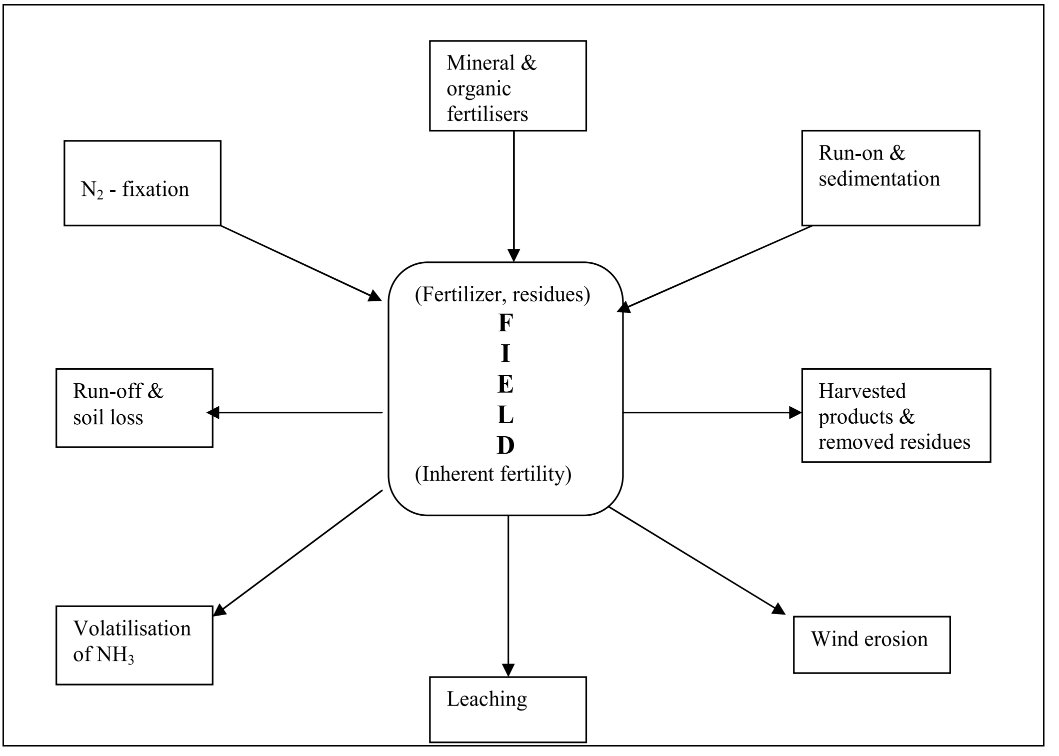


Fig. 1. External components of nutrient balance (Source: *modified from Hekstra, 1996*)

Phosphorus (P) originates from the parent material and soils formed from acid igneous rocks are generally low in phosphorus (Stevenson, 1985). The P cycle is complex and involves the storage of P in living organisms, dead organic matter and inorganic forms. There are comparatively low levels of total P in the soils compared to other elements. Organic P in the soil comprises more than half of the total soil P (Brady, 1984). Microbial activity simultaneously consumes and releases P to the soil solution, while the native P is not easily available and any P applied as fertilizer is prone to fixation. Soil pH also controls the fixation rate. In acid soils, as is the case with the Makoholi 5G soils, phosphate is readily precipitated as the highly insoluble Fe- and Al-phosphates or absorbed to oxide surfaces. This makes the available P in soils very low.

The original sources of potassium (K) are primary minerals such as micas and potash feldspars. The total quantity of potassium is generally greater than that of any other major nutrient element. Most of the potassium is in the primary mineral and non-exchangeable forms (Brady, 1984; Stevenson, 1985). Organic matter constitutes only a small amount and is not as important in determining the potassium content and its availability as is the case with nitrogen and phosphorus. The three main forms of potassium according to Brady (1984) are: unavailable K, which is found in primary minerals (90 - 98% of total K); slowly available K, which is the non-exchangeable (fixed) form (1 - 10% of total K) and readily available K, which is exchangeable including the amount in the soil solution (1 - 2% of total K).

In Zimbabwe, as is the case with many Sub-Saharan countries, soil nutrient losses as a result of erosion and leaching have not been given much consideration in the past. While data may be available on nutrient losses due to plant uptake, the relative significance of one type of loss to the other losses is not known. The information on the different forms of individual nutrients and their amounts in the soil serves to show how important it is to evaluate total nutrient losses as compared to the available/ exchangeable forms (sheet erosion). It is a fact that for annual soil fertility interpretation, the total quantity of a nutrient in a soil is much less important than the amount that is available or that can be made readily available for plant uptake. However, the degree to which the soil is impoverished may remain masked if only the available/ exchangeable forms of nutrients are considered. The nutrient losses through sheet erosion can mainly be reduced through conservation tillage. The main advantages of conservation tillage are the reduction of run-off and soil loss from a field by increasing infiltration. However, when infiltration is increased, it invariably increases drainage, which may lead to increased leaching of the nutrients. It is, therefore, of utmost importance to evaluate the different tillage systems on their nutrient losses with leachate and to ascertain whether in fact the systems that conserve soil and water also lead to more leaching of available nutrients than conventional tillage systems. This Chapter, therefore, focuses on the quantification and comparison of nutrients lost with sheet erosion (sediments and run-off), plant uptake and leaching under different tillage systems.

## 2. Materials and methods

### 2.1 Site characteristics

The research work was carried out at Makoholi Research Station, situated 30 km north of Masvingo town, Zimbabwe. It is the regional agricultural research centre for the sandveld soils in the medium to low rainfall areas. The station lies within Natural Region IV at an altitude of about 1200 m (Thompson, 1967; Anon, 1969). Characteristic of this region is the erratic and unreliable rainfall both between and within seasons (Anon, 1969). Average annual rainfall is between 450 and 650 mm (Thompson and Purves, 1981). The soils at Makoholi are inherently infertile, pale, coarse-grained, granite-derived sands, (Makoholi 5G) of the fersiallitic group, Ferralic Arenosols (Thompson, 1967; Thompson and Purves, 1978). Arable topsoil averages between 82 and 93% sand, 1 and 12% silt and 4 and 6% clay (Thompson and Purves, 1981; Vogel, 1993). The small amount of clay present is in a highly dispersed form and contains a mixture of 2:1 lattice minerals and kaolinite (Thompson, 1967). The organic matter content is also very low, about 0.8%, while pH ($CaCl_2$) is as low as 4.5. The soils are generally well drained with no distinct structure (Thompson and Purves, 1981), but some sites have a stone line between 50 and 80 cm depth. The low infiltration rates and water holding capacities are due to the soil texture characteristics. The agricultural potential of these soils is fair (Grant, 1981; MNRT, 1987) and their productivity is likely to decline under intensive continuous cropping (Thompson and Purves, 1978). Therefore increased production can only be achieved through good management as well as application of fertilisers or animal manure (MNRT, 1987).

### 2.2 Experimental design and tillage treatments

The treatments were laid out in a randomised complete block design replicated three times. The blocks were located at different positions along the slope (Down-slope, Middle-slope and Up-slope), which could differ in fertility, erodibility as well as moisture levels. At the beginning of the nutrient loss study, all trial plots had been under cultivation and the same treatment for a period of five years. All tillage operations were carried out soon after harvest before the soil dried out.

### 2.2.1 Conventional tillage

The land was ox-ploughed to 23 cm depth, using a single-furrow mouldboard plough and thereafter harrowed with a spike harrow. All crop residues were removed from the plots (Plate 1), as is the practice in the communal areas. This tillage system is the most commonly used tillage system in the communal areas and was chosen as a standard primary tillage method, including this treatment provides a baseline for assessing the merits of other treatments.

Plate 1. Conventional tillage system

### 2.2.2 Mulch ripping

The land was ploughed to the recommended depth of 23 cm in the first year and rip lines were opened. During the subsequent seasons, crop residues from the previous season were left to cover the ground and only rip lines, 23 cm deep, were opened between the mulch rows, using a ripper tine (Plate 2). The rip lines acted as crop rows and were alternated every year, to allow roots ample time to decay. Two basic conservation tillage components – minimum tillage and mulching - are realised here. The main aim was to maximise infiltration through rainfall interception provided by the mulch, thus minimising run-off. These parameters are the most important in the semi-arid regions, where soil moisture is the most limiting factor in agricultural production (Hudson, 1992). This treatment is one of the basic conservation tillage systems, which has shown great potential in protecting the soils, without compromising the production.

Plate 2. Mulch ripping system

#### 2.2.3 Tied ridging

The land was ploughed to the recommended depth of 23 cm in the first year and crop ridges constructed at 1 in 250 grade, using a ridger. The ridges were about 900 mm apart and small ties were put at about 700-1000 mm along the furrows between the crop ridges (Plate 3). These ties were between one half to two thirds the height of the crop ridges allowing for run-off to flow over the ties and not over the ridges (Elwell and Norton, 1988). The ridges were maintained several years through re-ridging so as to maintain their correct size and shape. This treatment has been found to reduce run-off, and the soil losses to satisfactorily low level.

Plate 3. Tied ridging system

### 2.2.4 Bare fallow

Ploughing, up to 23 cm depth, was done using a tractor disc plough and disc harrow. The plots were kept bare and weed free, by spraying the germinating weeds during the season. This treatment is important for soil erodibility assessment and modelling purposes, as it gives the highest possible soil loss values and will probably give the lowest nutrient loss values as no fertilisers are applied.

### 2.3 Collection of run-off and sediments

Soil loss and run-off measurements were from 30 m x 10 m run-off plots, with 5 m border strips on either side. All treatments were laid out at 4.5% slope. The length of the plots was orientated up-slope. Tillage operations were done across the slope. For the tied ridging treatment, the collection area was 150 m long and 5 crop rows wide (4.5 m), with 2 guard rows above and below. The crop ridges were laid at 1% slope and the length of the plots was orientated across the slope. Polythene strips were dug in to form the boundary around each 30 m x 10 m and 150 m x 4.5 m plot. This is the standard soil erosion methodology for Zimbabwe – Soil Estimation Model for Southern Africa (SLEMSA).
Surface run-off and soil loss from each plot were allowed to collect in a gutter at the bottom of the plot. From the gutter these were channelled through a PVC delivery pipe into the first 1500 litre conical tank. The collection tanks were calibrated and run-off was measured using a metre-stick. Once the first tank was full its overflow passed through a divisor box with ten slots, which channelled only one tenth of the overflow into the second tank. Nine tenths of this overflow was allowed to drain away, thus increasing the capacity of the second tank. Due to the larger net plots of the tied ridging treatment, three tanks were installed, so as to capture the anticipated larger volume of sediments. Tanks were emptied at the end of each storm unless the interval between storms was too short to allow emptying. Sediments and run-off (including the suspended material) collected from run-off plots were treated as two different entities. Suspension was pumped out and sub-sampled for the determination of soil concentration in run-off, using the Hach spectrophotometer DL/2000. The sludge was transferred into 50 liter milk churns, topped up with water to a volume of 50 litres and weighed. The mass of oven dry soil, $M_o$ (kg) was calculated using the following equation (Wendelaar and Purkis, 1979; Vogel, 1993):

$$M_o = 1.7 \times (M_s - M_w) \tag{1}$$

where $M_s$ = mass of fixed volume of sludge (kg)
$M_w$ = mass of the same volume full of water (kg)
1.7 = constant for the soil type

### 2.4 Agronomic details

Maize (*Zea mays* L.) is the staple food in Zimbabwe. For this reason, maize was chosen as a trial crop, so as to make the research project relevant to the communal areas. Due to the dry conditions prevailing at Makoholi, maize variety R 201, which tolerates moisture stress and is short seasoned, was used. All weeding operations were done using a hand-hoe. The problems of nematodes, very common in the sandy soils and that of maize stalk borer were controlled, so as to minimise the influence of factors other than those imposed by treatments. On all plots, except the Bare fallow, planting holes of about 10 cm depth and diameter were opened before the onset of the rains. The crop spacing of 900 mm inter-row and 310 mm in-row were used resulting in a plant population of about 36 000 plants/ha. Thereafter Carbofuran, was applied into these planting holes at a rate of 20 kg/ha. Compound D (N:P:K=8:14:7) was also applied into the planting holes at a rate of 200 kg/ha to give a final ratio of 16 kg N: 12 kg P: 12 kg K. The nematicide and fertiliser were then slightly covered with soil and left until adequate rainfall had been received. Once the profile of the ridges was wet throughout, maize was planted, two seeds per station. Ten days after planting, crop emergence count was carried out followed by weeding. The crop was then thinned out to one plant per station. When the crop was about six weeks, ammonium nitrate top-dressing fertiliser was applied at 100 kg/ha, amounting to 34.5 kg N/ha. The ammonium nitrate application coincided with the second weeding and the application of Thiodan, to control maize stalk borer.

Plant growth parameters (plant height, number of leaves, tasselling and silking) were recorded on selected twenty plants per plot and an average taken to indicate plant growth for the different tillage systems. The measurements started at two weeks after planting and continued up to physiological maturity, about 18 weeks after planting. Sub-plots of 3.6 m x 6 m were marked out with each plot having two subplots for all treatments except for tied ridging, which had four sub-plots per plot. The crop within this area was harvested, grain samples weighed and dried at $105^0$ for 16 hours, while the stover samples were weighed and dried at $65^0$ for 24 hours in the Memmert Universal ovens (Model UL 80). The yield was calculated at 12% grain moisture, while stover was given as dry matter. On all trial plots, soil samples were taken at 0-250 mm using a split auger. The soil samples were air dried and passed through a 2 mm sieve.

### 2.5 Leachate measurement

A technically simple and cheap methodology, as described by Hagmann (1994) was used for the collection of leachate. Nine percolation lysimeters were installed on a 4.5% slope under the three tillage systems. For each tillage system, two lysimeters were installed in plots, which were later cropped and one in a bare plot. The percolation lysimeters were made out of galvanised metal sheets and were 1.5 m long, 0.9 m wide and 0.5 m deep. They were installed at a depth of 0.25 m below the soil surface to allow for undisturbed tillage and weeding operations as well as undisturbed inter-flow. The 0.25 m above the lysimeter box together with the depth of the percolation lysimeter ensured that the base of the lysimeter was below the rooting depth, which has been found to be 0.70 m (Moyo and Hagmann, 1994). Thus the water, which reached the bottom of the lysimeters could well be defined as drainage water, as it would have left the rooting zone and would not benefit the crop. Soil was excavated and piled according to soil horizon. Lysimeter boxes were installed in the pits, filter packages inserted at the outlets and drainage pipes were connected to the outlets. A layer of 50 mm of fine gravel was added at the bottom of the boxes to allow for better percolation. The excavated soil was refilled in layers corresponding to the soil horizon. To regain the natural bulk density (1.4 - 1.5 $Mg/m^3$) the refilled soil was slightly compacted. The pipes from the outlets were laid out at a gentle slope to enable drainage water to gravitate into the collection pit. Leachate was collected daily.

### 2.6 Laboratory analyses

An analysis of the sediments for macro-nutrients was carried out, where the different sediment components (water, suspended material and sludge) were treated as different entities. The main aim being to quantify nutrient losses as a result of erosion and to ascertain which sediment component carries the most nutrients. Total nutrients were determined in an effort to capture all forms of nutrients and therefore give a clear picture of how much was lost with erosion, rather than giving a mere fraction of the available form. The soil samples and sediments were air dried and passed through a 2 mm sieve, while the plant material (stover and maize grain) were oven dried at $35^0$C and ground before they were analyzed for the macro nutrients N, P and K. Nitrogen was determined using the microkjeldahl method as described by Bremner and Mulvaney (1982) and the ignition method as described by Olsen and Sommers (1982) was used to quantify phosphorous. Potassium content was determined using the wet digestion method using perchloric acid as described by Knusden, *et. al.*, (1982).

Run-off was filtered and as with leachate, the aliquot treated as soil extract. Run-off and leachate were analysed for the dissolved nutrients, where the nutrient concentration was either titrated with boric acid, for N determination, read from an Atomic Absorption Spectrophotometer for the determination of P or read from a flame-photometer in the case of K. It should be noted that only $NO_3^-$- N was quantified during this study, as very low levels of $NH_4^+$-N are lost due to leaching, even under intensive maize production systems (Kladivko, *et. al.*, 1991; Prunty and Montgomery, 1991; Drury, *et. al.*, 1993).

### 2.7 Soil nutrient balance

A method that was developed by Stoorvogel and Smaling (1990) on balancing of nutrients was used. This method looks at balancing the inputs and outputs as the sum of the inputs versus the sum of the outputs in a given system as follows:

$$\text{Nutrient balance} = [IN1 + IN2 + IN3 + IN4 + IN5 + IN6] - [OUT1 + OUT2 + OUT3 + OUT4 + OUT5 + OUT6] \quad (2)$$

Where: IN1 = mineral fertilisers; IN2 = animal manure; IN3 = atmospheric deposition; IN4 = biological nitrogen fixation; IN5 = sedimentation; IN6 = uptake by deep rooted plants; and OUT1 = harvested products; OUT2 = crop residues; OUT3 = leaching; OUT4 = gaseous losses; OUT5 = soil erosion OUT6 = losses in deep pit latrines.

According to Scoones and Toulmin (1999), some of the parameters may relatively be easy to assess, e.g. fertiliser and manure inputs and crop outputs, while leaching, volatilisation and erosion present more measurement difficulties. The trend has been that the leaching, volatilisation and erosion losses were estimated. In this study however, these, with the exception of volatilisation, have been quantified. Volatilisation presents negligible losses under these conditions, as it is most prevalent in anaerobic and mainly alkaline conditions (Stoorvogel and Smaling, 1990). The biological N fixation in maize production systems is also negligible. Atmospheric deposition of N is the only worthwhile input that could have been considered in the balancing of N, however that data was not available and as such the N balance may be slightly inaccurate. No manures were used in this study and the only inputs into the system were limited to inorganic fertilisers.

Thus the equation was modified as follows:

$$\text{Nutrient balance} = [IN1] - [OUT1 + OUT2 + OUT3] \quad (3)$$

Where: IN1 = mineral fertilisers; OUT1 = plant uptake; OUT2 = soil erosion; OUT3 = leaching

The differences in soil loss, run-off, plant growth parameters, yield and nutrient losses attributed to treatment were analysed with the analysis of variance (ANOVA) procedure of Genstat 5 Release 1.3 statistical package. An independent t-test was used to compare the means of different populations. Unless otherwise indicated, significance is indicated at $P < 0.05$ (*), 0.01(**) to 0.001 (***).

## 3. Results

### 3.1 Plant nutrient losses with sheet erosion

Soil erosion removes topsoil enriched in nutrients by natural formation processes and fertilization. To be able to quantify the nutrient losses with sheet erosion one has to measure run-off and the rate of soil erosion. Previous research has put much emphasis on the importance of N and P in plant nutrition. For example Barisas *et. al.* (1978), Sherwood and Fanning (1981) and Arnimelech and McHenry (1984) also examined N and P and disregarded other nutrients. N is without doubt the most significant nutrient for high maize yields and its deficiency limits production more than any other nutrient (de Gues, 1973). A maize crop uses N throughout its growing period, unlike P which mainly promotes root development and is an important part of the proteins. Thus P application is only confined to basal application, while N can be applied up to three times during the growing season. P deficiency also has drastic effects on the maize yields. Singer and Munns (1987) reported that the loss of phosphorus and nitrogen is most serious, because they are often deficient. Elwell and Stocking (1988) also quantified the loss of N and P with erosion, while disregarding cations, especially K, which is always applied as a basal fertilizer (together with N and P) in all maize production systems in Zimbabwe. The importance of N and P may well be emphasized if only plant nutrition is of importance, but no consideration is given to the soil condition after erosion (erosion effect). It is arguable that the loss of cations is also equally important as it leads to soil acidification and consequently to soil degradation. Potassium is relatively abundant in the soil, especially when compared to P (Stevenson, 1985) and if its loss is left unabated, the soils become impoverished and this enhances soil degradation. K, classified as a primary nutrient with N and P, is essential for crop growth (de Geus, 1973). Assessing its loss is important, as it has a role in plant nutrition and is also implicated in soil degradation studies.

#### 3.1.1 Run-off and soil loss

Run-off and soil losses were highest under the bare fallow followed by the conventional tillage while mulch ripping and tied ridging recorded low run-off volumes and soil losses (Table 1). These differences among the different tillage treatments were significant at $P < 0.001$. To properly evaluate the effectiveness of the conservation tillage treatments (MR and TR), the mean of conventional tillage and that of the two conservation tillage treatments were compared using an independent t-test. The results showed that conventional tillage differed significantly from the mean of the two conservation tillage systems (at $P < 0.001$), while the two treatments did not differ significantly from one another.

In the semi-arid conditions, rainfall is a very important parameter in agricultural production. It is not only the amount that matters but also the nature of rainfall received, i.e. its distribution and intensity. High intensity rainfall of up to 132 mm/h was received mainly during the months of October and November. During this period the soils in the communal areas would mainly be bare due to the nature of the predominant farming system here, where all crop residues are removed from the fields or left for the livestock to graze on during the winter months (Hudson, 1964; Elwell and Stocking, 1974). Most soils are said to best absorb rainfall of 12 mm/h intensity (Wrigley, 1992) and the very high intensities measured during this study show that the erosion potential is very high. The potential damage is aggravated by the absence of crop cover (Hudson, 1964) during this period. According to Wrigley (1992) heavy intensity of tropical storms is frequently in excess of the receptive capacity of the soil and the beating of soil by raindrops seals the soil surface thus hindering infiltration. Run-off is initiated, which is the initial stage of erosion. These findings imply that high intensity rains are received at the beginning of the season, before enough crop cover has been attained and the soils would be most vulnerable to erosion.

| Treatment | Run-off (mm) | Soil loss (kg/ha) |
|---|---|---|
| CT | 104.4 | 33.7 |
| MR | 39.8 | 1.7 |
| TR | 34.1 | 2.2 |
| BF | 161.1 | 93.4 |
| Signif. level | *** | *** |
| s.e.d. | 8.07 | 4.00 |

Table 1. Run-off (mm) and soil loss (kg/ha) as affected by tillage at Makoholi Contill site during three seasons

Run-off was dependent on seasonal rainfall and it differed significantly among the three years as follows: Year 1 received 483 mm of rainfall and had an overall mean run-off of 60 mm; Year 2 received 384 mm and recorded 31 mm of run-off while rainfall recorded in Year 3 was 765 mm, with 165 mm of run-off. Another factor, which influenced run-off, was the tillage system. The conservation tillage treatments reduced run-off drastically when compared to the conventional systems, ranging from <1 - 15% under mulch ripping; 1 - 11% under tied ridging; 13 - 22% under conventional tillage and 17 - 39% under the bare fallow. Overall the treatments recorded annual averages of 40 mm (MR); 34 mm (TR); 104 mm (CT) and 161 mm (BF). The month of January, during Year 3 received 419 mm of rainfall, (55% of total seasonal rainfall), which increased run-off drastically, especially under mulch ripping treatment, as the soil was saturated and water logging was experienced. Run-off decreases with the increase in soil moisture up to saturation point, after which it increases drastically (Le Bissonnais and Singer, 1992; Olsen, 1994). For this reason run-off from mulch ripping rose steeply to 15% of total seasonal rainfall, tied ridging recorded 11%, while under conventional tillage run-off rose slightly to 22%.

The natural equilibrium of the soil under the bare fallow was disturbed through cultivation and the lack of crop, weed or mulch cover further aggravated the situation. This practice accelerates the rate of organic matter mineralisation through disruption of soil aggregates and increased aeration (Schroeder, 1984; Salinas-Garcia, Hons and Matocha, 1997; Angers, N'dayegamiye and Cote, 1993). Organic matter improves water infiltration and storage (Follet *et al.*, 1987) thus its reduction results in high run-off. Furthermore, when there is no soil protection and rainfall energy is not intercepted surface crusting is promoted (Troeh *et al.*, 1980), which further accelerates run-off. According to Le Bissonnais and Singer (1992), soil crusts reduce water infiltration rate and induce the erosion process by increasing runoff. The cropped treatments recorded lower average percentages of run-off due to the crop cover effect. Continuous inversion of the soil (through ploughing and harrowing), as has been highlighted for the bare fallow and conventional tillage, increase organic matter mineralisation resulting in the reduction of water stable aggregates (Beare, Hendrix and Coleman, 1994). The soil structure therefore, deteriorates and enhances run-off (Elwell, 1990), which increases as the number of years of cultivation increase. Effective reduction of run-off was realised under mulch ripping and tied ridging. The mulch under mulch ripping intercepts rainfall energy, thus increasing infiltration (Adams, 1966; Braithwaite 1976; Elwell 1986). The rotting stover adds organic matter to the soil, which contributes to improved water infiltration and storage and generally to the maintenance of a good soil structure (Hargrove, 1991; Reicosky *et al.,* 1996). The lower run-off under tied ridging was a result of water ponding in the micro-dams, which increased infiltration drastically. The tied ridging treatment was described by Hudson (1992), as a useful compromise between water storage and drainage, as water is retained in the basins to soak into the soil, resulting in very little water leaving the system.

The soil losses in sandy soils were, as expected, very high (Table 1) as these soils are much more susceptible to erosion as compared to other soils with higher contents of clay and/or organic matter. As soil erosion is a function of run-off, soil loss was also dependent on seasonal rainfall and tillage treatment. The very high losses under the bare fallow (93t/ha/yr) were a result of continuous ploughing of the soil and leaving the soil bare throughout the year. Aeration was high leading to mineralisation of organic matter and there was no ground cover to protect the soil from the impact of climatic factors (rainfall, solar radiation). No impediments were constructed to slow down the velocity of run-off at any given time during the growing seasons. The combination of all these factors is a deteriorated soil structure, low organic matter content, few water stable aggregates and accelerated soil loss. Gerzabek, Kirchmann and Pichlmayer (1995) found that bare fallow plots had lower soil stable aggregates compared to other treatments. Since the water stable aggregates are a measure of the resistance of soil particles against disruptive forces of water (Beare *et al.*, 1994), it therefore follows that the lower the water stable aggregates the higher the soil loss as soils would not be able to resist erosion.

The soil losses under conventional tillage were very high and averaged 34t/ha/yr. Although this value is low when compared to the 50 - 75 t/ha/yr estimated by Elwell (1975), it must be noted that his estimation may still be valid for some seasons characterised by high rainfall amount and intensities, e.g. the 54 t/ha soil losses recorded during Year 3. One other factor, which influences the amount of soil loss is the period of cultivation, as these results are from fairly new fields, having only been cultivated for a period of nine years, while some soils in the communal areas have been under cultivation for over fifty years. Continuous ploughing and removal of crop residues under this treatment has led to the deteriorating soil structure. Elwell, (1990) reported that cultivation leads to mineralisation of organic matter, which is important in the soil aggregation and that the removal of crop residues together with organic matter mineralisation leads to poor soil structure. Poor soil structure profoundly increases run-off and soil loss. The very high topsoil losses with conventional tillage will eventually result in reduced plant available water and nutrients and thus productivity, as the soil depth is limited due to the presence of a stone line at around 50 - 80 cm depth (Vogel, 1993). This suggests that the soils become shallower, have less organic matter and become relatively less permeable to air, water and roots. Although plant nutrients can be compensated by additions of fertiliser or manure, in rain-fed agriculture, plant available water cannot be ameliorated. The physical properties therefore, altered (e.g. water holding capacity) by soil erosion, are the most long term yield limiting factors (Lowery & Larson, 1995).

The most effective systems in reducing soil erosion were mulch ripping and tied ridging, which recorded 1.7 t/ha/yr and 2.2 t/ha/yr respectively. Under these treatments the soil losses were maintained at very low levels even during seasons with extremely high rainfall amounts and intensities. The observed conservation potentials of these two systems are through (a) the reduction of run-off and enhancement of rainfall infiltration; (b) minimum disturbance of the soil, meaning less aeration and thus reduced mineralisation of organic matter and maintenance of soil structure; (c) the soil protection provided by mulch which reduced the impact of rainfall energy and solar radiation. The negligible soil losses under mulch ripping and tied ridging, guarantee the maintenance of soil tilth, organic matter content, plant nutrients and thus soil productivity.

### 3.1.2 Total Nitrogen, Phosphorus and Potassium losses with sheet erosion

The amounts of nutrients lost with erosion were found to vary greatly with each nutrient, as affected by tillage system and amounts of soil lost. The nutrient losses were calculated using the following equation:

$$Nut_{los} = Soil_{los} \times Nut_{conc} \tag{4}$$

where $Nut_{los}$ = any nutrient lost with sediments (kg/ha)
$Soil_{los}$ = mass of soil lost by erosion (kg/ha)
$Nut_{conc}$ = the concentration of a nutrient in the sediment (ppm or %)

The nutrient status of the soils was determined before the assessment of nutrient losses through the three different ways. Table 2 shows the nutrient status of the soils and that the most abundant nutrient in the soil is potassium followed by nitrogen and the least abundant is phosphorus. It is also clear that these soils are inherently infertile. As total nutrients are considered, it is expected that the nutrient with the highest concentration in the soil will also result in the highest losses and vice versa. Thus, comparing the amount of different nutrients lost with the sediments may not be very meaningful but a method of evaluating and comparing the loss of different nutrients should also be based relatively upon the status of that nutrient in the soil. This method involves the determination of nutrient concentration in the soil and in the sediments and calculating the enrichment ratios.

| Treatment | Nutrient status of the soil | | |
|---|---|---|---|
| | N % | P ppm | K ppm |
| BF | 0.04 | 39.4 | 554.2 |
| CT | 0.05 | 52.0 | 616.7 |
| MR | 0.05 | 62.2 | 575.0 |
| TR | 0.05 | 91.8 | 487.5 |

Table 2. Nutrient content of the soils as at beginning of the study at Makoholi Contill site

Using equation 4 to calculate the amount of N lost with erosion, the highest total nitrogen losses were realized under bare fallow, at 28 kg/ha followed by conventional tillage (16 kg/ha), while they were least under mulch ripping (2.3 kg/ha), which was also barely different from tied ridging (2.7 kg/ha), see Table 3. Total nitrogen loss differed significantly

(P < 0.001) between the different treatments, different years and for the treatment * year interaction. These results follow the same trend that was established for soil loss (Table 1) and serve to confirm the dependence of nutrient losses on the amount of soil lost from a field. The maintenance of soil under the two conservation tillage treatments is also directly related to the lower N losses. Although nitrogen losses were highest under the bare fallow, the actual nutrient concentration in the soil was least under this treatment because no fertilizers were applied and the sediments under this treatment comprised mainly the non-reactive coarse particles.

The overall phosphorus loss (of 0.5 kg/ha) was much lower than nitrogen loss (12.3 kg/ha), due to the generally low P status in the sandy soils. The bare fallow had the highest P loss of 0.9 kg/ha followed by conventional tillage with 0.8 kg/ha, tied ridging 0.2 kg/ha and the least P losses were recorded under mulch ripping (0.09 kg/ha) (Table 3). This trend was to be expected, as nutrient losses are a function of soil loss. Despite the low losses, the treatments and years were significantly different at P < 0.001. The two conservation tillage treatments were not significantly different from one another. Potassium was lost in greater quantities when compared to the other elements (overall 17.3 kg/ha). It has been highlighted that K is the most abundant element in the soils' mineralogy (Table 3), and this explains the high losses.

| Treatment | N (kg/ha) | P (kg/ha) | K (kg/ha) |
|---|---|---|---|
| CT | 15.81 | 0.750 | 24.5 |
| MR | 2.25 | 0.091 | 0.6 |
| TR | 2.70 | 0.169 | 4.3 |
| BF | 28.42 | 0.861 | 39.8 |
| Signif. level | *** | *** | *** |
| s.e.d. | 1.341 | 0.0667 | 5.49 |

Table 3. Total N, P and K losses (kg/ha) as a result of erosion under different tillage systems over three years at Makoholi Contill site

The same trend that was established for N and P was also found for K, where more K was lost with bare fallow (40 kg/ha) and conventional tillage (25 kg/ha) as compared to the conservation tillage systems (0.6 and 4 kg/ha for mulch ripping and tied ridging respectively), see Table 3. The overall treatment differences were significant at P < 0.001 mainly due to significantly different soil losses between the treatments. The different years also gave rise to different K losses, which were significant at P < 0.001. These differences show the conservation merits of mulch ripping and tied ridging, implying that potassium is also conserved effectively through the ability of these treatments in reducing erosion.

Soil erosion is a selective process that renders the soil of its fine particles –clay and organic matter- leaving less productive coarse sand and gravel behind. Moyo (1998) found that the sediments contained between 2 and 4 times more clay and between 5 and 7 times more organic matter than the original soils. The affinity of the nutrients to the fine soil particles is well known and documented. The exchange sites on the clay minerals and organic matter are the basis for this affinity, as nutrients are held at these exchange sites and organic matter is also crucial in the cycle of P and N (Brady, 1984; Stevenson, 1985, Singer and Munns 1987). Tiessen, Cuevas and Salcedo (1998) and Stocking (1984) also reported that soil organic matter provided plant nutrients in low-input agriculture and that N and P release depended on the mineralisation of organic matter. Brady (1984) reported that organic matter was the major indigenous source of N while 65% of total P in the soil was found in the form of organic compounds. Clay more than organic matter, is the main source of fixed K and other cations and their losses are therefore associated with clay loss. Due to the selective nature of sheet erosion, high affinity of P to adsorption, fixation of K and ammonium ions, as well as the presence of Ca and Mg ions in clay minerals, erosion is the main source of nutrient and productivity loss in agricultural lands. This is why the loss of top soil is detrimental to any soils' productivity as there is a close association between clay, organic and the plant nutrients. The proximity and concentration of organic matter near the soil surface and close association with plant nutrients, make the erosion of soil organic matter and clay a strong indicator of overall plant nutrients resulting from erosion (Follet *et al.*, 1987).

There is evidence that a substantial amount of nutrients is lost with erosion, as shown by the overall averages of 12.3 kg/ha N; 0.5 kg/ha P and 17.3 kg/ha K. The amount of nutrient lost was found to be strongly dependent on the nutrient status of the soil, i.e. the higher the status of a particular nutrient in the soil, the higher its loss with erosion. The nutrient status of the soils showed the following trend K > N > P and the overall nutrient loss with erosion also showed exactly the same trend. This explains why soils with higher fertility status lose much more nutrients relative to those with a lower fertility status (Stoorvogel and Smaling, 1990). According to Rose *et al.* (1988), the amount of a nutrient lost with erosion is dependent upon the soil type, tillage practice and the type of erosion. From this study it was found that the amount of soil loss and the sediment fraction were important in determining the amount of nutrient loss (Table 4), especially on these sandy soils, where the amount of clay and organic matter are critical as sources of plant nutrients.

| Treat /Year | Element | Element kg/1t SL | Standard error | % variance accounted for | P value | Correlation SL:Element |
|---|---|---|---|---|---|---|
| Pooled | N | 0.360 | 0.019700 | 94.5 | *** | 0.980 |
| Pooled | P | 0.010 | 0.002090 | 38.3 | *** | 0.719 |
| Pooled | K | 0.767 | 0.104000 | 80.0 | *** | 0.908 |
| Treat /Year | Element | Element kg/1t Susp. | Standard error | % variance accounted for | P value | Correlation Susp:Element |
| Pooled | N | 1.589 | 0.0416 | 95.4 | *** | 0.977 |
| Pooled | P | 0.058 | 0.00722 | 40.6 | *** | 0.654 |
| Pooled | K | 4.201 | 0.271 | 86.5 | *** | 0.932 |
| Treat /Year | Element | Element kg/1t Sludge | Standard error | % variance accounted for | P value | Correlation Sludge:Element |
| Pooled | N | 0.186 | 0.0137 | 76.5 | *** | 0.879 |
| Pooled | P | 0.005 | 0.000302 | 80.0 | *** | 0.904 |
| Pooled | K | 0.390 | 0.0198 | 92.1 | *** | 0.960 |

SL = Soil loss; Susp = suspended material; Treat = Treatment;

Table 4. Nutrient loss as affected by soil loss, sludge and suspended material at Makoholi Contill site

Regression analysis was carried out to relate nutrient loss to the amount of soil lost for the different tillage systems. Firstly, a general regression analysis was carried out, where all the data collected was pooled, without specifying the treatments or the years and soil loss, suspended material and sludge were considered independently (Table 4). Data was then split according to the different treatments (disregarding years) and again the different elements were regressed with soil loss. From the regression output, each element was then calculated in relation to a tonne of lost soil. Correlation coefficients were also worked out for the relationship between each element and soil loss (Table 4 and 5). Pooling the data gave moderate nutrient losses for every tonne of soil lost. All the nutrients were below 1 kg for every 1 tonne of soil lost when total sediments were considered and ranged from 0.01 for P to 0.7 kg for K. The amounts of the nutrient losses were somewhat related to the losses under bare fallow but these amounts would under estimate the losses under cropped treatments. Generally for the pooled estimates, K was the most abundant element in the sediments and the sequence could be summed up as follows: K > N > P. The variance accounted for in the estimates was also very high for N and K and low for P.

The sediment fraction also influenced the amount of nutrients per unit of soil loss, with more nutrients lost with suspended than with coarse material. Table 4 shows that an average of 1.589 kg N was lost with one tonne of suspended material compared to 0.186 kg N lost with one tonne of sludge (8.5 times less), 12 times more P was lost with one tonne of suspended material than with sludge, while K loss was 11 times more in suspended material than in sludge. This finding further consolidates the fact that much more nutrients are lost with suspended material regardless of tillage treatment and plant element. The loss of coarse soil particles should have implications on soil productivity mainly due to the reduction of soil tilth and not soil fertility. The different treatments also showed that the conservation tillage treatments lost more nutrients per unit soil loss than conventional tillage systems (Table 5), due to the low sludge: suspension ratio in the former. For the same reason, conventional tillage also lost more nutrients (all elements) per tonne of soil loss than the bare fallow. Between the two conservation tillage treatments, more nutrients were lost under tied ridging than under mulch ripping, though the differences were not significant.

The type of soil determines first and foremost the status of a particular nutrient in the soil, with the sandy soils having lower nutrient contents than the clay soils (Stoorvogel and Smaling, 1990). This therefore, means that for the same amount of eroded soil, the clay soils are bound to lose more nutrients than the sandy soils. One should not overlook the fact that sandy soils have higher enrichment ratios of clay and organic matter contents and thus higher nutrient enrichment ratios than clay soils, although nutrients lost on sandy soils may be less. The nutrient losses with erosion are closely associated with the rate of soil loss Elwell and Stocking (1988), Kejela (1991) and Zoebisch *et al.,* (1995). Due to the fact that plant nutrients sorbed to the soil are transported with eroding sediments, the amount of soil lost with erosion becomes very important in determining the amount of nutrients lost. The conservation tillage systems dramatically reduced losses of soil and total nutrients when compared to conventional tillage systems, however the nutrient concentrations per unit soil loss are higher than for conventional tillage systems. Furthermore the more extensive loss of the topsoil under the conventional tillage systems results in fertility loss as the topsoil is rich in nutrients due to the high amount of the nutrient reserves like clay and organic matter (Troeh *et al.*, 1980; Follet *et al.*, 1987; Tanaka, 1995). The conservation of clay and organic matter under conservation tillage, therefore implies nutrient conservation (Barisas *et al.*, 1978; Tiessen, Cuevas and Salcedo, 1998). The concentration of nutrients in the sediments was much higher under the conservation tillage systems as compared to conventional tillage, obviously as a result of a high percentage of fine particles in the sediments compared to the later and thus the high affinity of nutrients to fine soil particles (Barisas *et al.* 1978). However, the advantage of low amount of sediments in conservation tillage also resulted in lower average losses under this system. By conserving the soil, nutrients are conserved and soil fertility sustained. It should be emphasized, however, that the loss of organic matter and clay and resultant physical degradation of the soil, also leads to poor tilth, low available water holding capacity and high bulk density (Munodawafa, 2007).

| Treat. | Element | Element kg/1t SL | Standard error | % variance accounted for | P value | Correlation SL:Element |
|---|---|---|---|---|---|---|
| BF | N | 0.305 | 0.030000 | 70.9 | *** | 0.842 |
| BF | P | 0.008 | 0.001270 | 29.9 | *** | 0.614 |
| BF | K | 0.700 | 0.105000 | 72.0 | *** | 0.958 |
| CT | N | 0.434 | 0.044200 | 54.2 | *** | 0.891 |
| CT | P | 0.017 | 0.005070 | very low | ** | 0.339 |
| CT | K | 1.199 | 0.073400 | 95.1 | *** | 0.977 |
| MR | N | 1.242 | 0.041400 | 98.7 | *** | 0.994 |
| MR | P | 0.028 | 0.002420 | 89.5 | *** | 0.966 |
| MR | K | 4.600 | 0.659000 | 80.1 | *** | 0.951 |
| TR | N | 1.437 | 0.150000 | 79.0 | *** | 0.900 |
| TR | P | 0.059 | 0.016700 | 11.7 | * | 0.496 |
| TR | K | 5.155 | 0.359000 | 95.7 | *** | 0.981 |

Table 5. The relationship between nutrient loss and soil loss under different tillage systems at Makoholi Contill site

### 3.1.3 Nutrient enrichment ratios

In this study the nutrient enrichment ratio is defined as the ratio of the nutrient concentration in the soil to the nutrient concentration in the sediments. Overall the enrichment ratios for the different nutrients were not very different from one another (Table 6). These were as follows: N: 4.3; P: 3.8 and K: 4.2. Although the amount of P lost with erosion was only a fraction of N and K amounts, it is clear that relative to the amount of P in the soil, all nutrients were lost in near equal proportions. The highest enrichment ratios were recorded under the conservation tillage systems, where the ratios ranged between 6.0 (P) and 7.3 (K). Under conventional tillage the sediments were enriched as follows: 2.0 for N, 1.9 for P and K. The bare fallow recorded the least nutrient enrichment ratios of about 1.0 for all nutrients except P, which recorded a ratio of 2.7. The difference in enrichment ratios was only recorded for the different tillage systems and not for the plant nutrients, as these showed a similar trend within these tillage systems.

| Treatment | Nutrient concentration in the sediments | | | Enrichment ratios nutrient in soil: nutrient in sediments | | |
|---|---|---|---|---|---|---|
| | N % | P ppm | K ppm | N | P | K |
| Year 1 | | | | | | |
| CT | 0.05 | 39.8 | 803.9 | 1.3 | 1.0 | 1.5 |
| MR | 0.07 | 104.6 | 1351.1 | 1.4 | 2.0 | 2.2 |
| TR | 0.41 | 570.9 | 5397.7 | 8.2 | 9.2 | 9.39 |
| BF | 0.28 | 447.6 | 5110.6 | 5.7 | 4.9 | 10.5 |
| Year 2 | | | | | | |
| CT | 0.03 | 14.4 | 318.7 | 1.0 | 0.8 | 0.6 |
| MR | 0.12 | 61.9 | 961.5 | 3.0 | 2.5 | 1.6 |
| TR | 0.40 | 156.6 | 1875.0 | 8.0 | 5.2 | 4.0 |
| BF | 0.25 | 104.8 | 1813.5 | 5.06 | 2.5 | 4.0 |
| Year 3 | | | | | | |
| CT | 0.05 | 15.0 | - | 1.7 | 0.9 | - |
| MR | 0.06 | 33.4 | - | 1.5 | 1.2 | - |
| TR | 0.22 | 124.4 | - | 4.3 | 4.8 | - |
| BF | 0.52 | 326.1 | - | 10.3 | 10.3 | - |

Table 6. Nutrient concentrations in the sediments and enrichment ratios for different tillage systems at Makoholi Contill site

### 3.2 Nutrient losses as a result of leaching

For all the nutrients, treatment differences were first analysed as influenced by year and as influenced by month. The first option at times led to no significant differences between the treatments although the years were significantly different from each other. The rain months on the other hand often resulted in significant differences in nutrient losses implying that seasonal rainfall tended to mask the influence of rainfall on nutrient loss as a result of leaching.

#### 3.2.1 Drainage water

Before quantifying nutrient losses as a result of leaching, it is important to first of all examine the medium which transports these nutrients and that is drainage water. As nutrient losses with erosion were dependent on the amount of run-off and soil loss, it is expected that nutrient losses with leaching will be dependent on drainage. During the three years of data collection, no drainage was experienced during the months of October (beginning of the season) and March/April (end of the season). Drainage was collected as from November, with one or more lysimeters recording some drainage and continued up to February. The average drainage recorded over three seasons showed that for cropped treatments, more drainage was recorded under the conservation tillage treatments (78 mm under mulch ripping and 77 mm under tied ridging), when compared to conventional tillage with 64 mm (Figure 2). However, the bare fallows generally recorded more drainage than the cropped treatments due to crop transpiration under cropped treatments. Since three types of bare fallow were set up (under conventional tillage, mulch ripping and tied ridging), the data further showed that even under bare fallow, conservation tillage resulted in more drainage (136 mm under MR and 129 mm under TR) than conventional tillage (95 mm). This is a result of the little run-off, previously highlighted, under conservation tillage due to enhanced infiltration and therefore drainage.

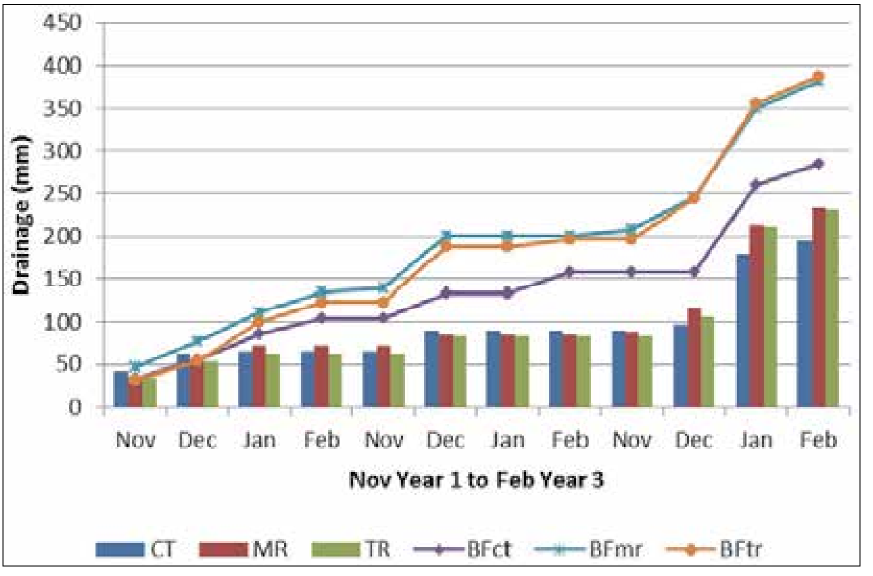


Fig. 2. Cumulative drainage from lysimeters under three treatments (cropped and bare) at Makoholi Contill site, over three seasons

The rainfall amount received during the effective 12 months of recording (November, December, January and February for three seasons) varied substantially among the different months, which differed significantly at $P < 0.001$. Drainage was highest (> 98 mm) in January of the third year, when the highest rainfall (419 mm) was recorded. Analysis of variance gave a significant difference ($P < 0.001$) between the different treatments, with conventional tillage recording the least drainage. There was however, no significant difference between the two conservation tillage treatments (cropped). The bare fallow under conventional tillage also differed significantly from bare fallow under conservation tillage, while there was also no significant difference between the bare fallows under conservation tillage. The different years differed significantly ($P < 0.001$) from one another, due to the different amounts of rainfall recorded during those years. Drainage was lowest when rainfall was also lowest and the highest drainage was recorded during the wettest season. A significant interaction between rainfall and treatment was found ($P < 0.009$), showing that different rainfall regimes resulted in significantly different drainage amounts from the different treatments. There was no interaction however, between treatment and year. This means that when the treatments are compared based on total seasonal rainfall, they show the same trend during the different years. The higher drainage under the bare plots compared to cropped treatments was obviously a result of transpiration by the crops, as Marshall (1967) reported that water in the soil was subject to removal by drainage, evaporation and transpiration by plants. As can be seen from these results, the conservation tillage systems recorded higher drainage than conventional tillage systems. This is a result of higher infiltration rates under mulch ripping and tied ridging, as evidenced by low run-off. The high run-off under conventional tillage translated into low drainage (Xu, Prato and Ma, 1995).

### 3.2.2 Nitrogen, Potassium and Phosphorus losses with leachate

The overall mean of nitrogen loss over the three seasons was highest under the three bare fallows, followed by cropped conventional tillage and was lowest under the cropped conservation tillage systems. Nitrogen losses among the different months ranged from 4.1 to 10.0 kg/ha. The month with the lowest mean nitrogen loss was November of the third year, which surprisingly did not have the lowest mean rainfall. Also the highest overall mean N loss was recorded in December of the second year, which had the second highest rainfall amount. This finding shows that nitrogen loss may not be directly dependent on drainage amount but also on the soil $NO_3^-$ concentration at that time. The differences between the treatments were highly significant at $P < 0.001$, with the bare fallows recording almost the same amount of N (5.7 for CT; 5.3 for MR and 5.7 kg/ha/yr. for tied ridging). For the cropped treatments however, conventional tillage lost more N (3.1 kg/ha) than the conservation tillage systems (MR with 1.26 and TR with 2.77 kg/ha/yr.). The rain months also gave rise to significantly different nitrogen losses ($P < 0.001$). Furthermore, the significant interaction between treatment and rain that was found, serves to highlight that different treatments responded differently to monthly rainfall, as can be seen from Figure 3.
Very little amounts of phosphorus were lost with drainage, with a grand monthly mean of 4.1 g/ha. P lost over the three years ranged from 0.93 g/ha to 10.58 g/ha (Figure 4). Under the cropped treatments more P was lost under tied ridging (an average of 17 g/ha) followed by conventional tillage (14 g/ha) and least under mulch ripping (13 g/ha), however the differences were not significant.

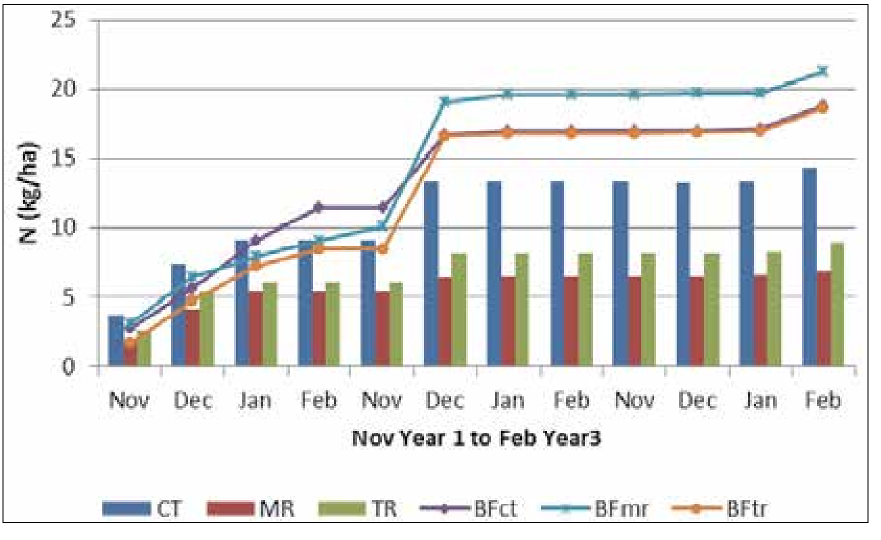


Fig. 3. Cumulative nitrogen loss with drainage water under three tillage systems (cropped and bare) at Makoholi Contill site, over three seasons

The bare treatments generally recorded more P loss than the cropped treatments, however, the trend was reversed as mulch ripping recorded highest loss (37 g/ha) followed by tied ridging (18 g/ha) and the least amount was lost under conventional tillage 15 g/ha). The higher P loss under mulch ripping may be due to high organic matter content and returned crop residues, thus also shifting the equilibrium towards soluble P, however the amounts and differences recorded are also small that they have no significant effects. Despite the little amounts of P lost, the treatment differences for the bare fallows were significant at $P < 0.001$, while the cropped treatments did not differ significantly from one another. The three years and the rain months each gave significantly different P losses at $P < 0.001$, with the wettest month recording the highest P loss. The different treatments reacted differently to rainfall amount received, thus resulting in highly significant interaction ($P < 0.001$) between treatment and rainfall.

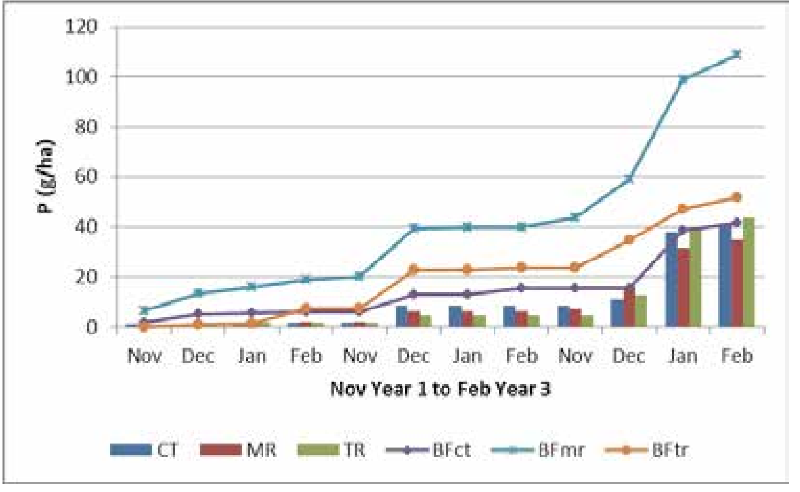


Fig. 4. Cumulative phosphorus loss with drainage water under three tillage systems (cropped and bare) at Makoholi Contill site, over three seasons

More potassium was lost with leachate as compared to nitrogen, a grand monthly mean of 2.1 kg/ha was recorded for potassium (Figure 5) compared to 1.1 kg/ha found for nitrogen. Among the cropped treatments more potassium was lost under conventional tillage (9 kg/ha) than under the two conservation tillage treatments (5 kg/ha under MR and 3 kg/ha/yr. under TR), which also differed significantly from each other, with mulch ripping losing more potassium than tied ridging.

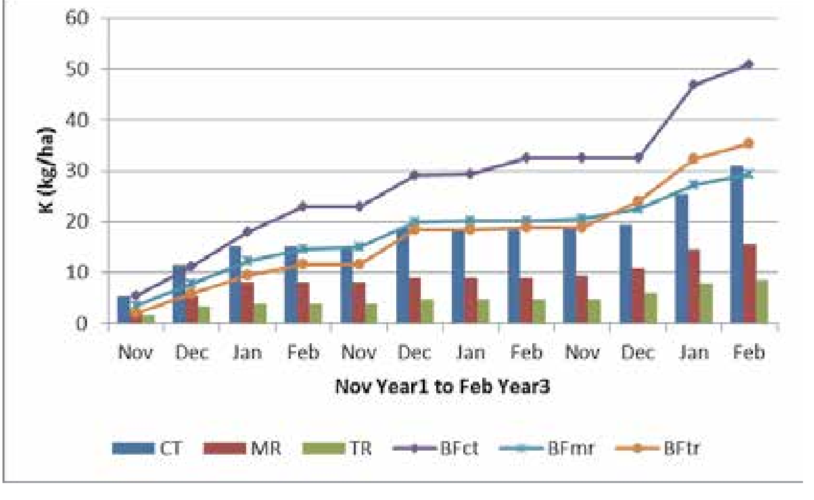


Fig. 5. Potassium loss with drainage water under three tillage systems (cropped and bare) at Makoholi Contill site, over three seasons

Due to lack of water and nutrient uptake by crops under the bare treatments, more drainage was realised and the nutrients that would otherwise be taken up by the crop were leached, leading to generally higher nutrient losses. Again the bare conventional tillage recorded the highest K loss (17 kg/ha) compared to uncropped mulch ripping (10 kg/ha) and tied ridging (12 kg/ha), which did not differ significantly from each other. The month with the highest rainfall amount also recorded the highest K loss and vice versa. It is clear therefore, that K loss was to a large extent influenced by the amount of rainfall received or drainage. Potassium loss differed significantly among the different treatments at $P < 0.001$, with the conservation tillage systems recording less K than conventional tillage. The different years differed significantly at $P = 0.003$, while the rain months were more significantly different at $P < 0.001$. The interactions between year and treatment and month and treatment were not significant.

Nutrient losses with leachate were generally highest under the bare plots, where there was no nutrient uptake by the crop. As nutrient losses are dependent on the nutrient concentration in the soil profile (Kolenbrander, 1981), this means that the concentration of nutrients in the soil solution was kept relatively high under the bare plots, thus leading to higher leaching losses. Under the cropped treatments, conventional tillage lost more nutrient than the conservation tillage systems. The higher drainage under the conservation tillage systems did not result, as was anticipated, in higher losses of nutrients. This is attributed to the improved soil structure, under the conservation tillage systems, leading to water percolation through macro-pores and thus lower nutrient losses (Kolenbrander, 1981). N and K presented the highest losses as they are mobile in the soil and their high concentrations make these elements susceptible to leaching (Stoovogel and Smaling, 1990; Drury *et al.*, 1993). N loss was not significantly dependent on the rainfall amount, showing that there are other factors that influence N leaching other than rainfall. Kolenbrander (1981) found that the amount of N lost through leaching depended on the nitrate concentration in the soil profile, while Stoorvogel and Smaling (1990) reported that the leaching of nutrients depended on rainfall amount and fertility of the soil. Rainfall intensity also affects the nutrient losses with leachate as according to Havis and Alberts (1993) low rainfall intensities produce higher leachate but lower release rates than high intensities. The contact time between the soil particles and thus nutrient desorption is lower during high intensity rainfall and higher during low intensity rainfall, thus affecting the concentration of nutrients in the leachate. There was a highly significant difference between the treatments, where the cropped treatments lost less N than bare plots and conservation tillage systems also lost less than conventional tillage. This finding also confirms the improved soil structure under the conservation tillage systems, which allows water to percolate through macro-pores or inter aggregate pore space (Follet *et al.*, 1987), thus reducing nutrient loss under conservation tillage.

Conventional tillage recorded higher K losses (9 kg/ha) compared to mulch ripping (5 kg/ha) and tied ridging (3 kg/ha). This, again, has a bearing on the improved soil structure under the conservation tillage treatments. The higher amount of K loss under MR compared to TR may be the result of K accumulation under this treatment, as according to Stoorvogel and Smaling (1990) the maize crop has a high amount of K in the stover than in the grain and according to Drury *et al.*, (1993), precipitation leaches nutrients from decomposing plant matter. P was lost in very small amounts (< 50 g/ha/yr.) as it is not mobile in the soil. The very low P status in the soil together with its high affinity for fixation in acidic tropical soils

(Stoorvogel and Smaling, 1990), were other reasons why P leaching was very low, as P in the soil solution is kept very low. Nitrogen and potassium thus presented the highest losses with grand seasonal means of 0.86 kg/ha and 2.02 kg/ha respectively, while negligible losses (< 50 g/ha) were recorded for phosphorus: K > N > P. This may be due to the abundance of labile nitrogen (nitrate) as a result of high fertiliser application as compared to the other elements and the abundance of potassium in the soils. However it is also important to consider that $K^+$ and $NO_3^-$ ions are very mobile in the soil. Stoorvogel and Smaling (1990) reported that K and N were vulnerable to leaching, while P was tightly bound by the soil. This explains the negligible loss of phosphorus as it is generally immobile in the soil (Brady, 1984; Singer and Munns, 1987).

### 3.3 Maize grain yield and nutrient losses due to plant uptake

Generally yield is a function of many parameters including climate, soil productivity and management. In this study the climate parameter, rainfall and its effect on the quality of the season are discussed. It is assumed that the quality of the season should influence crop emergence, establishment, growth and yield. An optimal season should have no major mid-season droughts (the period during the growing season, when rainfall amount is less than half of the potential evapotranspiration, $P \leq 0.5$ PET) (FAO & Agritex, 1992). During this period there is definite moisture deficit and crop growth is negatively affected.

During Year 1 and Year 3, no mid-season droughts were experienced, although the typical dry spells (periods where none of the three consecutive five-day periods (15 days) have rainfall exceeding 20 mm) occurred. In Year 2, two mid-season droughts and a series of dry spells were experienced following low rainfall and poor rainfall distribution. Crop emergence was dependent on the amount and distribution of initial rainfall. Generally when rainfall distribution was good, good crop emergence was realised. Due to the higher evaporative losses and resultant moisture stress under tied ridging (Moyo and Hagmann, 1994), crop emergence was generally lower under this treatment than with the other treatments. Both emergence and plant population did not differ significantly between treatments. However, the different years gave rise to significant differences ($P < 0.05$) for both parameters, since the years were characterised by different rainfall (amount and distribution). Rainfall, rather than treatment is an important factor in determining emergence and plant population.

Although mulch ripping outperformed the other treatments during the drier year (2), overall, there were no significant treatment differences in yield ($P = 0.449$), see Table 7. Independent t-tests also showed no significant treatment differences. However, yields varied significantly during different years, ($P < 0.001$). As a result there was a significant interaction between year and treatment ($P < 0.05$), where conservation tillage treatments performed better when rainfall was less and conventional tillage was better when it was wet. Independent t-tests for the different years also confirmed this significant difference as the means of all treatments differed significantly at ($P < 0.001$). Rainfall, more than treatment, proved to be the most yield limiting factor in this region.

During Year 1, there were no significant differences in yield between the three treatments both within the group and independently. However, during Year 2, the treatments performed significantly different at $P < 0.01$ (Table 7). This season was characterised by low and poorly distributed rainfall, which brought about the advantages of mulch - reduction in evaporative losses and generally maintaining the soil moisture during the major dry spells.

Mulch ripping thus realised the highest yield. Conventional tillage had a significantly lower yield than the conservation tillage treatments ($P < 0.001$). The conservation and subsequent production merits of the conservation tillage treatments, especially mulch ripping, are apparent during drier seasons. Comparing the conservation treatments against each other showed that mulch ripping yielded almost twice as much as tied ridging resulting in a significant difference at $P < 0.05$. In Year 3 the overall treatment differences for yield were not significant. However, conventional tillage had the highest yield, which differed significantly from the mean of mulch ripping and tied ridging.

| Treat/Year (Rainfall) | Year 1 (483mm) | Year 2 (384mm) | Year 3 (765mm) | Overall mean (kg/ha) | Source of variation | Yield |
|---|---|---|---|---|---|---|
| CT | 2415 | 860 | 4642 | 2639 | Treat | NS |
| MR | 2623 | 2203 | 3923 | 2916 | Year | *** |
| TR | 2969 | 1132 | 3736 | 2612 | Treat x Year | * |
| Overall mean | 2669 | 1398 | 4100 | 2722 | MR vs TR | NS |
| n = 9 (Treatment) | | s.e.d. = 259.8 | $s^2 = 303730$ | | CT vs (MR, TR) | NS |
| n = 9 (Year) | | s.e.d. = 259.8 | df = 18 | | Year1 vs Year 2 | *** |
| n = 3 (Treatment x Year) | | s.e.d. = 450.0 | | | Year 3 vs (Years 1,2) | *** |

Table 7. Analysis of variance for yield (kg/ha) as affected by tillage and year (rainfall) and their interactions

Nutrient uptake of different elements was varied, with the highest uptake being that of nitrogen (grand mean of 44.4 kg/ha) and the least uptake was recorded for phosphorus, with a grand mean of 8.08 kg/ha. Potassium uptake was 33.7 kg/ha. While the uptake of nutrients is determined by the crop physiology, it is interesting to find out if the different treatments had any effect on the amount and ratio of the different elements, which were taken up. The uptake of nitrogen was higher than for the other nutrients, probably due to the high amount of N fertiliser that was applied. Conventional tillage recorded the highest mean N uptake of 50 kg/ha, while the uptake under mulch ripping and tied ridging was not significantly different as means of 40.6 and 42.6 kg/ha N were recorded respectively (Figure 6). N uptake under the different treatments was significantly different at $P < 0.05$. The different years influenced the N uptake resulting in significantly different values at $P < 0.001$. This was expected as nutrient uptake is a function of yield. Yield, which varied significantly among the different treatments, was found to be dependent on rainfall amount. More than six times less phosphorus was taken up as compared to N uptake, however, P uptake did not differ significantly among the three treatments. Grand means of ~ 8 kg/ha P uptake were recorded for all the treatments (Figure 6). Analysis of variance showed no significant difference among the treatments. During Year 3 significantly high P was taken up by the crop compared to Years 1 and 2, which did not differ significantly from each other. The overall P uptake as influenced by the year was significantly different at $P < 0.001$. This finding shows the dependence of nutrient uptake/yield on rainfall amount. K uptake varied only slightly among the different treatments and was generally higher under conservation tillage treatments compared to conventional tillage. This variation was, however, statistically not significant. The highest K uptake was recorded during Year 3, while Year 1 and Year 2 differed only slightly.

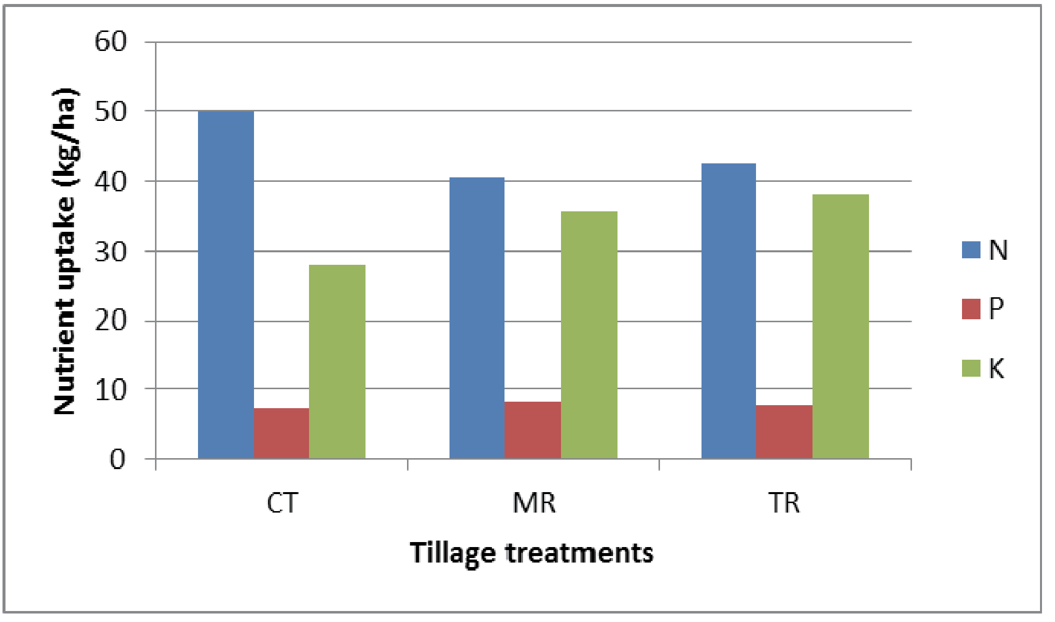


Fig. 6. Uptake of different nutrients by the maize crop on erosion plots, under three tillage systems at Makoholi Contill site (mean of three seasons)

Yield was found to be more dependent on seasonal rainfall than on tillage treatment. The treatment differences were minimal with mulch ripping recording an overall average of 2.9 t/ha; tied ridging and conventional tillage both at 2.6 t/ha. While conventional tillage recorded the highest yield during the wet year, the conservation tillage treatments had better yields during the drier seasons. This clearly highlights the moisture conservation potential of the conservation tillage treatments during drier years, while the higher yields under conventional tillage are only possible during wet years. There was a greater variation of yield under conventional tillage (0.9 - 4.6 t/ha) compared to the conservation treatments (2.2 - 3.9 t/ha) under mulch ripping and 1.1 to 3.7 t/ha under tied ridging. Mulch ripping was also found by Moyo and Hagmann (1994) to have the highest crop yield per mm of growth effective rainfall (water use efficiency). Tanaka (1995) reported that crop residue is not only the key to soil erosion control but also the key to ameliorating the eroded sites. It is expected that with high soil and runoff losses from conventional tillage further decline in productivity will be made apparent through reduction in the yield potential of this treatment. This is an indication that the two conservation tillage treatments can sustain yields better than the conventional tillage. The very high yields recorded under all the treatments during the wettest season confirm the direct positive relationship between yield and rainfall amount.

Nutrient uptake was dependent on yield, which also depended on rainfall amount. The element that was taken up most of all was N (44 kg/ha), followed by K (34 kg/ha), while only 8 kg/ha of P were taken up. Although the actual quantities of nutrients taken up by the crop varied from treatment to treatment, the differences were not statistically significant. Since nutrient uptake is directly dependent on yield, this was expected as yields did not differ significantly among the different treatments.

### 3.4 Comparison of nutrient losses due to erosion, leaching and plant uptake

Nutrient losses varied greatly depending on the tillage treatment, the element considered, the season in question and the type of loss that the element was subjected to (erosion, leaching or plant uptake). In general the highest losses were realised through plant uptake which is considered to be good since this is the only positive loss, when compared to the other two types of nutrient losses. There is, no doubt, a depletion in soil fertility, however, if only the harvested products are removed from the field and other crop residues are returned, in the absence of erosion and leaching, fertiliser requirements for the maintenance of soil fertility may be reduced. Erosion and leaching on the other hand are considered negative and should be minimised at all costs. Erosion losses were higher than those incurred with leaching for the conventional tillage, while the reverse was partly true for the conservation tillage systems.

A general overview of the nutrient losses showed that total nitrogen losses (through erosion, leaching and plant uptake) were least under mulch ripping with a range of 27 - 53 kg/ha/yr. followed by tied ridging (range of 25 - 64 kg/ha) and highest under conventional tillage (range of 44 - 87 kg/ha). The same trend was established for phosphorus loss although the amounts lost and the differences between the treatments were minimal (ranges of 7 - 10 kg/ha for CT; 6 - 8 kg/ha for MR and 6 - 10 kg/ha for TR). Potassium losses ranged from 29 - 92 kg/ha under CT; 30 - 43 kg/ha under MR and 22 - 57 kg/ha under TR. Table 8 shows the average losses of different elements found for the different treatments and years.

| Treatment | N | P | K |
|---|---|---|---|
| CT | 63.54 | 8.42 | 60.70 |
| MR | 38.21 | 7.41 | 36.48 |
| TR | 49.85 | 7.83 | 39.61 |
| Signif. level | * | NS | NS |
| Year | N | P | K |
| 1 | 52.0 | 7.40 | 64.20 |
| 2 | 31.6 | 6.77 | 27.00 |
| 3 | 68.0 | 9.49 | - |
| Signif. level | ** | * | *** |

- = missing data

Table 8. Average total nutrient losses under different tillage treatments during three seasons at Makoholi Contill site

Generally, more nutrients were lost under conventional tillage as compared to the conservation tillage system. Only N losses gave significant treatment differences at $P < 0.05$. However, when the mean of the conservation tillage systems was compared to conventional tillage, the systems were significantly different at $P < 0.01$. Although quantitatively more P was lost under conventional tillage than under the conservation tillage systems, the treatment differences proved not to be significant. Overall there were no significant treatment differences for K loss, however conventional tillage differed significantly from the mean of the two conservation tillage treatments. The different years affected the nutrient losses more than tillage, resulting in all the elements differing significantly between the years (Table 8).

After the total nutrient losses were calculated and assessed, each type of nutrient loss was evaluated relative to the total nutrient loss. Plant uptake contributed between 38 and 99% of total nutrient loss, erosion losses were between 0.1 and 51%, while leaching accounted for 0 to 16% of total nutrient loss. This is the reason why different treatments did not show significant differences overall because plant uptake was very high and masked the effects of the other forms of nutrient loss. However, this general finding shows clearly that after plant uptake, erosion is the main factor contributing to nutrient loss in agricultural arable lands.

The different nutrients as influenced by tillage were then evaluated and gave the following results:

Nitrogen: the losses were highest due to plant uptake (Figure7), however the percentage of N taken up under conventional tillage (56 - 84%) was lower than under mulch ripping (88 - 95%) and tied ridging (85 - 92%). This finding implies therefore, that the contribution of the losses by leaching and erosion were lower for the conservation tillage systems. Nitrogen losses as a result of erosion were thus highest under conventional tillage, contributing to more than a quarter of total N loss (range 16 - 29%). The conservation tillage systems on the other hand realised minimum losses of less than one percent and highest losses of 11% (MR) and 8% (TR). The leaching losses were in all cases lower than the erosion losses and the highest percentage leaching loss of 15% was recorded under conventional tillage, while mulch ripping had a maximum of 7% and tied ridging of 10% of total N loss.

Phosphorus: Plant uptake contributed to most of the P loss even under conventional tillage (Figure 7). A range of between 82 and 97% was found under conventional tillage, while the conservation tillage systems had higher percentages of between 97 - 99.8% (MR) and 96 - 99.8% (TR). Erosion losses were high under conventional tillage (between 3 and 18%), while these were maintained at below 5% under both MR and TR. The results also prove that P is not very susceptible to leaching as all the treatments recorded less than 1 % of total P loss.

Potassium: The crop uptake of potassium had a low percentage under conventional tillage, where it ranged from 38 - 77% of total K loss. This was as a result of very high losses of K due to erosion, ranging from 23 - 46%. K leaching was almost as high as N leaching under this treatment (0 - 16%). The percentage of K uptake under the conservation tillage treatments was relatively higher than under conventional tillage (85 - 96% MR; 79 - 95% TR). This means that the contribution of erosion and leaching was also less than under conventional tillage. K loss as a result of erosion under mulch ripping remained very low, < 1 - 2%, while it was higher under tied ridging, 1 - 14% of total K loss. The contribution of leaching to total K loss was higher when compared to that of N and P. Between 3 and 13% of total K loss was lost through leaching under mulch ripping and 4 - 6% under tied ridging.

Total nutrient losses were significantly lower under conservation tillage treatments than under conventional tillage. Although nutrient uptake did not differ significantly among the different treatments, erosion and leaching losses accounted for the differences. After plant uptake, erosion presents higher nutrient losses than leaching. While N and K are mobile in the soil and vulnerable to leaching (Stoorvogel and Smaling, 1990), the N and K losses attributed to this type of nutrient loss were lower than the amounts lost with erosion and plant uptake, except for leached K under mulch ripping, where the losses were higher than erosion losses. This may be due to the high amount of K in solution, leached from the decomposing stover (Havis and Alberts, 1993). There could be an over supply of K in the soil and the high uptake may be because K is available, “luxury uptake” rather than due to demand. P loss with leaching was lowest and this was expected as P is immobile in the soil and many researchers have highlighted the affinity of P to fixation in the soil. This study has shown that P is not prone to leaching, but is susceptible to erosion, while N and K are mobile in the soil and are very susceptible to leaching and erosion. Although plant uptake contributes the highest percentage of total nutrient loss, the low percentages of 38, 49 and 56% under conventional tillage are a cause for concern as this means that leaching and erosion sometimes contribute up to > 50% of total nutrient loss.

### 3.5 Soil nutrient balance

The main purpose of this section is to try and balance the nutrients lost from the field with added inputs. The total nutrients from the field have been quantified in the previous section (erosion, leaching and plant uptake) and the inputs have only been limited to applied fertilisers. The results of this nutrient balance for the different treatments and elements, calculated using Equation 3, have been summed up in Table 9.

| Tillage | IN1 kg/ha | OUT1 kg/ha | OUT2 kg/ha | OUT3 kg/ha | Balance kg/ha |
|---|---|---|---|---|---|
| Nitrogen | | | | | |
| CT | 51.5 | 44.63 | 15.81 | 3.10 | - 12.04 |
| MR | 51.5 | 34.70 | 2.25 | 1.26 | + 13.29 |
| TR | 51.5 | 44.38 | 2.70 | 2.77 | + 1.65 |
| Phosphorus | | | | | |
| CT | 12.0 | 7.65 | 0.75 | 0.01 | + 3.59 |
| MR | 12.0 | 7.30 | 0.09 | 0.01 | + 4.60 |
| TR | 12.0 | 7.65 | 0.17 | 0.02 | + 4.16 |
| Potassium | | | | | |
| CT | 12.0 | 34.05 | 24.50 | 9.31 | - 55.86 |
| MR | 12.0 | 35.18 | 0.60 | 4.58 | - 28.36 |
| TR | 12.0 | 40.67 | 4.30 | 2.99 | - 35.96 |

Where: IN1 = mineral fertilisers; OUT1 = plant uptake; OUT2 = soil erosion; OUT3 = leaching

Table 9. Nutrient balance assessment of the three tillage systems, average over three years at Makoholi Contill site

Under conventional tillage, there is evidence of soil mining (negative balance) for all elements except P. The conservation tillage systems only recorded negative nutrient budgets for K losses, due to very high losses as a result of crop uptake, which were generally very high to be compensated by the amount applied. Under conservation tillage soil mining was mainly as a result of crop harvests, while under conventional tillage substantial amount of nutrients were also lost as a result of soil erosion and leaching. Thus under this treatment, maintenance of soil fertility means to first of all replace nutrients lost with leaching and erosion before replacement of the amount taken up by the crop. The sum of N losses under conventional tillage is in excess of the amount of N applied annually (an equivalent of 125% of applied N) and some of the losses are directly from the soils' nutrient reserves (Table 10). Under mulch ripping an equivalent of 76% of applied N was also removed annually. The depletion rate was lower and was contained well below the annual N application rate. Soil fertility is thus better maintained under mulch ripping. An equivalent of 99% of applied N was lost under tied ridging, indicating that the rate of N applied under tied ridging is literally just adequate to offset the nutrient losses, however, great care has to be exercised to ensure that the losses do not exceed the fertiliser applied. P losses under all the treatments were maintained well below the equivalent amount of P applied, an average of 70% under CT; 62% under MR and 75% under TR (Table 10). This implies that part of the applied fertiliser was retained in the soil, showing the extreme fixation of P under these conditions. Such low levels of P fertilisation may not show any yield response due to the high rate of fixation. The crop also took up relatively low amounts of P when compared to the other elements. Potassium on the other hand showed that much, more K was lost when compared the amount applied (CT 505%; MR 304% and TR 330%), see the negative balance on Table 9.

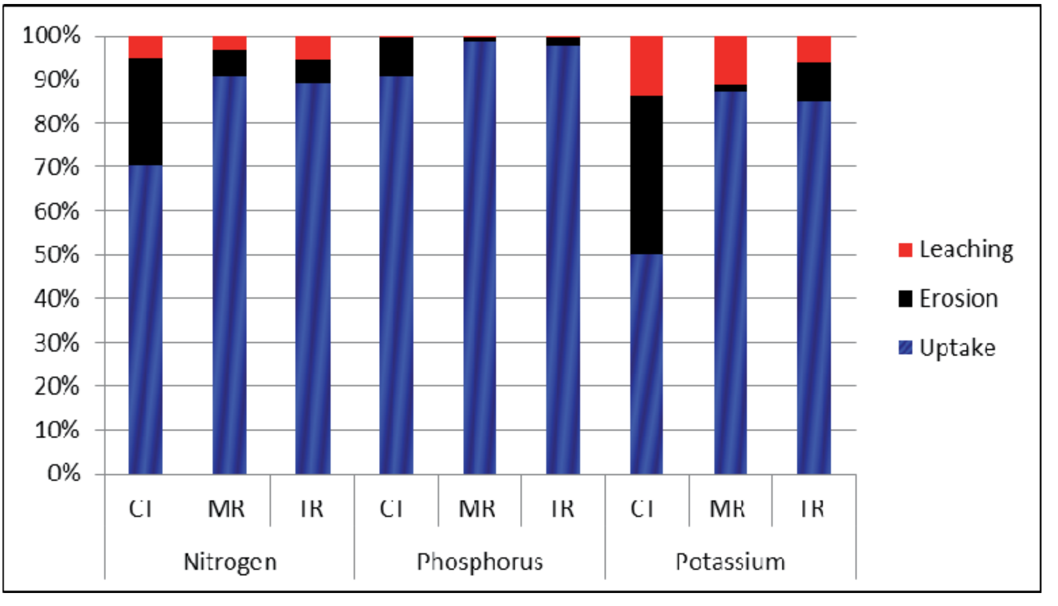


Fig. 7. Comparison of nutrient losses (N, P, K) through leaching, plant uptake and erosion under different tillage systems over three seasons at Makoholi Contill site

Nutrient budgets should be employed and fertiliser application should aim at replacing lost nutrients as well as meeting the crop requirements, for the following season. Conservation tillage is highly recommended as it reduces the danger of soil mining drastically, but the problem of soil mining may not be wholly solved through the introduction of soil and water conservation systems if inputs and outputs are not well balanced. While most of the nutrients may be conserved, even up to accumulation, other nutrients, especially those not applied as fertilisers may need replacement once in a while. Nitrogen is the only element that is significantly affected by the atmosphere, increasing its amount in the soil through biological fixation and natural deposition (Hach, 1979). Due to these factors, changes resulting from the N losses may not be easily quantifiable within a few seasons.

| Treatment | Elements | Year 1 | Year 2 | Year 3 | Average |
|---|---|---|---|---|---|
| | | Total loss % | Total loss % | Total loss % | % |
| CT | N | 119.80 | 86.32 | 171.37 | 125.8 |
| MR | N | 68.63 | 52.91 | 105.45 | 75.7 |
| TR | N | 120.34 | 48.59 | 127.23 | 98.7 |
| CT | P | 66.67 | 57.17 | 86.58 | 70.1 |
| MR | P | 53.58 | 62.50 | 69.08 | 61.7 |
| TR | P | 64.75 | 79.50 | 81.58 | 75.3 |
| CT | K | 768.67 | 243.00 | | 505.8 |
| MR | K | 358.75 | 249.17 | | 304.0 |
| TR | K | 477.50 | 182.58 | | 330.0 |

Table 10. Relationship between total nutrient loss and applied fertiliser over three seasons at Makoholi Contill site

Nutrient losses with plant uptake, erosion and leaching result in the depletion of nutrient status of the soil, i.e. "soil mining". Balancing the nutrients applied in a system with the outputs from the same system gave negative nutrient budgets, confirming the above hypothesis. What is important however, is that under conservation tillage, mainly the nutrients lost through plant uptake need to be replaced, while under conventional tillage, a significant amount of nutrients lost with erosion and leaching also need to be replaced. The maintenance of soil fertility under this system is more costly than under conservation tillage. Although the conservation tillage systems have proven without doubt that they conserve soil, water and nutrients, the nutrient budgets have also shown that it is of utmost importance to balance the inputs and the outputs. To maintain soil fertility and ensure sustained productivity of the soils, fertiliser rates have to address both the crop needs of the following season and replace the nutrients lost during the previous season.

## 4. Conclusions

The losses of nutrients with sheet erosion are primarily dependent on the content of each element in the soil, the amount of soil loss (determined by the type of tillage system) and the type of sediment fraction. Plant nutrient losses are higher when the nutrient status in the soil and/ or soil loss from a field is high. There is a very high association between nutrients and fine soil particles. This makes the suspended material the most detrimental sediment fraction with very high concentrations of nutrients. Drainage is dependent on rainfall and on tillage system. The bare fallows tend to have more drainage than cropped treatments, while the conservation tillage systems have more drainage than conventional tillage. Nutrient losses with leachate are highest under bare fallows compared to cropped treatments. Despite the higher drainage, the conservation tillage systems however, have lower nutrient losses compared to conventional tillage, due mainly to improved soil structure. N and K are mobile in the soil and are prone to leaching, while P is immobile and thus not susceptible to leaching. N and K are mobile in the soil and are prone to leaching, while P is immobile and thus not susceptible to leaching. Nutrient uptake by the crop is dependent on yield and nutrients are taken up in the following order: N > K > P. More nutrients are lost with plant uptake, followed by erosion and then leaching. For the conventional tillage system erosion and leaching contribute to a substantial amount of total loss, while under the conservation tillage systems most of the nutrients are taken up by the crop and a negligible amount is attributable to erosion and leaching losses. The nutrient balance of inputs versus outputs is negative for N and K under conventional tillage, while under conservation tillage systems, negative nutrient budgets are confined to K, which is subject to higher losses due to its high concentration in the soil. The optimal applications are adequate to offset all nutrient losses under conservation tillage treatments, while under conventional tillage these are inadequate and indicate soil mining. The conservation tillage treatments significantly reduce nutrient losses from agricultural lands due to their ability to reduce soil erosion and leaching.

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# 10

# Effect of Slope Position and Land-Use Changes to Bio- Physical Soil Properties in Nakasongola Pastoral Rangeland Areas, Central Uganda

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## 1. Introduction

Land degradation, which includes degradation of vegetation cover, soil degradation and nutrient depletion, is a major ecological problem in Uganda. It is estimated that fertile top soil is lost at a rate of one billion cubic meters per year, resulting in massive environmental degradation and constituting a serious threat to sustainable agriculture and forestry (Yost and Eswaran, 1990).

Forests and the benefits they provide in the form of wood, food, income, and watershed protection have an important and critical role in enabling people to secure a stable and adequate food supply. Deforestation and land degradation, however, are impairing the capacity of forests and the land to contribute to food security, and to provide other benefits, such as fuel wood and fodder in Uganda (Siriri *et al*., 2000). Ugandan environmental issues are causing a great concern because the expansion rate of such major problems as drought, desertification, water pollution, is reaching an alarming stage (NEMA, 2001).

In Nakasongola district, the presence of steep slopes subject to cultivation since many years, has led to serious soil erosion. The lag in agricultural productivity advancement behind population growth has caused intense land use conflicts, particularly between the agricultural and the forestry sectors. To compensate for the low agricultural productivity, deforestation for arable land expansion has been the principal land use conversion employed in Uganda and in particular in Nakasongola for centuries. There are several repercussions of such land use conversion, the most important in Uganda's and in particular in Nakasongola context being accelerated soil erosion and deterioration of soil nutrient status (FAO, 1986). However, Nakasongola is known not only for the severity of land degradation, but also, since the last decades, for the concentrated efforts taking place to redress these problems including construction of stone terraces and soil bunds, protected areas and afforestation (Osiru and Hahn, 1994).

The protected areas, which are a type of land management implemented on degraded, generally open access land, are a mechanism for environmental rehabilitation with a clear biophysical impact on large parts of the formerly degraded commons. In places where protected areas are established, particularly in the western part of the country, they constitute green spots with considerable species diversity. The ability of rehabilitation areas to recruit and sustain new life forms is a true measure of their contribution to biodiversity and forest resource conservation (Yost and Eswaran, 1990; Tucker and Murphy, 1997).

In protected areas, it is generally believed that the land resources such as soil, wild flora and fauna, or water will be protected from degradation. Although the restoration ecology and buffering effect of protected areas have been well studied, there are relatively few studies in the region, which would provide a measure of the success or failure of protected areas as one strategy to help prevent decline of soil fertility and adverse effect of water erosion, thereby increasing agricultural productivity. Above all there are no quantitative studies that analyse the effectiveness of protected areas in improving soil chemical and physical properties.

Therefore, the objectives of this study were to asses the impact of protected areas on soil properties, and investigate the relationship between age of protected areas and their effectiveness in improving soil chemical and physical properties.

## 2. Materials and methods

### 2.1 Study area

Nakasongola District is located on the Bombo - Gulu highway 114 Km north of Kampala. It covers an area of 3,424 km$^2$ of land and boarders with Apac district in the northeast, Mukono district in the east, Masindi in the west and Luwero district in the south (Figure 1) . The area lies on the central plateau between 1000 and 1400 m above sea level. The topography is characterized by extensive uniform undulating plains with broad seasonal swamps.

The soils are mainly weathered basement complex formations of the precambrian age, which consists mainly of metamorphic and igneous rocks, largely composed of gneisses and granites. Remnants of the older mid-tertiary surface are found as relic murram and iron stones in some places. The annual rainfall varies from 875 – 1120 mm with two marked dry seasons. The main vegetation types are woodland and woodland savanna, thicket and soft wood plantations. Much of the cultivated land exists as patches with in the woodland.

The district has a total population of 528,126 people, the population density is about 230 persons per km$^2$, and the growth rate is 2.7 %. There are three ethic groups; the Baganda (70%) and Baruli (28) and others (2%). Subsistence agriculture is the major economic activity employing about 89% of population.

Rocks outcropping in the study area belong to the Mesozoic sedimentary series and Tertiary basalt flows. Soils of the specific study sites are developed on lime rich parent material. Using the FAO-UNESCO soil classification system, they were classified as calcaric Cambisols. In the study area, the erosion rate is extremely serious and it is common to observe ground surfaces that have been incised strongly by rill and gully erosion.

The study area has a semi arid continental climate with an average annual rainfall of 615 mm/yr with extreme values of 290 and 900 mm (Figure 2). The rainy season starts in June, peaks in July and August and trails off in September. The study sites have 53-104 rain days per year with mean of 80 days. According to the agro-climatic classification system, which is traditionally used in Uganda, all the study sites are classified as mid altitude.

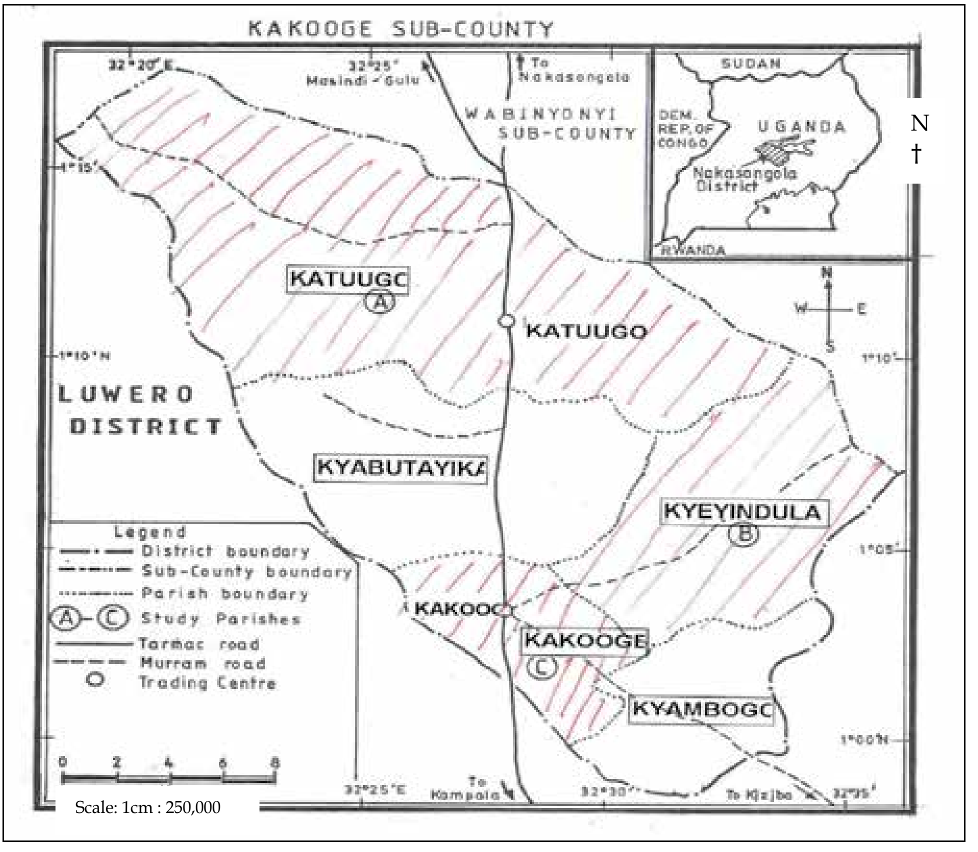


Fig. 1. Location of the study district and specific study site at Kakooge sub-county

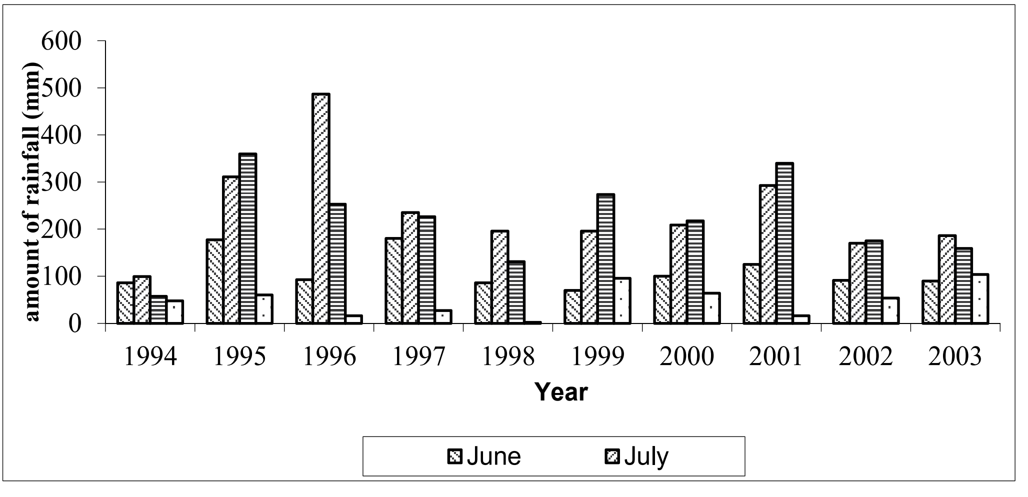


Fig. 2. Monthly rainfall of the study sites (source: Uganda Meteorological Division, Ministry of Water, Lands and Environment, Kampala, Uganda).

The typical land use in the study area is range land and protected areas on the steep slopes and crop land in the flats. The vegetation of the study area is largely dominated by *Acacia etbiaca*, and *Euclea shimperi*. Under-storey vegetation of the study sites is dominated by a very diverse assemblage of grass and herbs most of which are palatable for livestock. Crops are mainly cassava, maize, and sweet potatoes. Crop yields are low (500kg/ha on average), with great variability depending on rainfall and the location of the cultivated lands. According to the district's Agricultural office, the average land holding is 0.46-0.76 ha per household.
Within the Nakasongola district, twelve hillslopes ranging from 3.5 to 48.5 ha were selected based on similarities in lithology, soils, climate and land use. Out of the twelve sites selected, six are protected areas and the remaining six are free grazing lands. To examine the influence of age since protection five and ten year's old protected areas each having three replicates were selected for the study. Features of the specific study sites are included in table 1.

| Land use | Slope class | Year of closing | Area (ha) | Slope Steepness (%) | Slope Length (m) | Vegetal canopy | | Ground vegetation | | Estimated soil loss t/ha/year (USLE)[1] |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | Type | Cover (%) | Type | Cover (%) | |
| 10 AC | US | 1994/95 | 20.3 | 24 | 125 | BU (2m) | 43 | TWSBG | 58 | 16.3 |
| | MS | | 40.8 | 19 | 274 | BU (2m) | 31 | TWSBG | 60 | 6.8 |
| | FS | | 34.5 | 10 | 290 | BU (2m) | 32 | TWSBG | 52 | 2.6 |
| 5 AC | US | 1999/00 | 6.8 | 37 | 80 | BU(.5 m) | 28 | GTWSB | 23 | 98.5 |
| | MS | | 6.8 | 33 | 73 | BU(.5 m) | 30 | GTWSB | 38 | 68.7 |
| | FS | | 5.4 | 18 | 86 | BU(.5 m) | 29 | GTWSB | 38 | 27.7 |
| $C_1$ | US | -- | 12.7 | 22 | 147 | SBU | 4 | TWSB | 5 | 57.4 |
| | MS | | 10.2 | 25 | 81 | SBU | 5 | TWSB | 5 | 80.2 |
| | FS | | 6.5 | 11 | 111 | SBU | 10 | TWSB | 9 | 27.9 |
| $C_2$ | US | -- | 10.7 | 27 | 82 | SBU | 6 | TWSB | 6 | 121.5 |
| | MS | | 7.7 | 29 | 64 | SBU | 7 | TWSB | 11 | 96.2 |
| | FS | | 7.3 | 13 | 80 | SBU | 4 | TWSB | 6 | 25.0 |

10AC, 5 AC: 10 and 5 years old protected areas respectively. $C_1$ and $C_2$ are free grazing lands used as a control for 10 and 5 years protected areas. US, MS, and FS are Upper, Middle and Foot slope. TWSBG, GTWSB, and TWSB; tall weeds and short bushes with grasses, grasses with tall weeds and short bushes and tall weeds with short bushes respectively. BU (2m) = bushes of 2 meter effective height; Bu (0.5) = bushes of 0.5 meter effective height and SBU= short bushes.
[1] = predicted through application of Universal Soil Loss Equation (USLE);.

Table 1. Average characteristics (n = 3) of the specific study sites

### 2.2 Experimental design

Five and ten year's old protected areas and free grazing lands were selected in the middle altitude agro-ecological zone (mid latitude) for this study. Each of the sites was replicated three times over the study area. Each of the study sites (protected area and free grazing lands) was divided into three slope positions, upper slope (US), middle slope (MS) and foot slope (FS). The US position is the uppermost portion of each study site. It receives little or no overland flow but may contribute runoff to downslope areas. The MS position receives overland flow from the upperslope and contributes runoff to the FS. The FS represents the lower part of each study site.

### 2.3 Soil sampling and laboratory analysis

Within each position in protected areas and free grazing land, transects were laid out and eight soil sampling points along transect were selected using systematic random sampling. Samples of 0-15 cm soil depth were taken from the eight points of each transect. The eight samples were combined in a large bucket and mixed thoroughly to form a composite soil sample; 36 composite samples were collected from all the sites. Major live plant materials (roots and shoots) in each sample were separated by hand and discarded. The soil samples were air dried, grinded and passed through a 2 mm sieve before the determination of soil nutrients. Standard soil test procedures for observation and analysis in Kyambogo University Soil Science Laboratory were used for all nutrients and physical soil parameters. Organic matter was determined by Walkley Black method and total Nitrogen was determined by Kjeldahl method (Bremmer and Mulvany, 1982). Available Phosphorus was determined by Olsen method (Olsen and Sommers, 1982). Ammonium and Sodium acetate methods were used to determine exchangeable bases and cation exchange capacity (Osiru and Hahn, 1994). Calcium and Magnesium values were then red using Atomic Absorption Spectrophotometer while Potassium and Sodium were determined by flame photometer. The pH and EC of the soil samples were determined by pH and conductivity meter using supernatant suspension of 1:5 soil water ratios. Particle size analyses of the sampled soils were determined in soil suspension by Hydrometer method (Gee and Bauder, 1982). Bulk density was determined by core method through measurement of volume and mass (Blake and Hartge, 1986). Total porosity was determined experimentally by the water displacement method and calculation using bulk density values.

Moreover, the Universal Soil Loss Equation (USLE) was used (Wischmeier, 1976, Table 1) for the purpose of characterizing the severity of erosion in the study sites and generates supplemental data (Yost and Eswaran, 1990). Individual interviews using structured questionnaires were also conducted to understand farmers' perception on the role of protected areas.

### 2.4 Statistical analysis

Statistical analyses were performed to test the influence of land use conversion and age of protected areas on soil chemical and physical properties using one-way analysis of variance (ANOVA). Mean comparisons were made using the Tukey Honest Significant Difference (HSD) test with $p < 0.05$. Two - way ANOVA was also used to see whether there is an interaction effect between the two main independent variables: age and slope gradient. The independent variables used in this study were land use, slope gradient and age of protected areas. All the analyses were conducted through Statistica 6.0 program.

## 3. Results

### 3.1 Soil chemical properties and land uses

The soils of free grazing lands and protected areas differed considerably in content of soil organic matter (SOM), total Nitrogen (TN), available Phosphorus (AP) and exchangeable bases and cation exchange capacity (CEC) (Table 2), with significantly higher ($p < 0.05$) values found in protected areas than in free grazing lands. Two - way analysis of variance also revealed that age of protected areas was highly significant ($p < 0.01$). Slope effect was not significant. There was a tendency of significant interaction effect ($p = 0.07$) indicating that the influence of slope changes with age.

| | Soil nutrients | | | | | | |
|---|---|---|---|---|---|---|---|
| Land use & slope position | SOM (%) | TN (%) | AP (ppm) | K cmol(+)/kg | Ca cmol(+)/kg | Mg cmol(+)/kg | CEC cmol(+)/kg |
| 10 AC US | 2.4$^{ab}$ | 0.27$^{ab}$ | 4.5$^{ab}$ | 0.32$^{ab}$ | 9.2$^{ab}$ | 0.4$^{a}$ | 52.2$^{ac}$ |
| 10 AC MS | 2.8$^{ab}$ | 0.34$^{ab}$ | 5.8$^{ab}$ | 0.34$^{ab}$ | 5.7$^{a}$ | 0.5$^{a}$ | 54.5$^{a}$ |
| 10 AC FS | 3.0$^{a}$ | 0.34$^{ab}$ | 6.0$^{ab}$ | 0.36$^{ab}$ | 8.7$^{ab}$ | 0.8$^{ab}$ | 55.5$^{a}$ |
| 5 AC US | 2.8$^{ab}$ | 0.35$^{ab}$ | 4.6$^{ab}$ | 0.59$^{ab}$ | 11.0$^{ab}$ | 1.8$^{b}$ | 56.4$^{a}$ |
| 5 AC MS | 3.1$^{a}$ | 0.37$^{ab}$ | 5.1$^{ab}$ | 0.63$^{ac}$ | 11.0$^{ab}$ | 0.8$^{ab}$ | 56.6$^{a}$ |
| 5 AC FS | 3.2$^{a}$ | 0.57$^{a}$ | 6.8$^{a}$ | 0.70$^{ac}$ | 5.1$^{a}$ | 0.4$^{a}$ | 58.0$^{a}$ |
| $C_1$ US | 1.5$^{b}$ | 0.20$^{b}$ | 3.0$^{b}$ | 0.22$^{b}$ | 15.4$^{ab}$ | 1.3$^{ab}$ | 39.6$^{b}$ |
| $C_1$ MS | 1.5$^{b}$ | 0.21$^{b}$ | 3.7$^{ab}$ | 0.23$^{b}$ | 17.6$^{ab}$ | 1.4$^{ab}$ | 40.9$^{b}$ |
| $C_1$ FS | 1.9$^{ab}$ | 0.24$^{b}$ | 4.1$^{ab}$ | 0.22$^{b}$ | 27.0$^{b}$ | 1.4$^{ab}$ | 41.0$^{b}$ |
| $C_2$ US | 1.8$^{ab}$ | 0.23$^{b}$ | 3.7$^{ab}$ | 0.76$^{c}$ | 2.7$^{a}$ | 0.6$^{a}$ | 48.3$^{dc}$ |
| $C_2$ MS | 1.8$^{ab}$ | 0.26$^{ab}$ | 2.8$^{b}$ | 0.70$^{c}$ | 18.0$^{ab}$ | 0.5$^{a}$ | 45.5$^{db}$ |
| $C_2$ FS | 1.4$^{b}$ | 0.25$^{ab}$ | 3.6$^{ab}$ | 0.55$^{bc}$ | 6.4$^{a}$ | 0.2$^{a}$ | 45.4$^{db}$ |
| F value | 5.398$^{**}$ | 2.506$^{*}$ | 2.473$^{*}$ | 7.633$^{**}$ | 3.488$^{**}$ | 1.733$^{*}$ | 9.015$^{**}$ |

Different letters along the column indicate significant differences between mean values of the different land use types at $p < 0.05$ (Tuckey HSD).

$^{*}$, $^{**}$ significant at 0.05 and 0.01 level of significance respectively.

SOM - soil organic matter; TN -total Nitrogen; AP – available Phosphorus; and CEC – cation exchange capacity.

5 AC – five years protected areas; 10 AC – ten years protected areas; $C_1$ – free grazing land used as a control for 10 years protected areas and $C_2$ – free grazing land used as a control for 5 years protected areas. US, MS and FS are upper, middle and foot slopes respectively.

Table 2. Average soil nutrient content (n = 3) of 0-15 cm soil depth for the land use types and slope position

The mean SOM content varied between 1.4 and 3.2 % (Table 2). Multiple comparison of SOM revealed that SOM levels under five and ten years protected areas were significantly higher than in $C_1$ and $C_2$. The mean CEC of the sampled soils varied between 39.6 and 58.0 cmol (+)/kg and had a similar pattern to SOM. Mean TN content varied between 0.20 and 0.57 %. Comparison of means revealed that the protected areas had significantly higher total Nitrogen content than $C_1$ and $C_2$. Mean AP content varied between 2.8 and 6.8 ppm and displayed similar pattern to TN. The mean value of exchangeable K, varied between 0.22

and 0.76, Ca between 2.7 and 27 and Mg between 0.2 and 1.8 cmol(+)/kg-soil. Except Ca, the level of exchangeable bases in the sampled soils showed the same trend as the other soil parameters for multiple comparisons.

The result of pH tests revealed that 75 % of the sampled sites had moderately alkaline soils (pH between 7.4 and 8.4) and 25 % of the sampled sites had neutral soils (pH 6.6-7.3). The level of soluble salts in the sample sites was also low, that is less than 1 mmhos/cm. Analysis of variance shows that there were no a significant differences between protected areas and free grazing lands (at $p < 0.05$) for pH and EC values.

In protected areas, an increasing trend was observed in soil organic matter, total Nitrogen, available Phosphorus and exchangeable Potassium from US to FS position (Figure 3). However, such trend was not clearly seen in free grazing lands.

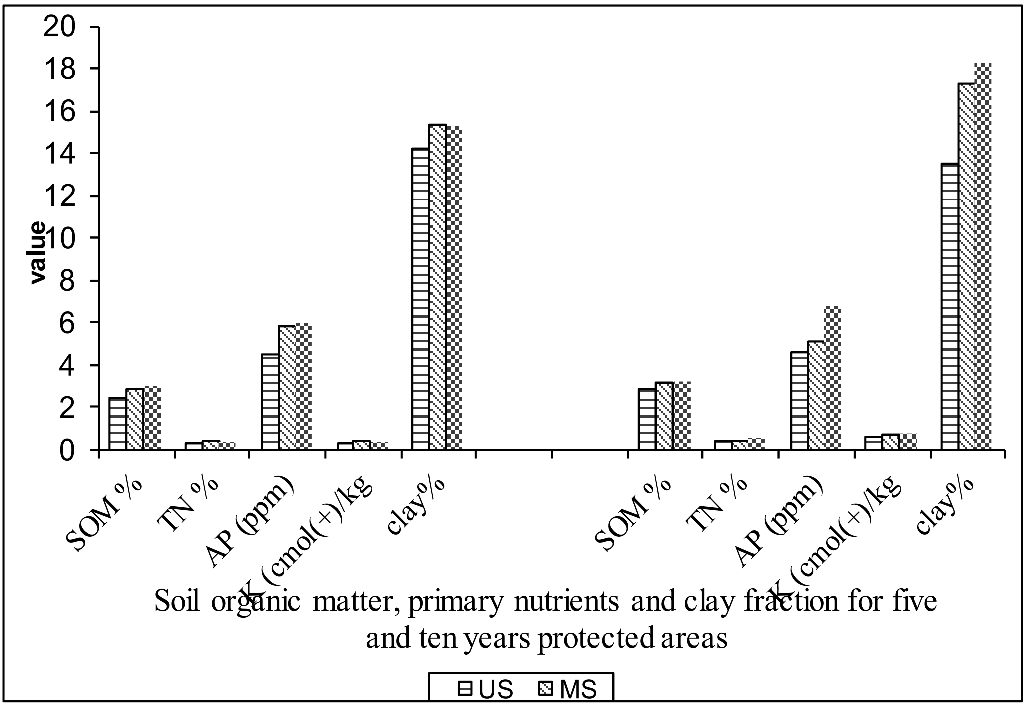


Fig. 3. Soil organic matter, primary soil nutrients and clay fraction in relation to slope positions in protected areas.

### 3.2 Soil nutrient stocks and land uses

In this study, total amount of nutrients stored at 0-15 cm soil depth were determined using bulk density values of each slope position of each site (Table 3). There were considerable differences in total amount of nutrients stored in the upper 15 cm soil between protected areas and free grazing lands. The mean value of SOM stock of ten year protected areas was higher by 40%, TN by 31% and AP by 40% than its control $C_1$. Mean value of SOM, TN, and AP stock of five year protected areas was higher by 44%, 39%, and 40% than its control $C_2$. However, the nutrient stocks in five and ten years protected areas were nearly similar.

| Land use and slope position | Soil nutrient stocks (kg/ha) | | | | | |
|---|---|---|---|---|---|---|
| | SOM | TN | AP | K | Ca | Mg |
| 10 AC US | 45810 | 4938 | 8.2 | 223 | 3286 | 84 |
| 10 AC MS | 50598 | 6097 | 10.7 | 256 | 2126 | 110 |
| 10 AC FS | 54119 | 6073 | 11.7 | 237 | 3175 | 157 |
| Mean for the land use type | **50176** | **5703** | **10.2** | **239** | **2863** | **117** |
| 5 AC US | 53548 | 6658 | 8.7 | 446 | 3743 | 377 |
| 5 AC MS | 52289 | 7204 | 9.0 | 411 | 4176 | 167 |
| 5 AC FS | 62037 | 9255 | 12.0 | 512 | 1985 | 90 |
| Mean for the land use type | **55958** | **7705** | **10** | **456** | **3301** | **211** |
| $C_1$ US | 29211 | 4714 | 4.4 | 177 | 6062 | 305 |
| $C_1$ MS | 27607 | 3880 | 6.3 | 172 | 6715 | 316 |
| $C_1$ FS | 36671 | 4024 | 7.7 | 167 | 10474 | 322 |
| Mean for the land use type | **31163** | **4206** | **6.1** | **172** | **7750** | **314** |
| $C_2$ US | 35544 | 4588 | 6.0 | 598 | 1084 | 149 |
| $C_2$ MS | 30446 | 4621 | 5.0 | 498 | 6578 | 108 |
| $C_2$ FS | 27125 | 4829 | 7.0 | 416 | 2302 | 57 |
| Mean for the land use type | **31038** | **4679** | **6** | **504** | **3321** | **105** |

See table 2 for abbreviations

Table 3. Average (n = 3) soil nutrient stocks of 0-15cm soil depth for the land uses and slope positions

### 3.3 Soil nutrient correlations

Some values found in this study were significantly correlated with each other (Table 4). Soil organic matter was correlated positively and significantly with total Nitrogen (TN), AP, and CEC (at $p < 0.05$).

| | SOM | TN | AP | CEC | Ca | K |
|---|---|---|---|---|---|---|
| TN | 0.54* | | | | | |
| AP | 0.36* | 0.45* | | | | |
| CEC | 0.55* | 0.55* | 0.56* | | | |
| Ca | -0.31 | -0.28 | -0.21 | -0.52* | | |
| K | 0.21 | 0.30 | 0.18 | 0.36* | -0.21 | |
| Mg | 0.12 | -0.11 | -0.18 | -0.32 | 0.51* | -0.10 |

Numbers are correlation coefficients (r). N = 36
* significant at 0.05 level of significance

Table 4. Correlation between soil nutrients

### 3.4 Land use and soil physical properties

The mean values of soil bulk density (BD) at a depth of 0-15 cm varied between 1.11 and 1.34 g/cm$^3$ (Table 5). The mean values of total porosity varied between 49 and 58 %.
Analysis of variance shows that there was no significant difference between land uses (at $p< 0.05$) on their BD and porosity values. Texture of the 0-15 cm soil depth was sandy loam at all sampled sites (Table 5).

## 4. Discussion

### 4.1 Effects of protected areas

The higher SOM, TN and AP contents in protected areas compared to those from free grazing lands can be explained by the difference in soil erosion and biomass return. Reduced erosion is expected to occur in well developed protected areas because the canopy formed by the mature shrubs and under-story vegetation shields the soil from the erosive energy of the falling raindrops and thereby protects it from splash erosion and surface or sheet erosion. The lowest soil loss (8.5 ton/ha /yr) calculated using USLE was found in ten years old protected areas and is mainly due to the relatively high vegetation canopy cover ( 31-43%) by 2 m effective height bushes, and ground vegetation cover of 52 – 60% (Table 1). In contrast, the higher soil loss (81 ton/ha /yr) predicted for free grazing land is the result of very low vegetation cover (4 - 7%) and ground vegetation cover of less than 10% (Table 1).
Our result is in agreement with other studies. The study conducted by Jiang *et al.*, (1996), showed that soil loss decreases exponentially with increasing degree of cover by mulch. Siriri and Bekunda (2001) revealed that the median rate of soil erosion in polygons of different land uses decreased in the following pattern: bare soil > open canopy forest > pasture > protected canopy forest. A report by Mullar-Harvey *et al.*, (1985), indicates that under a tropical monsoon climate, the establishment of forest on

eroded slopes reduced annual soil erosion from about 15000 to 3000 $m^3$/ $km^2$ over a period of 10 years.

| Land use types | slope position | Bulk density ($gm/cm^3$) | Porosity (%) | Particle size distribution (%) | | | Textural class |
|---|---|---|---|---|---|---|---|
| | | | | Sand | Silt | Clay | |
| 5 AC | Upper slope | 1.11 | 58.02 | 57.9 | 28.6 | 13.5 | SL |
| | Middle slope | 1.26 | 52.33 | 53.1 | 29.6 | 17.3 | SL |
| | Foot slope | 1.28 | 51.51 | 52.1 | 29.6 | 18.3 | SL |
| | **Average value** | **1.22** | **53.96** | | | | |
| 10 AC | Upper slope | 1.19 | 54.75 | 61.23 | 24.5 | 14.22 | SL |
| | Middle slope | 1.28 | 51.62 | 62.3 | 22.4 | 15.3 | SL |
| | Foot slope | 1.20 | 54.4 | 59.9 | 24.8 | 15.3 | SL |
| | **Average value** | **1.23** | **53.60** | | | | |
| $C_1$ | Upper slope | 1.34 | 49.4 | 62.3 | 18.68 | 19.02 | SL |
| | Middle slope | 1.26 | 52.33 | 62.83 | 21.34 | 15.82 | SL |
| | Foot slope | 1.28 | 51.68 | 62.3 | 21.34 | 16.35 | SL |
| | **Average value** | **1.29** | **51.14** | | | | |
| $C_2$ | Upper slope | 1.34 | 49.29 | 59.64 | 27.9 | 12.44 | SL |
| | Middle slope | 1.20 | 54.5 | 61.24 | 25.24 | 13.51 | SL |
| | Foot slope | 1.26 | 52.25 | 58.57 | 27.38 | 14.04 | SL |
| | **Average value** | **1.27** | **52.02** | | | | |
| | F value | $0.933^{ns}$ | $0.936^{ns}$ | | | | |

SL = Sandy loam; ns = non significant (at $p < 0.05$, Tukey HSD).

Table 5. Average (n = 3) soil physical properties of 0-15cm soil depth of the land uses and slope positions

The higher SOM in protected areas also improves the soil physical properties such as soil structure and total soil porosity. This in turn increases the amount of water infiltrated into the soil and decreases the amount of runoff that can be generated from a given amount of rainfall. Water infiltration in the soil is enhanced by both preferential flow along trees roots and accumulation of absorbent humus on the soil surface, thereby significantly reducing the volume, velocity, and erosive and leaching capacity of surface runoff (Jiang *et al*., 1996). However, land use for free grazing land is related to soil management practices that have commonly been very destructive to the soil and have caused serious erosion. Therefore, differences in soil erosion control contribute to the significant difference in nutrients in protected areas and free grazing land soils.

In addition, less biomass return causes the reduction of SOM, TN and AP in free grazing lands (Mullar - Harvey *et al*., 1985; Shariff *et al*., 1994). The most evident impact of grazing on the rangeland ecosystem is removal of a major part of above ground biomass by livestock. Therefore the input of aboveground litter to the soil decreases. Any reduction in litter inputs may have important consequences for soil nutrient conservation and cycling (Shariff *et al*., 1994). Grazing may also have an indirect effect on soil characteristics through change in plant species composition. This is mainly because plant species has a significant impact on decomposition and nutrient cycling at ecosystem levels.

The similar pattern displayed among SOM, TN and AP for multiple comparison of means can be explained by the influence of soil organic matter in nutrient retention and supply (Brubaker *et al*., 1993). Among soil properties, total organic carbon is a sensitive soil quality indicator suggesting that within a narrow range of soil, it may serve as a suitable indicator of soil quality (Mullar-Harvey *et al*., 1985). Moreover, the soil organic matter fraction may offer further insight into soil fertility changes and the sustainability of management history.

The insignificant differences in soil bulk density and porosity among protected areas and free grazing lands can be explained by the coarse texture nature of the study sites and low amount of preexisting soil moisture. Preexisting soil texture and moisture conditions are important variables to consider when investigating the relationship between soil compaction and grazing intensity. The effect of grazing intensity in increasing bulk density is pronounced in wet and fine textured soils. The slight difference found in this study can be explained by their difference in SOM content and compaction of the sites by livestock trampling effect.

The role of soil organic matter in storing and supplying nutrients explains the significant correlation between SOM and soil nutrients such as TN and AP and CEC. Despite the fact that one of the effects of organic matter is to retain cations and protect them from leaching and from removal by runoff, organic matter is inversely correlated with Calcium. This is mainly due to the effect of soil erosion and parent material. In free grazing areas, where soil erosion is more severe than in protected areas, most of the top soil is removed and the soils that were sampled partly included subsoils, highly influenced by lime rich parent material.

The positive impact of protected areas on the amount and availability of soil nutrients has many influences on the livelihood of the local people near the study sites. The amount of grass produced in a given area increases and the local people get supplemental animal feed and thatching material. The regeneration of indigenous trees is also improved by the increased availability of water and nutrients. Besides, farmers benefit from the protective effect of protected areas. This is mainly because protected areas reduce runon and soil erosion in farm lands located below the protected areas. This in turn leads to an increase in crop production (Siriri *et al*., 2000).

### 4.2 Effect of slope position

Because the different slope positions have similarities in vegetal canopy and ground vegetation cover (table 1), the variation in soil nutrients at different slope positions in protected areas can only be explained through the difference in soil erosion rates. Results from this study showed that upper slope positions in protected areas had higher rate of soil erosion. This is mainly due to the steepness of the US positions compared to MS and FS positions. Nutrients in the upper slope positions can be dissolved and washed by runoff and might be deposited in the middle and foot slope positions.

However, slope position effect in free grazing lands was inconsistent. This can be explained by differences in vegetation removal, related to accessibility by livestock and human beings; depending on the position of villages nearby. This difference creates a difference in C input which influences SOM content, and thereby related soil nutrients such as total Nitrogen and available Phosphorus. The amount of predicted soil loss was not also consistent like in protected areas where it showed a decreasing trend from US to FS position. Hence, there is no clear trend among the slope positions in free grazing lands.

## 5. Conclusions

This study assessed the effects of land-use conversion and slope position on soil chemical and physical properties. Significant differences (at $p < 0.05$) between land uses were found for SOM, TN and AP and exchangeable bases and CEC. Five and ten years' protected areas had higher levels for SOM, TN, AP and CEC compared to free grazing lands. The difference in soil nutrients content between the land use types is mainly due to differences in soil erosion and biomass return.

From the technical point of view, under the present land use management and climate conditions of the study area, free grazing areas in hilly lands should be changed to protected areas before soil organic matter and other nutrient contents are depleted more. Besides, the erosion processes may be very active resulting in further degradation. However, the socio-economic dimensions of protecting free grazing lands and changing them into protected areas (for example, impact on livestock and crop production in short run) should also be considered before making a decision.

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# Part 4

## Erosion Control

# 11

# Pastoral Hill Slope Erosion in New Zealand and the Role of Poplar and Willow Trees in Its Reduction

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## 1. Introduction

### 1.1 The geology of New Zealand and its contribution to erosion

New Zealand is a predominantly hilly and mountainous country. An area of 18 million hectares (69% of the country) has slopes greater than 12°, and is commonly called 'hill country'. This is further divided into 'hill-land' (12–28°) and 'steepland' (if slope exceeds 28°) (DSIR 1980). The range of slope and elevation, coupled with a wide latitudinal range, a mid-oceanic setting encompassing subtropical to cool temperate climates, and complex geologic and tectonic regimes means that New Zealand's 'hill country' is physically diverse. As a consequence of this diversity, the productive potential of New Zealand's' hill country, and its response to climatic events, land use pressure and environmental change varies significantly across the country.

Erosion is a significant environmental issue facing agricultural and forestry land uses in large parts of the hill country of New Zealand. It causes both on-site (loss of soil productive capacity and water holding capacity) and off-site (declining water quality, river aggradation, increased vulnerability of infrastructure to severe climatic events) effects. As well as the environmental costs there are major financial risks and social consequences, at every organisational level, of repeated erosion and flooding events.

Of the 6.3 million hectares in the North Island, the majority is developed on soft rock and crushed soft rock terrain in the south-east and west of the island (2 825 000 ha, 45%). Volcanic ash and loess-mantled terrain comprises another 23% (1 456 000 ha), largely on the periphery of the Central Volcanic Zone. Hard rock hill country, exclusive of the igneous hard rock hill country, is largely concentrated on the margins of the axial ranges (919 000 ha, 14.5%), whereas the hill country developed on deeply weathered sedimentary and igneous rocks (863 000 ha, 13.6%) is located predominantly in Northland and on the Coromandel Peninsula. In the North Island approximately 200 000 ha has a mapped erosion severity of severe, very severe or extreme. The North Island hill country is dominated by mass movement erosion. Shallow landslide (soil slip) and sheet erosion are the most widely distributed followed by earthflow and gully erosion. Sediment yield is highest in the northern East Coast Region, and high throughout much of the rest of the East Coast, inland Taranaki and parts of Northland. Mean sediment yields are highest in the crushed soft rock hill country and hilly steeplands, and steep soft rock hill country (Basher et al., 2008).

In contrast to the North Island, 56% of the South Island's 3.7 million hectares of hill country is developed on hard rock terrain. Only 569 000 ha is developed on soft rocks. Steep weathered hill country is associated with both schist and greywacke. In the South Island less than 103 000 ha has an erosion severity ranking of severe, very severe or extreme. Unlike the North Island, the South Island is dominated by surface erosion types. Sheet and soil slip erosion are the most common forms of erosion in the South Island hillcountry. Tunnel gullying is far more common in the South Island while gully and earthflow erosion are far less common. Sediment yields tend to be far lower in the South Island and are highest on the West Coast because of the higher rainfall. Mean sediment yields are highest in the steep soft rock hill country (Basher et al. 2008).

### 1.2 Erosion processes on Hill country

New Zealand is widely recognised as having high rates of both natural and man-induced erosion. The extent and type of natural erosion is determined by the complex interplay between geology (rock type, weathering, structure, and regional plate tectonics), climate (particularly rainfall amounts and intensities, frequency of large storms) and vegetation, producing regional variation in the susceptibility of the land to erosion. Severe earthflow and gully erosion are closely related to the extensive areas of mudstone and crushed rock terrain in the North Island, whereas in the South Island gully erosion is largely associated with hill country developed on easily eroded soft mudstone, sandstone, weakly consolidated conglomerate, and regional fault and crush zones. Annual rainfall is the dominant factor influencing rates of erosion, as measured by suspended sediment yields, and numerous studies point to the importance of large storms as triggers for widespread landsliding (Griffiths 1981, 1982; Hicks et al. 1996).

Imposed on these natural drivers of erosion is the effect of historical deforestation and land use which has increased erosion to a greater or lesser degree depending on the lands' inherent susceptibility to erosion. Numerous studies demonstrate the linkage between increased sedimentation rates, European settlement and deforestation (e.g. Page et al. 1994a, b). Deforestation can lead to at least a 10-fold increase in erosion both in the long term and during large storms, while short-term sediment yield increases of up to 100 times after forest harvesting have been recorded (e.g. Fahey et al. 1993). Much of the increase occurs when the tree roots lose their strength, about 2–5 years after forest removal. However, the impact of deforestation persists in many deforested areas with thick regolith or soft rocks which continue to erode faster than similar forested areas for at least a century (Page et al. 2000).

Landslides are the best studied erosion process, are very widespread, and a major contributor to hillslope erosion and sediment yield. Crozier (2005) concludes that most common landslide events are triggered by storm rainfall with critical intensities governed by the prevailing antecedent moisture conditions, rainfall duration and amount. Since the onset of European deforestation, increased sediment production over much of New Zealand has largely been determined by landslides (Glade 2003). On unstable slopes, thousands of landslides can be triggered by high-magnitude/low-frequency climatic events during storms with estimated return periods in excess of 50 years (e.g. Glade 1997, 1998, 2003).

In contrast, gully (e.g. De Rose et al. 1998, Betts et al. 2003; Marden et al. 2005; Parkner et al. 2006, 2007, Fuller & Marden 2011, Marden et al. 2011) and earthflow erosion (e.g. McConchie 1986; Zhang et al. 1991, 1993; Trotter 1993, Marden et al. 2008) are far less well studied. However, in at least some East Coast areas gullies cover much less area than landslides but make a far larger contribution to sediment yield as low-magnitude/high frequency rainfall events activate gully erosion (Page et al. 2000, Marden et al. 2008).

## 2. Hill country erosion control practices

New Zealand is unique in the way it uses its steep and often unstable hill country for pastoral farming, grazing predominantly sheep and beef cattle but also deer. Stock remain on the hills all year round, except in the high country during winter.

### 2.1 History of using poplar and willow for erosion management in pastoral hill country

Following deforestation (1.1), pastoral hill country erosion became so severe, the Water and Soil Conservation Act was passed by the New Zealand Government in 1941. This Act set in place an administrative structure of Catchment Boards tasked with controlling soil erosion and flooding, directed by central government policies. Central government funded applied research and grants to Catchment Boards till 1988. In 1988 the Catchment Boards became Regional or Unitary Councils responsible for managing a wide range of environmental activities including soil erosion and flood control. A new act, the Resource Management Act (RMA) of 1991, is now the guiding legislation for erosion control.

Erosion control measures for pastoral hill country were required to be established in the presence of the grazing animal, with permanent retirement from grazing recommended only in the most extreme situations. Only tree-based control measures were affordable on the scale required. The main beneficial effects of trees on the mass stability of the slopes are well described by Gray and Sotir (1996, pp 59-61). The two most suitable tree species proved to be poplar (*Populus* spp) and willow (*Salix* spp). Both species are readily established from large poles in the presence of stock, with a minimum of protection, are able to be produced cheaply in nurseries, and are easily transported and planted on steep unstable hill country. They grow quickly, can tolerate wet soil conditions for long periods, and do not shade pasture growth to any degree unless planted at close spacing. They are protected from stock browse by plastic sleeves placed over the poles at planting time which are removed after 4 or 5 years when the bark is able to resist stock rubbing and stripping. Their added values to the pastoral site through shade, shelter, quality fodder (especially during drought periods) and carbon sequestration, increase their utilisation by landowners.

### 2.2 Using poplar and willow for erosion control on pastoral hill country

This section describes recommended practices employed by landowners or land managers to prevent or reduce the more prevalent forms of erosion on pastoral hill land. Usually the erosion control is operating where earlier erosion events have occurred. Almost all pastoral hill slopes in New Zealand show signs of past erosion events. Landslide erosion control is given most attention for the reasons given in 1.1. Landslides affect much greater areas of land, are more widespread geographically, and have significant economic impact (see 3.1). The focus of the root research activity described later in this chapter is on slope protection against landslides. Control of other erosion types will be covered briefly.

### 2.2.1 Landslide erosion control

Landslides of the nature seen in figure 1 (L) follow a major rainstorm event and can happen at any stage of the year. Herbaceous plant cover alone on these pastoral hill slopes offers insufficient resistance to landsliding. To prevent the occurrence of landslide, or to prevent further landsliding, willow or poplar poles are planted up to 15 m spacings across the slope in a regular pattern or, more usually, where further landsliding is considered most likely to occur (Figure 1 R). Landslide prone soils are often shallow and exposed to climatic extremes.

Fig. 1. Severe slipping on soft sandstone (L); young space planted poplars reduce the risk of further erosion on an unstable slope (R).

The recommended approach is to plant a mix of clones and actively manage them (e.g. protection, replace dead trees, no cattle exposure for 2-3 years) for the first five years of growth to ensure a healthy tree develops. Willows are usually preferred in the wetter areas and lower on the slope where they are better adapted and the soil moisture is higher, while poplars are favoured further up the slope. Spacings are reduced where there is water ponding (Hathaway 1986). Pole planting is done in winter, either by ramming the sharpened pole directly into the moist soil or planting into an augured hole. Newly planted poles are revisited in summer to ram contracted soil around the pole. Survival rates of 90% are often achieved for poplars and willows planted from poles. Any dead poles are replaced the following winter. Tree death from animal damage is reduced by covering the poles with a plastic sleeve and excluding cattle from planted areas up to three years. Where the potential for slip erosion is very severe to extreme a closed canopy tree cover is recommended. However, this requires permanent retirement from pasture, a change many farmers are not willing to undertake.

### 2.2.2 Earthflow erosion control

Willows are preferable to poplars for slowing earthflows as their roots form thick mats when in a wet environment. Earthflow control measures rely on removing surface water to minimise infiltration by surface smoothing and constructing diversion banks, tying together the surface with willow roots, pair planting willows where gullies are likely to form and planting at the toe of the movement to hold up the toe (essential when the toe is being undercut by a waterway). Spacing is closer for wetter or more active earthflows.

### 2.2.3 Slump erosion control

These events are usually sufficiently large that tree planting alone will not control them. They require surface water to be drained, the surface smoothed, retired from grazing and close planted in trees. Once in trees they need to be managed as continuing movement will overturn or topple trees requiring clearing to keep drainage open, and replanting. Over time movement will reduce.

### 2.2.4 Gully and tunnel gully erosion

Discontinuous gullies are very common in pastoral hill country. Large erosion events fill the valley bottoms and smaller events begin to scour them. Control of further erosion is achieved by pair or single planting willows up the valley bottom at 15-20 m spacings and, at points with serious erosion, developing a planted block with 2-3 rows of willows planted at 1.5m spacings across the valley bottom and in an area fenced out from grazing. Erosion from small gullies has been successfully arrested by construction of debris dams at intervals up the gully coupled with pair planting of willows. The willow root mats growing across the gully bed and over the surface of the dam cover and isolate the eroding surface. Drainage lines supplying the gully are pair planted with tree willows at 15-20 m spacings and measures taken to ensure the grassed surface is not broken, allowing a gully head to form. Terrace edges are protected by either grassed waterways or constructed flumes. To stabilise very large gullies often requires the whole catchment to be retired from grazing and planted in closed canopy trees. Where climate allows trees to grow, tunnel gully erosion is controlled by planting a poplar or willow pole in the collapsed tunnels. Further tunnel erosion is prevented by planting poles in the characteristic sunken drainage lines supplying the tunnel. The tree roots hold up the sediment gradually filling up the holes.

### 2.2.5 Stream bank erosion

In hill country, storm events create flash floods resulting in bank scour and deepening the channel to form a gully. Stream bank protection is achieved by planting willows or poplars along the banks. The roots stabilise the banks and protect the stream bed from degrading. Large limbs are removed from mature stream bank trees to rejuvenate them, reduce their size and reduce risk of bank destabilisation.

## 3. Effectiveness of trees in reducing erosion on pastoral slopes

Knowledge of the effectiveness of trees in stabilising soil and reducing erosion potential on slopes is important when considering time scales, costs, landowner attitudes to topsoil displacement and loss and the achievement of catchment scale objectives for managing consequences of silt incursion into waterways.

### 3.1 Effect of hill slope erosion on pasture production

Landslide erosion results in immediate and often dramatic reductions in pasture production on steep hill country (Lambert et al. 1984). Reductions in land productivity from soil erosion occur directly through the loss of topsoil, and indirectly through reduced pasture yields on eroded ground (Blaschke et al. 1992). Quantification of economic and biophysical losses associated with landslide erosion is crucial to justify the need for soil conservation and erosion control activities on steep hill country. An understanding of the rate of recovery of production on landslide scars is essential for implementing improved farm management techniques and land uses. Measured pasture dry matter yields on young landslide scars were ~20% of the yields produced on un-eroded ground, and while such scars revegetated rapidly over the first 20 years and could attain 70–80% of original productivity, further recovery was slow (Lambert et al. 1984, Rosser & Ross 2010). Maximum pasture recovery occurred within about 20 years of landsliding and further recovery beyond 80% of un-eroded level is unlikely (Lambert et al. 1984, Rosser & Ross 2010).

Loss of pasture production reflects loss of soil physical and chemical properties. Recovery of pasture production on landslide scars follows similar recovery to soil physical (soil depth, particle density, etc.) and chemical properties (Total C, total N, etc.). Topsoil depths on eroded sites were roughly a third of topsoil depths on uneroded sites, indicating reduced profile available water capacity on eroded soils (Rosser and Ross 2010). However, data were inconclusive on whether surface total C would recover to values on uneroded sites in the long term. Other soil properties (C/N, pH, Mg, Na, and CEC) are expected to recover to uneroded values within human time scales and are not the cause of permanent reductions in pasture growth on older landslide scars. The implications of this and previous research are that the sustainability of pastoral agriculture on steeper east coast hill country, underlain by poorly consolidated parent materials, will come increasingly under threat from the progressive reduction of pasture production through cumulative erosion.

The economic costs of pastoral hill country erosion are under-researched and under-reported in New Zealand.

### 3.2 How effective are *Populus* and *Salix* in reducing erosion?

Erosion control plantings of trees hold many advantages over herbaceous ground cover in their capacity to bind soil at deeper levels, improve slope drainage, anchor the soil to bedrock through root penetration and establishment, reduce the impact of rain on the soil surface, provide a barrier to downward movement of soil, redirect rainfall via flow to parts of the slope with high protection (i.e. near the stem), re-evaporate rainfall from leaves and branches. However, the presence of soil conservation poplar or willow trees on pastoral slopes does not ensure that erosion by slippage or earth flow will not occur. The trees might be described as effective in preventing erosion only when there is no slippage or if slippage is restricted to very short movements follow prolonged rainfall and soil saturation. Factors determining effectiveness include planting density, tree spacing, tree root length density, tree root mass density, location of trees in relation to slope topography and water movement, age of trees.

Few published data are available on the effectiveness of space-planted *Populus* and *Salix* trees in reducing hillslope erosion. Hawley and Dymond (1988) using digital image analysis of aerial photographs, and a derived relationship between the fraction of ground eroded (landslide scars) and distance from a tree, estimated the degree to which individual trees of *P.* × *euramericana* aged 14-17 years reduced shallow landsliding following a recent severe storm. The tree spacing was 25 sph, though with some mortality. They concluded that an average *Populus* tree saved 8.4 $m^2$ of ground from failure. thereby reducing pasture production losses by 13.8% compared with equivalent untreed sites. At spacings of 10 m x 10 m (100 sph) and assuming 100% tree survival, they estimated that immediate pasture production losses attributable to landslides would have been reduced by at least 70%, compared with 13.8% at a tree spacing of 25 sph. Hawley (1988) also predicted that for two trees spaced 11.5 m apart (75 sph), land slippage around one tree, including the contribution from a neighbouring tree, would be reduced from 8.2% to about 1.45%, that is, by about 82% of that previously. A more comprehensive study by Douglas et al. (2011) covering 53 sites with *Populus* trees, 6 sites with *Salix* trees and 6 sites with *Eucalyptus* trees recently exposed to a severe storm event showed that over all sites, trees reduced the extent of slippage by an average of 95% compared with slippage on nearby pasture control sites. The treed sites contained groups of trees varying in number from 4-10, and at densities ranging between 32 sph and 65 sph. Slippage occurred at 10 of the 65 sites, and the greatest extent of slippage occurred where trees had a diameter at breast height (DBH) of <30 cm. They concluded that spaced *Populus* and *Salix* trees dramatically reduced the incidence and severity of soil slippage on erodible slopes, and that they were even more effective when their average DBH was 30 cm or greater. Mature plantings of 30-60 sph (13 m to 18 m spacing) were very effective in controlling soil slip erosion.

There is a paucity of data on the effectiveness of spaced *Populus* and *Salix* trees with DBH <30 cm in reducing slope erosion in New Zealand hill country. Firm recommendations can be made on planting density for mature trees, but it is clear that higher planting densities are needed for younger trees if they are to reduce slippage to the same extent and in fact they are not likely to be effective at all until some years after planting, depending on growth rate and exposure to severe storm events.

## 4. Root behaviour of poplar and willow trees on slopes

One of the key ways trees and other woody vegetation contribute to slope stability and control a range of erosion processes is through their development of root networks that enhance the mechanical reinforcement of soil (Phillips and Watson 1994; Genet et al., 2008; Stokes et al., 2009; Schwarz et al., 2010). Roots provide reinforcement to soil through a combination of their tensile strength, frictional resistance, and soil bonding properties (Schmidt et al., 2001; Bischetti et al., 2005, 2009; Genet et al., 2005; De Baets et al., 2008). The major factors that determine the degree to which tree roots modify soil reinforcement are root system architecture and tree density, which are influenced by factors including species, tree size, topography, soil characteristics (e.g. depth, texture, bulk density, water content), and above-ground management (Roering et al., 2003; Reubens et al., 2007; Stokes et al., 2009).

Although poplars and willows have been used in New Zealand for a range of erosion control programmes for more than 50 years (Thompson and Luckman, 1993; Wilkinson 1999; Douglas et al., 2011), before 2000, few studies were conducted or reports written to gain knowledge and understanding of their root systems and their strength and how they vary spatially and temporally (Hathaway 1973, 1986; Hathaway and Penny 1975; Vine 1980; Luckman et al., 1981; Hughes 1992; Wilkinson 1999). Laboratory testing found that the tensile strength of poplar and willow roots collected from 1-year-old trees ranged from 36.3 to 45.6 MPa and samples from different seasons varied in strength possibly because of variations in specific gravity and the ratio of lignin to cellulose (Hathaway and Penny 1975). At Palmerston North in the southern North Island, Vine (1980) excavated the roots of six species or clones of poplar aged 3-6 years and examined roots greater than 5 mm diameter. The root systems were asymmetric, exhibiting strong growth into unplanted areas, and root grafting was observed between adjacent trees. There was considerable variation in vertical and lateral root growth between species or clones, and between trees within clones. Hathaway (1986) reported that few data were available on root system morphology and root density of soil conservation plants under New Zealand conditions. Consequently, recommendations on appropriate species and clones for erosion control on specific sites were based almost entirely on survival and above-ground characteristics, and practitioner experiences. Near Palmerston North, roots of poplar trees aged 5 or 6 years growing at a site with a free-draining soil and another site with restricted drainage were sampled by intensive coring to a depth of 1 m (Hughes 1992). Very few large woody roots (> 2.5 mm diameter) were found at the poorly-drained site compared to the well-drained site, because a number of large roots were likely killed during periodic wet periods at the moister site. The few roots present at depth and at the periphery of the poorly-drained tree rooting volumes were non-woody fine roots. Across diameter classes, woody roots were detected up to 1 m depth in the free-draining soil whereas in the soil with poor drainage, they occurred at less than 0.7 m depth except for a root < 2.5 mm diameter found at 0.75 m depth and about 0.3 m from the tree. Woody roots were found up to 4.3 m from trees in the poorly-drained site and up to 5.3 m from trees in the well-drained site. Wilkinson (1999) listed nine principal reasons for using poplars and willows for erosion control, one of which was their extensive root systems capable of rapidly stabilising large soil masses. He reported that annual extension of their roots across and downslope was similar to their annual height increment (m) while upslope extension occurred at about half of that rate. No further details were presented and it is uncertain if these findings were from research conducted in New Zealand or overseas.

### 4.1 Recent studies

With changing research emphases and enhanced funding in sustainable land management and related areas over the last decade, the distribution and other characteristics of roots of poplar and willow, and factors that influence them, have been determined (McIvor et al., 2005, 2008, 2009; Douglas et al., 2010a, b; Marden and Phillips 2011). On a research farm near Palmerston North, the distribution of coarse/structural roots (> 2 mm diameter) of young trees of *Populus deltoides* x *nigra* 'Veronese' (diameter at breast height (DBH) < 30 cm) of different age and size (McIvor et al., 2008), and position on a slope prone to shallow landslides (McIvor et al., 2009), was determined by whole-tree excavation. The trees were established by planting 3 m poles (vegetative cuttings), the usual establishment method for this species (Wilkinson 1999), on an east-facing slope of 15-25°. The distribution of radial roots of individual trees aged 5, 7 and 9.5 years growing lower on the slope (< 20°) on an accumulation zone (relatively deep soil) was variable around the trees at each age and differed between upslope and downslope sides of the trees (Figure 2, McIvor et al., 2009). Roots of trees at each age were distributed asymmetrically with those of trees aged 5 and 9.5 years aligned in a similar direction of left/upslope to right/downslope. In contrast, the roots of the tree aged 7 years were concentrated in the left/downslope quadrant. Across all trees, radial roots were generally found within 0—40 cm soil depth and often within 0-15 cm depth. Vertical roots extended to a depth of about 1.0 m, where a fragipan occurred. Total root dry weights (excluding root crown) were 0.57 kg (tree aged 5 years), 7.8 kg (7 years) and 17.9 kg (9.5 years), and total root length was 79.4 m for the tree aged 5 years and 663.5 m for the 9.5 year-old tree (Table 1). A linear relationship was established between root mass and DBH (Root mass = 1.16*DBH - 7.56) and between root length and DBH (Root length =

45.1 *DBH - 293.4). The results indicated that root development of the trees was minimal in the first 5 years but then increased rapidly. It was suggested that poplar trees established from poles on erosion-prone slopes needed to attain at least 5 years to develop a structural root network that binds soil effectively.

| Tree Age yr | Height m | DBH cm | Position on slope | Slope angle | Above ground mass kg | Below ground mass kg | | Coarse Root length m |
|---|---|---|---|---|---|---|---|---|
| | | | | | | root | crown | |
| 5 | 7.3 | 8.4 | Lower | 22.0 | 10.49 | 0.57 | x | 79.4 |
| 7 | 9.3 | 14.4 | Lower | 21.3 | 43.52 | 7.8 | x | 349.3 |
| 9.5 | 13.3 | 21.3 | Lower | 22.1 | 132 | 17.9 | 3.3 | 663.5 |
| 11 | 13.4 | 29 | Lower | 21.8 | 260.79 | 81.35 | 18.18 | 1611.3 |
| 11 | 12.95 | 27.2 | Mid | 28.6 | 210.87 | 38.77 | 16.5 | 1131.3 |
| 11 | 11.15 | 18.9 | Upper | 32.0 | 61.48 | 8.15 | 6.6 | 293.2 |

Table 1. Dimensions of six *Populus* × *euramericana* 'Veronese' trees excavated on a pastoral hillslope.

In a sequel investigation in the same planted block, McIvor et al. (2009) determined the effect of slope position (upper (slope angle 32$^{o}$), mid (28.6$^{o}$) and lower (21.8$^{o}$)) on root distribution and other root characteristics of 'Veronese' poplar aged 11.5 years (Table 1). At each position, trees (sample size = 1) had DBH of 18.9 cm (upper slope), 27.2 cm (mid) and 29.0 cm (lower). Most of the > 2 mm diameter roots occurred within 0-40 cm soil depth.

Radial distribution of roots varied between slope position with distribution at mid- and lower slopes being more symmetrical than at upper slope where roots were mainly directly upslope of the tree and west of the tree (Figure 2). Roots spread at least 8 m in one direction around each tree, and in some cases more than 10 m. They changed direction and depth frequently and crossed each other regularly. Roots growing downslope were mostly located within 30 cm of the ground surface and changed depth less often than roots in the upslope direction. Growth of vertical roots varied with position and depended primarily on soil depth to a fragipan, ranging from 0.35 m at the upper slope to 1.4 m at the lower slope. Roots penetrated the fragipan at the upper and mid-slope positions but not at the lower slope position where soil depth was greatest. Total root length ranged from 287.9 m (upper slope) to 1,611.3 m (lower) and total root dry weight (excluding root crown) ranged from 8.15 kg (upper) to 81.35 kg (lower). Earlier relationships between root mass and DBH, and root length and DBH (McIvor et al., 2008), were revised with inclusion of data collected in this study to give exponential relationships of Root mass = $0.0003*DBH^{3.62}$ and Root length = $0.6582*DBH^{2.26}$. Both studies by McIvor et al. (2008, 2009) involved trees that were likely far enough from adjacent trees to minimise any above- or below-ground interactions with them. The findings were valuable because they provided knowledge and understanding of essentially isolated trees on slopes, and hence full expression of the measured root traits under the prevailing topographic and climatic conditions.

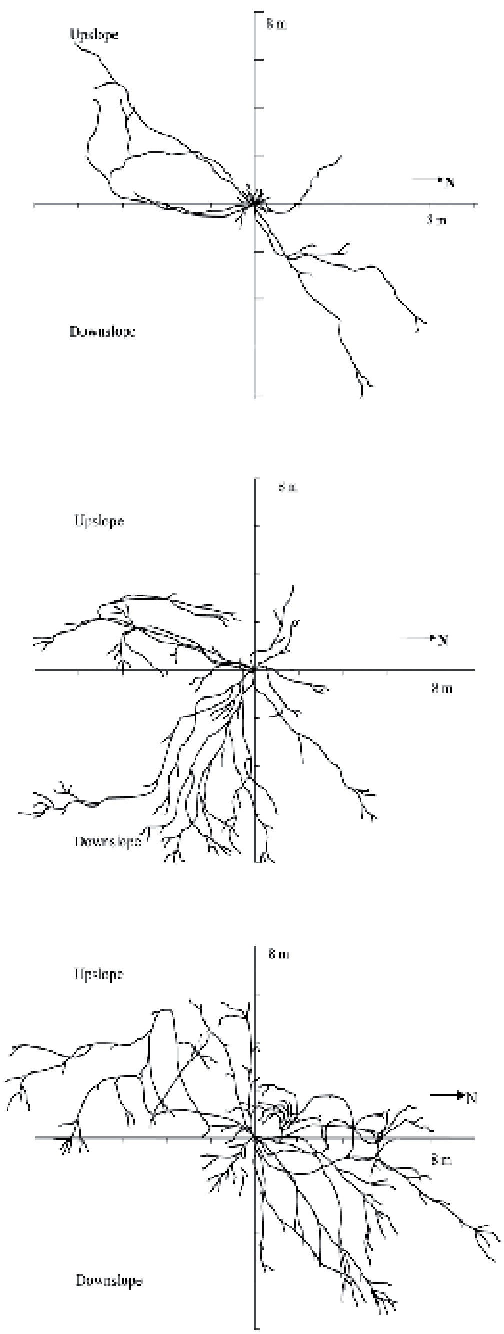


Fig. 2. Radial distribution of structural roots (>2 mm diameter) of 'Veronese' poplar trees aged 5 yr (top L), 7 yr (top R) and 9.5 yr (bottom) growing on hill country near Palmerston North (from McIvor et al. 2008).

### 4.2 Density effects

The next phase of research determined the effect of tree density or spacing on root distribution of poplar. Planting densities of 25-256 stems ha$^{-1}$ have been recommended but little is known of how the root system develops chronologically and what root densities different planting densities achieve. Two studies were conducted near Palmerston North involving 9-11 year old poplar, mostly *Populus deltoides* x *nigra* 'Tasman', arranged in a Nelder planting design, and involved excavating trenches between adjacent trees to a soil depth of 90 cm. In the first study, trenches were dug in four directions around single trees at densities of 84 and 770 stems per hectare (sph) growing on slope angles of 3.5-5.5$^{o}$ (McIvor et al., 2005). Most of the roots were within 0-45 cm soil depth and the number of roots decreased exponentially with depth. For example, at a distance of 0.9 m from the tree at 770 sph, root number decreased from 398 roots/m$^2$ at 0-15 cm depth to 6 roots/m$^2$ at 75-90 cm depth. At 770 sph, the tree root network at any point was contributed by more than one tree. At the low density planting, there were no roots found at the midpoint between adjacent trees. Around both trees, there were no significant differences between root number and root cross sectional area at each of the four directions, suggesting that the root systems were distributed symmetrically on these low slope angles. It was suggested that for tree plantings at this age, the ideal planting density for both pasture production and erosion control is likely to be between 84 and 770 sph.

Additional tree densities were included in the study by Douglas et al. (2010a), and involved 84, 89, 160, 210, 237 and 770 sph. More than 80% of roots were < 5 mm diameter and root number and root area ratio (RAR) were higher in shallow soil layers e.g. 0-15 cm depth than deeper in the profile. Trees at 770 sph had 3-12 times more roots and 3-9 times greater RAR than those at other densities. Mean cross-sectional area per root was 3.5-4.8 mm$^2$ and did not vary significantly between densities. Densities of 160, 210 and 237 sph had moderate to high root occupancy of soil layers and satisfactory root number and cross-sectional area. They were therefore recommended as options to enhance soil strength whilst likely enabling satisfactory understorey pasture production. Using the 160 sph density rather than the two higher densities would reduce planting material and labour requirements and potentially increase pasture growth.

### 4.3 Root length density effects and seasonal changes

Current research is investigating the root occupancy of the root network to complement the radial distribution of whole tree excavations and the vertical distribution from trench studies. Root length density (RLD) is a way of describing root occupancy. Coring provides data on both coarse and fine roots for a particular volume of soil from which RLD can readily be calculated, and has led to understanding of the contribution of fine roots to the root network and calculation of RLD was determined from cores for wide spaced trees of *Salix matsudana* × *alba* 'Tangoio' and *Populus* × *euramericana* 'Veronese' growing on pastoral slopes of varying steepness from 14° to 21°, and from 21° to 32° respectively. Cores of diameter 200 mm were taken in 150 mm intervals to 600 mm depth at fixed distances of 2 m and 3 m (willow) and 2 m and 4 m (poplar) from the stem and at positions 120° apart to allow for asymmetrical root distribution. For the willows, in the summer of 2009 and 2010 respectively, 71.3% and 83.5% of root length density (RLD) was found in the upper 300 mm with as much as 67.4% of RLD being in the top 150 mm of soil (Table 2). While the % of RLD from 300-600 mm depth decreased from 2009 to 2010 in absolute terms the RLD had changed little (Table 2). The % contribution of fine roots to total RLD varied little between depths and sample times, and ranged from 89 to 93. RLD is contributed largely by fine roots whereas root mass density is contributed by coarse roots (Figure 3). Fine RLD was much higher in the upper 150 mm of soil in 2010, though not at lower depths. Fine root production is particularly responsive to soil moisture. For the poplars % RLD in the upper 150 mm ranged from 65% down to 17%, and in the top 300 mm ranged from 78% down to 43%. RLD was similar at the different distances from the stem for both species. Vertical distribution pattern of RLD in willow from core data and RAR in poplar from the trench data are very similar, whereas there are notable differences in root distribution between trench and core data for poplar. On steeper slopes RLD is possibly more evenly distributed through the soil profile. Further studies are needed to verify this hypothesis. To fully understand root occupancy distribution RLD should be measured to 10 m from the stem. There was a significant reduction in RLD during the dormant season in both *P.* × *euramericana* 'Veronese' and in *S. matsudana* × *alba* 'Tangoio'. The reduction in RLD was almost entirely through loss of fine root, with little change in coarse root RLD and was more pronounced in the top 300 mm for both species. For example, the ratio of fine RLD to coarse RLD for *P.* × *euramericana* 'Veronese' poplar reduced from ~15x at the end of summer to close to 1x during the dormant period of the year (Figure 4). The replacement of fine roots lost during the dormant period happens quickly once the trees become active, which happens earlier for willows than for poplars. The reduction of fine RLD during the dormant season provides more pores for water storage and drainage, but reduces the root-soil contact significantly, and the cohesive soil-root network strength will reduce accordingly. Further research is needed to know what constitutes sufficient soil-root network strength for slope stability in the pastoral hill country situations where these trees are being planted.

| Year | Soil depth | Root Length density $mm^{-3}$ | | | % allocation of RLD |
|---|---|---|---|---|---|
| | mm | Fine Roots | Coarse roots | All roots | |
| 2009 | 0-150 | 1769 | 210 | 1979 | 41.5 |
| | 150-300 | 1374 | 43 | 1418 | 29.8 |
| | 300-450 | 601 | 47 | 648 | 13.6 |
| | 450-600 | 654 | 71 | 725 | 15.1 |
| 2010 | 0-150 | 4303 | 321 | 4624 | 67.4 |
| | 150-300 | 1013 | 90 | 1103 | 16.1 |
| | 300-450 | 558 | 57 | 615 | 8.9 |
| | 450-600 | 478 | 41 | 519 | 7.6 |

Table 2. Mean root length density ($mm^{-3}$) of mature *Salix matsudana* × *alba* 'Tangoio' measured at 2 m from the stem in two consecutive years (DOY 49 2009, DOY 47 2010)

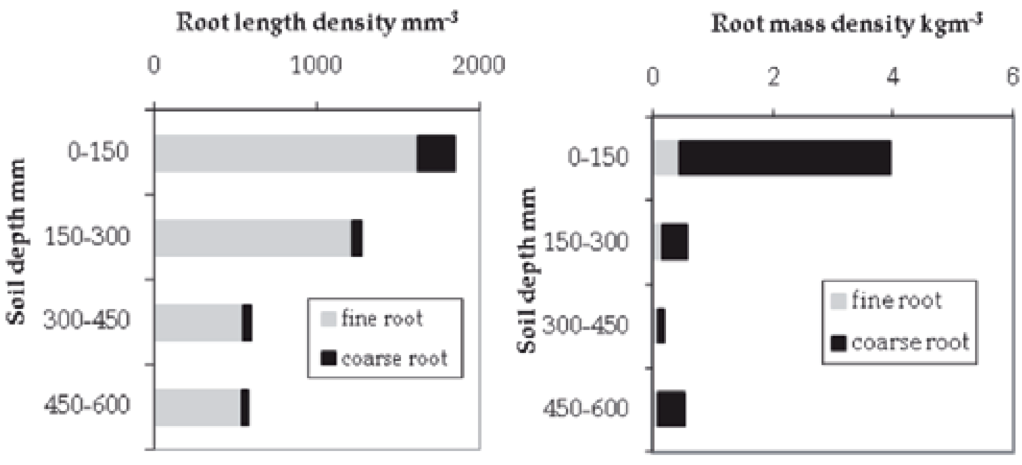


Fig. 3. Fine (< 2 mm diameter) and coarse (≥ 2 mm diameter) root length density (L), and fine and coarse root mass density (R) of *Salix matsudana* × *alba* 'Tangoio' varying with soil depth.

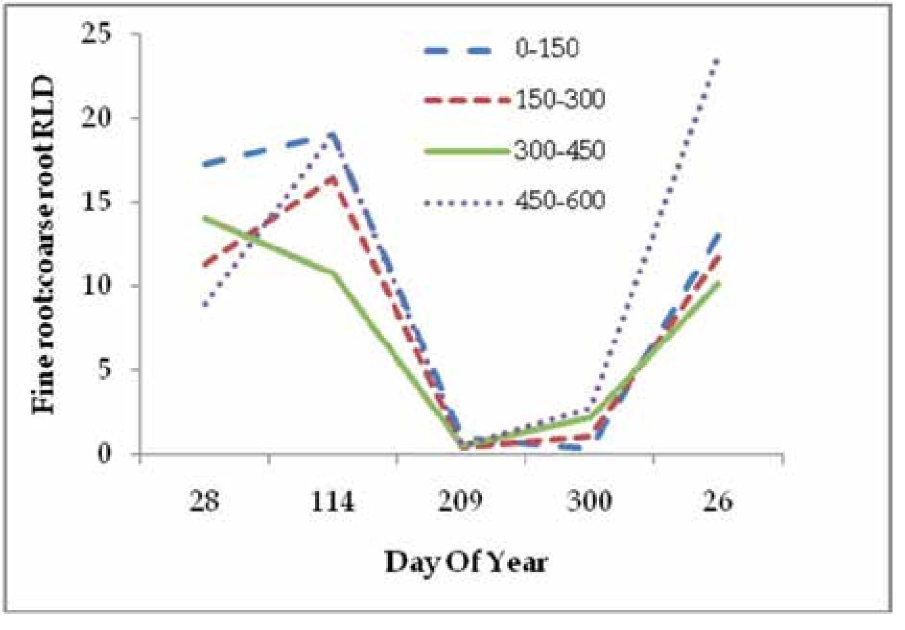


Fig. 4. Fine root RLD: coarse root RLD for *P.* × *euramericana* 'Veronese' poplar at different soil depths (mm) over one year.

#### 4.4 Methodology

Studies of poplar and willow roots in New Zealand have only used whole-tree excavation (e.g. McIvor et al. 2009), trenching (e.g. Douglas et al., 2010a), and coring (McIvor et al. 2010) methods. Whole-tree excavation has many advantages including enabling complete understanding of root distribution, insights into growth and external morphological changes in roots outwards from a tree, classification into diameter classes, determination of total root mass and total root length, and providing data for use in models e.g. 3-D architecture models. However the method is very time consuming e.g. 10-15 person days for excavating a young tree, difficult on steep slopes, and tree removal weakens a site prone to various erosion processes. Trenching and coring provide samples from the whole root system, and therefore are prone to sampling variation, including for coring, obtaining samples without roots. Both methods exclude vertical roots extending from the central tree axis, which are important for potential penetration into underlying bedrock/fractures and anchorage. There has been limited investigation of the potential of chemical analysis methods e.g. dyes, and ground penetrating radar for studies on poplar and willow roots.

#### 4.5 Root tensile strength of *P. × euramericana* 'Veronese' poplar

Roots sampled from 'Veronese' poplar trees aged < 10 years and growing on a slope angle of 23-27° (McIvor et al. 2008) were tested for tensile strength (Watson et al., 2007). Undamaged roots with over-bark diameters of 1.16-12.63 mm (under-bark diameter 0.90-8.51 mm) and 150-250 mm long, collected within 0-250 mm soil depth, were tested. A power function relationship was developed between live root-wood tensile strength (Y; MPa) and under-bark diameter (X; mm) using 123 samples, of $Y = 80.79 {*} X^{-0.82}$ with an $r^2$ value of 0.69.

Mean tensile strength decreased rapidly over the range 1-3 mm diameter, being 90.8 MPa for roots < 1mm, 56.9 KPa for roots 1-2 mm, and 40.1 MPa for roots 2-3 mm diameter (Figure 5). Root diameters of 3-9 mm had mean tensile strengths of 19.0-24.3 MPa. The decrease in tensile strength with increase in diameter may be a function of the changing material properties of the root-wood (e.g. cellulose) with increase in root size (age), an increase in the number or severity of defects with increasing root size, or other factors.

#### 4.6 Current research

There is increasing awareness of the need to manage poplar and willow trees planted for soil conservation. As trees age, they can become large and prone to limb breakage and toppling e.g. under high winds, potentially damaging farm infrastructure (tracks, fences, buildings etc.), injuring livestock, and creating debris that can hinder livestock mustering and other operations. The practice of pollarding, involving the removal of the entire canopy, has been advocated for numerous poplar and willow plantings, to prevent large trees developing, and supplying supplementary fodder for livestock during feed shortages such as summer/autumn drought. The implications of managing tree canopies on root distribution and development, and its impact on soil stabilisation functions, are being determined for poplar (Douglas et al., 2010b) and willow (McIvor et al. 2010) in the southern North Island. It is possible that the density of managed conservation trees will need to be increased to achieve similar levels of effectiveness for erosion control as unmanaged trees.

Root growth of a range of poplar and willow germplasm and vegetative material (cuttings stakes, poles) is being determined at two sites – one near Palmerston North and the second near Gisborne on the east coast of the North Island. Growth after one year was determined in April/May 2010 by excavation (e.g. Marden and Phillips 2011) and further excavations are scheduled for 2011 and 2012.

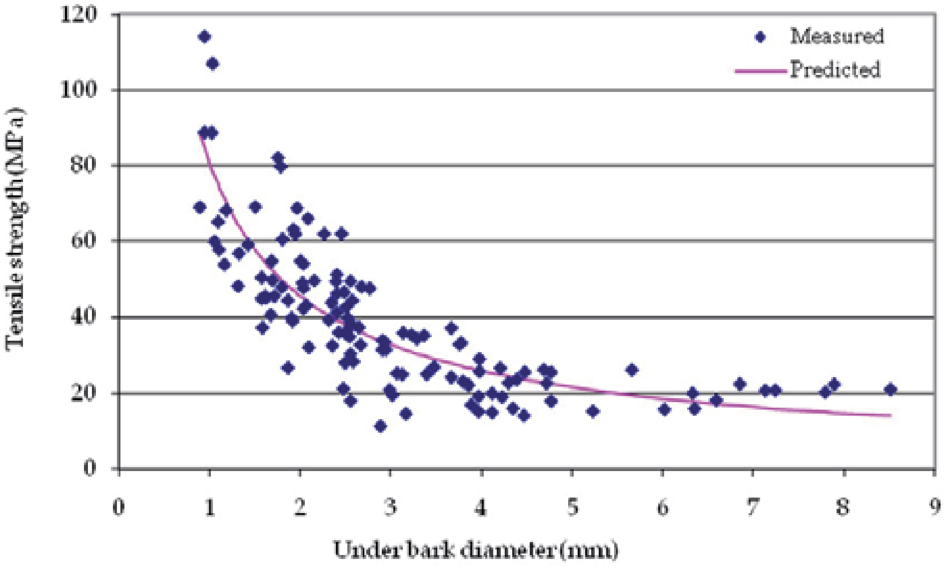


Fig. 5. Relationship between live root-wood tensile strength and under-bark root diameter for 'Veronese' poplar roots.

## 5. Modelling of hill country erosion and environmental outcomes

Effective programmes for controlling pastoral hill country soil erosion depend on close cooperation between the organisation responsible for environmental monitoring and protection and the landowner. The development of individual Whole Farm Plans (WFPs), incorporating physical, farm management and business plans, are designed to achieve sustainable land use, both environmental and economic. Whole farm plans also sit within a wider environment and have environmental outcomes at the catchment or greater level. This includes reduction in runoff and sediment entering waterways and river systems (the receiving environment), with consequent decrease in nutrients, particularly N and P, and reduction of flood risk through stabilisation of river-bed levels.

An effective monitoring programme is necessary to measure and analyse the impact of the approach at both the farm and broader scales. It will therefore have to provide information to the landowner for decision making and also be suitable for scaling up to the catchment scale. Monitoring to date of the effectiveness of WFPs has been largely limited to monitoring of the implementation of the conservation works programme, so that it has been generally activity based. Soil conservation activities include the planting of conservation poplar and willow trees at recommended densities/spacings, fencing off waterways, building sediment traps or wetlands along waterways, retiring steep land or planting it in forestry trees. Shifting the emphasis from actions/tasks to environmental outcomes will provide regional councils (see 2.1) with direct measures of the achievements towards the target goals. It will also indicate the rate at which progress is being made towards them, both at the WFP level and in larger areas of the region, through appropriate amalgamation of data from several or many WFPs.

Attributes that should be assessed or measured at any stage in the implementation of works in a WFP are:

i. vegetation type and cover (extent),
ii. the area of each works (at the individual site level), and
iii. age of vegetation.

These can all be assessed using aerial photography at an appropriate scale e.g. 1:3000 or (preferably) higher resolution, complemented with planting plans describing location and timing, and other knowledge of the farmer/land manager. Detection of areas of forestry, whether exotic or indigenous, is easier than those with spaced plantings of conservation trees on pasture, particularly when the trees are young, and field checking of sites is recommended e.g. for survival. Also, tree growth of the same species/clone can vary considerably with aspect, position on a slope, and other factors, so that knowledge of tree age is only a preliminary indication of the likely size of trees.

For conservation works involving forestry (exotic or indigenous), canopy cover can be estimated from appropriate aerial photographs, but a sequence over time e.g. every 3-5 years is required to make this a useful approach to determine critical times when effectiveness for erosion control (and sediment reduction) is increased significantly e.g. at canopy closure. Estimates of canopy cover per site with an accuracy of plus or minus 10% should be more than adequate to describe canopy development over time. Sites with spaced trees on slopes or in gullies are more difficult to assess for canopy cover because they almost always have lower and more variable tree densities than forestry. It is unwise to compare canopy covers of forestry with those of spaced trees because the same canopy cover in both vegetation systems will usually not indicate the same stand density - hence the implications for root distribution patterns and erosion control potential are quite different. For spaced trees, it is recommended that at least one measurement of tree size – dbh –is measured on a sample of trees at an individual site e.g. 1-5%, to complement data on tree age. This might be conducted on the same trees once every 3-5 years until tree maturity so that the task does not become overly onerous. Collection of such data will be useful to define tree size and will enable a better description of the status of trees which survive or die during future storms, and those which hold or fail to hold soil on eroded slopes.

### 5.1 Estimation of sediment export from farm with a whole farm plan

It is assumed that in the farm plan is well designed set of soil conservation works, including

i. Exotic afforestation
ii. Planting of spaced trees on pasture (2-tier farming; agroforestry, tree-pasture system)
iii. Regeneration of indigenous forestry (retired land from scrub to indigenous forestry).

#### 5.1.1 Exotic forestry

Close-tree planting will reduce erosion on pastoral areas by 90% once trees are mature (Dymond *et al*., 2006). For trees less than 20 years old, a maturity factor may be defined by

$$M_f = Age_f / 20$$

where $M_f$ is the maturity factor of forestry, and

$Age_f$ is the age of the trees in years (for trees older than 20 years $M_f$ is set to 1).

The long-term mean erosion rate is reduced by $M_f$ x 0.9 for exotic forestry.

#### 5.1.2 Spaced trees on pasture

If the pastoral land contains significant areas of highly erodible land (HEL), then there will be recommended soil conservation works as part of the WFP, designed to significantly reduce mass-movement erosion. The recommendations will involve

i. spaced tree planting, and
ii. gully tree planting.

It is assumed that these conservation works will reduce mass-movement erosion by 70% once plantings are mature ((Hawley and Dymond, 1988; Thompson and Luckman, 1993; Hicks, 1995). The maturity factor of the soil conservation works may be calculated by

$$M_p = f \times Age_p / 15$$

where $M_p$ is maturity factor of the soil conservation trees,
$f$ is the proportion of trees in the plan that have actually been planted, have survived, and are well maintained, and
$Age_p$ is the age of the soil conservation trees (for trees older than 15 years $Age_p$ is set to 15).
The long-term mean erosion rate is reduced by $M_p$ x 0.7 for spaced trees on pasture.

#### 5.1.3 Indigenous forestry

Vegetation on retired land is assumed to be at one of five phases with maturity factor $M_r$:

i. reverting pasture; $M_r = 0.0$
ii. incomplete scrub canopy closure (early stage); $M_r = 0.1$
iii. incomplete scrub canopy closure (intermediate stage e.g. 3 years); $M_r = 0.5$
iv. complete scrub canopy closure; $M_r = 0.9$ (usually after 5 years), and
v. indigenous forest. $M_r = 1.0$

The long-term mean erosion rate is reduced by $M_r$ x 0.9 for indigenous forestry.

### 5.2 Calculation of sediment export from farm

For simplicity, it is assumed that all sediment lost from eroded land enters waterways and leaves the farm. However significant quantities of sediment may be retained on-farm through appropriate structures e.g. dams, or through natural means e.g. wetlands, but these are not considered here.

1. Sediment export (tonnes/yr) from land with **exotic forestry** is $e_f$ x [1-$M_f$ x 0.9] x $A_f$
where $e_f$ is the mean erosion rate (tonnes/km$^2$/yr) of the exotic forestry land if it was in pasture (from NZeem®: Dymond et al., 2010) and
$A_f$ is the area of exotic forestry in the farm plan (km$^2$).
2. Sediment export (tonnes/yr) from land with **spaced trees** is $e_p$ x [1-$M_p$ x 0.7] x $A_p$
where $e_p$ is the mean erosion rate (tonnes/km$^2$/yr) of the pastoral land with trees, if trees were not planted (from NZeem®), and
$A_p$ is the area of land with trees on pasture in the farm plan (km$^2$).
3. Sediment export (tonnes/yr) from land with **indigenous forestry** is $e_r$ x [1-$M_r$ x 0.9] x $A_r$
where $e_r$ is the mean erosion rate (tonnes/km$^2$/yr) of the indigenous forestry land if it was in pasture (from NZeem®), and
$A_r$ is the area of indigenous forestry in the farm plan (km$^2$).

$e_f$, $e_p$ *and* $e_r$ may or may not be the same.

The **total sediment export** $S$ (tonnes/yr), from a farm which contains land use/ conservation works with a range of maturities, may be calculated by estimating the mean

**Case study Farm in hill country near Palmerston North, southern North Island**

These formulae were applied to the Whole Farm Plan (WFP) for a commercial farm. According to NZeem® the farm currently exports 2640 tonnes of sediment per year on average. If the following soil conservation methods (from the WFP) were implemented

Year 1 - 200 space-planted poplars
Year 2 - Afforestation of 3.4 ha; 200 space-planted poplars
Year 3 - Afforestation of 8.6 ha; 130 space-planted poplars; 70 poplars for gully control
Year 4 - Afforestation of 12.0 ha; 130 space-planted poplars; 70 poplars for gully control
Year 5 – Afforestation of 6.6 ha; 200 space-planted poplars.

then the sediment export from the farm would reduce gradually from 2640 tonnes/yr to 820 tonnes/yr over 20 years.

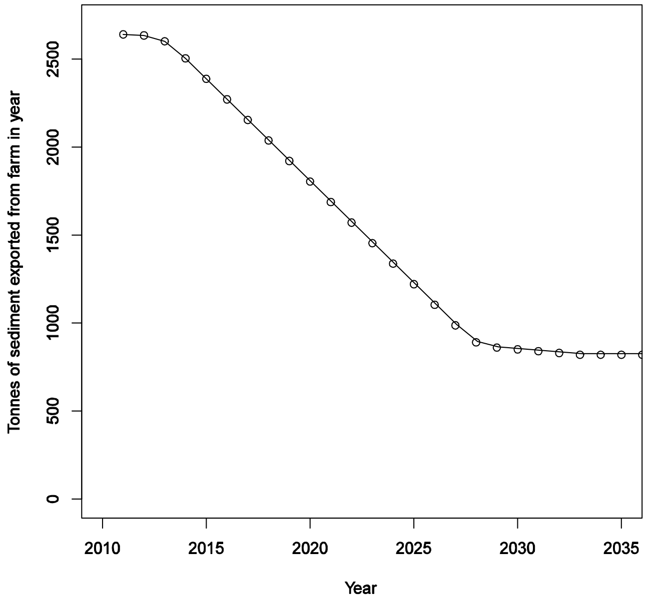


Fig. 6. Sediment export from hill country farm after implementing soil conservation works in the case study.

$S = e_f(1-0.9\bar{M}_f)A_f + e_p(1-0.7\bar{M}_p)A_p + e_r(1-0.9\bar{M}_r)A_r$ where $\bar{M}_f$ is the mean maturity factor for the exotic forestry sites, $\bar{M}_p$ is the mean maturity factor for the spaced tree sites, and $\bar{M}_r$ is the mean maturity factor for the indigenous forestry sites.In a WFP for a 20 yr period, for example, there may be three sites with exotic forestry planted in Yr 0, Yr 5 and Yr 15, 20 sites with spaced trees on pasture (annual plantings of 150 poles), and two sites where growth of indigenous vegetation is encouraged through retirement – the first in Yr 5 and the second in Yr 15. The formula can be applied at any stage of the implementation of the conservation works in a WFP ranging from entirely new, through partial implementation, to completed implementation. Therefore it can be applied to WFPs implemented already.

The case study (above, figure 6) shows how these formulae are applied to estimate the sediment export of the three major vegetation covers/land uses at sites within a farm.

## 6. Climate change and soil erosion in New Zealand pastoral hill country

Climate change will have measurable effects at the farm scale in our lifetimes, whether through pasture composition changes, increased productivity of some pastures, increased loss of productive soil from underprotected landscapes, a greater frequency of drought years or general changes in water availability. It is not clear at this stage where the balance between positive and negative impacts will lie. However, it appears very plausible that proactive adaptation to these changes will help landowners to shift the balance more towards the positive side of effects. Ongoing research will help address some of these issues and provide a better basis for informed decision-making.

There is an increasing need for collaboration between New Zealand scientists working on erosion rates, climate change scenarios, soil conservation methods, and sediment management as the problems become more complex. Coupled with this increased collaboration is the equally important requirement to communicate scientific findings to both policy makers and landowners.

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# 12

# Erosion Control in Furrow Irrigation Using Polyacrylamide

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## 1. Introduction

Currently gravity irrigation remains the most widely used in agricultural areas and 90 percent of plantings, the water is applied to the plots by gravity. One of the main problems with these systems is erosion, which is a by product of the erosive forces of water over the row, which brings soil loss and therefore decreases in crop yield.

Erosion is the removal of surface soil material caused by water or wind (Kirkby, 1984). It is caused by several factors, such as steep slopes, climate, inadequate use of soil, vegetation cover and natural disasters; however, human activities can greatly accelerate erosion rates. This phenomenon is considered a severe problem because it is associated with inappropriate agricultural practices, overgrazing, poor utilization of forests, thickets, grasslands, forests, and changes in land use from forest land primarily for agricultural purposes.

According to Becerra (2005), the two main types of erosion are geological erosion and accelerated erosion. Geological erosion includes both training and erosive processes, which maintain the soil in a favorable balance, suitable for plant growth. Accelerated erosion and loss of soil degradation is a result of human activities.

Geological erosion when the soil is found in its natural environment under the cover of native vegetation. This type of erosion is responsible for the formation of soils and its distribution on the Earth's surface. The long-term effect of this type of erosion has led to larger landscape features such as canyons, meandering rivers, and valleys. In other words, this type of erosion is the result of the action of water, wind, gravity and glaciers.

Accelerated erosion, soil loss is usually associated with changes in vegetation and soil conditions and is caused mainly by water and wind. The forces involved in accelerated erosion are: (1) attack forces, which remove and transport the soil particles, (2) resistance forces, which limit the erosion.

Soil erosion is the main source responsible for the gradual decrease in fertility and therefore the productive capacity of soils. Erosion caused by hydric erosion, include the action of rain and runoff.

In general, water erosion is divided into erosion or splashing raindrops, sheet erosion, rill erosion, gully erosion and irrigation channels.

Soil splashing occurs when raindrops fall directly onto the soil particles or very thin areas of water, spraying huge amounts of soil due to the kinetic energy of impact. In plane soils, the dispersion of soil is more or less uniform in all directions, but on sloping soil will be a net transport downhill. If overland flow occurs, the removed particles are incorporated into the water flow and will be transported downstream before being deposited on the surface again.

Sheet erosion removes soil evenly in thin layers, due to the laminar surface flow in thin layers that runs along the soil. Raindrops cause the detachment of soil particles, increasing the sediment movement by filling the pores of the surface layer, reducing the rate of infiltration. The abrasive force and the drag of the laminar flow are a function of the depth and speed of runoff for soil particle or aggregate size, shape and density determined.

Rill erosion occurs when surface flow is concentrated, the water acts on the soil detaching and causing channels or small streams. These types of channels become stable and are easily seen. The detachment and transport are more severe because the speeds of moving water are higher runoff and hydraulic shear stress increases with the degree of slope and hydraulic radius of the section of the channel.

Gully erosion in open channels above the grooves, which collects water during or immediately after rainfall. Gully erosion, which causes a late stage that produces rill erosion, just as it is a post-sheet erosion.

The erosion in irrigation channels is due to soil detachment and transport of it are more severe, thus the flow velocity in the channel is greater than that caused by rain effects. Soil detachment increases with the degree of slope and hydraulic radius of the section of the channel.

One of the main mechanisms that cause water erosion is the formation of surface sealing when the soil is exposed to the action of the impact of raindrops and concentrated flows in the rills (Orts et al., 2000).

Seal formation is the result of two complementary mechanisms (Yu et al., 2003): a) physical disintegration of surface soil aggregates, and b) the physicochemical dispersion of clay, moving to deeper soil layers by infiltrating water. These block the pores below the surface and form a low permeability layer called "washing area".

Given these aforementioned problems, many forms are viable and economic alternatives, being the application of polyacrylamide (PAM) which is one of them. This has been as soil conditioning since 1950; however, the expansion of its use was not seen until the last decade (Green & Stott, 2001). When applied to soil it, increases the aggregate stability, reduce the release and transport of sediments, flocculate the suspended sediment, increases infiltration (Norton et al., 1993; Lentz et al., 2001; Leib et al. 2005) and is a non-toxic product whose by mechanical interaction degrades into CO2, water and nitrogen.

The PAM is a water soluble polymer with the ability to enhance soil stabilization. It is grouped in a class of compounds formed by polymerization of acrylamide (Lentz et al., 2001). Pure PAM is a homopolymer of identical units to that of acrylamide. The molecular weight gain increases the length of the polymer chain and consequently the viscosity of the PAM solution. It is currently used in the construction and agriculture, as a soil conditioner it on the anionic polymer of high molecular weight (10 - 20 mg $mol^{-1}$), whose structure is shown in Figure 1.

The mechanism responsible for reducing runoff and soil loss, and therefore the final increase in infiltration is related to the ionic strength of the PAM in the soil solution (Norton et al., 1993; Santos et al., 2000). Therefore, in the soil solution decreased clay dispersion and flocculation aid, according to the theory of diffuse double layer (Van Olphen, 1977).

The diffuse double layer is compressed to the surface of clay when the electrolyte concentration is increased and decreases the separation of clay particles. Due to compression of the double layer, the range of the repulsive forces is greatly reduced (Van Olphen, 1977), thereby promoting flocculation.

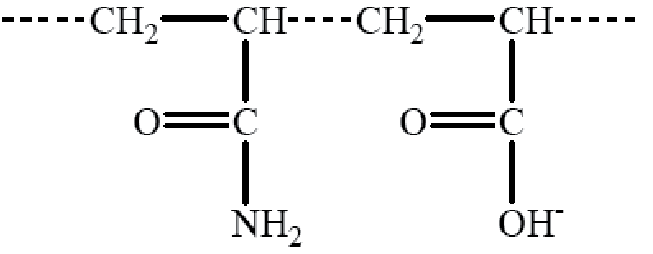

Fig. 1. Molecular structure of anionic polyacrylamide

Several studies have shown that PAM dissolved in irrigation water at a rate of 10 kg $ha^{-1}$ improves water infiltration (Leib et al., 2005; Chávez et al., 2010) and may be increased from 7 to 8 times the final infiltration as compared with the control (Ajwa & Trout, 2006). However, at rate of 20 kg $ha^{-1}$ applied in granular was this has proven to be effective in controlling erosion (Lentz & Sojka, 2000; Chávez et al., 2009), if applied in the first 5 mm of soil, it will reduce runoff by up to 30% as compared with the control plot (Yu et al., 2003), and may be increased by up to 54% of the aggregates stability (Lentz et al., 2001; Shrestha et al., 2006).
According to Leib et al. (2005) polyacrylamide application prior to irrigation control the erosion caused by concentrated flow in furrow irrigation systems, reducing soil detachment caused by hydraulic shear stresses. Wetting fronts are broader and infiltration is higher in the rows treated with polyacrylamide as compared with those that are untreated (Yu et al., 2003).
The use of PAM as alternative practice of soil conservation has been repeatedly proven to be an effective and viable (Bjorneberg et al., 2000; Santos et al., 2000; Lentz et al., 2001; Yu et al., 2003; Kornecki et al. 2005; Shrestha et al., 2006; Chávez, 2007; Chavez et al., 2009). However, this effectiveness depends on the type and charge density and molecular weight (Green et al., 2001). PAM with high molecular weight and low concentrations in the irrigation water has given better results for erosion control (Lentz & Sojka, 2000; Bjorneberg et al., 2003; Shrestha et al., 2006; Chávez, 2007; Chavez et al., 2009).
The application of PAM to soil is a viable way to control erosion, however, there are no current studies comparing the methods of implementation; therefore, the aim of this study was to evaluate three forms of application of PAM in a groove to find the one that is most effective in controlling soil erosion caused by concentrated flows, using a rate of 20 kg $ha^{-1}$.

## 2. Theory

Soil erosion process is associated with the action of two forces: hydraulics and resistance: the first break and remove the particles and carry them through the channels and the second, due to electrochemical nature, somehow prevents the detachment. The shear stress acting on the bottom of a river or channel, or on the soil surface, is one of the most significant variables of hydraulic power. Calculation of this is derived from the momentum equation for uniform flow in an open channel (Chow et al., 1988).
Rill erosion is a phenomenon that involves the detachment of soil particles and transports them due to the drag force of flowing water. The deposit of sediment is the result of the

previous two phases, which are hauled through the grooves and led to the boundaries of the land, bringing with it problems of sedimentation in drainable networks, which are then translated into economic losses for users because the rehabilitation work needed for optimal functioning is expensive.

The basic equation in the process of erosion in rills and interrill is the sediment continuity equation for unsteady flow with little depth (Foster, 1982):

$$\frac{\partial q_s}{\partial x} + \rho_s \frac{\partial (cy)}{\partial t} = D_r + D_i \tag{1}$$

where $q_s$ is the sediment load [$ML^{-1}T^{-1}$], x the downstream distance [L], $\rho_s$ the particles density [$ML^{-3}$], c the sediment concentration [$ML^{-2}T^{-1}$], y the flow depth [L], t the time [T], $D_r$ the rill erosion [$ML^{-2}T^{-1}$], and $D_i$ the interrill erosion [$ML^{-2}T^{-1}$]. The erosion parameters $q_s$, $D_r$, and $D_i$ are measured per unit width of the channel.

For shallow flows and gradually varied the term $\rho_s \partial (cy)/\partial t$ can be neglected, resulting in the continuity equation that is widely used for permanent flows:

$$\frac{dq_s}{dx} = D_r + D_i \tag{2}$$

The shear stress along the boundaries of the flow, leads to incision of the walls of a channel as long as such efforts exceed the tractive force or critical shear stress. The detachment of soil particles wetted perimeter can be well described by:

$$D = K(\tau_s - \tau_c)^{\alpha} \tag{3}$$

where D is the detachment of soil in a wet perimeter point [$ML^{-2}T^{-1}$], K the soil erodibility factor [$L^{-1}T$], $\tau_s$ the hydraulic shear stress acting on a surface wetting perimeter [$ML^{-1}T^{-2}$], $\tau_c$ the critical shear stress [$ML^{-1}T^{-2}$] and $\alpha$ has a value of 1.05.

Soil detachment in the rills due to incision is proportional to excess shear stress with respect to its critical value, ie $\alpha$ takes the value of 1.0, and K = $K_r$, called erodibility factor in the rill [$L^{-1}T$], this is (Foster, 1982):

$$D_r = K_r(\tau - \tau_c) \tag{4}$$

where $D_r$ is the detachment of soil in the furrow [$ML^{-2}T^{-1}$], ie the mass of loosened soil in unit time per unit area, $\tau$ the hydraulic shear stress in the rill bed [$ML^{-1}T^{-2}$], $\tau_c$ the critical shear stress ensures that the soil particles are detached [$ML^{-1}T^{-2}$].

Soil resistance to shear forces of flowing water is called the critical shear stress ($\tau_c$), or also tractive force, this value is the value of the regression line when it crosses the x-axis, ie when soil detachment begins by concentrated flow effect. For cohesionless soils, the Shields diagram is the method used to describe the tractive force of the individual particles.

According to Alberts et al. (1989) for cohesive sediment, the individual grains of sediment lie and remain in the background because of their own weight and resist horizontal movement due to friction with the adjacent grains. Therefore, the stabilizing force is associated with the submerged weight of individual grains. Whereas from a critical shear stress, the sediment may start to move, the Shields parameter is an expression that denotes the situation where the sediment is about movement, where the drag force equals the friction velocity. The sediment starts to move when the cutting speed of flow is greater than the critical shear rate.

For non cohesive materials, the critical shear stress has been associated with many soil properties, including the cutting force, salinity and moisture content (Alberts et al., 1989), and the percentage of clay, the average particle size, percentage of dispersion, organic matter content, cation exchange capacity, ratio of calcium - sodium and plasticity index (Prosser & Rustomji, 2000).

The typical range of critical efforts to cut agricultural soils is 1 to 3 Pa; however, Foster & Meyer (1975) recommended an average of 2.4 Pa. On the other hand, Alberts et al. (1989) developed a regression equation using an extension of the Water Erosion Prediction Project (WEPP model) with field data and found that the critical shear stress for agricultural soils can be estimated based on: the fine fraction of sand, fraction calcium carbonate, sodium adsorption ratio, specific surface area, clay fraction dispersed in water and clay fraction.

In agricultural soils with a clay fraction larger than 0.30 mm, Alberts et al. (1989) found that the tractive force of the soil can be predicted from the volumetric water content. Other relationships have been developed from data obtained from the WEPP model and the results are different from the original relationship.

Conceptually, the critical shear stress total of flow in a channel can be divided into two components: the roughness of the grain and form roughness (Graf, 1971). The effort hydraulic roughness of the grains is responsible for erosion and sediment transport. The total soil hydraulic effort is a combination of grain roughness and form, however, the form roughness is larger than the roughness of the grains; therefore, in the case of detachment in a channel it is necessary to know the stress distribution along the borders, ie the stress on the bed $\tau$ channel for uniform flow as given by:

$$\tau = \gamma_w R_h S_o \tag{5}$$

where $\gamma_w$ a is the specific weight of water [$ML^{-3}$], $R_h$ the hydraulic radius [L]; and $S_o$ the slope of the furrow [$LL^{-1}$].

Because the shear stress distribution in the bed of the rill is not uniform, use of an average value thereof, which is considered as a potential detachment, but this can result in significant errors in the estimation of $\tau_c$ (Foster , 1982).

## 3. Application

The experimental work was developed in the hydrological module of the Faculty of Engineering of the Universidad Autonoma de Queretaro. PVC pipe class 14 of 30 cm in diameter was used and cut in half to form the simulated rill. The dimensions of the circular channel half-circle formed were: 0.30 m wide x 0.25 m deep at the center and 6 m in length, which was settled the soil at depth of 0.20 m at the center, with a bulk density similar to that observed in the field. Given the channel slope was 3%, the same as was achieved by three supports placed at the ends of the channel and in the center of it, see Figure 2 (Chávez, 2007).

The soil used, according to the FAO-UNESCO classification (1988), is a Pelic Vertisol representative of the study area.

The inflow water was provided to the system by a constant head tank of 60 liters, placed 2 m above the reference level, and water was supplied by a ¾ HP pump connected to a tank with capacity of 1000 liters. Inflow water in the furrow was regulated by a butterfly valve and measured by a flow meter, see Figure 3. The initial flow was 75 l $h^{-1}$, the second flow was 100 l $h^{-1}$, then further increases were 50 l $h^{-1}$ until the maximum flow of 250 l $h^{-1}$ was reached.

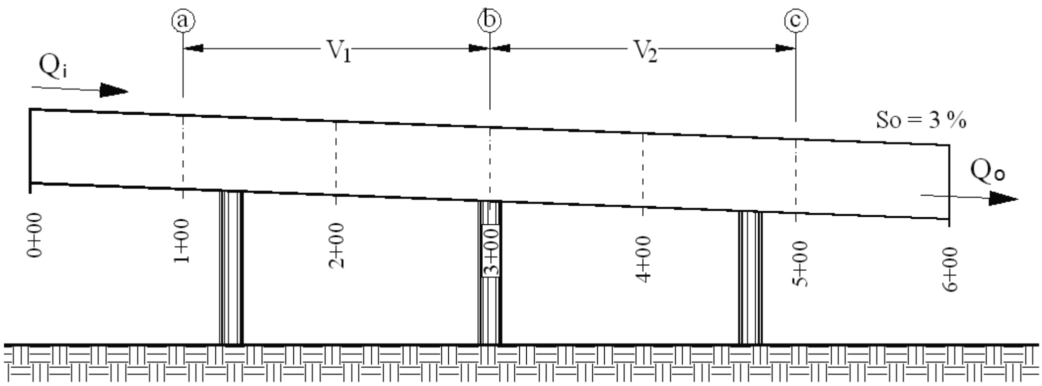


Fig. 2. Scheme of channel

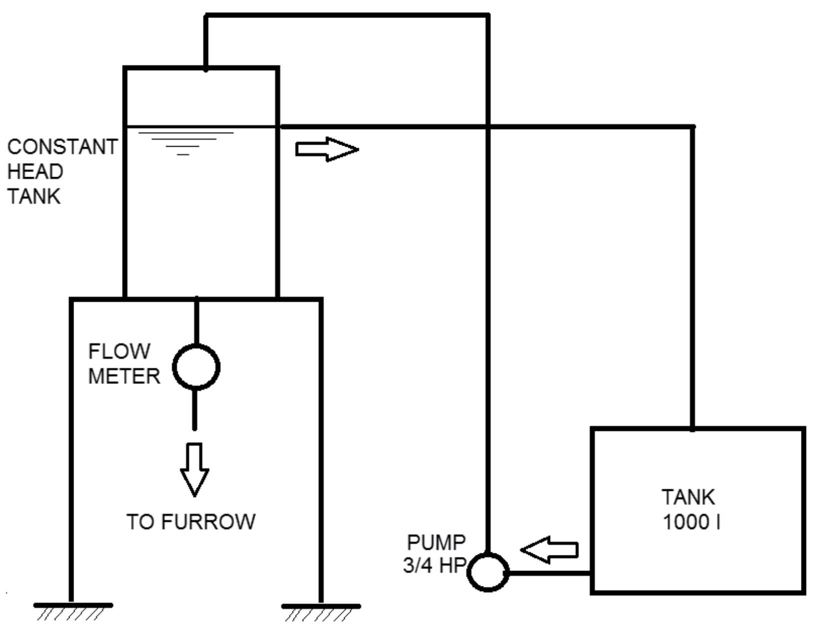


Fig. 3. System of water flow in the experiment

### 3.1 Treatments applied

Three treatments of PAM were applied on the soil at rate of 20 kg $ha^{-1}$: PAM applied as a granular, diluted PAM and injected into the inflow and diluted PAM and sprayed on the furrow bed, which were compared with measurements made in a furrow without a PAM application.

### 3.2 Measurement of variables

For the analysis of soil detachment $D_r$, critical shear stress $\tau_c$ and the shear stress $\tau$ were measured flow parameters such as speed and width of the water surface, which serves to identify areas of flow, wetted perimeter, hydraulic radius, and other parameters needed to determine the amount of detachment and forces acting on the rill. Plastic bottles was placed at the end of the rill in order to collect the runoff and sediment samples for laboratory evaluation. The velocity and width water surface parameters were measured three times directly into the rill at points a, b and c, see Figure 2, and for calculations the averages in each of flow quantified were used. The boundaries of these three sections correspond to specific sites where measurements of water velocity ($v_1$ and $v_2$) were made.

In this research it was assumed that the channel is rectangular shape, based on the progressive erosion of the flow before reaching a non-erodible soil layer takes this form (Lane & Foster, 1983). The calculation of flow depth was from the continuity equation: Q=Av, where Q is the flow and v the flow velocity in the channel. Considering that the area is given by A=bh, where b is the channel width and h the head, and it will have h=Q/bv. The speed is obtained as the average $v = \frac{1}{2}(v_1 + v_2)$. These data were used in the calculation of shear according to equation (5).

### 3.3 Calculation of the detachment rate Dr and critical shear stress $\tau_c$

The calculation of the detachment rate $D_r$, was made with the amount of sediment collected in the containers for each of the measurements made in the different treatments, at a time and a rill area known as ($gm^{-2}s^{-1}$). The calculation of shear stress [Pa] was made using equation (5), where specific weight $\gamma_w = 9879$ $Nm^{-2}$, was taken assuming a constant temperature 20°C.

### 3.4 Measurement of erosion

The runoff and sediment samples were taken once the inflow was stable, using plastic bottles wide necked of a one liter capacity. The collected samples were weighed, flocculated with 10 ml of a saturated solution of aluminum sulfate, decanted and dried in an oven at 105 °C to constant weight.

## 4. Results

Table 1 shows the values of parameters obtained by fitting the experimental data of detachment and shear stress by the method of least squares of equation (4), ie: erodibility factor in the furrow $K_r$, the critical shear stress $\tau_c$, the coefficient of determination $r^2$, and the value of the soil detachment rate in the furrow $D_r$ with maximum flow of 250 l $h^{-1}$.

### 4.1 Detachment rate

The furrow detachment rate $D_r$ obtained for the control was 6.9 $gm^{-2}s^{-1}$, but this value was significantly reduced in 67.6% with PAM applied in granular form to register a detachment of 2.3 $gm^{-2}s^{-1}$. This reduction was more pronounced when the polymer was applied diluted and injected into the inflow which is a common practice carried out in irrigation and sprayed on the soil; therefore, obtaining a reduction of 85.1% and 96.2% respectively, which is similar to results obtained by Lentz et al. (2001).

| Treatments | $D_r$* | $K_r$ | $\tau_c$ | $r^2$ |
|---|---|---|---|---|
| | [$gm^{-2}s^{-1}$] | [$m^{-1}s$] | [Pa] | |
| **Control** | 6.9436 | 4.218 | 0.6498 | 0.9328 |
| **Granular PAM** | 2.2521 | 2.116 | 1.0770 | 0.9108 |
| **Injected PAM** | 1.0371 | 1.261 | 1.1084 | 0.9027 |
| **Sprayed PAM** | 0.2662 | 0.757 | 1.3923 | 0.9178 |

Table 1. Summary of results obtained from the application of PAM. Linear Equation. (*Maximum flow 250 l $h^{-1}$).

The reduction in soil detachment is due to the length of the chain anionic PAM presented, which when in contact with the negative charges of clays binds to it forming stronger bonds, providing greater resistance to soil evolution (Lentz, 2003). Consequently, when irrigation water comes in contact with the soil, and the PAM has been diluted and previously applied it has already reacted to the soil by providing greater cohesion, otherwise, the polymer applied granular form, which interacts with soil particles until it is dissolved with water.

Erosion data obtained from the furrow are represented in Figure 4 with a linear fit, where the reference furrow recorded an erosion rate of 0.30 g $l^{-1}$ with initial flow and increasing the flow rate 100 l $h^{-1}$ increased the release to 18.30 g $l^{-1}$, whereas with 150 l $h^{-1}$ the increase was only 5.7 g $l^{-1}$, an increase of 150 l $h^{-1}$ at 200 l $h^{-1}$ had a low impact on the rate of sediment (6.3 g $l^{-1}$), and with a flow of 250 l $h^{-1}$ there was an increase of 16.3 g $l^{-1}$ as compared to the previous test.

Soil detachment began treatment with the granular PAM with a flow of 150 l $h^{-1}$ and an erosion rate of 0.4 $gl^{-1}$. With the same flow, this initiates the release with the injected PAM in the inflow, but reported only 0.1 g $l^{-1}$, while with the sprayed PAM on the soil there was no detachment. An important difference was observed in the behavior of $D_r$. The values were approximately 1 which are very similar in the furrow control applying a flow of 100 l $h^{-1}$ and the furrow treated with injected PAM in the inflow applying a flow of 250 l $h^{-1}$, while values of $D_r$ for treatments with the sprayed PAM on the soil are still below these values.

The critical shear stress, defined as the point from which particles star to detach and transport the soil, present significant differences using a significance level $\alpha$ of 0.05 in student t-test between the control and treatments ($p < 0.0001$). Therefore, to control the critical shear stress by 0.65 Pa was obtained, whereas treatment with granular PAM is 1.12 Pa, which means that the application of PAM increases the soil resistance to erosion in furrows to increase the value of $\tau_c$. This increase was less pronounced in treatment with injected PAM in the inflow, with 1.11 Pa the critical shear stress; however, the furrow treated with sprayed PAM on the soil had a greater effect recorded value of 1.39 Pa. This increase is approximately double that of the strength of soil resistance to detachment caused by concentrated flows into the furrows with respect to the control treatment. The detachment rate with the inflow of 250 l $h^{-1}$ can be seen in Figure 5.

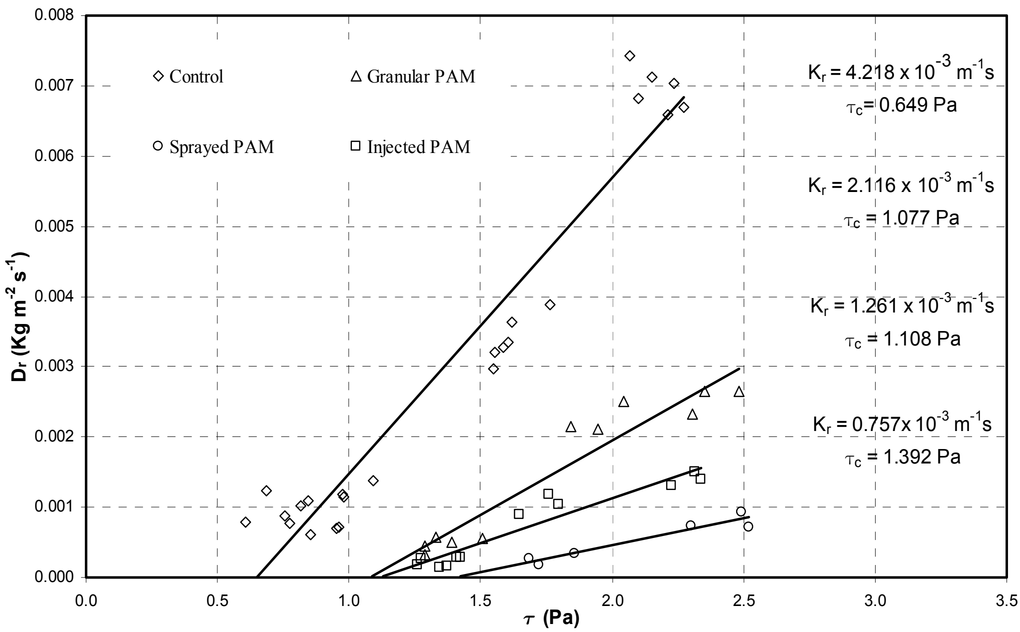


Fig. 4. Linear relation of the soil detachment with the treatments

### 4.2 Rill erodibility factor Kr

The erodibility factor $K_r$ decreases by 50% as compared to the control as a result of the addition of PAM to soil in granular form. This change is associated with the PAM changes the physical and chemical properties of soil, with greater cohesive strength of particles and improving the stability of the aggregates. However, the soil response to the application of PAM is different in each treatment. Considering only the three forms of implementation of PAM shows that the average value of $K_r$ for granular application is 2.11 m $s^{-1}$, this decreases to 1.26 m $s^{-1}$ and 0.75 m $s^{-1}$ with sprayed PAM on the soil and injected in the inflow, respectively, see Figure 6. Therefore, the regression slope indicates that the application of polymer sprayed on the soil, is the most efficient way to control erosion.

The data ($\tau$, $D_r$) were fitted to a straight line, but it is possible that the erodibility factor itself depends on the shear stress at the furrow bed. Consequently, the data also were fitted with equation (3). Equations (3) and (4) are equivalent, if the erodibility factor is the next unit:

$$K_r = K\left(\tau - \tau_c\right)^{\alpha-1} \quad (6)$$

Estimates of coefficient values K [ $M^{1-\alpha}L^{-2+\alpha}T^{-1+2\alpha}$ ] and from the exponent $\alpha$ of equation (3) were obtained with the method of least squares, using equation (3) in the form $\tau = \tau_c + \left(1/K\right)^{1/\alpha} D_r^{1/\alpha}$ . Values are reported in Table 2 and shown graphically in Figure 7.

Control

Granular PAM

Injected PAM Sprayed PAM

Fig. 5. Detachment in the furrows treatments with PAM and the control

Making a comparison between the linear and the potential model shows that the critical shear stress for untreated furrow decreases from 0.649 to 0.0035 Pa, which indicates that the erosion starts when the irrigation water comes in contact with the surface soil. The critical shear stress in the soil application of granular PAM increased 0.267 Pa, while injected PAM into the inflow increased 0.221 Pa. In addition, with the relationship potential, sprayed PAM on the soil decreased by 0.237 Pa soil strength in relation to the linear model.

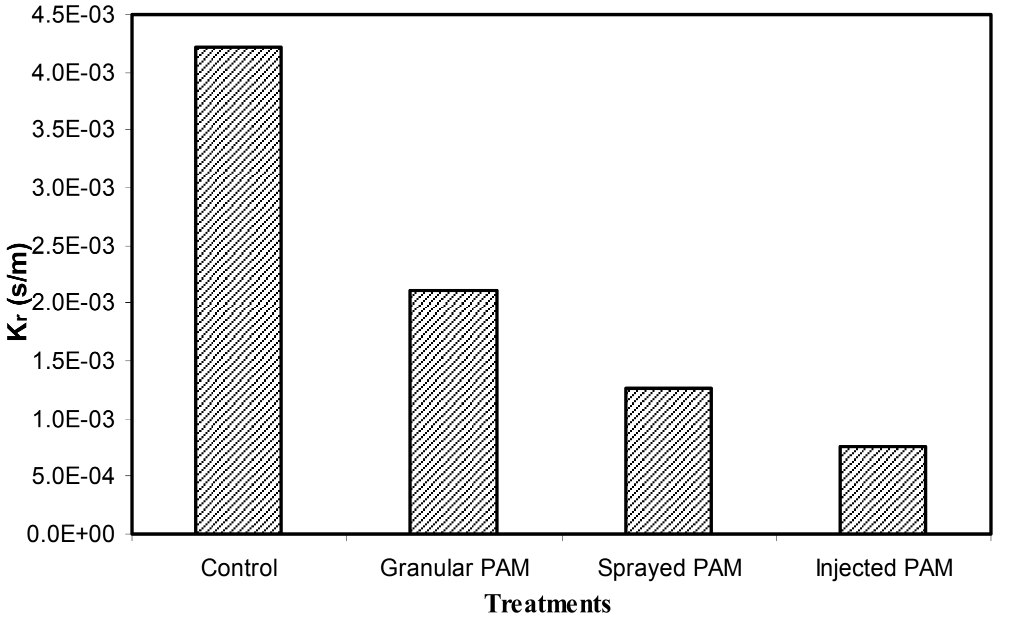


Fig. 6. Erodibility factor in the furrows treatments with PAM and the control

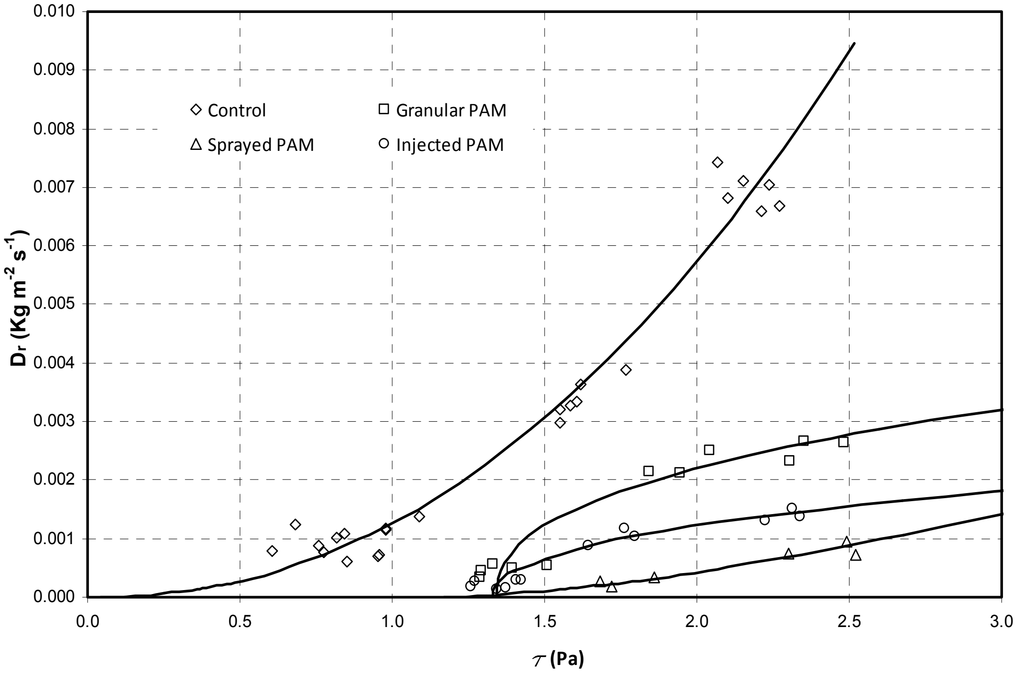


Fig. 7. Potential relation of the soil detachment with the treatments

If the linear model is taken as the valid model, the erodibility factor $K_r$ in the furrow remains constant; however, as shown in the potential model, this value is dynamic, ie varies in function of the applied flow.

| Treatments | $D_r^*$ | K | $\tau_c$ | $\alpha$ | $R^2$ |
|---|---|---|---|---|---|
| | $gm^{-2}s^{-1}$ | $g^{1-\alpha}m^{-2+\alpha}s^{-1+2\alpha}$ | Pa | | |
| **Control** | 6.9436 | 1.2727 | 0.0035 | 2.1763 | 0.9560 |
| **Granular PAM** | 2.2521 | 2.6121 | 1.3443 | 0.4024 | 0.9324 |
| **Injected PAM** | 1.0371 | 1.4598 | 1.3290 | 0.4319 | 0.9489 |
| **Sprayed PAM** | 0.2662 | 0.5539 | 1.1546 | 1.5408 | 0.9225 |

Table 2. Summary of results obtained from the application of PAM. Potential Equation. (*Maximum flow 250 l $h^{-1}$).

### 4.3 Sediment loss

Sediment loss as shown in Figures 8 and 9 , where the difference between the control furrow with respect to the furrows treated with PAM is significant. For a flow of 100 l $h^{-1}$, the control furrow lost about 20 g per liter of water passing through the furrow, while the PAM-treated furrows have no losses.

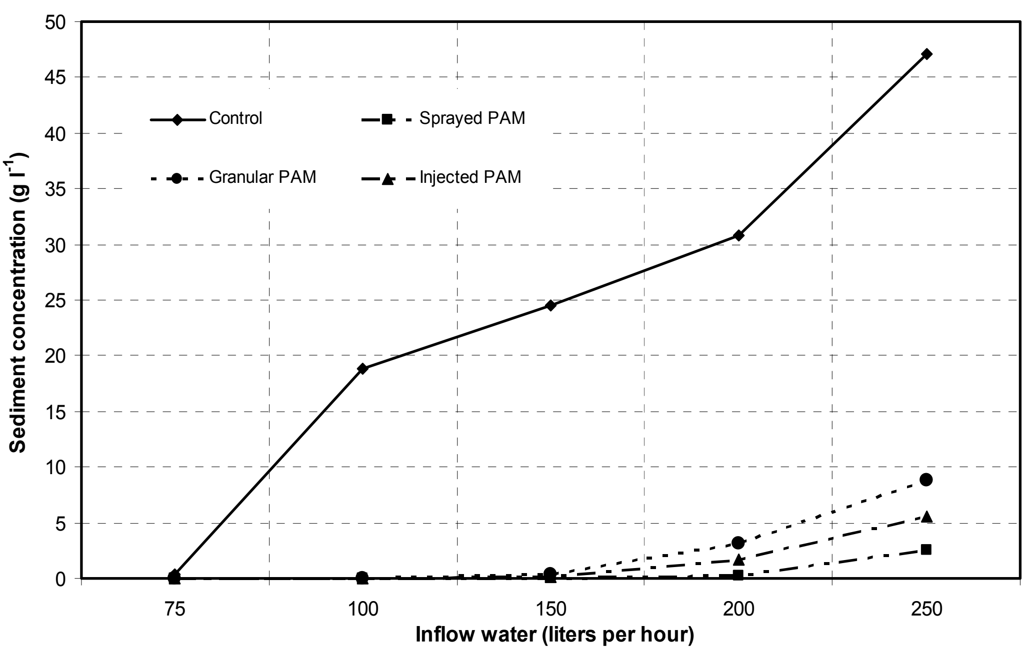


Fig. 8. Soil loss associated with different flow rates in all treatment

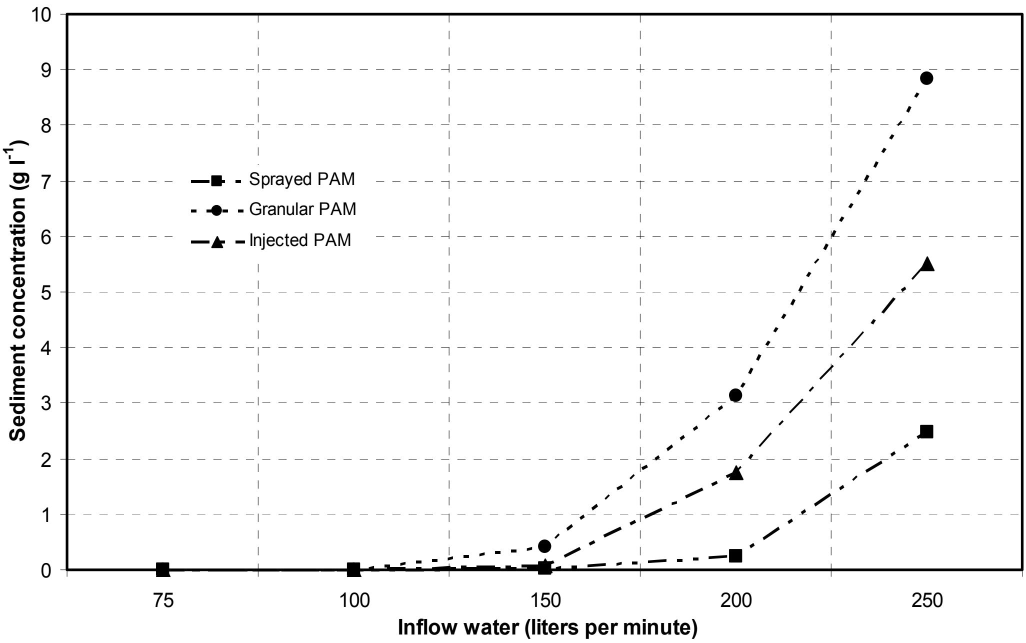


Fig. 9. Soil loss associated with different flow rates in the furrow with PAM

The same figures show that with an inflow of 200 l $h^{-1}$ the furrow treated with sprayed PAM has a sediment concentration of only approximately 0.20 g $l^{-1}$, injected PAM 1.80 g $l^{-1}$ while the injected PAM and sprayed PAM lost 3.10 g $l^{-1}$, as compared with the reference furrow loses on average 30.8 g $l^{-1}$, for an efficiency of 90-99% in the control of erosion. Orts et al., (2000) obtained a 97% reduction in soil erosion by applying PAM; however, the flow rate used was 23 l $h^{-1}$.

On the other hand, with flow of 250 l $h^{-1}$ the control furrow lost about 45 g $l^{-1}$ while the furrows treated with granular PAM saw loses 9 g $l^{-1}$, and sprayed PAM on the soil saw loses of less than 5 g $l^{-1}$. Therefore, reducing sediment loss is 80-94%. The efficiency of PAM decreased, but due to the lack of measurement with higher flow rates to 250 l $h^{-1}$, one can not infer that this decrease is progressive. It would be necessary to carry out investigations to see if the trend is the same.

## 5. Conclusion

The application of polyacrylamide to the soil in any of the forms of application helps to reduce the detachment of soil particles caused by the hydraulic efforts and the critical shear stress. However, liquid application is more effective in controlling erosion.

With the PAM implementation the sediment loss in the control furrow (45 g $l^{-1}$) was reduced to 10, 5 and 0.2 g $l^{-1}$ with applications of granular PAM, diluted and injected in the inflow, and diluted and sprayed on the soil, respectively. This is because the PAM provided the cohesion between soil particles, increasing by more than one order of magnitude resistance to detachment.

The values of critical shear stress and rill erodibility factor are different between the linear model and potential model; however, the linear model over estimates the value of critical shear stress of control treatment, and for treatments with PAM, this value is sub estimate except with the PAM sprayed treatment. The furrow erodability factor obtained with the model potential is not constant; however, there is a need to experiment with higher flow rates than those applied in this experiment to see if the trend continues or there is a change.

Finally, the properties of the soil type and amount of clay, type of ions in solution and the ionic strength of soil solution, and the pH affect the efficiency of polyacrylamide to control soil erosion. Therefore, if in addition to applying the polymer, combined with other soil conservation practices, the results obtained will be better.

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# Soil Erosion Control on Arable Lands from North-East Romania

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## 1. Introduction

Since the beginning of its formation, the crust of the Earth has evolved under the action of the morphogenetic processes occurring differently as periodicity and intensity as a result of the interacting processes that change the surface of the earth at the levels of interference of the lithosphere with the atmosphere, hydrosphere and biosphere.

As part of the exogenous morphogenetic processes, soil erosion plays an important role in the dynamics that shape the crust of the Earth. Erosion is the process by which soil or rock particles detach themselves from the land surface, being carried from their place of origin and deposited elsewhere (Selby, 1993).

The three stages of natural erosion are performed by two main agents: water and air in motion, whose inexhaustible kinetic sources are solar energy and gravity.

Soil tillage using tools also makes the human factor an erosional agent because the three phases of erosion can be distinguished in this activity. Unlike other soil erosion agents, human activity can be directly controlled and rationally directed.

Soil erosion is one of the main causes of vast agricultural and forest degradation on the Earth.

In Romania, the theoretical and applied study of water erosion is of particular concern as the physical and geographical conditions of the greater part of the territory are influenced by this process.

About one third (i.e. 4918.8 thou ha) of the total area of agricultural lands is affected by erosion and landslides.

Among the agricultural uses, orchards are the most severely affected (65.6%), followed by natural grasslands (58.3%); the arable surface and landslides is about 20% of the total use category.

Soil erosion prevention and control still prevails on arable land as there are approximately 2.6 million hectares (i.e. 26% of the total arable land) whose slope is greater than 5%.

On the sloping lands where there are potential conditions for increased erosion, the first goal of the anti-erosion action is to reduce the annual loss to levels that can be compensated by the natural process of soil recovery.

In Romania, for the arable lands consisting of medium soils, the annual admissible erosion is considered 6 $t \cdot ha^{-1} \cdot year^{-1}$.

The decrease in the annual soil loss favours the preservation of nutrients and rain water retention, thus improving soil fertility of the sloping lands.

There are other important and inexpensive measures that can be taken for the sloping arable lands in order to maintain the soil losses within allowable limits: *crop structure and rotation*, *crops location on the slope*, *soil tillage* and *fertilization*.

*Crops structure.* Different plant combinations should be grown on the sloping arable lands in order to prevent erosion to exceed the limit. Research performed under various conditions allowed the calculation the best rate between the rows crops in the crop structure so that soil loss is maintained within tolerable limits. Thus, on an arable land with a slope of 10-12% and a loamy-clay soil in northern Moldavian Plateau, the annual soil loss was below 6 $t{\cdot}ha^{-1}$ at the highest rate of 62% row crops of the assortment of cultivated plants.

*Crops emplacement on the slopes*. For soil, water and nutrients loss to be insignificant, the selection of the appropriate range of plants to be grown on sloping arable land and their allocation within rational rotation should be associated with special systems for the location of plants growing on the slopes.

Strip cultivation and cover crop cultivation are the most frequently used anti-erosion systems.

Contour and strip cropping is a simple, effective and convenient system to control soil erosion on the arable land with slopes greater than 5-6%. It consists of cultivation on the slope, parallel to the level curves row crops on land strips alternating with the best protective plants (winter grains, annual legumes, etc.). Thus, the kinetic energy of the water runoffs along the width of strips grown with row crops is dissipated in the downstream area which is grown with crops planted in dense rows.

The control of surface runoff and the reduced erosion intensity depend on the width of the cultivated strips, which in turn depends on the slope, soil erosion susceptibility, rainfall aggression, etc. Determining the indicative value of the strip width can be made by using empirical relationships based on either the critical erosion velocity criterion, or the annually averaged admissible erosion.

Considering the admissible erosion of 6 $t{\cdot}ha^{-1}{\cdot}year^{-1}$, the values regarding the strip width are presented in Table 1.

| Slope (%) | Strip width for the soil with erodability: | | |
|---|---|---|---|
| | low | medium | high |
| 5 - 10 | 117 - 83 | 100 - 71 | 79 - 56 |
| 11 - 15 | 78 - 59 | 66 - 50 | 52 - 40 |
| 16 - 25 | 55 - 30 | 47 - 25 | 37 - 20 |

Table 1. Indicative values of cultivated strips width (m), depending on slope and soil resistance to erosion

The results of the research carried out at the Perieni Research and Development Centre for Soil Erosion Control emphasize that, irrespective of slope size, soil erosion decreases by 2 - 8 times in the strip planting system, compared with a maize-only crop grown on the slope (Table 2).

There was no significant yield increase in the early years of applying the strip planting system; however, as erosion decreased, the effects were cumulative and production increased significantly.

The location of the planted strips remained unchanged and, after 5-6 years, uneven forms occurred at the boundary between the consecutive strips which, if properly cultivated with grass and maintained, can become slopes agricultural terraces.

| Slope (%) | Crop and strip width (m) | Eroded soil | |
|---|---|---|---|
| | | t/ha | % |
| 12 | Wheat + maize 4 x 40 | 25,9 | 28,6 |
| | Wheat + maize 2 x 80 | 35,6 | 39,4 |
| | Wheat 120 + maize 40 | 15,6 | 17,2 |
| | Maize 120 + wheat 40 | 63,3 | 69,7 |
| | Wheat 160 | 5,8 | 6,4 |
| | Maize 160 | 90,3 | 100,0 |
| 14 - 16 | Wheat + maize 60 | 35,9 | 69,1 |
| | Wheat + maize 2 x 40 | 27,2 | 52,4 |
| | Wheat 60 + maize 20 | 13,3 | 25,6 |
| | Wheat 80 | 6,9 | 13,3 |
| | Maize 80 | 51,9 | 100,0 |

Table 2. Influence of strip planting system on soil erosion (data from RDCSEC Perieni)

The application of the stripe planting system is expected to result in about 8% extra work, compared with a single crop grown on the slope.

Contour buffer strips used to retain soil and reduce erosion. This type of buffer strip is simply a strip of perennial vegetation that is alternated with wider cultivated strips of cropland. Buffer strips system is recommended on arable lands with a slope greater than 8-10%, especially in areas of annual rainfalls exceeding 500 mm; however, the system can be applied successfully in dry areas as well, by selecting the appropriate grass assortment for strips sowing.

The width of land between the buffer strips is determined so that soil loss is limited to acceptable values.

The buffer strips are generally 4-6 m in width. Sometimes, width may increase to 8-10 m in the lowest third of the deep slopes with a convex profile and soils that are less resistant to erosion. Common practice uses strips whose width is equal to one or two working widths of the sowing machines employed in the area.

In case the slope surface is uneven, grass strips of variable width are recommended to achieve consistent width on the entire length of the cultivated strip between the strips, we recommend buffer strips of variable width.

The hydrological function of a grass strip is to intercept, partially detain and disperse the water runoffs from upstream. A rough grass reduces the transport capacity of the flow, and some part of the solid material carried by the water currents is discharged on the grass-covered area. Over the years, successive deposits contribute to transformation of the buffer strips into slopes of agricultural terraces provided that their location remains unchanged.

Many researchers have found positive effects of the buffer strips system of cultivation on leakage reduction and soil erosion. After twenty years of research in the Moldavian Plateau, for example, soil loss was found under the admissible limit on the protected arable areas, i.e. 3-4 times lower than on the lands with no buffer strips (Popa et al., 1984).

Better results can be obtained by applying a combination of strip and buffer strips systems. Among the advantages, there are:

- significant decrease in soil loss by the retention of water and eroded material both in the grass-covered strips and in the cultivated strips;

- the width of the grown strips can increase, thus leading to increased efficiency of the agricultural machinery and decreased production expenses per surface unit;
- the strips planted are better delimited, as they are bordered by the grass covered strips.

By applying the buffer strips system, the arable area decreases by 3-5 %; however, the loss is compensated and exceeded by the increased yield resulting from the grown strips and the hay yield obtained from the grass covered strips (3.5-4.0 t/ha).

*Soil tillage*. Of all the soil tillage works, ploughing is of the greatest interest in terms of anti-erosion actions. Concerning the direction, it was observed that ploughing along the steepest slope and, consequently, sowing downhill favour soil erosion to a great extent. By comparison, ploughing along the level curves helps reduce water loss by 75% and soil loss by 2-9 times. The best results were obtained when ploughing was performed with a reversible plough.

The experiments performed at the Perieni Research and Development Centre for Soil Erosion Control to determine the influence of the ploughing depth on the erosion of the arable lands points out that at moderate values of slope and medium soils with good natural drainage, increasing the ploughing depth over 20 cm is not justified in terms of erosion or increased yield in wheat and peas. However, in maize, ploughing at 30 cm depth reduced erosion by 28% while the water reserves available in the soil increased by over 300 $m^3$/ha, compared with 20 cm ploughing. These results prove the opportunity for deep ploughing in the hoed crops grown on the slopes.

Finally, the sloping land planting with no tillage, according to the *minimum tillage* or *no tillage* system was effective not only economically but also in terms of erosion as reducing the number of machine passes results in less soil compaction, structure destruction, and lower soil carried downstream through repeated mobilization, etc.

*Fertilization* of sloping arable land indirectly contributes to reducing soil loss by erosion. The rational application of fertilizers ensures a vigorous root system development and increases plant mass on the surface. Under these circumstances, water infiltration into the soil is significantly improved because the kinetic energy of the raindrops and surface runoff is reduced by enhancing the roughness created by the increasing volume of vegetation.

*Sial et al.* (2007) found that the average soil and humus loss in summer grains and potatoes were 3-4 times higher on the unfertilized land, compared with the fertilized variants.

The anti-erosion effect produced by the fertilization of sloping arable lands is considered higher in the plants of long vegetation time that are sown in thick rows.

In the Moldavian Plateau, located in north-eastern Romania, the relief is predominantly hilly. The average altitude is 250 m, with dominant slopes below 20%; however, there are also slopes whose geodeclivity varies between 25% and 35% or even more. The relief has evolved on complex lithological successions composed of sandy-loamy and marly rocks with intercalations of sands, sandstones and limestones.

The pedological cover is dominated by agricultural use, as types and subtypes are successive in the area - from less developed, in the East and South-East (mollisoils) to highly developed in the North-East (argiluvisoils). Here, the local soil formation factors have determined the occurrence of several insular or much elongated intrazonal areas.

The climate is temperate continental with large annual thermal amplitude; rainfalls occur mainly in the growing season and are unevenly distributed, usually as torrential rains.

Such natural conditions are favourable to slope processes, therefore it is necessary to take measures and apply works to maintain the tolerable limits of water erosion, and to contribute to the stabilization of the potential sliding surfaces in the risk areas.

## 2. Non-structural soil erosion control measures

### 2.1 Protecting the soil from erosion by cropping systems and fertilization

Protecting the soil from erosion is the first step toward a sustainable agriculture. The major costs to the farm associated with soil erosion come from the replacement of lost nutrients and reduced water holding ability, accounting for 50 to 75% of productivity loss (Pimentel D. et al., 1995).

In all the countries, the investigations carried out in the last period have followed the establishment of some technological solutions that maintain the productivity of agro-ecosystem and the protection of environment factors. The Soil Protection Framework Directive of EU includes the necessary legislative proposals, taken into account by all the member states concerning the three main threats on the decline in organic matter, soil erosion and contamination and some additional aspects regarding compaction, diminution of biodiversity, floods and landslides. In the EU, more than 150 million hectares of soil are affected by erosion and 45% of the European soils have a low content of organic matter (Russell et al., 2006).

In Austria, during 1994-2007, the mean soil losses in three locations dropped from 6.1 to 1.8 $t \cdot ha^{-1}$, by using conservation tillage in cover crops, and until 1.0 $t \cdot ha^{-1} \cdot year^{-1}$ with direct drilling. Nitrogen (9.2, 3.7 and 2.5 $kg \cdot ha^{-1} \cdot year^{-1}$) and phosphorus (4.7, 1.3, 0.7 $kg \cdot ha^{-1} \cdot year^{-1}$) losses showed similar tendencies (Rosner et al., 2008).

Of the total Italian area, 51.8% is considered to be at potential risk of desertification (Marchetti et al., 2008). Soil erosion is the most relevant soil degradation system that affects at least 19% of the territory at the potential risk of desertification, while aridity is the second desertification risk (19.0%) (Moraru et al., 2010).

The Directive 2006/42/EC proposes the identification of zones with erosion-degraded soils and organic matter in decline, for meeting the requirements of the United Nations Convention to Combat Desertification (UNCCD) in Northern Mediterranean and Central and Eastern European Country Parties.

In Romania, soil erosion is the most expensive degradation process, which affects almost 63% of the total area and 56% of the arable area from Romania. Investigations on the potential erosion, conditioned by geomorphologic, soil and climatic factors, have shown that in NE Romania, the mean soil losses by erosion were of 18.3 $t \cdot ha^{-1}$. The studies, carried out on the effective erosion, based on direct determinations and complex analyses, have shown that in the entire NE zone, the effective erosion had a mean value of 4.8 $t \cdot ha^{-1} \cdot year^{-1}$. The north-eastern region has 15.45% (2,131,421 ha) of the farming area of Romania (14,836,585 ha) and includes huge areas with soils affected by erosion (over 60%), acidification, compaction, landslides and other degradation forms (Bucur et al., 2007).

In Bulgaria, the investigations showed that the mean annual rate of erosion on the arable lands was 4.76 $t \cdot ha^{-1}$ and 2.69 $t \cdot ha^{-1}$ on improved arable lands. Soil losses by erosion on the fields ploughed on the upstream-downstream direction, which are cultivated with maize, are 7.48 $t \cdot ha^{-1}$. In sunflower, cultivated with conventional tillage, the annual eroded soil was 3.044 $t \cdot ha^{-1}$, and by wheat straw and green fertilizer incorporation into soil, erosion has decreased at 2.327 $t \cdot ha^{-1}$ and 0.937 $t \cdot ha^{-1}$, respectively (Totka et al., 2006).

The favorable influence of reduced tillage system and of crop residues on soil erosion was also shown by other scientists (Jha et al., 2010). In no-tillage system, soil losses by erosion were close to the ones found in case of soil protection with 6 $t \cdot ha^{-1}$ of mulch. On 8.5% slope fields from SW Finland, annual soil losses by erosion are 5- 6 $t \cdot ha^{-1}$ and leached nitrogen and phosphorus amounts are 15.0 and 1.1 kg·$ha^{-1}$·$year^{-1}$, respectively (Muukkonen et al., 2007). The investigations conducted in Minnesota, USA, have shown that 927, 1853 and 3706 kg·$ha^{-1}$·$year^{-1}$ of crop residues, applied in maize crops, have decreased soil erosion until 6.177, 1.730 and 0.988 t·$ha^{-1}$ respectively, and water runoff until 35.6, 25.4 and 22.9 mm, respectively (Lindstrom, 1986). The results concerning erosion in the Coshocton, USA, showed that in the areas annual mean soil losses by erosion were 1.18 t·$ha^{-1}$ (range, 0.35 t·$ha^{-1}$ in wheat and 7.36 t·$ha^{-1}$ in maize) (Izaurralde et al., 2007).

Investigations conducted during 1980-2010 on a cambic chernozem at the Agricultural Research and Development Station of Podu- Iloaiei, Iasi County, followed the influence of different crop rotations on water runoff and nutrient losses, due to soil erosion. Experiments were conducted on the hydrographical basin of Scobalteni, with a reception area of 159 ha, a mean altitude of 119.4 m and a mean slope length of 250 m. The area of the watershed has been anti-erosion set up since 1983, being used combined cropping systems made of sod rewetting and strip cultivation. The width of cultivated strips is 200-250 m on 5-10% slopes, 100-150 m on 10-15% slopes and 50- 100 m on 15-18% slopes.

The determination of runoff and soil losses by erosion was carried out by means of loss control plots with a collecting area of 100 $m^2$ (25 m x 4 m) and by means of a hydrological section equipped with spillway and limn graph and devices for sampling water and soil loss by erosion. Total nitrogen, nitrate, phosphorus and potassium contents were determined in soil and water samples, lost by erosion in different crops, thus establishing the losses of nutrient elements. The climate is temperate continental with large thermal amplitude and uneven and commonly torrential rainfall prevalent during the vegetative season. The climatic conditions in the Moldavian Plateau were characterized by a mean multiannual temperature of 9.6°C and a mean rainfall amount, on 80 years, of 559.2 mm, of which 161.2 mm during September-December, and 398 mm during January-August. Within the experiment, the following rotation scheme was followed: wheat and maize continuous cropping, 2-year crop rotation (wheat-maize), 3-year crop rotation (pea-wheat-maize) and 4-year crop rotation + outside field cultivated with legumes and perennial grasses (*Medicago sativa* + *Lolium perenne*).

The content of organic carbon was determined by the Walkley- Black method, to convert SOM into SOC it was multiplied by 0.58. The content in mobile phosphorus from soil was determined by Egner-Riehm-Domingo method, in solution of ammonium acetate- lactate (AL) and potassium was measured in the same extract of acetate-lactate (AL) by flame photometry. ANOVA was used to compare the effects of treatments. In wheat, we used Gabriela variety and in maize Podu-Iloaiei 110 Hybrid.

On slope lands, soil nutrient losses being very high, due to leaching, runoff and element fixing, the establishment of rates and time of fertilizer application must be done differently, according to soil characteristics, cultural practices and climatic conditions. On eroded slope lands, the growing systems ensure the reduction in soil losses below 4 t·$ha^{-1}$·$year^{-1}$ and allow getting efficient yields from the economic point of view.

The results on water runoff and soil losses in different crops from the Moldavian Plateau, determined by control plots, have shown that, during 1980-2010, of the total amount of 570.6 mm rainfall, 366.7 mm (64.3%) produced water runoff, which was between 6.4 mm in perennial grasses, in the second year of vegetation, and 30.6-36.4 mm, in maize and sunflower crops (Table 3).

| Crop | Rainfall causing runoff *(mm) | Runoff (mm) | Eroded soil ($t\cdot ha^{-1}$) | Mean turbidity ($g\cdot l^{-1}$) | Humus ($kg\cdot ha^{-1}$) |
|---|---|---|---|---|---|
| Bare fallow | 366.7 | 60.5 | 18.17 | 30.0 | 622 |
| Sunflower | 366.7 | 36.4 | 8.93 | 24.5 | 310 |
| I year perennial grasses | 366.7 | 19.1 | 1.91 | 10.0 | 65 |
| II year perennial grasses | 289.6 | 6.4 | 0.25 | 3.8 | 8 |
| Maize | 366.7 | 30.6 | 8.39 | 27.4 | 287 |
| Peas | 366.7 | 21.6 | 3.72 | 17.2 | 127 |
| Wheat | 337.9 | 11.5 | 1.62 | 14.1 | 56 |
| Beans | 366.7 | 24.6 | 4.56 | 18.5 | 156 |
| Soybean | 366.7 | 20.6 | 3.89 | 18.9 | 133 |

* Mean annual rainfall, recorded during 1980 - 2010 - 570.6 mm, rainfall causing runoff – 366.7 mm

Table 3. Mean annual runoff and soil losses by erosion, recorded in different crops

The annual soil losses due to erosion, recorded at the same period, were between 0.25 $t\cdot ha^{-1}\cdot year^{-1}$ in perennial grasses, and 8.39-8.93 $t\cdot ha^{-1}\cdot year^{-1}$ in maize and sunflower crops. The obtained results on the potential erosion (conditioned by geo- morphological, soil and climate factors) have shown that on the fields uncovered by vegetation from the Moldavian Plateau, the mean soil losses due to erosion were 18.17 $t\cdot ha^{-1}$, values corresponding to a moderate erosion risk. The protection degree of soil against erosion, expressed by the ratio between the value of the effective erosion (under specific technological conditions) and of the potential erosion (soil eroded under conditions of uncovered soil, which was not set up with soil erosion control works) is an indicator of erosion risk that shows soil vulnerability to erosion. It is given by the ratio between the value of the effective erosion and that of mean allowable erosion, which corresponds to soils from the studied watershed. Taking into account that the erosion process cannot be avoided and the tolerance level of soil annual losses is 6 $t\cdot ha^{-1}\cdot year^{-1}$, which corresponds to the annual rate of soil renewal, the mean annual soil losses due to erosion, recorded during 1980-2010 in maize (8.39 $t\cdot ha^{-1}$) and sunflower (8.93 $t\cdot ha^{-1}$), may result in destructing the fertile soil layer in a few decades.

Erosion has affected soil fertility by removing once with eroded soil, high amounts of organic carbon and mineral elements, which reached 16.23-17.61 $kg\cdot ha^{-1}$ nitrogen, 1.15-1.29 $kg\cdot ha^{-1}$ phosphorus and 2.02-2.39 $kg\cdot ha^{-1}$ potassium, in maize and sunflower crops (Table 4). On 16% slope lands, the mean annual nitrogen, phosphorus and potassium leachates, due to erosion, recorded during 1980-2010, were between 9.77 and 21.29 $kg\cdot ha^{-1}$ in row crops (soybean and sunflower) and between 4.48 and 9.39 $kg\cdot ha^{-1}$ in wheat and pea crops (Table 4).

The obtained results on erosion in different crop rotations have shown that under conditions of 16% slope lands from the Moldavian Plateau, the diminution in soil losses below the 4 $t\cdot ha^{-1}\cdot year^{-1}$ was done only in 3-4 year crop rotations with one or two outside fields, cultivated with perennial grasses and legumes that protect soil against erosion better (Table 5).

The results concerning water runoff, soil and mineral element losses from crops, placed in different rotations, have shown that on 16% slope lands, the use of pea-wheat-maize rotation + 3 outside fields, cultivated with legumes and perennial grasses, resulted in soil losses, which diminished by 39.4% (1.98 $t\cdot ha^{-1}$), as compared to wheat-maize rotation (Table 5). On 16% slope lands, the mean annual losses of nitrogen due to erosion were comprised, during 1980-2010, between 16.23 $kg\cdot ha^{-1}$ in maize continuous cropping and

| Crop | Organic carbon and mineral elements lost by erosion, kg·ha$^{-1}$ | | | | | | |
|---|---|---|---|---|---|---|---|
| | Organic carbon | N at water runoff | N in eroded soil | Total N | P-AL | K-AL | Total NPK |
| Bare fallow | 360 | 5.397 | 25.989 | 31.386 | 2.090 | 4.362 | 37.838 |
| Sunflower | 180 | 4.841 | 12.768 | 17.609 | 1.286 | 2.393 | 21.288 |
| I year perennial grasses | 38 | 2.158 | 2.828 | 4.986 | 0.220 | 0.476 | 5.682 |
| II year perennial grasses | 5 | 0.723 | 0.360 | 1.083 | 0.027 | 0.061 | 1.171 |
| Maize | 166 | 4.315 | 11.917 | 16.232 | 1.150 | 2.022 | 19.404 |
| Peas | 74 | 2.700 | 5.544 | 8.244 | 0.406 | 0.744 | 9.394 |
| Wheat | 32 | 1.518 | 2.370 | 3.888 | 0.182 | 0.406 | 4.476 |
| Beans | 91 | 3.247 | 6.653 | 9.900 | 0.501 | 0.911 | 11.312 |
| Soybean | 77 | 2.802 | 5.679 | 8.481 | 0.443 | 0.848 | 9.772 |

Table 4. Mean water runoff, soil, organic carbon and mineral element losses, due to erosion, in the Moldavian Plateau

| Crop rotation | Water Runoff | | Erosion | | Organic carbon kg·ha$^{-1}$ | Row plants % |
|---|---|---|---|---|---|---|
| | mm | % | t·ha$^{-1}$·year$^{-1}$ | % | | |
| Mcc * | 30.6 | 100 | 8.39 | 100 | 166 | 100 |
| B W M Sf W | 22.9 | 75 | 5.01 | 60 | 100 | 60 |
| W M | 21.1 | 69 | 5.03 | 60 | 99 | 50 |
| P W M Sf + G | 21.9 | 72 | 4.67 | 56 | 93 | 40 |
| P W M | 21.2 | 69 | 4.58 | 55 | 91 | 33 |
| P W M Sf + 2 G | 19.9 | 65 | 3.99 | 48 | 80 | 33 |
| B W M + 2 G | 17.1 | 56 | 3.60 | 43 | 63 | 40 |
| P W Sf + 2 G | 17.7 | 58 | 3.18 | 38 | 62 | 20 |
| S W M + 2 G | 16.3 | 53 | 3.12 | 37 | 60 | 40 |
| P W M + 3 G | 15.4 | 50 | 3.05 | 36 | 52 | 17 |

* Mcc= Maize continuous cropping, B W Sf M W = Beans-wheat-sunflower-maize-wheat rotation, W M= Wheat-maize rotation, P W M= Peas-wheat-maize, P W M Sf + G = Peas-wheat-maize -sunflower + reserve field, with legumes and perennial grasses, B W M+ 2 G = Beans-wheat-maize + 2 reserve field, with legumes and perennial grasses, S W M = Soybean- wheat-maize + 2 reserve field, with legumes and perennial grasses

Table 5. Average annual water and soil runoff by erosion in different crops rotation

5.757 kg·ha$^{-1}$·year$^{-1}$ in pea - wheat - maize rotation + three outside fields cultivated with perennial grasses (Table 6). If phosphorus and potassium losses are low (1.15-2.02 kg·ha$^{-1}$·year$^{-1}$), the nitrogen losses should be diminished by using rotations with crop structures that protect soil against erosion.

The highest losses of nutrients were recorded in bean-wheat- sunflower-maize-wheat rotation (10.304 kg·ha$^{-1}$ nitrogen and 12.192 kg·ha$^{-1}$ total NPK) (Table 6). These amounts decreased very much at the same time with the increase in the rotation structure of cover crops, such as pea, wheat, alfalfa and perennial grasses. Data are very important for establishing and regulating the fertilizer rates applied in crops and for controlling the environment pollution with nitrogen, phosphorus and potassium. During 1980-2010, the use of crop rotations until 20% of row plants, which also included outside fields cultivated with perennial grasses, reduced soil and mineral element losses by 36.7% (1.85 t·ha$^{-1}$) and 33.7% (3.987 kg·ha$^{-1}$), respectively, as compared to 2-year crop rotation (wheat- maize). On 16% slope lands during 1980-2010, the crop structure which reduced mean soil losses by erosion until 3.12 t·ha$^{-1}$·year$^{-1}$ included 20% straw cereals, 20% annual legumes, 20% row crops and 40% perennial grasses and legumes.

| Crop rotations | N at water runoff | N in eroded soil | Total N | P-AL | K-AL | Total NPK |
|---|---|---|---|---|---|---|
| Mcc * | 4.315 | 11.917 | 16.232 | 1.150 | 2.022 | 19.404 |
| B W MSf W | 3.088 | 7.216 | 10.304 | 0.660 | 1.228 | 12.192 |
| W M | 2.917 | 7.144 | 10.061 | 0.666 | 1.214 | 11.941 |
| P W M Sf + G | 2.891 | 6.715 | 9.606 | 0.621 | 1.146 | 11.373 |
| P W M | 2.844 | 6.610 | 9.454 | 0.579 | 1.057 | 11.090 |
| P W M Sf + 2 G | 2.590 | 5.759 | 8.349 | 0.531 | 0.983 | 9.863 |
| B W M+ 2 G | 2.249 | 4.579 | 6.828 | 0.399 | 0.734 | 7.961 |
| P W Sf + 2 G | 2.245 | 4.527 | 6.772 | 0.407 | 0.775 | 7.954 |
| S W M+ 2 G | 2.160 | 4.384 | 6.544 | 0.387 | 0.721 | 7.652 |
| P W M + 3 G | 1.963 | 3.794 | 5.757 | 0.330 | 0.611 | 6.698 |

Table 6. Average mineral elements lost by erosion (kg·ha$^{-1}$) in different crops rotations

During 1980-2010, on 16% slope fields, the increase from 20 to 40% of row crops (maize and sunflower) used in rotations increased mean annual losses of eroded soil by 46.7% (1.486 t·ha$^{-1}$) and the use of crop rotations with 60% row crops increased by 57.5% (1.829 t·ha$^{-1}$) mean annual quantities of eroded soil. According to these results concerning the contribution of melioration plants to the diminution of soil and mineral element losses due to erosion, the technical elements were established for anti-erosion works, such as width of cultivated strips and sod rewetting, crop structure, crop rotations and assortment of legumes and perennial grasses used on slope lands. This scientific information is a source for creating a database necessary for the elaboration of land improvement projects, watershed setting up and protecting soil and water resources.

The crop rotation is also important under conditions of an intensive technology, being the main measure for soil protection, crop and efficient capitalization of all technological factors. The investigations conducted in long-term experiments at Rothamsted have shown that only at high fertilizer rates (>$N_{192}P_{35}K_{90}Mg_{35}$), a significant increase was found in total organic carbon and stable carbon in soil [1]. In clayey-loam mollisol from Kanawha, the organic carbon increased from 33.3 to 37.3 g·kg$^{-1}$ soil, when using the rate of 270 kg·ha$^{-1}$ nitrogen against the unfertilized control, only in maize-oats-alfalfa-alfalfa rotation (Li et al., 2009).

On soils from the Moldavian Plateau, most of them situated on slope fields, poor in organic matter and nutrients, the proper use of different organic resources may replace a part of rich technological consumption (mineral nutrients), determine the improvement in the content of organic matter from soil and ensure better conditions for the valorisation of nitrogen fertilizers. Crop rotations with annual and perennial grasses and legumes have increased the biodiversity of agro-ecosystems, diminished the quantity of nitrogen-based fertilizers, contributed to the increase in soil fertility and diversified the options of farming management.

The total carbon from cambic chernozem in the Moldavian Plain significantly increased at higher fertilization rates than $N_{160}P_{100}$ , in case of organic and mineral fertilization and in 4-year crop rotation, which included ameliorative perennial grasses and legumes (Table 7).

| Treatment | Mcc | W M | P W M | P W M Sf + G | Average | Difference |
|---|---|---|---|---|---|---|
| $N_0P_0$ | 15.0 | 15.2 | 16.5 | 16.8 | 15.9 | 0 |
| $N_{80}P_{60}$ | 15.5 | 14.8 | 16.9 | 17.1 | 16.1 | 0.1 |
| $N_{120}P_{80}$ | 15.8 | 16.2 | 17.3 | 18.2 | 16.9 | 0.9 |
| $N_{160}P_{100}$ | 16.8 | 17.0 | 18.5 | 19.7 | 18.0 | $1.9^{x}$ |
| $N_{80}P_{60}$+30 t·ha$^{-1}$ manure | 19.0 | 19.0 | 20.1 | 21.4 | 19.9 | $3.9^{xxx}$ |
| Average | 16.4 | 16.4 | 17.9 | 18.6 | 17.3 | |
| Difference | 0 | 0.0 | $1.5^{x}$ | $2.2^{xx}$ | | |
| | Crop rotation | | Fertilizer | | Interaction | |
| LSD 5% | 1.4 | | 1.5 | | 1.2 g·kg$^{-1}$ | |
| LSD 1% | 1.8 | | 2.1 | | 1.6 g·kg$^{-1}$ | |
| LSD 0.1% | 2.4 | | 2.7 | | 2.1 g·kg$^{-1}$ | |

Table 7. Influence of long-term fertilization and crop rotation on mass of carbon from soil (C, g·kg$^{-1}$)

In maize continuous cropping and wheat-maize rotation, very significant values of the carbon content were found only in the organic and mineral fertilization, in 4-year crop rotation + reserve field, cultivated with perennial legumes and in $N_{160}P_{100}$ fertilization.

In cambic chernozem, on the slope lands from the Moldavian Plain, a good supply in mobile phosphorus of field crops (37-46 mg·kg$^{-1}$) was maintained in annual application of a rate of $N_{120}P_{80}$ and a very good supply (69-78) at the rate of $N_{80}P_{60}$+30 t·ha of manure, applied in crops from 3- or 4 -year crop rotations with perennial grasses and legumes (Table 8).

| Treatment | Mcc | WM | PWM | PWMSf+G | Average | Difference |
|---|---|---|---|---|---|---|
| $N_0P_0$ | 13 | 10 | 14 | 15 | 13.0 | 0 |
| $N_{80}P_{60}$ | 29 | 26 | 35 | 40 | 32.5 | $19.5^{xxx}$ |
| $N_{120}P_{80}$ | 41 | 38 | 49 | 56 | 46.0 | $33.0^{xxx}$ |
| $N_{160}P_{100}$ | 58 | 52 | 63 | 69 | 60.5 | $47.5^{xxx}$ |
| $N_{80}P_{60}$+30 t·ha$^{-1}$ manure | 67 | 58 | 69 | 78 | 68.0 | $55.0^{xxx}$ |
| Average | 41.6 | 36.8 | 46.0 | 51.6 | 44.0 | |
| Difference | 0 | $-4.8^{0}$ | $4.4^{x}$ | $10.0^{xxx}$ | | |
| | Rotation | | Fertilizer | | Interaction | |
| LSD 5% | 3.8 | | 3.3 | | 4.3 mg·kg$^{-1}$ | |
| LSD 1% | 5.1 | | 4.4 | | 5.7 mg·kg$^{-1}$ | |
| LSD 0.1% | 6.7 | | 5.8 | | 7.5 mg·kg$^{-1}$ | |

Table 8. Influence of long-term fertilization and crop rotation on the content of mobile phosphorus from soil (P-AL, mg·kg$^{-1}$)

After 43 years of testing, the lowest rate of mobile phosphorus accumulation in soil was recorded in wheat-maize rotation, and the highest one, in 3- and 4- year crop rotations, including annual and perennial legumes, which leave in soil easily degradable crop residues.

### 2.2 Influence of tillage practices on yields of maize and wheat and on some soil properties

Soil is basic medium for seed germination, seed emergence, root growth and ultimately crop production. In the last period, the investigations conducted in different countries followed the influence of improving technological elements on fertilization, soil tillage and crop rotations with legumes and perennial grasses, which determine the increase in the content of organic carbon from soil and the reduction of $N_2O$ emissions. The $N_2O$ emissions from soil increase linearly with the amount of mineral nitrogen applied by fertilization (0.0119 kg $N_2O$-N $\cdot ha^{-1} \cdot year^{-1}$). The application of manure determines the diminution in nitrogen protoxide emissions (0.99 kg $N_2O$-N $\cdot ha^{-1} \cdot year^{-1}$), compared with the application of liquid manure (2.83 kg $N_2O$-N $\cdot ha^{-1} \cdot year^{-1}$) or mineral fertilization (2.82 kg $N_2O$-N $\cdot ha^{-1} \cdot year^{-1}$) (Gregorich et al., 2005). The conventional system with annual ploughing, carried out at the same depth and with repeated treatments for seedbed preparation with disk-harrows, has negative consequences on soil physical characteristics: mechanical and water stability of aggregates, porosity, infiltration capacity, hydraulic conductivity, water holding capacity, stratification of organic matter and nutrients, activity and diversity of edaphic flora and fauna, carbon biomass, soil water and temperature regime (Jitareanu et al., 2007). Environment deterioration is mainly caused by soil erosion, compaction, soil structure damage due to human activities and loss of organic matter, as well as by extreme climatic conditions, influenced by world changes. Because the farming conventional systems have caused soil degradation in many countries, the technologies concerning the mechanization of agricultural practices must be adapted to the requirements concerning soil and water protection, and in the areas with soils more sensitive to degradation, soil conservation practices are necessary (Flanagan et al., 2009). Soils tilled at time, differentiated according to the requirements of crop rotations, to climatic conditions, contribute to the improvement of soil physicochemical characteristics, to diminish weed infestation degree and allow manure and crop residue incorporation. By diminishing the soil intensity and mobilization depth, the aggregation condition is improved and, therefore, the losses of organic carbon are diminished, as a result of humus decay, due to less aired environment and to better soil protection against erosion (Jug et al., 2007, Romaneckas et al., 2010). The content of organic carbon in the shallow layer (0-10 cm) on Gleyic Luvisol with sandy loam texture from Halle, Germany, after 37 years of applying different soil tillage systems, differentiated from 10.0 $g \cdot kg^{-1}$ under the conventional system, with 25 cm ploughing, to 14.9 $g \cdot kg^{-1}$ at 25 cm chisel tillage and, respectively, to 13.2 $g \cdot kg^{-1}$ under no-tillage. At the depth of 0-20 cm, the content of organic carbon was of 10.2 $g \cdot kg^{-1}$ at 25 cm ploughing, 12.7 $g \cdot kg^{-1}$ at chisel tillage and 11.6 $g \cdot kg^{-1}$ under no-tillage. The use of legumes for soil protection against erosion has resulted in fixing 105 kg from the atmospheric nitrogen and annual increase in the carbon content from soil by 1055 kg, which is twice higher than the accumulation in the no-till system (Farahbakhshazada et al., 2008). In Austria, between 1994 and 2007, the mean soil losses at the three locations dropped from 6.1 to 1.8 $t \cdot ha^{-1} \cdot$year-1 with conservation tillage in cover crops, and to 1.0 $t \cdot ha \cdot year^{-1}$ with direct drilling. Nitrogen (9.2, 3.7, 2.5 $t \cdot ha \cdot year^{-1}$) and phosphorus (4.7, 1.3, 0.7 t·ha·year $^{-1}$) losses showed similar tendencies (Colyer, 2005, Ghuman & Sur, 2001).

The aim was to study the influence of soil tillage systems and fertilization on yield of soybean, wheat and maize crops and soil agrochemical characteristics in long-term field experiments.

The influence of soil tillage systems and fertilization on yield of soybean, wheat and maize crops and soil agrochemical characteristics was studied in field experiments conducted during 1998-2010 on a Cambic Chernozem at the Agricultural Research and Development Station of Podu-Iloaiei, Iasi County.

The typical Cambic Chernozem from Podu-Iloaiei was formed on a loessy loam, has a mean humus content (3.1-3.4%), is well supplied with mobile potassium (215-235 ppm) and moderately with phosphorus (28-35 ppm) and nitrogen (0.160-0.165%). Soil has a high clay content (39-41%) being difficult to treat when soil moisture is close to the wilting point (12.2%). In wheat, we used Gabriela variety and in maize Oana hybrid.

Experiments were set up in split-split plots with four replicates, tillage treatments in main plots and fertilization in subplot. Tillage treatments were seedbed preparation by 20 cm ploughing + disking, 30 cm ploughing + disking, chisel tillage + disking, paraplow tillage and disking and one year ploughing + one year disking. Fertilization treatments were unfertilized ($N_0P_0$), N 120 kg·ha$^{-1}$ ($N_{120}$) + 80 kg·ha$^{-1}$ $P_2O_5$ ($P_{80}$), N 160 kg·ha$^{-1}$ ($N_{160}$) + 80 kg·ha$^{-1}$ $P_2O_5$ , N 80 kg·ha$^{-1}$ ($N_{80}$) + 80 kg·ha$^{-1}$ $P_2O_5$ + 30 t·ha$^{-1}$ manure and $N_{80}$ + 80 kg·ha$^{-1}$ $P_2O_5$ + 6 t·ha$^{-1}$ wheat straw. In the field there was a 3-year crop rotation (soybean-wheat-maize),

Physical and chemical analyses of soil samples were carried out according to the methods established by the Research Institute of Bucharest, which are applied by all agrochemistry laboratories from Romania.

Soil on which physical and chemical analyses were done was sampled at the end of plant growing period. Soil response was determined in water suspension by potentiometric means with glass electrode. The content of organic carbon was determined by the Walkley-Black method, the content in mobile phosphorus from soil was determined by Egner-Riehm-Domingo method, in solution of ammonium acetate-lactate (AL) and potassium was measured in the same extract of acetate-lactate (AL) by flame photometry.

Rainfall during January-June (1998-2010) assured normal conditions for wheat growing in 5 years. Rainfall amounts were lower, compared to the multiannual mean on 80 years (248 mm) in 5 years, when rainfall deficit was between 53.3 and 119.0 mm. The climatic conditions during 1998-2010 were favourable to maize growing and development in 5 years and unfavourable, due to low rainfall amounts, in the other 6 years. In the last 13 years, the deficit of rainfall during January-August, compared to the multiannual mean of the area, was between 31.4 and 136.9 mm in 5 years. The drought in autumn and during January-August required the adjustment of soil preparation practices to the requirements of water conservation from soil. Most of soils in the Moldavian Plateau situated on slope fields, poor in organic matter and nutrients, the proper use of different organic resources may replace a part of rich technological consumption (mineral nutrients), determine the improvement in the content of organic matter from soil and may assure better conditions for the capitalization of nitrogen fertilizers.

The mean wheat yields obtained during 1998-2010 at 20 cm ploughed variant were of 3265 kg·ha$^{-1}$ and in case of seedbed preparation by chisel + disk and by repeated disking, the yields were lower by 4.0% (122 kg·ha$^{-1}$) and 6.0% (223 kg·ha$^{-1}$), respectively, as compared to

20 cm ploughing (Table 9). At the same period, the mean wheat yields obtained under unfertilized were of 1818 kg·ha$^{-1}$ and the rates of $N_{120}$ + 80 kg·ha$^{-1}$ $P_2O_5$ or $N_{160}$ + 80 kg·ha$^{-1}$ $P^2O^5$ resulted in getting yield increases of 94% (1714 kg·ha$^{-1}$) and 110% (2006 kg·ha$^{-1}$), respectively. The application in wheat of a rate of $N_{80}$ + 80 kg·ha$^{-1}$ $P^2O5$ + 30 t ha$^{-1}$ manure resulted in getting yield increases of 117% (2122 kg·ha$^{-1}$).

In wheat, found in a 3-year crop rotation (soybean-wheat-maize), the percentage of hydrostable aggregates was less influenced by soil tillage system (54.7-62.8%) and more by applied fertilizers (50.3-61.3%) (Table 10).

In wheat crop, the percentage of hydrostable aggregates varied, according to applied fertilizer rates, between 49.2 and 58.6% at 20 cm ploughing, between 53.3 and 59.9% at 30 cm ploughing and between 56.2 and 68.8% at chisel treatment. The highest percentage of hydrostable aggregates was at the rate of $N_{160}$ + 80 kg $P_2O_5$ (58.2%) and at organo-mineral fertilization (61.3%).

The content of organic carbon in the shallow layer (0-20 cm) comprised between 18.43 and 20.08 g/kg at the fertilization with $N_{120}$ + 80 kg·ha$^{-1}$ $P_2O_5$ and, respectively, $N_{80}P_{80}$+30 t·ha$^{-1}$ manure (Table 11). At chisel and paraplow works, the content of organic carbon from soil was higher by 0.33 g·ha$^{-1}$, as compared to 20 cm ploughing system.

Applying for 13 years moderate mineral fertilizer rates ($N_{80}P_{80}$), together with 6 kg·ha$^{-1}$ wheat straw, has resulted in increasing the organic carbon from soil by 1.7 g·ha$^{-1}$, compared to unfertilized variant. The highest content of organic carbon was found at the rate $N_{80}P_{80}$+30 t·ha$^{-1}$ manure, where it increased by 2.67 g·ha$^{-1}$, compared to the unfertilized control.

| Soil tillage | $N_0P_0$ | $N_{80}P_{80}$+6 t·ha$^{-1}$ wheat straw | $N_{120}P_{80}$ | $N_{160}P_{80}$ | $N_{80}P_{80}$+30 t·ha$^{-1}$ manure | Mean | % | Differ. kg·ha$^{-1}$ |
|---|---|---|---|---|---|---|---|---|
| 20 cm ploughing + disk | 1860 | 2960 | 3520 | 3890 | 4097 | 3265 | 100 | 0 |
| 30 cm ploughing + disk | 1950 | 3213 | 3880 | 4232 | 4331 | 3521 | 108 | 256 |
| Chisel + disk | 1870 | 2587 | 3540 | 3840 | 3880 | 3143 | 96 | -122 |
| Paraplow + disk | 1690 | 2823 | 3380 | 3640 | 3620 | 3031 | 93 | -235 |
| One year ploughing, one year disking | 1720 | 2860 | 3340 | 3520 | 3770 | 3042 | 93 | -223 |
| Mean | 1818 | 2889 | 3532 | 3824 | 3940 | 3201 | | |
| % | 100 | 159 | 194 | 210 | 217 | | | |
| Difference kg·ha$^{-1}$ | 0 | 1071 | 1714 | 2006 | 2122 | | | |
| | | Soil tillage (A) | Fertilizer (B) | Interaction A x B | | | | |
| LSD 5% | | 198 | 150 | 330 | | | | |
| LSD 1% | | 321 | 201 | 481 | | | | |
| LSD 0.1% | | 516 | 264 | 689 | | | | |

Table 9. Influence of soil tillage system and fertilization on wheat yield (kg·ha$^{-1}$).

| Soil tillage | $N_0P_0$ | $N_{80}P_{80}$+6 t·ha$^{-1}$ wheat straw | $N_{120}P_{80}$ | $N_{160}P_{80}$ | $N_{80}P_{80}$+30 t·ha$^{-1}$ manure | Mean | % | Differ. |
|---|---|---|---|---|---|---|---|---|
| 20 cm ploughing + disk | 49.2 | 55.2 | 53.1 | 55.2 | 58.6 | 54.7 | 100 | 0.0 |
| 30 cm ploughing + disk | 53.3 | 58.3 | 56.8 | 58.3 | 59.9 | 57.5 | 105 | 2.8 |
| Chisel + disk | 56.2 | 65.1 | 61.9 | 65.1 | 68.8 | 62.8 | 115 | 8.1 |
| Paraplow + disk | 51.8 | 59.9 | 57.9 | 59.9 | 64.2 | 58.8 | 108 | 4.1 |
| One year ploughing, one year disking | 40.9 | 52.4 | 47.3 | 52.4 | 55.2 | 49.6 | 91 | -5.1 |
| Mean | 50.3 | 58.2 | 55.4 | 58.2 | 61.3 | 56.7 | | |
| % | 100 | 116 | 110 | 116 | 122 | | | |
| Difference | 0 | 8 | 5 | 8 | 11 | | | |
| | | Soil tillage (A) | Fertilizer (B) | Interaction A x B | | | | |
| LSD 5% | | 3.1 | 3.0 | 5.4 | | | | |
| LSD 1% | | 5.5 | 5.4 | 7.5 | | | | |
| LSD 0.1% | | 9.3 | 9.1 | 10.3 | | | | |

Table 10. The influence of soil tillage and fertilization on hydrostability of soil aggregates greater than 0.25 mm (%)

| Soil tillage | $N_0P_0$ | $N_{80}P_{80}$+6 t·ha$^{-1}$ wheat straw | $N_{120}P_{80}$ | $N_{160}P_{80}$ | $N_{80}P_{80}$+30 t·ha$^{-1}$ manure | Mean | % | Differ. |
|---|---|---|---|---|---|---|---|---|
| 20 cm ploughing + disk | 17.33 | 19.07 | 18.27 | 18.97 | 19.97 | 18.72 | 100 | 0.000 |
| 30 cm ploughing + disk | 17.27 | 19.03 | 18.43 | 18.93 | 20.00 | 18.73 | 100 | 0.013 |
| Chisel + disk | 17.67 | 19.33 | 18.73 | 19.23 | 20.53 | 19.10 | 102 | 0.380 |
| Paraplow + disk | 17.80 | 19.33 | 18.73 | 18.97 | 20.40 | 19.05 | 102 | 0.327 |
| One year ploughing, one year disking | 16.97 | 18.77 | 18.00 | 18.73 | 19.50 | 18.39 | 98 | -0.327 |
| Mean | 17.41 | 19.11 | 18.43 | 18.97 | 20.08 | 18.80 | | |
| % | 100 | 110 | 106 | 109 | 115 | | | |
| Differ. | 0.00 | 1.70 | 1.03 | 1.56 | 2.67 | | | |
| | | Soil tillage (A) | Fertilizer (B) | Interaction A x B | | | | |
| LSD 5% | | 0.351 | 0.341 | 0.784 | | | | |
| LSD 1% | | 0.510 | 0.455 | 1.080 | | | | |
| LSD 0.1% | | 0.765 | 0.599 | 1.487 | | | | |

Table 11. Evolution of the organic carbon content from soil, 10 years after applying different rates of fertilizers and soil tillage systems

Many investigations conducted in different countries have shown that applying low rates of mineral fertilizers with nitrogen, phosphorus and potassium in wheat and maize continuous cropping and wheat-maize rotation has determined the diminution in the content of organic matter from soil. The diminution in the content of organic carbon from soil, due to mineral fertilization, was found in loam-sandy fields from Nashua, USA, where lower rate than 180 kg N $\cdot ha^{-1}$ was applied in maize-soybean rotation (Russell et al., 2006) and in clay loam soils from Rothamsted, England, where lower rates than $N_{192}P_{35}K_{90}Mg_{35}$ were applied (Blair et al., 2006).

#### 2.2.1 Long-term effect of nitrogen and phosphorus fertilizer and crop residue on production and soil fertility in the Moldavian Plateau

In many countries, the investigations conducted on eroded soils have followed the establishment of crop rotations and soil tillage and fertilizing systems, which contribute to maintaining and recovery of soil fertility (Campbell et al., 2005, Yadav & Malanson, 2008).
The negative impact of continuous cropping on the content of organic carbon from soil was shown by many specialists (Lal, 2006, Liu et al., 2006, Rusu et al., 2006, Wesley et al., 2006). In many areas, applying crop residues, together with moderate nitrogen rates, have resulted in improving physical, chemical and biological soil characteristics (Carter et al., 2002, Naderloo et al., 2009). The investigations conducted on maize by Lindstrom, in Minnesota, USA, show that applying crop residues at rates of 927 and 3706 kg $\cdot ha^{-1}$ has determined the decrease in soil erosion from 6.18 to 0.99 t $\cdot ha^{-1}$ and in water runoff from 35.6 to 22.9 mm. Other studies show that applying crop residues, together with nitrogen fertilizers, under conventional soil tillage with ploughing, did not result in increasing the organic carbon content after 30 years of experiencing (Osvaldo, 2006). These studies show that establishing the amounts of crop residues, which must be applied for maintaining the content of organic carbon and for diminishing soil erosion, should have in view the interactions between crop rotation, soil tillage, fertilization and soil and climate conditions. The amounts of applied crop residues must contribute to diminishing soil erosion, maintaining the content of organic carbon from soil and determining yield increases.
On weakly eroded lands, the mean maize yields obtained during 1997-2010, were comprised between 3287 kg $\cdot ha^{-1}$ (100%) at the unfertilized control and 7188 kg $\cdot ha^{-1}$ (119%) at rates of 70 kg N + 70 kg $P_2O_5$ + 60 t $\cdot ha^{-1}$ manure (Table 12). In maize, the application of mean rates of mineral fertilizers (70 kg N + 70 kg $P_2O_5$) with 60 t $\cdot ha^{-1}$ manure has resulted in getting yield increases of 134% (3275 kg $\cdot ha^{-1}$), compared to the unfertilized variant. Applying rates of 100 kg N + 100 kg $P^2O^5$ resulted in getting yield increases of 84% (2748 kg $\cdot ha^{-1}$) in maize, placed on weakly eroded lands, and 94% (2306 kg $\cdot ha^{-1}$) in maize placed on highly eroded soil, compared to the unfertilized variant.
In maize placed on weakly eroded lands, the mean yield increases obtained for each kg of a.i. of applied fertilizers have varied according to applied fertilizers rates, between 7.2 and 14.1 kg grains ($N_{40}P_{40}$-$N_{140}P_{100}$). On highly eroded lands, the mean maize yield obtained under unfertilized was of 2452 kg $\cdot ha^{-1}$, while the mean yield increases, obtained by applying 40 or 60 t $\cdot ha^{-1}$ manure, were of 36.4-34.2 kg grains per ton of applied manure. The mineral fertilizers ($N_{40}P_{40}$-$N_{140}P_{100}$) resulted in getting mean yield increases of 8.4- 11.7 kg grains $\cdot ha^{-1}$ a. i. of applied fertilizer. Very close yield results were also obtained by applying, for 43 years, rates of 70 kg N + 70 kg $P_2O_5$ $\cdot ha^{-1}$ +3 t $\cdot ha^{-1}$ stalks of pea or soybean, variants at which yield increases have varied, according to soil erosion, between 2550 and 2615 kg $\cdot ha^{-1}$ (78-80%) on weakly

eroded lands and between 2161 and 2223 kg·ha$^{-1}$ (88-91%) on highly eroded lands (Table 12). The analysis of results obtained has shown that the erosion process, by decreasing soil fertility, has determined the differentiation of the mean maize yield, according to slope and erosion, from 5756 (100%) to 4538 kg·ha$^{-1}$ (78.8%). Mean annual losses of yields registered in maize in the last 13 years, caused by erosion, were of 1218 kg·ha$^{-1}$ (21.2%).

| Fertilizer rate | Weakly eroded soil | | | | Highly eroded soil | | | |
|---|---|---|---|---|---|---|---|---|
| | Maize | | Wheat | | Maize | | Wheat | |
| | yield | diff. | yield | diff. | yield | diff. | yield | diff. |
| $N_0P_0$ | 3287 | 0 | 1697 | 0 | 2452 | 0 | 1163 | 0 |
| $N_{70}P_{70}$ | 5159 | 1872 | 3192 | 1495 | 4120 | 1668 | 2478 | 1315 |
| $N_{100}P_{100}$ | 6035 | 2748 | 4078 | 2381 | 4758 | 2306 | 3248 | 2085 |
| $N_{140}P_{100}$ | 6660 | 3373 | 4523 | 2826 | 5263 | 2811 | 3665 | 2502 |
| $N_{70}P_{70}K_{70}$ | 5285 | 1998 | 3384 | 1687 | 4251 | 1799 | 2710 | 1547 |
| $N_{100}P_{100}K_{100}$ | 6324 | 3037 | 4398 | 2701 | 5020 | 2568 | 3570 | 2407 |
| $N_{140}P_{100}K_{100}$ | 6816 | 3529 | 4797 | 3100 | 5475 | 3023 | 3923 | 2760 |
| 20 t·ha$^{-1}$ manure | 4150 | 863 | 2761 | 1064 | 3252 | 800 | 2165 | 1002 |
| 40 t·ha$^{-1}$ manure | 5199 | 1912 | 3445 | 1748 | 3909 | 1457 | 2813 | 1650 |
| 60 t·ha$^{-1}$ manure | 5953 | 2666 | 4018 | 2321 | 4505 | 2053 | 3294 | 2131 |
| $N_{70}P_{70}$+20 t·ha$^{-1}$ manure | 6119 | 2832 | 4102 | 2405 | 4719 | 2267 | 3304 | 2141 |
| $N_{70}P_{70}$+40 t·ha$^{-1}$ manure | 6545 | 3258 | 4619 | 2922 | 5261 | 2809 | 3669 | 2506 |
| $N_{70}P_{70}$+60 t·ha$^{-1}$ manure | 7188 | 3901 | 4894 | 3197 | 5727 | 3275 | 4011 | 2848 |
| $N_{70}P_{70}$+6 t·ha$^{-1}$ hashed straw | 5733 | 2446 | 3770 | 2073 | 4646 | 2194 | 3041 | 1878 |
| $N_{70}P_{70}$+6 t·ha$^{-1}$ stalks of maize | 5656 | 2369 | 3578 | 1881 | 4500 | 2048 | 2929 | 1766 |
| $N_{70}P_{70}$+3 t·ha$^{-1}$ stalks of pea | 5902 | 2615 | 4010 | 2313 | 4675 | 2223 | 3237 | 2074 |
| $N_{70}P_{70}$+3 t·ha$^{-1}$ stalks of soybean | 5837 | 2550 | 3911 | 2214 | 4613 | 2161 | 3164 | 2001 |
| Mean | 5756 | 100 | 3834 | 100 | 4538 | 78.8 | 3081 | 80.4 |
| LSD 5% | | 315 | | 340 | | 336 | | 310 |
| LSD 1% | | 444 | | 450 | | 450 | | 430 |
| LSD 0.1% | | 593 | | 580 | | 605 | | 570 |

Table 12. Influence of mineral and organic fertilizers on maize and wheat yields (kg·ha$^{-1}$), in weakly and highly eroded soils, after 43 years of experiments

In wheat, the application of mean rates of mineral fertilizers with 60 t·ha$^{-1}$ manure has resulted in getting yield increases of 188% (3197 kg·ha$^{-1}$), compared to the unfertilized variant. In wheat placed on weakly eroded lands, the mean yield increases obtained for each kg of a. i. of applied fertilizers varied, according to fertilizers rates applied, between 9.15 and 11.8 kg grains. The mean annual yield losses, registered in wheat, caused by erosion, were of 753 kg·ha$^{-1}$ (19.6%).

The analysis of agro-chemical data shows that nitrogen fertilizers (ammonium nitrate) have determined the pH decrease. A significant diminution was registered in the ploughed layer, at rates of 140 kg·ha$^{-1}$ N, where pH value has reached 5.7, after 43 years (Table 13).

The analyses carried out on the evolution of soil response, after 43 years of experiencing, have shown that the significant diminution in the pH value was found at higher rates than 100 kg N·ha$^{-1}$. The lowest pH values were found in maize at rates of $N_{140}P_{100}$ and 70 kg N + 70 kg $P_2O_5$·ha$^{-1}$ + 6 t·ha$^{-1}$ stalks of maize, which can be explained by the unfavourable conditions in which the processes of nitrification and crop residue decay, developed.

| Fertilizer rate | Weakly eroded lands | | | | Highly eroded lands | | | |
|---|---|---|---|---|---|---|---|---|
| | pH ($H_2O$) | Org. C g/kg | P-AL (ppm) | K-AL (ppm) | pH ($H_2O$) | Org. C g/kg | P-AL (ppm) | K-AL (ppm) |
| $N_0P_0$ | 7.3 | 16.5 | 17 | 216 | 7.2 | 14.2 | 8 | 192 |
| $N_{70}P_{70}$ | 6.9 | 16.9 | 54 | 186 | 6.8 | 14.3 | 41 | 186 |
| $N_{100}P_{80}$ | 6.3 | 17.5 | 86 | 178 | 6.2 | 15.5 | 62 | 174 |
| $N_{140}P_{100}$ | 5.8 | 18.2 | 89 | 174 | 5.7 | 16.6 | 64 | 156 |
| 60 t·ha$^{-1}$ manure | 7.4 | 21.3 | 74 | 276 | 7.1 | 19.9 | 66 | 259 |
| $N_{70}P_{70}$ + 60 t·ha$^{-1}$ manure | 7.2 | 21.8 | 96 | 292 | 6.9 | 20.2 | 74 | 289 |
| $N_{70}P_{70}$ + 6 t·ha$^{-1}$ hashed of wheat | 7.0 | 18.9 | 64 | 234 | 6.8 | 18.4 | 57 | 216 |
| $N_{70}P_{70}$ +6 t·ha$^{-1}$ stalks of maize | 6.6 | 18.8 | 58 | 239 | 6.6 | 18.1 | 52 | 196 |
| $N_{70}P_{70}$ +3 t·ha$^{-1}$ stalks of pea | 6.9 | 18.5 | 47 | 228 | 6.8 | 18.0 | 48 | 184 |
| $N_{70}P_{70}$ +3 t·ha$^{-1}$ stalks of soybean | 6.8 | 18.3 | 51 | 234 | 6.7 | 18.0 | 49 | 182 |
| Mean | 6.8 | 18.7 | 63.6 | 225.7 | 6.7 | 17.3 | 52 | 203 |
| LSD 5% | 0.27 | 0.07 | 5.9 | 16.1 | 0.24 | 0.10 | 4.8 | 15.7 |
| LSD 1% | 0.38 | 0.10 | 8.2 | 24.0 | 0.35 | 0.14 | 6.9 | 22.6 |
| LSD 0.1% | 0.55 | 0.15 | 11.5 | 36.2 | 0.56 | 0.20 | 9.9 | 33.4 |

Table 13. Effect of soil erosion and fertilization system on the organic carbon and mineral element content in 16% slope fields

## 3. Conclusion

Mean annual losses of soil by erosion, recorded during 1980-2010, were of 0.25 t·ha$^{-1}$ in perennial grasses in the second growth year, 4.56 t·ha$^{-1}$ in bean, 8.39 t·ha$^{-1}$ in maize and 8.93 t·ha$^{-1}$ in sunflower.

Erosion affects soil fertility by removing together with eroded soil, significant mineral element amounts, which in maize and sunflower crops reach 16.2-17.6 kg·ha$^{-1}$ nitrogen, 1.2-1.3 kg·ha$^{-1}$ phosphorus and 2.0-2.4 kg·ha $^{-1}$ potassium, representing, on the average, 12-14% of the chemical fertilizers necessary for these crops.

On 16% slope fields, the use of soybean - wheat - maize rotation + two outside fields, cultivated with perennial grasses, determined the diminution by 62.8% (5.27 t·ha$^{-1}$) in the mean annual losses of eroded soil and by 59.7% (9.688 kg·ha$^{-1}$) in nitrogen leakages, compared with maize continuous cropping.

From the results obtained on erosion in different crop rotations, we found that in 16% slope fields from the Moldavian Plateau, soil losses by erosion diminished below 4 t·ha$^{-1}$ only in case of 3- or 4-year crop rotations with two or three reserve fields, cultivated with legumes and perennial grasses, which protect soil.

The 43-year use of 4-year crop rotations + reserve field, cultivated with perennial grasses and legumes, has increased soil total carbon 13.4% (C 2.2 g·kg$^{-1}$) in comparison with maize continuous cropping.

Soil preparation without furrow inverting has resulted in improving soil physical and hydro physical characteristics and allowed a better capitalization of technological factors and, especially, of fertilizers, which determined greater yield increases in wheat by 3.6% (271 kg·ha$^{-1}$) ($N_{160}$ +80 kg·ha$^{-1}$ $P_2O_5$), compared to 20 cm ploughing.

Soil tillage by chisel and disk allowed soil treatment under better conditions for wheat growing in dry autumns, which are very frequent in the area.

The results obtained made us assess that soil tillage system must be adjusted to plant requirements from crop rotation and to soil and climatic conditions of the area. Establishing the systems of soil tillage for the whole crop rotation (disking or chisel + disk work in wheat crop, 20 cm ploughing for soybean and 25-28 cm ploughing for maize) resulted in a better capitalization of the other technological factors, water conservation in soil, maintaining soil physical condition and reduction, on the entire rotation cycle, in fuel consumption.

Applying moderate rates of mineral fertilizers ($N_{80}P_{80}$), together with 6 t·ha$^{-1}$ wheat straw or 30 t·ha$^{-1}$ manure, has determined, 10 years after using chisel, the increase in organic carbon content from soil by 1.66 and, respectively, 2.86 g·kg$^{-1}$.

On highly eroded lands, in maize after wheat, in a three year rotation, the mean yield obtained under unfertilized was of 2452 kg·ha$^{-1}$, the mean yield increases obtained by applying 60 t·ha$^{-1}$ manure every two years, were of 34.2 kg grains per ton of manure applied and mineral fertilizers ($N_{140}P_{100}$) resulted in obtaining mean yield increases of 11.7 kg grains·kg-1 a.i. of applied fertilizer.

On slightly eroded lands, keeping a good supply in soil nutritive elements was done by the annual use of some fertilizer rates of at least $N_{100}P_{100}$ or $N_{70}P_{70}$+ 40 t·ha$^{-1}$ manure applied once in two years or $N_{70}P_{70}$ + 6 t·ha$^{-1}$ straw; on highly eroded lands, keeping a good plant supply in mineral elements was done at rates of $N_{140}P_{100}K_{70}$ or $N_{70}P_{70}$ + 40 t·ha$^{-1}$ manure.

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# 14

# Terracing as a Measure of Soil Erosion Control and Its Effect on Improvement of Infiltration in Eroded Environment

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## 1. Introduction

Soil erosion by water is commonly recognized as the one of main reasons of land degradation worldwide (e.g. Ananda & Herath, 2003; Beskow et al., 2009; Valentin et al., 2005). Most of the involved areas are occupied by various agricultural activities but pastures, forestry, unpaved roads as well as construction sites are also endangered by water erosion (Ananda & Herath, 2003; Arnanez et al., 2004; Harbor, 1999; Jungerius et al., 2002). Exemplary, in Europe, excluding European part of Russia, human-induced soil erosion develops on approximately 114 milion hectrares, which is 17 % of total land area (Gobin, 2004). According to information presented by Pimentel and Kounang (1998) more than 80% of world's agricultural land suffers soil erosion, from moderate to severe level. The mean noted yearly soil erosion rate on cropland worldwide reaches the level of approx. 30 Mg $ha^{-1}$, while reported values vary from 0.5 to over 400 Mg $ha^{-1}$ $yr^{-1}$ (Pimentel & Kounang, 1998). Soil erosion rates, caused by water, are highest in agrosystems located in hilly or mountainous regions of Asia, Africa and Southern America, especially in less developed countries. According to studies reported by Kuhlman et al. (2010) construction sites in Europe are endangered by soil erosion rate higher than 2.0 Mg $ha^{-1}$ $yr^{-1}$, while soil erosion rate noted on approx. 70% of European arable lands is lower than 0.5 Mg $ha^{-1}$ $yr^{-1}$ or ranges from 0.5 to 2.0 Mg $ha^{-1}$ $yr^{-1}$.

Several factors influencing the rate of soil erosion by water were reported (e.g. Amore et al. 2004; Askoy and Kavvas, 2005; Basic et al., 2004; Zhang et al., 2008):

- Climatic conditions: precipitation, frequency of extreme rainfall events and thickness of snow cover as well as rate of its melting;
- Terrain surface morphology determining the rate of surface runoff generation and flow velocity: hillside slopes' length, inclination and exposure;
- Soil characteristics: its particle size composition and erodibility;
- Soil usage: manner of agricultural, forestry, engineering or constructional activities.

Soil erodibility is understood as a measure of its susceptibility to detachment and transport by water (Hammad et al., 2006). Among soil characteristics several properties influencing erodibility may be determined: aggregate stability, organic matter and clay particles content. Various types of soil erosion may be triggered by rainfall, thaw and runoff water: sheet, rill, and gully erosion (Askoy & Kavvas, 2005; Grønsten & Lundekvam, 2006; Valentin et al., 2005). Sheet erosion occurs when detached by impact of rain drop approximately uniform layer of soil particles is being transported downslope in unconcententrated flow, as a sheet. Rill and gully erosion are triggered by small concentrated flows along initially ephemeral rills or channels, which may, in time, involving higher rates of concentrated surface flows, develop larger morphological forms, even, in addition to soil crusts, slope inclination and tunneling creating erosional gullies (e.g. Askoy & Kavvas, 2005; Valentin et al., 2005).

Thus, soil erosion by water may trigger various negative changes in soil cover of eroded basin. According to literature reports, these effects of soil erosion may be divided into several main groups: changes in mechanical and mineral compositions of soil, changes and transformations of ground surface morphology, amendments in water balance of eroded catchments and reduction of soils' fertility (e.g. Fullen et al., 1998; Lado & Ben-Hur, 2004; Pimentel & Kounang, 1998; Valentin et al., 2005; Widomski et al., 2010; Widomski & Sobczuk, 2007).

The reported changes in mechanical and mineral compositions of soils are reflected by decreased content of organic matter and alerted content of all soil fractions (i.e. removal of clay fraction and increase of coarse ones), which in turn may be reflected in decreased water permeability and water capacity of eroded slope and increased volume of generated surface runoff. The volume of infiltration rate of surface water into deeper layers in the eroded profile may be reduced. Thus, the water balance of eroded basin may be significantly changed, generally shortened - the resultant water balance of eroded catchment usually presents reduced inflow of water into underground aquifer, thus limiting water availability for plants (Valentin et al., 2005; Widomski et al., 2010). Increased run-off may also result in increased removal of nutrients form top layer of soil, thinning and even, partial or complete, removal of top soil layer. The noticeable site effects of soil erosion such as rills and erosional gullies together with changes in soils fertility and water-storage capacity may drastically reduce its agricultural or forestry productivity. Removal of erosion effects covering repairing works, workload and materials is often costly (e.g. Kuhlman et al. 2010; Valentin et al., 2005). Therefore, the development and practical application of soil erosion control systems is obvious.

Literature reports show that soil erosion control system should (e.g. Govers et al., 2004; Nyssen et al., 2004; Valentin et al., 2005):

- Reduce destructive processes occurring in top soil;
- Improve infiltration of surface water into soil profile, thus improving water balance of eroded basins and increasing the amount of water available to plants;
- Limit the soil fertility deterioration caused by soils' composition changes and removal of nutrients and organic matter from soil.

Many techniques of soil conservation and erosion control have been developed in agricultural areas, starting from various types of soil tillage and vegetation cover (Basic et al., 2004; Cerdan et al., 2002; Robinson, 1999; Zheng, 2006), to different types of terraces, check dams and stone bunds (e.g. Govers et al., 2004; Valentin et al., 2005).

This chapter covers presentation of terracing as a method of soil erosion control and its efficiency assessment based on literature reports and numerical prediction of soil erosion rate for non-terraced and terraced system. The effects of various types of terraces on soil moisture and infiltration rate were also discussed. Special attention was paid to numerical modeling of infiltration rate for terraces-based system of erosion control on steep slopes developed and tested in Olszanka, Poland, where improvement of infiltration was secured by additional drainage elements filled with sand.

## 2. Terracing – types and brief description

Terracing is an agricultural technique for collecting surface runoff water thus increasing infiltration and controlling water erosion known from an ancient history and used to transform landscape to steeped agrosystems in many hilly or mountainous regions of the world (Zuazo et al., 2005). The well known regions of frequent application of terraces in Europe cover Spain, Italy, France, Portugal, Hungary (basically for vineyard cultivation) but they are also employed in such countries like Norway and Poland (Cots-Folch et al.,2006; Widomski et al. 2010). Terracing is also commonly used in agriculture in Northern and Southern America, Asia (e.g. Chinese Loess Plateau, Thailand, India etc.) and in developing countries in arid environment in Africa, i.e. Ethiopia, Rwanda, Tanzania and others (e.g. Dabney et al., 1999; Fu et al., 2003; Nyssen et al., 2000; Ramos et al., 2007; Sang-Arun et al., 2006; Tenge et al. 2005). Terraces are usually used to cultivate, manually or with mechanization application, different plants – from grains to grapes and various fruit trees e.g. apples, avocado, mango, loquat, litchi and others (Zuazo et al., 2005). The main purpose of terracing application was to improve the usefulness of steep slope and to increase its agricultural potential. This function is realized by creating the level surfaces according to contour lines of transformed slope (Cots-Folch et al., 2006). The level, bench platform allows to spread the surface runoff water, decreases its speed and thus allows more time for water infiltration into soil profile. Terraces are usually reported as a successful soil erosion control manner in regions endangered by soil erosion by combinations of steep slopes, climatic conditions and erodible soils. But in some cases this effectiveness is limited, especially with combination of sparse vegetation (Zuazo et al., 2005). They are also, in some cases, found to be expensive to construct and maintain (Ramos et al., 2007).

The main, worldwide known types of terraces are: various bench terraces, back-sloping bench terraces, stone-wall terraces and Fanya juu terraces (e.g. Tenge et al., 2005).

Bench terraces (Fig. 1) usually consist of a series of level or nearly level platforms constructed along the contour lines of terraced slope (e.g. Ramos et al., 2007; Tenge et al., 2005.) Platforms are separated by embankments known as risers. The main task of level platforms (also known as benches) is to reduce the length of the slope and its steepness, so the amount and velocity of surface runoff is also being reduced and the nearly level platforms retain surface water and allow infiltration into top soils. Thus the erosion control and increased infiltration of rain water as well as limiting soil fertility loss are possible. Bench terraces also allow mechanized farming operations and improvement water management (irrigation). The main observed disadvantages are construction and maintenance costs as well as observed reduction of cropping area (Ramos et al., 2007).

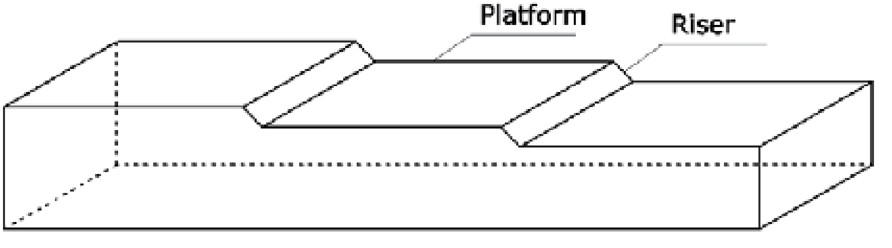


Fig. 1. Scheme of typical bench terraces

Generally bench terraces may be used on slopes up to 55% steep, with deep slopes and stable soils (landslides risk). Ramos et al. (2007) give more precise construction limits for lad slope – circa 20° for terraces constructed by machinery and 25° for terraces build by hand. Benches usually have width of 2,5-5 m for handmade and 3-8 m for terraces constructed by machinery, and slope gradient of the riser should be equal to 0.75:1 or 1:1.

Increase of infiltration rate on traditional bench terraces may be achieved by the additional infiltrational elements filled by permeable material e.g. sand (Widomski et al., 2010). The novel system of terracing increasing infiltration rate and improving the water balance of eroded catchment was developed by Rubaj (2002) on loess soils at fruit farm in Olszanka, Poland (Fig. 2). The system built on 6 to 15% upland slopes consists of terraces equipped with sand-filled ditches. Geometrical dimensions of constructed platforms are 4.0 m width, inclination downslope approx. 1% and 0.3 – 0.5 m of risers height, while sand filled ditches are 0.30 m wide and 0.80 m deep. Riser inclination is close to 70%. The level platform span was adjusted to the dimensions of tractors and other agricultural equipment used in farming operations. Draining ditches dimensions were determined by Rubaj (2002) based on estimations utilizing water capacity of sand filling and mean rainfall amounts contributed per precipitation event. Bench terraces in Olszanka are used to mechanically cultivate apple trees (*Malus domestica Borkh*) with platforms' surface strengthen by cover of the natural mixture of grasses. Despite numerous reports presenting information that prolonged infiltration resulting in increased saturation of loess soils may trigger geotechnical slope instability (Derbyshire, 2001; Wang & Sassa, 2003; Zhao et al., 2000), installation of draining ditches at Olszanka has not altered slope stability in the treated area (Widomski et al., 2010).

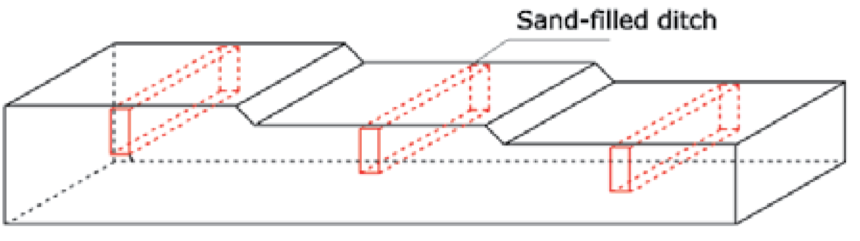


Fig. 2. Scheme of bench terraces with sand-filled ditches

Back-sloping bench terraces, presented at Fig. 3, are consisting of a riser, a compacted toe drain, located close to the riser, and a bed sloped back towards the toe drain (van Dijk & Bruijnzeel, 2003). The reported terrace risers have slopes of 35-50°, with terrace bed reaching the back-slope of 12°. The toe drain of approx. width of 0.3-0.4 m is usually slightly inclined (1-3°) towards the end of the terrace. Thus, the surface runoff water is redirected – the parallel flow, through the central drain, parallel to the contour line of the slope is possible. This type of terracing is usually suggested for regions of heavy precipitation.

Sone wall terraces (Fig. 4) are usually level or nearly level terraces based on stone walls (or stone bunds) reinforcing the risers. Stone walls or bunds are deployed along the slope. Then, with time, sediments deposition creates the terraces (Nyssen et al., 2000). Stone terraces may be used on steeper slopes, with more shallow soil cover. Moreover they have more permanent structure then ordinary bench terraces, with the ability of self-stabilizing but their construction costs are higher. These constructions are generally linked with regions of Mediterranean or Africa.

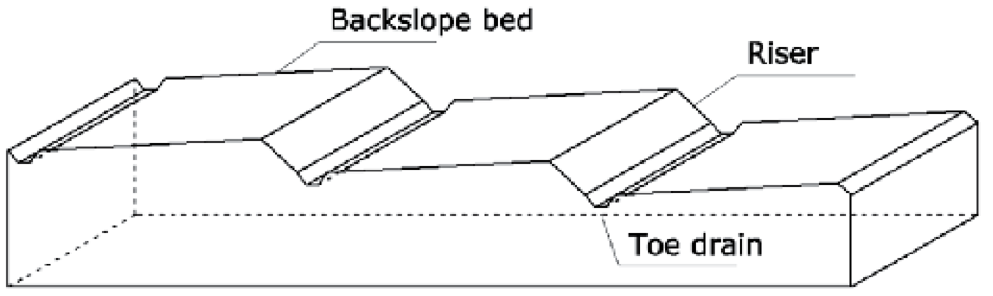


Fig. 3. Back-sloping bench terraces (modified after van Dijk & Bruijnzeel, 2003)

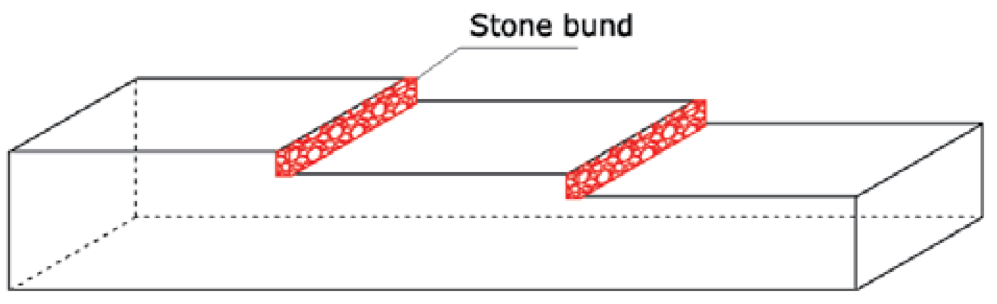


Fig. 4. Stone bund terraces

Fanya juu terraces (Fig. 5) are made by digging the trenches along the contour lines of terraced slope – the excavated soil is being thrown uphill to form an embankment, which may be also strengthened by grass cover. Then, sediments are being slowly accumulated by the upper part of the ditch and form the terrace. Thus, long slope is divided into shorter segments and surface runoff accumulates in the ditch and slowly infiltrates into the soil profile (Tenge et al., 2005). Generally, Fanya juu terraces require less labor then the bench ones. They are also applicable in the regions of thin soil cover, where benching is not suitable.

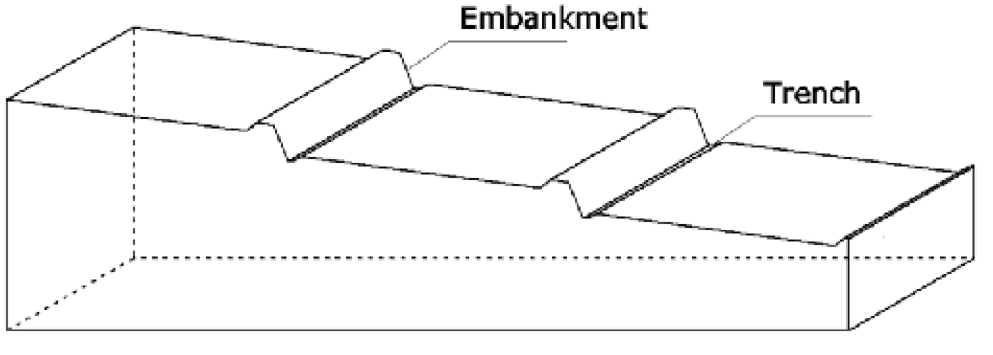


Fig. 5. Fanya juu terraces

## 3. Terracing as a measure of soil erosion control

### 3.1 Reported efficiency of terracing in soil erosion control

Terracing is generally reported as successful in limiting the soil erosion by water rate. Its efficiency in limiting the soil erosion rate is connected to reducing the volume and speed of rain surface runoff because the amount of lost soil is directly related to surface water flow (Zuazo et al., 2005). There are available several literature reports concerning the efficiency of terracing in limiting soil erosion compared to erosion rate for untransformed slopes in the same soil and climatic conditions for various regions of the world.

Researchers conducted by Hotta et al. (1967, as cited by Nakao, 2000) on Japanese fruit farm cultivating Satsuma mandarin seedlings compared clean culture and five different methods of soil erosion control: grass cover, straw mulch, grass strips, terraces with bare soil and stone wall terraces for the same slope, soil and climatic conditions for period of 23 months during 1963-1965. The results of the studies are presented in Table 1.

| Conservation measure | Soil loss, Mg ha$^{-1}$ Observed total precipitation: 1902 mm |
|---|---|
| Clean culture (no control) | 157.08 |
| Grass cover | 11.32 |
| Straw mulch | 1.18 |
| Grass strips | 81.63 |
| Bare soil bench terraces | 18.49 |
| Stone wall bench terraces | 11.98 |

Table 1. Different soil control measures tested in Japan (modified from Nanko, 2000)

The results of in-situ measurements presented in Table 1 show that tested terracing methods were a successful measure of erosion control, resulting in reduction of soil los rate by approx. 88.2% for bare soil bench terraces and 92.4% for stone wall terraces. The higher obtained reduction for stone wall terraces may be explained by the fact that the steeper elements in terracing systems, the risers, are made of stones, not soil. The presented results also indicate the high efficiency of grass cover, grass stripe and straw mulch application, which, combined with terracing, may increase its capability in reduction of soil erosion.

Exhaustive studies concerning soil erosion control by bench terracing application in Kerinci uplands of Sumatra, Indonesia were conducted by Siebert and Belsky (1990). Terracing located on complex red-yellow podzolics soils is used to plant *Setaria* grass. The observed annual precipitation in the region reached the level of 1945 mm, most of which falls in rainy season between September and May. Observed sediments loss, generated during the rainy season of 1987-88, reached the level of 3.81 Mg ha$^{-1}$ for uncontrolled sampling site and 1.13 Mg ha$^{-1}$ for bench terracing. The reduction of soil loss by approx. 70.3% obtained by terracing application is visible.

Tests of different types of terraces were conducted in African conditions by Tenege, De Graff and Hella (2005). Studies of erosion control system in Kawalei region, West Usambra, Tanzania, covered bench terraces, Fanya juu terraces and grass stripes as the most effective in local farmers' opinion. This region is characterized by an annual mean precipitation of 1000 – 1200 mm during two rainy seasons (March – May and September – November). The results of in-situ measurements conducted in 2002 and 2003 are presented in Table 2.

| | Soil loss, Mg ha$^{-1}$ | |
|---|---|---|
| Soil erosion control measure | Short rains - season 2002 | Long rains – season 2003 |
| None | 9.6 | 15.0 |
| Bench terraces | 3.0 | 3.1 |
| Fanya juu | 1.9 | 0.8 |
| Grass stripes | 6.0 | 8.3 |

Table 2. Erosion control methods tested in Tanzania (modified from Tenge et al., 2005)

According to the presented results (Tab. 2) Fanya juu terraces showed the highest efficiency in limiting soil erosion rate – the observed decrease of soil loss reached the level of 80.2% for 2002 season and 94.7% for season 2003. Slightly lower effectiveness of erosion control was observed for traditional bench terraces – 68.6% and 79.3%. Implementation of both terracing methods should be assessed as successful.

Very interesting information concerning efficiency of stonewalled terracing in Palestine, Mediterranean, according to various rainfall events categories and different types of erosion, was reported by Hammad et al. (2006). Tests were conducted for 50 years old stone wall terraces constructed on silt loam topsoil and silt clay loam subsoil, with annual mean precipitation of 580 mm, of which 90% occurs during the period of October – April. The selection of the most interesting reported results is presented in Table 3.

| | | Rainfall events category, mm | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | 0-10 | 10-20 | 20-30 | 30-40 | 40-50 | 50-60 | 60-70 |
| Non-terraced system | Interrill erosion, Mg ha$^{-1}$ | 0.0056 | 0.1391 | 0.1929 | 0.3418 | 0.2675 | 0.3695 | 1.0177 |
| | Splash erosion, Mg ha$^{-1}$ | 0.0749 | 0.1974 | 0.2838 | 0.3122 | 0.2816 | 0.5065 | 0.4898 |
| Stone wall terracing | Interrill erosion, Mg ha$^{-1}$ | 0.0009 | 0.0404 | 0.0384 | 0.099 | 0.1375 | 0.2024 | 0.3092 |
| | Splash erosion, Mg ha$^{-1}$ | 0.0687 | 0.1823 | 0.2476 | 0.2857 | 0.2612 | 0.4379 | 0.4449 |
| Reduction | Interrill erosion, % | 83.9 | 71.0 | 80.1 | 71.0 | 48.6 | 45.2 | 69.6 |
| | Splash erosion, % | 8.3 | 7.6 | 12.8 | 8.5 | 7.2 | 13.5 | 9.2 |

Table 3. Interrill and splash mean erosion for non-terraced system and stone wall terraces (modified after Hammad et al., 2006)

The results reported by Hammad et al. (2006) show that, according to variable rainfall event category, the share of two observed different types of soil erosion is also variable. Generally, the lower height of observed precipitation, the higher share of splash erosion in total observed soil loss is. Splash erosion is dominant up to 50-60 mm of precipitation in non-terraced environment and in all ranges of presented rainfall events categories for stone wall terraces. Application of terracing allowed reducing soil loss by splash erosion by 7.2-13.5%, while interrill erosion was successfully reduced by 45.2-83.9%. These results prove the ability of stone wall terraces to significantly reduce soil loss rate, even after very long period of operation – 50 years.

The above presented exemplary results of various types of terracing tests proved their efficiency in limiting soil erosion rate in local soil and climatic conditions. The highest level of soil loss reduction was achieved by Fanya juu and stonewalls terraces. This achievement is directly resulting from construction of these models of terraces – creating the embankments and enforcing the risers by stone walls. Traditional bench terraces, although also successful in significant reduction of soil loss, appeared to be less effective, according to the high rate of water erosion on short but steep risers.
High efficiency of all presented types of terracing may be additionally improved by application of proper vegetation cover. Various literature reports present high potential of different plants in limiting soil erosion rate, e.g. *Poaceae* and *Asteraceae* family, *Sarcopoterium spinosum* and *Sarcopoterium vercurosum*, *Avena sterilis*, *Lactuta virosa*, *Trifolium stellatum*, *Crupina crupinastrum*, *Vetiveria zizaniodes* and *Pennisetum purpureum* of various types of local grass (Kosmas et al., 2000; Mohammad & Adam, 2010; Sang-Arun et al., 2006).

### 3.2 Modeling of soil erosion on bench terraces

The efficiency of terracing in limiting soil erosion rate was presented by exemplary numerical calculations by WEPP model (Water Erosion Prediction Project) developed by EPA USDA (Environmental Protection Agency, US Department of Agriculture), for selected Polish soils, various vegetation cover types and local climatic conditions. WEPP is a well known and frequently positively verified model allowing soil erosion rate prediction basing on several groups of input data: soil properties e.g. particle composition, organic matter content, saturated hydraulic conductivity, mechanical properties and erodibility; climatic conditions e.g. daily precipitation, minimum and maximum air temperature; geometric characteristics of slope or catchment and, finally, vegetation cover description e.g. plant type or species, cover degree, plants height and leaf area (e.g. Amore et al., 2000; Beskow et al., 2009; Bhuyan et al., 2002; Grønsten & Lundekvam, 2006; Pandey et al., 2008; Van Lier et al., 2005).
Exemplary modeling calculations of soil loss prediction were conducted for traditional bench terraces constructed in 1980 - 90s in fruit farm in Olszanka, Poland on slope of 6-15 % inclination and Alfisols soil cover (e.g. Widomski et al., 2010). Required input data covering soil characteristics used in numerical prediction of soil loss on terracing system are presented in table 4.

| | Particle size distribution, % | | | Organic matter, % | Saturated hydraulic conductivity, m day$^{-1}$ |
|---|---|---|---|---|---|
| Depth, cm | Sand | Silt | Clay | | |
| 0-10 | 21 | 69 | 10 | 0.5 | 0.479 |
| 10-50 | 27 | 59 | 16 | 0.5 | 0.480 |
| 50-100 | 25 | 61 | 14 | 0.5 | 0.480 |

Table 4. Properties of soils used in prediction of water erosion for terraces in Olszanka, Poland

Input data covering climatic conditions were based on multiannual climatic measurements of daily precipitation and air temperature in the closest meteorological station, in Zamość, Poland. Numerical calculations were conducted for a section of terraces located in the top part of the slope (accordingly to WEPP limitations concerning the amount of points describing the slope) and non-terraced slope in the same conditions, 5 years time of simulation and three types of soil cover: bare soil, 50% and 100% grass cover. Modeled climatic conditions showed total precipitation of 2761.10 mm during 862 rainfall events and mean yearly precipitation equal to 552.22 mm. The obtained results of predicted soil loss for non-terraced and terraced slope are presented in Table 5.

| Conservation measure | | Mean annual soil loss, Mg $ha^{-1}$ | Soil loss reduction, % |
|---|---|---|---|
| Non-terraced | Bare soil | 4.576 | - |
| | 50 % grass cover | 2.054 | 55.1 |
| | 100% grass cover | 1.397 | 69.5 |
| Bench terraces | Bare soil | 2.211 | 51.7 |
| | 50 % grass cover | 1.531 | 70.5 |
| | 100% grass cover | 1.033 | 77.4 |

Table 5. Modeled efficiency of bench terraces constructed in Olszanka, Poland

Predicted rates of soil loss showed considerable reduction of sediments yield after application of bench terraces on modeled slope. But one should note that implementation of vegetation cover on untransformed slope may be also very effective – e.g. 50% grass cover gives comparable results than terracing with bare soil surface, 55.1% vs. 51.7%. Thus, combination of terracing and 100% cover allowed to obtain the highest level of soil loss reduction – 77.4% predicted. Presented calculations also showed specific characteristics of soil erosion by water on bench terraces. Figure 6 shows slope profile and relative soil erosion for modeled section of bench terraces. As it was described before, risers appeared the most prone element of terrace to soil erosion – calculated soil loss reached level of 3.124 kg $m^{-2}$ per year for the last, fourth riser in the section. On the other hand, the nearly level platforms trigger deposition of transported sediments. The predicted values of deposition were in range from 0.439 to 5.387 kg $m^{-2}$ per year. However, the presented predictive calculations lack empirical validation.

## 4. Soil moisture and infiltration on terraces

### 4.1 Reported effects of terracing on soil moisture and infiltration

Limiting surface runoff generation and speed resulting in increased infiltration and soil moisture content of soil profiles located on eroded slopes is one of the main tasks fulfilled by terracing (e.g. Tenge et al., 2005). Gathering rain water is accomplished by several constructional elements of various types of terracing: level, nearly level or back slope platforms, embankments, draining ditches or other draining elements. Exemplary report by Li et al. (1994, as cited by Lü et al., 2009) describing efficiency of terracing as soil conservation measure in selected watershed at Loess Plateau, China presented an increase of soil moisture by 20.7%, a decrease of soil removal and nutrients loss by 57.9%-89.9% and 89.3% - 95.9%, respectively, as the most important indicators of terracing effects.

Another studies conducted at Danangou catchment, Loess Plateau in China, by Fu et al. (2003) covered five land use structures and seven land use types, including terracing. Reported results of studies showed that mean soil water content for cropland on terraces was higher than that on slope orchard, fallow land, grassland and cropland on almost uniform slope – 15.2% vs. 11.15%, 11.09%,10.82% and 11.1%, respectively.

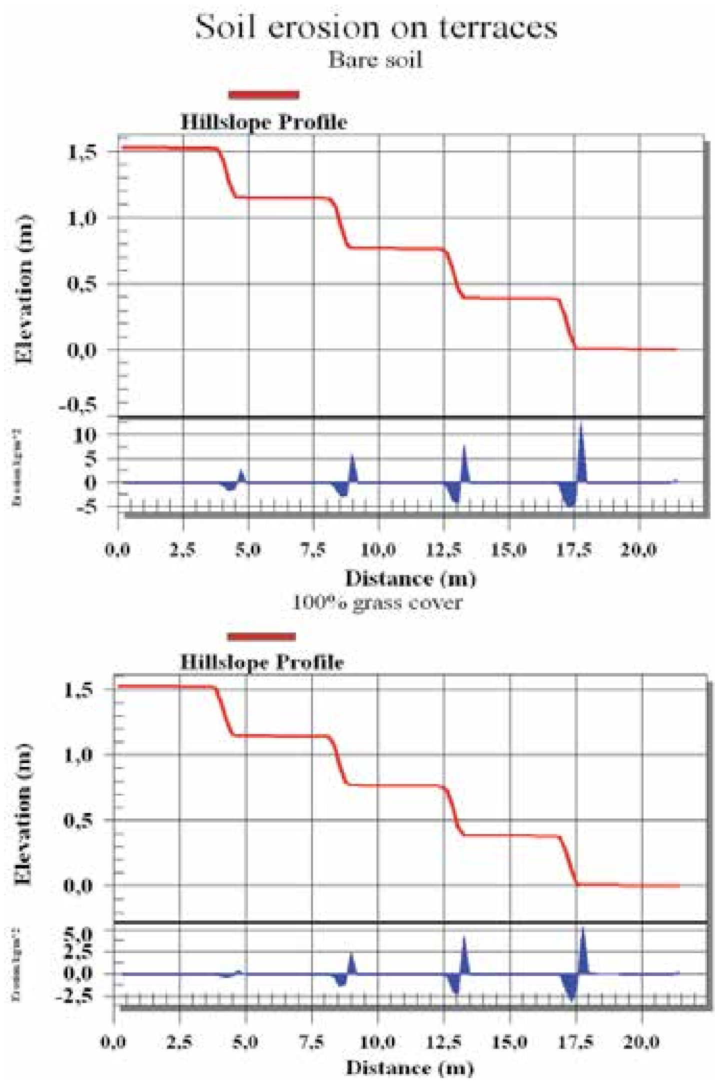


Fig. 6. Exemplary results of soil erosion prediction for bare soil and 100% grass cover bench terraces in Olszanka, Poland.

Effects of various types of terraces on soil moisture during short and long rains were tested in Kawalei region, West Usambra, Tanzania by already cited Tenege, De Graff and Hella (2005). Bench terraces appeared to be the most successful of the tested manners in retaining soil moisture, during both short and long rains. The mean soil moisture for bench terraces was 35.9% for short rains and 34.3% for long rains, while values noted for Fanya juu terraces, grass stripes and no-conservation were, respectively, 32.9% and 27.1%, 29.2% and 26.0% and finally 28.5% and 25.7% for short and long rains. The observed absolute increase of soil moisture for bench terraces was equal to 7.4% and 8.6% of water content which refers to relative increase by 26.0% and 33.5%. The corresponding reported values of absolute and relative increase of soil moisture for Fanya juu during short and long rainfall events were 4.4% and 1.4 % as well as 15.4% and 5.4%.

Interesting studies concerning various soil and water conservation techniques for cashew (*Anacardium occidentale L.)* cultivation were conducted in Puttur, Karnataka, India by Rejani and Yadukumar (2010) during the period of 2004-2010. Tested soil and water conservation techniques covered crescent bunds, coconut husk burial, reverse (back slope) terraces, catch pits and contour plot without any manner of conservation. Despite the fact, that according to the cited report, crescent bunds and coconut husk burial appeared to be the most effective in increasing soil moisture, terracing also showed some significant potential. The mean value of soil moisture for tested terraces system observed at three various depths (0-30 cm, 30-60 cm and 60-90 cm) during long-lasting experiment was equal to 14.6 % while soil moisture for system without any means of soil and water conservation was 11.6% (Rejani & Yadukumar, 2010). Thus, mean soil moisture of eroded slope with terraces was approx. 25.9 % higher than on non-terraced plot.

All the presented above, exemplary, scientific reports showed a significant potential of various types of terraces in increasing infiltration rate and resultant soil moisture content. Observed increases of soil moisture were dependant on slope inclination, local soil conditions, vegetation cover, terraces construction and climatic conditions.

### 4.2 Modeling of infiltration on terraces with sand-filled draining ditches

Modeling of infiltration efficiency on terraces was conducted for soil control system developed and tested on loess soils in Olszanka (Rubaj, 2002; Widomski et al., 2010) – Fig. 2. Numerical calculations were conducted to assess the increase of infiltration resultant from application of additional sand-filled drainage ditches to traditional level bench terraces. Commercial software FEFLOW by Wasy Ltd., Germany, based on finite elements method (FEM) was used in the numerical calculations. FEFLOW is a well known and repeatedly successfully verified model of groundwater movement, mass and heat transport in saturated or unsaturated porous media (Diersch & Kolditz, 2002; Mazzia & Putti, 2006; Trefry & Muffeles, 2007; Zhao et al., 2005). Numerical calculations were conducted on model consisting of a line of contiguous terraces (Fig. 2), with and without draining ditches, thus calculations were conducted for two various variants allowing to compare infiltration rate for ordinary bench terrace and terrace equipped with sand-filled draining ditch for two different locations on slope.

Numerical calculations of water movement in soil profile were based on standard form of Darcy's and Richard's equations (e.g. Pachepsky et al., 2003; Raats, 2001; Richards, 1931):

$$q = -K(\theta)\nabla\Psi \tag{1}$$

$$\frac{\delta\theta}{\delta t} = \nabla(K(\theta)\nabla\Psi) - S(\theta) \tag{2}$$

where: $q$ – groundwater flux, $\theta$ - volumetric water content, $t$ – time, $K(\theta)$ – hydraulic conductivity, $\Psi$- water potential, $S(\theta)$ – sink or source term.

The mathematical description of water retention curve shape adapted to presented calculations was presented by Mualem (1976):

$$\theta = \frac{\theta_s - \theta_r}{[1+(\alpha h)^n]^m} + \theta_r \tag{3}$$

where: $\theta_s$ – saturated volumetric water content, $\theta_r$ – residual volumetric water content, h – soil matric suction pressure, $\alpha$, $n$, $m$ – fitting parameters, $m = 1\text{-}n^{-1}$.

The above formula may be also presented in degree of saturation based form (i.e. Diersch 2005):

$$S = \frac{S_s - S_r}{[1+(\alpha\psi)^n]^m} + S_r, (\psi < 0) \tag{4}$$

where: $\psi$ – groundwater pressure potential.
Relative hydraulic conductivity of unsaturated soils was calculated in the presented model according to van Genuchten's formula (1980):

$$K = K_{sat} S^l \left[1 - \left(1 - S^{\frac{1}{m}}\right)^m\right]^2 \tag{5}$$

where: $K$ – relative unsaturated conductivity, $K_{sat}$ – saturated conductivity, $S$ – saturation fraction , $l$ – fitting parameter, $l$ = 0.5 (Diersch, 2005).
The presented numerical calculations required the following set of input data: geometric characteristics of slope, terracing and draining ditches; soil physical and water-transport characteristics, vegetation cover data and initial and boundary conditions. Required input data were obtained by in-situ and laboratory measurements and literature studies.

### 4.2.1 Input data

Soil physical and transport characteristics were studied in three layers at two different locations. Saturated hydraulic conductivity of soils was measured in situ by double-ring infiltrometer and in laboratory by soil permeameter, both manufactured by IMUZ, Poland. Water retention curves for tested soils were obtained by sand and plaster box, IMUZ, Poland. According to literature reports (Stauffer & Kinze, 2001; Werner & Lockington, 2003) single-valued mean retention curves were applied as adequate to time-average soil moisture profiles. Soil anisotropy ratio was obtained by cubic samples method (Iwanek, 2008). Input data covering characteristics of tested soils are presented in Table 6.

| Location | Depth, cm | Saturated conductivity $K_s$, m d$^{-1}$ | Anisotropy ratio | Saturated water content $\theta_s$, m$^3$ m$^{-3}$ | Residual water content $\theta_r$, m$^3$ m$^{-3}$ | Fitting parameter $\alpha$, cm$^{-1}$ | Fitting parameter $n$ |
|---|---|---|---|---|---|---|---|
| Top of slope | 0–10 | 0.479 | 2.770 | 0.3667 | 0.065 | 0.0029 | 1.8902 |
| | 10–50 | 0.480 | 1.150 | 0.3844 | 0.050 | 0.0039 | 1.3644 |
| | 50–100 | 0.480 | 1.150 | 0.406 | 0.050 | 0.0048 | 1.3816 |
| Bottom of slope | 0–10 | 0.492 | 2.302 | 0.3895 | 0.07 | 0.0051 | 1.4245 |
| | 10–50 | 0.620 | 0.870 | 0.4428 | 0.051 | 0.0082 | 1.3664 |
| | 50–100 | 0.620 | 0.870 | 0.4434 | 0.051 | 0.009 | 1.3572 |
| Sand in ditch | 10–80 | 7.690 | 1.000 | 0.42 | 0.010 | 0.0100 | 1.9600 |

Table 6. Characteristics of soils accepted to numerical studies

The following initial and boundary conditions were assumed to modeling:
- initial conditions covering soil moisture distribution in the tested terraces profile based on soil moisture in-situ measurements by manually operated TDR by Easy Test, Poland with accuracy of 2.0% of measuring scale;
- bottom boundary Dirichlet condition covering time dependent groundwater head obtained by water retention curves and volumetric soil moisture measurements;
- upper boundary Neumann condition describing inflow and outflow to the model, covering observer rainfall as well as runoff and evapotranspiration calculated by SWAP (Ben-Asher et al., 2006; Eitzinger et al., 2004; Sawar & Feddes, 2000);
- side Neumann gradient-type boundary condition allowing free movement of ground water.

Additional input data, such as climatic conditions covering air temperature and humidity, wind speed and precipitation, required to calculate mean daily evapotrnaspiration, were obtained by local weather station. Grass cover's roots distribution was measured in-situ, while value of Leaf Area Index (LAI) was based on literature studies (Mitchell et al., 1998). Values of measured precipitation and calculated evapotranspiration applied to numerical modeling are shown on a bar chart presented on Fig. 7.

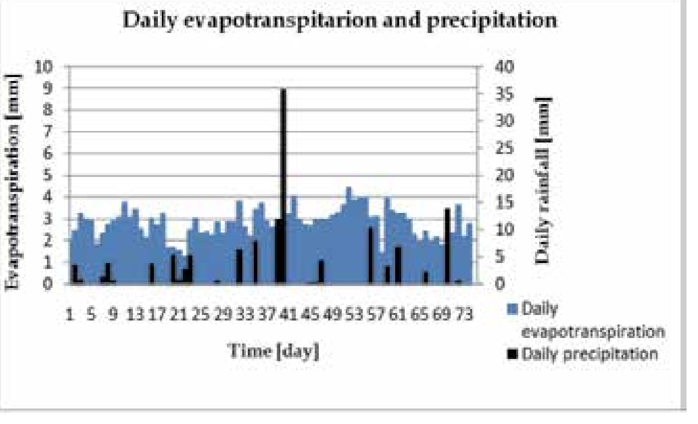


Fig. 7. Daily evapotranspiration and precipitation used as input data in modeling

### 4.2.2 Results of infiltration modeling

Modeling of infiltration on bench terraces and bench terraces with additional sand-filled draining ditches in soil and climatic conditions of Olszanka, Poland field site was conducted for the period of 74 days in summer of 2003 (17$^{th}$ June to 29$^{th}$ August). This period of simulation was selected according to various observed rainfall events – 24 events, precipitation from 0.5 mm to 34.8 mm. Assessment of infiltration was conducted for the elemental horizontal cross section located 1.0 below the ground surface and with span equal to the length of level platform of terrace, thus dimensions of the virtual surface were 1.0 m width and 4.0 m length.

Fig. 8 shows curves presenting accumulated infiltration for two tested terraces, with and without sand-filled draining ditch, for two various locations on the tested slope (top and bottom). In both tested cases the higher calculated infiltration was noted for the terraces equipped with draining ditch. This result may be explained by location of the tested terraces along the slope – terrace located at the base of the slope shows lower total infiltration because of different soil characteristics, higher run off speed and higher initial moisture content. In both cases the presented calculations indicate the proper reaction on rainfall appearance. The total volume of infiltrated water for all tested cases during the whole period of simulation is presented in Table 7.

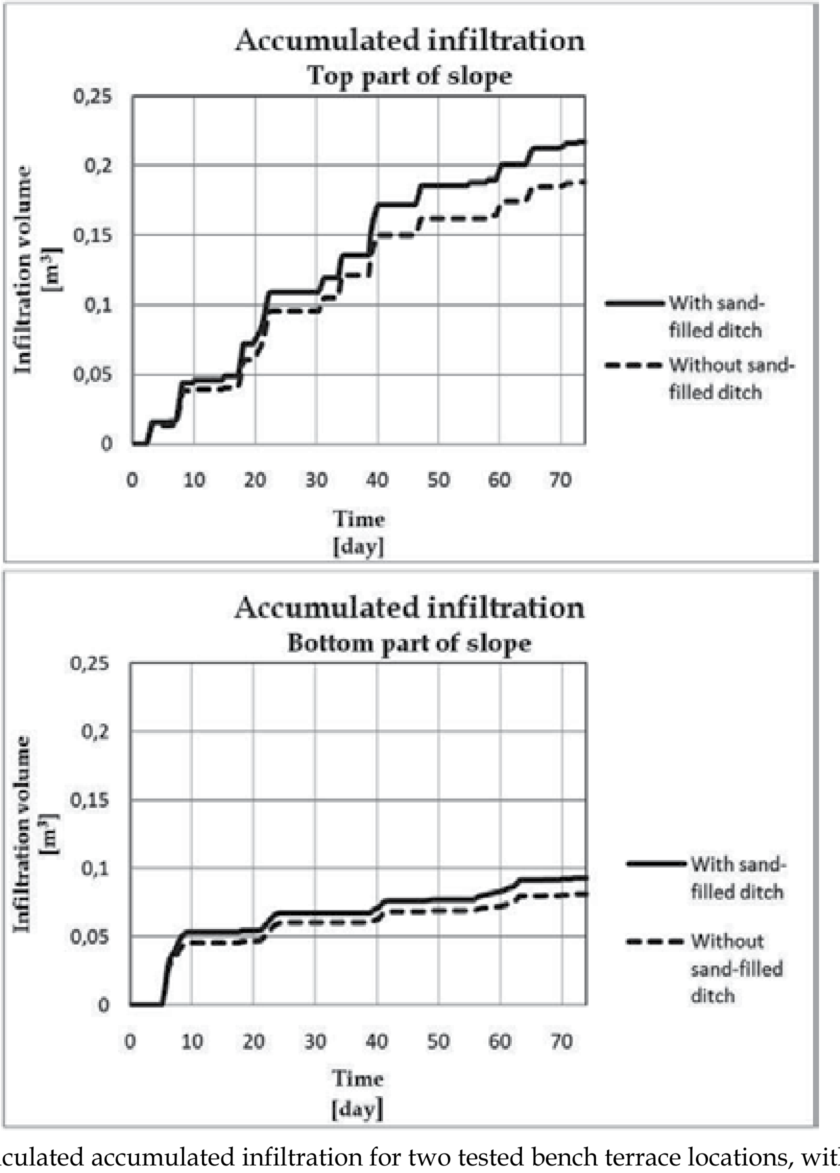


Fig. 8. Calculated accumulated infiltration for two tested bench terrace locations, with and without sand-filled ditches

The calculated total volume of infiltration for two studied types of bench terrace, with and without sand draining ditch, reached the values of 0.217 $m^3$ – 0.188 $m^3$ for top part of the slope and 0.093 $m^3$ – 0.082 $m^3$ for bottom part of slope. Thus, mean daily infiltration rate calculated for terrace located at top part of the slope for observation period was equal to 2.94 $dm^3\ day^{-1}$ for terrace with sand-filled ditch and 2.54 $dm^3\ day^{-1}$ without it. Results of the conducted calculations obtained for terrace located at bottom part of the slope showed daily mean infiltration rate of 1.26 $dm^3\ day^{-1}$ and 1.11 $dm^3\ day^{-1}$ for terraces with additional draining elements and without it. Despite the fact that total accumulated volume of infiltrated water and daily mean infiltration rate varies significantly for the two tested terraces, with and without sand ditches, located at different parts of the slope, the observed increase of calculated infiltration in both cases reaching the level of 15.60% – 14.08% is similar and clear.

| Location | Terracing type | Total infiltration volume, $m^3$ | Infiltration volume increase, % |
|---|---|---|---|
| Top part of slope | Standard bench terrace | 0.188041 | 15.60 |
| | Bench terrace with sand-filled draining ditch | 0.216909 | |
| Bottom part of the slope | Standard bench terrace | 0.081575 | 14.08 |
| | Bench terrace with sand-filled draining ditch | 0.093059 | |

Table 7. Calculated infiltration volume for tested terraces in two various on-slope locations

Empirical validation of presented numerical model was conducted by comparison of daily measured (19.00 p.m.) and calculated values of soil volumetric water content for the whole period of simulation (74 days), for four measurements locations – at the depth of 35 and 50 cm below the ground surface for each tested terrace. However, equipment failures and days off of the measuring staff resulted in reducing the number of applicable observations to 58 pairs. The observed coefficients of determination derived from linear regression of measured values of volumetric soil moisture vs. calculated ones were in range of $R^2$=0.702-0.799 (P=0.05). Thus, the values of soil moisture obtained by numerical modeling are in good agreement with the values measured in situ.

## 5. Summary

Various types of terracing are a common, worldwide known, method of soil erosion control. Among all described tasks of terraces the most important are: limiting the soil erosion rate and improving water balance of eroded basins by increased infiltration of surface water into deeper layers of soil profile. Presented literature reports proved that different types of terracing, in various local soil and climatic conditions, with different vegetation cover, are successful in decreasing soil removal by water erosion. The maximum level of soil loss decrease observed in the cited studies was equal to approx. 90%. Additionally, conducted numerical prediction of mean annual soil erosion for bench terraces constructed in fruit farm in Olszanka, Poland showed a significant decrease of calculated soil loss obtained due to terracing application. Moreover, numerical calculations showed the importance of vegetation cover strengthening the soil surface in limiting soil removal by surface run-off. Various studies reported in literature also proved the importance of terracing, especially additionally equipped with elements limiting run-off speed, in increasing the infiltration of surface water into the soil. Maximum reported increase of mean soil volumetric water content was approximately equal to 20%. The presented numerical assessment of efficiency of sand-filled ditches installed on bench terraces in Olszanka, Poland showed that total volume of infiltration increase in the tested terracing system was approx. 15% higher than for traditional bench terraces. Advantages of various types of terracing should entrant to their wider application, but one should also remember that terraces are relatively costly in construction and maintaining, especially terraces equipped with various elements redirecting surface run-off and increasing infiltration.

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The book deals with several aspects of soil erosion, focusing on its connection with the agricultural world. Chapters' topics are various, ranging from irrigation practices to soil nutrient, land use changes or tillage methodologies.

The book is subdivided into fourteen chapters, sorted in four sections, grouping different facets of the topic: introductive case studies, erosion management in vineyards, soil erosion issue in dry environments, and erosion control practices.

Certainly, due to the extent of the subject, the book is not a comprehensive collection of soil erosion studies, but it aims to supply a sound set of scientific works, concerning the topic. It analyzes different facets of the issue, with various methodologies, and offers a wide series of case studies, solutions, practices, or suggestions to properly face soil erosion and, moreover, may provide new ideas and starting points for future researches.

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