Best Practices in Evaluation and Restoration of Degraded Mediterranean Environments

Monografías do IBADER - Serie Territorio

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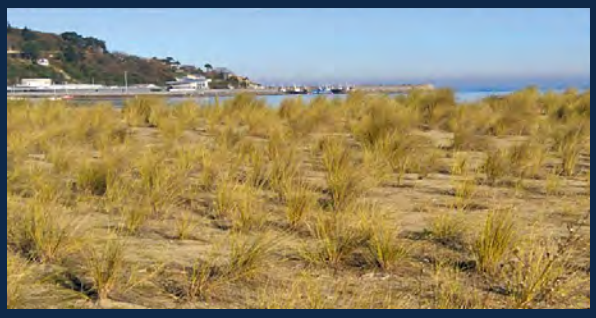

LANDCARE

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IBADER
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Monografías do IBADER

Instituto de Biodiversidade Agraria e Desenvolvemento Rural

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Contents

Preface

Let's face it. There is no way to restore an ecosystem, not to mention an entire landscape. The amazing diversity of organisms contained in an ecosystem, even the tiniest one, and the variety of the interactions needed to generate so many functions should be regarded as a unique wonder. As ecological restoration progresses, we are increasingly convinced that it can never substitute protection and conservation.

Ecosystem's complexity is so overwhelming, that we need to cut them into pieces before we can try to understand it. Not surprisingly, when it is time for rebuilding, we focus on dominant, key, charismatic species, hoping that the many bolts and nuts that are left aside will spontaneously join a machine that will run finely.

Furthermore, we want the ecosystem recovered in a legislative period or a few decades, disregarding the increasing amount of evidence showing that the effects of natural and anthropogenic disturbances can be detected after millennia of secondary succession.

Ecological restoration is about speed and acceleration, thus assuming that tempo is not an integral part of ecological processes. We use large amounts of exogenous energy in the form of fertilizers, organic amendments, physical structures, geomorphic profiling, seed banks, machinery, nurseries and labor to summarize a process that may last for centuries into a few years or decades. While doing this, we forget that biodiversity is inversely related to energy flow ($d^2B/dt^2 = -dD/dt$, where B is biomass, D is diversity and t is time, as Margalef (1968) suggested). Clearly, more studies are needed to understand the relationship between energy inputs, community assemblage and ecosystem function, that is, between resources committed and restoration success.

Yet, there are still plenty of reasons to engage in ecological restoration. Let's name three. On the one hand, there is no doubt that ecological restoration can help to protect biodiversity and increase the provision of ecosystem services. Even if the successional trajectory is not the one that was originally planned, accelerated increases in ecosystem integrity favored by human interventions can potentially bypass ecological filters and generate useful ecosystems to sustain nature and humans. On the other hand, urbanized civilizations increasingly need opportunities to reconnect with nature; the healthier relationship between nature and culture pursued by the Society for Ecological Restoration. Ecological restoration can play a crucial role in this respect, if its benefits are clearly understood and communicated. Finally, ecological restoration is not only an acid test for ecological theory (Bradshaw, 1986), but also an immense opportunity to generate new theory and increase our understanding of socio-ecological systems.

To advance in this direction, we must generate knowledge, and make it accessible for practitioners. Case studies are an efficient education and training tool: they promote meaningfully learning, present authentic contexts which integrate the complexity of socio-ecological systems, and integrate multiple sources of information. They stimulate the feeling that knowledge is applicable, encourage critical thinking, deliberation and discussion, and promote engagement (Yadav et al., 2007).

The volume Best practices in evaluation and restoration of degraded mediterranean environments, which I have the honor to introduce, reviews major drivers of degradation in Mediterranean agricultural systems, wetlands, coastal areas, forests and woodlands, and polluted soils, and describes the main approaches used to restore these environments. More importantly, it includes 29 case studies, describing different sites and restoration programs. It contributes to the emerging catalogue of examples of good practices in ecological restoration, which, as Chair of SER Europe, I can only welcome, and wish that will continue growing and increasing in accessibility for the benefit of the ecological restoration community. This book is the result of a 3-year international and multi-disciplinary effort to promote education and training in ecological restoration, and certainly it will contribute to these aims.

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Introduction

A large part of the Mediterranean region is affected by land degradation caused by anthropogenic pressure (overexploitation of agricultural and forest land, invasive tourism, wildfires, pollution, and desertification). The degradation is a challenge to the sustainable use of land and affects the provision of important services (food, water, biodiversity, wood, etc.) as well as the livelihoods of the inhabitants. This process has a strong effect on the region, where part of its economy (agriculture, fishing, tourism) and vital needs (food, water) depends on the conservation of its natural resources (EC, Land Degradation and Desertification). The European Directive 2011/2307(INI) "Our life insurance, our natural capital: an EU biodiversity strategy to 2020", includes as an objective for 2020 the restoration of at least 15% of degraded areas.

The application of appropriate knowledge and techniques can be used to recover disturbed land. However, land Rehabilitation training requires constant updating of knowledge and technologies, as well as study with real cases.

For its strong impact of economy and environment, land rehabilitation is generating an emerging labour market. According to European Directive 2011/2307(INI) "actions to restore ecosystems and biodiversity have significant potential to create new skills, jobs, and business opportunities". However, despite this demand and the opportunity to increase employment, there is a considerable shortage of skilled workers in this field. As a reflection of this problem, the "Green Employment Initiative" has been recently launched by the European Commission.

The current handbook is the result of a three-year Educational project in Southern Europe (2015-1-ES01-KA203-016214), whose overall objective has been to improve teaching and training capacities in relation to Land Degradation and Rehabilitation in this area to fulfil the demand of an emerging labour market and to contribute to the economy of the region. The material shown is different study cases of best practices discussed by the teachers, managers and students throughout the three intensive courses held in Lugo (Spain), Samos (Greece), Pisa (Italy) and Workshops in Lisbon and Santiago de Compostela.

The handbook was designed to gather most updated knowledge on land degradation processes and strategies of land restoration. It is focused on the main environmental problems of Southern Europe: overexploitation of agricultural and forest lands, pollution, wildfires, coastal degradation, and wetland-aquatic systems. One chapter for each of these topics is devoted. In addition, a chapter in which the impact on rural development is discussed has also been also included. All the chapters have a common structure: a) a general background explaining land degradation processes and welltested techniques of land rehabilitation and b) a series of selected case studies which show the application of technologies and tools in different regions.

http://www.ibader.gal/landcare-project.html

I Land use and agricultural practices

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1. Introduction

Soil sustains the 90 % of the production of food, feed and fiber; additionally, it contributes for the health of the entire ecosystem (FAO, 1994). The main processes of nutrient cycling are occurring within the soil, and it also plays an important role in the regulation of water flow and quality. Similarly, the soil represents an important reservoir for genes and elevated number of species from different *taxas*, and therefore supporting biodiversity (Maeder *et al*., 2002). On the other hand, because it is the main carbon sink of terrestrial ecosystems (Gallardo, 2016), it also plays an important role in the climate change mitigation. In summary, soils accomplish many services (Table 1).

Thanks to the modern agriculture, the food production has been increased to fulfil the huge needs of the current growth rate of human population, reducing the famine and improving the nutrition. About half of the global land is used for grassing or intensive agriculture (FAO, 1994). However, land cover and land use changes from natural forest or grassland to croplands or pasture and subsequent soil management have a strong negative impact on biomass, soil properties, and alter the soil ecosystem services (Gallardo, 2015).

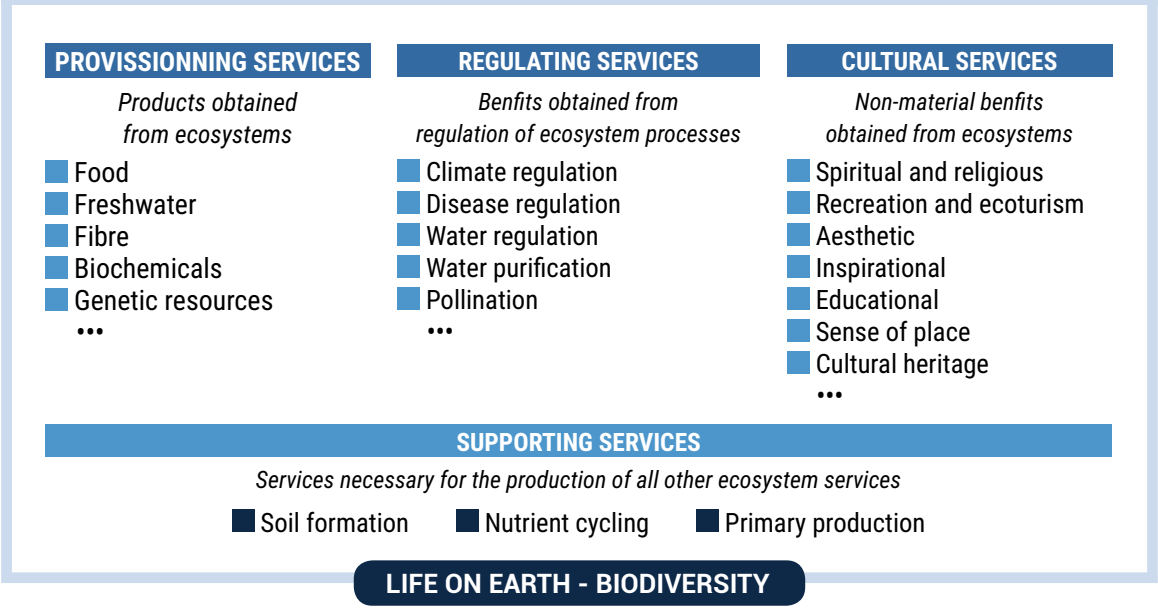
Land use changes and intensification agricultural practices affect different soil properties and the functions of the soil on the ecosystems. Soil organic matter (SOM) loss, degradation of soil physical properties, erosion, contamination, nutrient mining and salinization are the main degradation processes generated by intensification of agricultural practices (Table 2).

This chapter describes the five main soil properties affected by land use and management: (1) soil organic matter (SOM); (2) physical properties; (3) nutrients balance and supply; (4) salt balance (by salinization/alkalinisation); and (5) soil conservation from wind and water erosions).

In a subsequent section, the main techniques and strategies to correct the degradative processes are described.

Finally, four selected study cases of land restoration are discussed.

FIGURE 1
Ecosystem services categories



Source: Adapted from Ecosystems and human well-being: a framework for assessment by the Millennium Ecosystem Assessment. Copyright © World Resources Institute. Reprogrames by permission of Island Press, Washington. DC.

***Table 1**. Examples of ecosystem services provided by soil, and control factors.*

Ecosystem service	Control factors
Life support	Soil and associated terrestrial ecosystem development are a function of climate, topography, underlying surficial materials, functional groups of species, time, and land use.
Soil climate regulation and atmospheric CO_2 sequestration	Soil carbon sequestration capacity is function of moisture regimen, temperature, soil texture, structure, nutrient availability, associated vegetation and ecosystem disturbance regime The regulation of greenhouse gasses (carbon dioxide, methane, and nitrous oxides) emissions are controlled by soil moisture regime, temperature, nutrients, and microbial activity levels.
Flood regulation	Soil texture and structure, and organic matter content affect infiltration rate, erosion potential and water storage capacity.
Water purification and soil contaminant reduction	Soil texture (mainly silt and clay content), soil structure, organic matter content, and functional biodiversity of soil biota.
Food and fiber production	Nutrient cycling support plant growth, including food and fiber production.

***Table 2**. Key agricultural management practices and their consequences for soil quality and functioning.*

Agricultural practice	Specific management	Impact on the soil and related ecosystem functions
Cropping system	Harvest frequency Monoculture Rotation Intercropping	Reduction of variety of organic residue inputs to soil with negative effects for soil biodiversity and its function Degradation of physical properties Losses of organic rich layers of soil by erosion Pesticide accumulation Degradation of chemical properties
Use of agrochemicals	Over fertilization Abuse of pesticide use	Nitrate and phosphorus accumulations Large nitrous oxide and methane emissions Water pollution Exacerbate the water scarcity in arid and semi-arid regions Contamination of coastal ecosystems by large N inputs from the agroecosystem catchments
Irrigation	Irrigation system (traditional or modern) Fertirrigation	Erosion Water scarcity Secondary salinization
Livestock management	Overgrazing Industrial breeding	Soil erosion Soil compaction Soil degradation Loss of valuable species and nutrient losses Water pollution Accumulation of medicine and antibiotic residues
Agriculture in wetlands (including rice production)		Increase in carbon dioxide and methane emissions Changes of intensity and nature of emissions after drainage

2. Land management and soil conservation

2.1. Land uses/land cover and land use changes

Food production systems demands large areas of land. The information on land use/land cover shows that pastures and mosaics, and crops make up more than 40% of the total global land and European land surfaces (Figure 1). Also, large areas are covered by trees, including native forests and woodlands. And bare soil is the other most represented cover classes, mainly at global scale.

Over the history, European landscape and vegetation have been subjected to deep and continuous transformations, usually implying important degradation of native forest and grassland ecosystems (Bauer, 1991; Marty *et al*., 2007). During the 19th and 20th centuries, deforestation reached unprecedented rates linked to an increasing population pressure and an intensification of the agricultural practices. As a consequence of the rapid loss of trees and denudation of soils, serious problems of soil and water conservation (slipping, landslide, erosion, and flooding events) occurred from mainly 1860 on (Camacho *et al*., 2002; Araque, 2009).

As a result of the demographic exodus from rural to urban areas, and the implementation of National and European Afforestation Programs, from 1940 the trend to lose forest has changed (EEA, 2017). From this time onwards, and especially from 1970s, rural depopulation caused widespread land abandonment in many parts of Europe. In some areas, crops and pastures have been mostly neglected and a natural secondary succession occurred, increasing forest-land recovery (for example, in Spain: Merino *et al*., 2015). However, in recent years considerable losses of forest has occurred due to forest wildfires. In the Mediterranean countries the abandonment of agricultural lands was a complex phenomenon driven by socio-economic and ecological conditions (soil and climatic limitations), and in some cases, by land mismanagement (Rey Benayas *et al*., 2007; García-Ruiz & Lana-Renault, 2011).

However, from 1900, the adoption of measures aimed to reducing intensive agricultural methods involved afforestation schemes or conversion to grazing lands (Figure 2).

Another trend detected in the report by EEA (2017) is the decrease of farm structures of traditional rural landscapes, part of which has been transformed to a more intensive agriculture or have been abandoned.

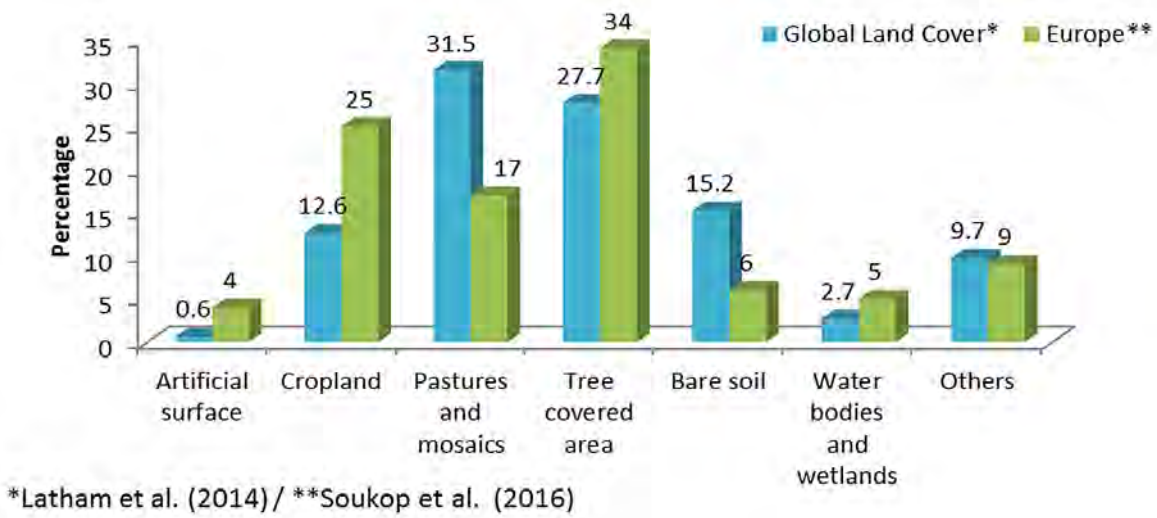


Figure 1. Proportions of global land cover types (GLC-SHARE data base; adapted from Latham et al., 2014) and land cover types in Europe (39 countries Corine Land Cover 2012 data set; adapted from Soukop et al., 2016).

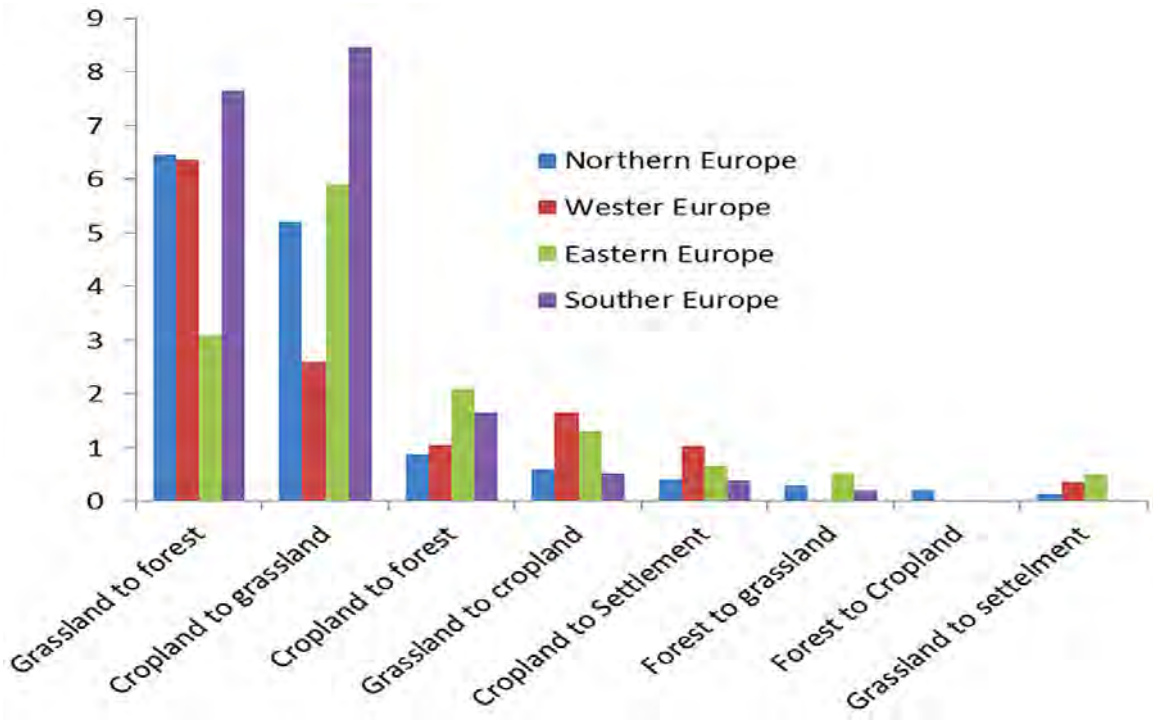


Figure 2. Main land transitions and relative amount of land changes per region for 1950-2010 (adapted from Fuch et al., 2012).

2.2. Soil organic matter in managed soils

Role and functions: Soil organic matter content and quality is a key factor for the fertility and conservation of soil, and then for sustainability of the crop production. Although SOM only constitutes a small fraction of the total mass of soil, it affects strongly the physical, chemical and biological properties of the soil (Gallardo, 2016). On the other hand, SOM plays a critical role in the carbon balance of terrestrial ecosystems, affecting the global carbon cycle (Intergovernmental Panel on Climate Change, 2013).

The net accumulation of carbon in the soil as SOM, is result of the balance between carbon inputs (biomass or necromass returned to the soil) and outputs (mineralization of the soil organic matter), which is determined by climate and certain geophysical controls (Figure 3). The amount of organic matter varies widely between different soils, from low contents in arid and sandy soils, to as much as 30 % in the first 10 to 20 cm of the soil profile under forests. This SOM storage can be abruptly altered also by changes in land cover and in land use (i.e., by human activities as the management of the resource), which can result in remarkable SOM losses. Thus, for the same climatic and soil conditions, the highest SOM contents are found in forests; pastures and grasslands usually have lower SOM contents, although peat soils and even soils found in humid areas show higher SOM contents. Croplands have the lowest SOM contents.

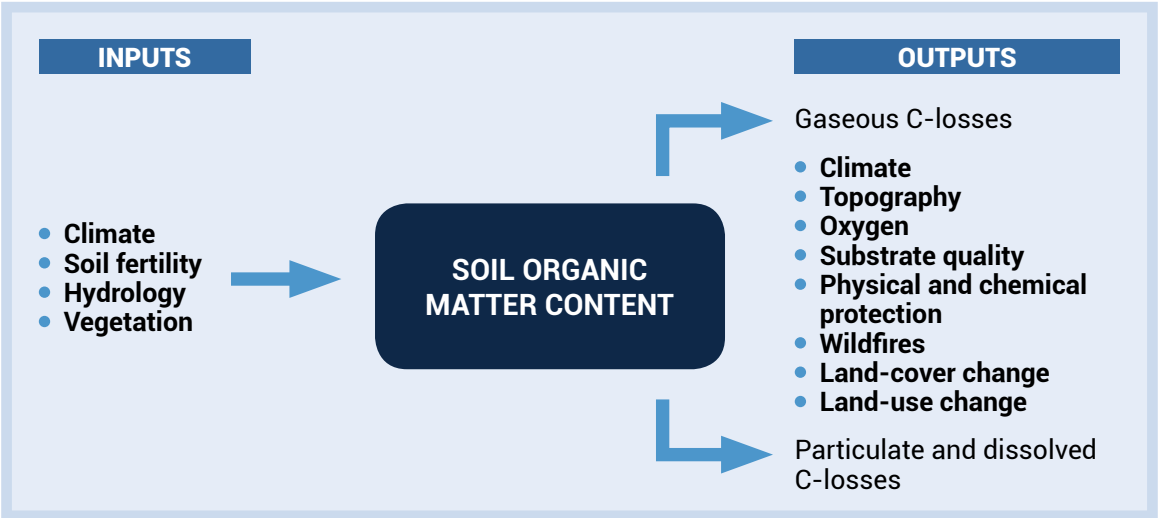


Figure 3. Environmental controls the organic carbon-balance in the soil.

Influence of intensive cultivation on soil organic matter (SOM): Transition from native forest to crop land usually includes an intermediate stage when the soil is covered by native or induced pasture, which cushions temporally soil and SOM losses. However, when conversion of native forests to croplands occurs directly a rapid loss of SOM is produced.

Unlike natural landscapes in which most of the organic residues produced by the vegetation is returned to the soil, in cultivated areas, large amounts of plant biomass are removed, and in some cases the soil is bare. Zero tillage and conservation agriculture are practices been designed to reduce the impact of agriculture on SOM loss.

Soil tillage (i.e., the mechanical preparation of soil for cultivation) favours aeration and breaks up organic residues, making them more accessible to microbial decomposition. However, this practice has negative consequence, leading to losses of N and SOM from the system.

The loss of SOM is illustrated in Figure 4, which shows the changes in this parameter throughout the period of 30 years since a natural meadow was tilled and a continuous crop was established. Even higher losses of SOM than those shown are observed in tropical humid forest regions, in which high temperatures and abundant rainfall amount favour the loss of SOM mineralization. Abandonment, afforestation or establishment of grasslands in arable land usually lead usually to important gains in the SOM content (see section 3.2.1).

Another land use change which affects SOM content is the drainage of peatlands and wetlands (Kayranli et al., 2010) and their conversion to agriculture. This is particularly important because the huge carbon dioxide (CO_2) emissions generated.

The soils from intensively managed forest plantations in tropical and temperate regions are also suggested to SOM losses (Guillaume et al, 2015), although in a lower extent than those in croplands.

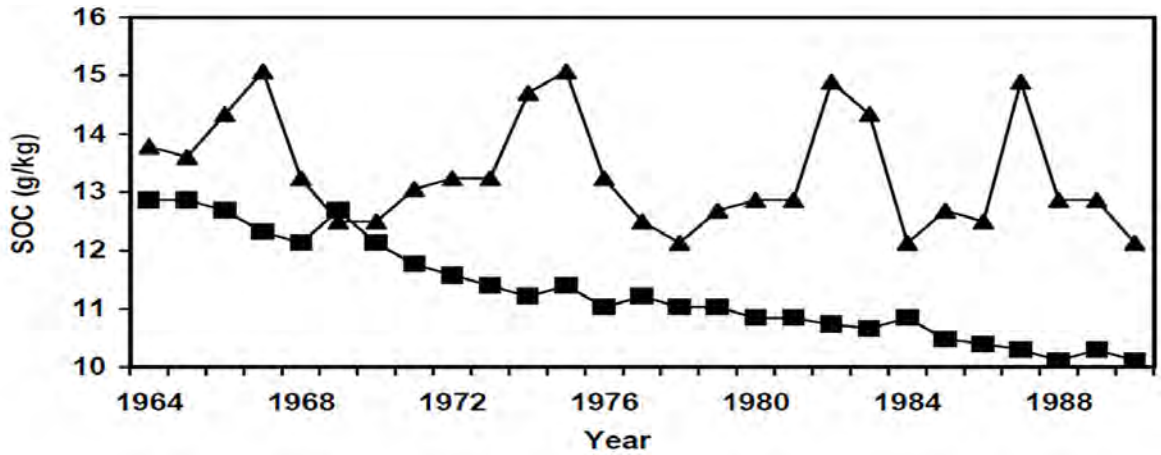


Figure 4. Changes in soil organic carbon (SOC) concentration (0-20 cm depth) in two contrasting systems with conventional tillage (García-Préchac et al., 2004). This figure shows the loss of SOC after transformation of a forest soil to an agricultural soil under continuous cropping. The inclusion of a pasture helps to maintain the SOC content; similar effect is obtained when manure is regularly applied to the soil.

2.3. Soil physical properties and soil management

Role and functions of the physical properties: The physical properties of soil influence the functions in the ecosystem and how the soil should be managed. These properties determine not only the movement of water and solutes through the soil and the susceptibility of the soil to erosion, but also the growth of crops.

Influence of tillage and crop residues management on physical soil properties: Tillage breaks aggregates, incorporates organic residues from previous crops to the soil, kills weed seeds, and creates favorable conditions for crop plant germination and growth. In addition, it reduces water losses by evaporation and enhances the mineralization of organic residues, releasing nutrients for crop demand. In the long term, however, tillage operations have a negative effect on the structure of the soil. During tillage soil mixture encourages oxidation of SOM, effect that is greater when crop residues are removed. In addition, the use of mechanized tillage requires consumption of fossil fuels that generates CO_2 emissions to the atmosphere.

When the soil loses SOM, the soil aggregates collapse. Continuous cultivation, especially on soil with high SOM content, often leads to a reduction of the large pore space or macropores. Tillage reduces the content of SOM, and then, it produces a reduction of the total pores as the most marked effect. An appropriate amount of macropores are necessary for water and air movement in the soil. A reduction in the pore size tends to reduce the infiltration (water is accumulated on soil surface; Figure 5) and the air content, decreasing the rate of organic residue decomposition and nutrient mineralization in the soil.

Tillage operations, especially those that use heavy machinery, tend to break the aggregates and to encourage soil compaction. Increases in bulk density and soil strength reflect the depletion in porosity and soil hydraulic conductivity with important negative effects on soil degradation and plant growth.

In addition, aggregates exposed on the surface of the soil are very vulnerable to destruction by rain drops. The smallest particles tend to enter the pores and seal them. Deposition of these particles within the pore spaces creates a superficial crust, a compacted layer of a few millimetres thick. The result is the formation of a sealed surface layer that prevents infiltration, increasing the surface runoff and the erosion of particulate organic matter (made up by partially decomposed plant debris which is not bound to mineral particles) and mineral soil (Figure 6). Furthermore, the presence of crusts greatly reduces the ability of seeds to germinate, thus prolonging the time during which the soil remains unprotected. In arid or semiarid soils, it reduces the amount of water available to crops. The crusting after planting allows only a small number of seedlings to emerge, so it is generally necessary to perform a new planting.

Modern agriculture what privileges monoculture that over traditional rotations and the use of heavy machinery as worsened the physical conditions of the soil, leading to an increase of the degradation and loss of SOM.

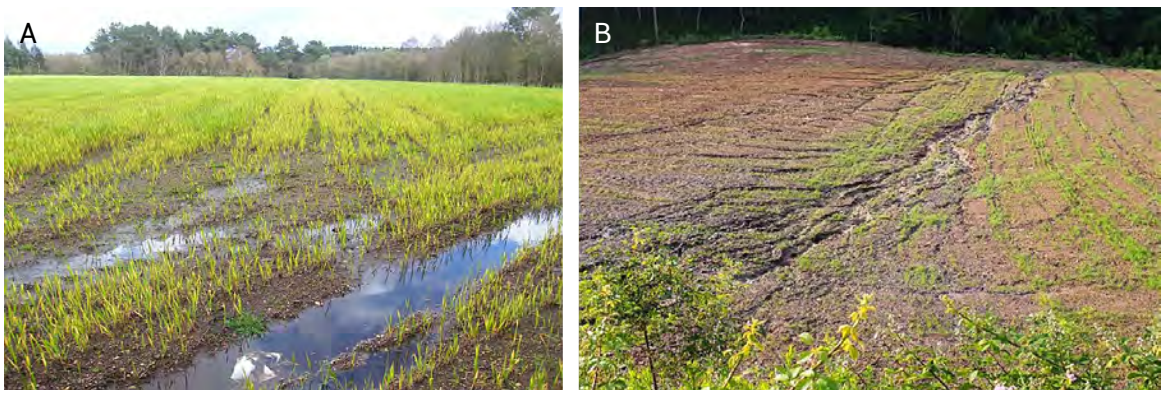


Figure 5. A) Cropland with low infiltration rate, because of low aggregate stability due to continuous cultivation; and B) A poor soil structure is quickly detected because of appearance of rills.

Figure 6. Ploughing and lack of plant cover usually lead to a sealed surface layer, which prevents water infiltration and seed germination.

Soil compaction due to mechanical operations and overgrazing: In addition to the reduction of macroporosity in the soil, due to the SOM loss, compaction occurs because of applying loads or pressure by farm machinery and overgrazing in pasture. Compacted soils are common in most agricultural and forest operations as a consequence of repeated passes of vehicles and land grading equipment. In such soils, bulk densities values of 1.5-1.8 g cm^{-3} are usually found, while the forest soil with a high SOM had soil density around 0.8 g cm^{-3}. Ideal bulk densities for growing agricultural crops range between 1.0-1.2 g cm^{-3}, while bulk densities higher than 1.5 g cm^{-3} are not recommended. Soils with the highest bulk density (e. g., 1.8-2.0 g cm^{-3}) are frequently found in industrial sites, playing grounds or popular areas, such as campsites.

Soil compaction by machinery usually causes substantial reduction in soil porosity, although wheel load, tire type, and inflation pressure play an important role in soil compaction. The extent of the soil compaction depends on the applied load, soil type, and moisture status (Strudley *et al*., 2008). The effect of the machinery is heavier in bare soil; therefore, removal of harvest residues increases considerably the effect of the machinery.

The most important compaction damage occurs when the soil is wet. The force required to compact a soil decreases exponentially with the increase in soil water content. Some plows, such as the moldboard plow, compact the soil in depth. The continued use of these tool promotes the development of a dense layer below the tilled layer, called plow plan (plow pan).

Intensively managed forest soils (clear cutting) are also subjected to compaction. This is favored by mechanical harvesting, using heavy tractors or specialized felling (harvester) and logging (skidder, forwarder) machines. Surface runoff and soil erosion can be important in steep slopes, especially when ruts are created during logging operations. In severe compacted soils, forest revegetation can be impeded for long time periods.

Continued grazing in high stocking rates and livestock trampling cause frequently soil compaction (Tracy and Zhang, 2008), decrease in standing vegetation or vegetative cover, depletion of soil organic carbon (SOC) and increases in erosion.

2.4. Soil nutrient balance and cycles

Nutrient supply constrains the soil productivity. However, in general productivity constrains are generally larger in cultivated lands than in non-cultivated ones. Particularly, nitrogen and phosphorus alone or combined limit plant growth in most of lands in the short-term (Elser *et al*., 2007). Nutrient availability to plants is controlled by soil chemical cycles.

Nutrient cycling in native ecosystems and, also, in agroecosystems, involves entry of nutrients, internal transfers between vegetation and soil microbes, and exchange with the soil matrix, and their loss. More than 90% of the nitrogen and phosphorus absorbed by plants of most native terrestrial ecosystems comes from the internal cycling of elements that return from vegetation to soils. The adequate management of the SOM is a key factor for reduce the need of external nutrient inputs in managed ecosystems as forest plantations and crops (Gallardo, 2016). Nutrients are lost from ecosystems by leaching, gas emission, wind and water erosion, fires, outflow, and in agroecosystems by the removal of materials along harvest products exported from the production units. Human activities tend to increase inputs and

outputs relative to the internal transfers from the regeneration of available nutrients from dead organic residues, generating different environmental problems. Most severe imbalances are generated by mono-cropping and use of excessive quantities of nutrients or, conversely, by addition of essential nutrients or amendments in lower amount than the required by crops.

Human disturbances such as forest conversion, harvest, and fires increase the proportion of the nutrient pool that is available and therefore vulnerable to loss. Some of these losses occur by leaching of dissolved elements to groundwater, and particulate (both in organic and inorganic particles) losses causing a depletion of soil fertility due to a slow reposition of nutrients exported from the site. Frequently an increase in soil acidity occurs by outputs of essential cations as potassium, calcium, and magnesium due to natural conditions or excessive addition on ammonium fertilizers.

However, chemical fertilizers and manure are key to maintain the agricultural intensification. Therefore, understanding the existing relation between plant demands and soil supply results critical to avoid imbalances. Generally, agriculture produces an excess of nutrients resulting from imbalances between nutrient inputs (fertilizers, leguminous crops which fix nitrogen in the soil, and atmospheric deposition), harvest removal and environmental losses. This excess has led to widespread nutrient pollution and the degradation of lakes, rivers and coastal oceans. In addition, the release of nitrous oxide from fertilized fields contributes to climate change. Finally, in many cases lands are abandoned due to the large losses of SOM and nutrients by soil erosion caused by overgrazing or bad agricultural practices (see Section 2.5. *Erosion in managed soils*).

Targeted policy and management are needed to improve the balance between yields and the environment. Such actions must include the coupled between plant requirements and soil inputs, improving manure management, and capturing excess nutrients through residues recycling, wetland restoration and other practices. Increasing SOM is an important issue.

There are two different ways to recover the SOM and nutrients after land use (Roa-Fuentes *et al*., 2015) to nearly the former levels (see Section *3.2.2. Managing abandoned landscapes*). In brief, one way is allowing the regeneration of native vegetation without human intervention by ending disturbance (***passive restoration***). Although this strategy is cheaper, results are limited either by the slow rate of SOM and nutrient accumulations. The second way is using tree plantings (***active restoration***). Studies of plantings have shown that active restoration could be the best way to speed up the recovery of vegetative covers and soil functions after land-use abandonment (Roa-Fuentes *et al*., 2015). During restoration practices, it is important to consider that plant species could affect the soil fertility in different degrees and timing, altering the rate of nutrient cycling and energy fluxes (see Table 3, in Section 3.2.2. *Managing abandoned landscapes*).

2.5. Soil salinization and alkalization

The salinization occurred when water-solute salts are accumulated at the topsoil (epipedon) and this salt accumulation can be higher to the plant tolerance (above of 3.000 mg salt kg^{-1}). Additionally, to the direct toxic effect over the plant species, the salts also decrease the osmotic pressure, reducing the water uptake by plants. Therefore, decreasing the soil water availability.

Salinization can be caused by natural factors associated with weathering of parent material rich in different types of salts (as NaCl, $CaSO_4$, $CaCO_3$, *etc.*) in sites where the evapotranspiration is higher than precipitation, as Mediterranean and semi-arid areas. In many Mediterranean regions, water is scarce and soil salinity is a major cause of desertification, mainly due to human activities (Daliakopoulos *et al*., 2016). Induced salinization affects large surfaces of Spain, Italy, Hungary, Greece, and other countries from Southern Europe.

Although the management can promote the salinization by watering or decreasing water table by pumping. Irrigation in dry land implies two types of problems. In dry regions, when the water is pumped in excess of recharge, it competes with uses for human and natural ecosystems, affecting the availability for urban use and streams for fisheries and recreational uses. Besides, in dry regions with salt content in the subsoil, using underground water source typically increases the soil salinity. Since drought period is increasing under the climate change, increased irrigation has occurred in many areas. In addition, flows from agricultural irrigation usually carry more salts, nutrients, and pesticides, impacting natural systems and drinking water. Unfortunately, in many salinized soils additional irrigation is needed to wash away the salts from the soil rooting layer, diminishing further the water resources and increasing the desertification (Figure 7).

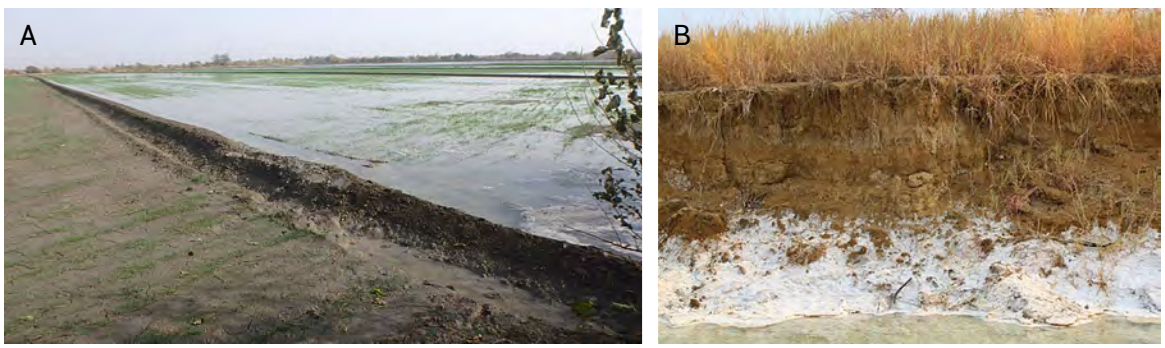


Figure 7. A) When salinization occurs, additional irrigation is used to wash up the salts beyond the soil rooting layer. Cotton crop in Bukhara, Uzbekistan; and B) Sometimes salts are too close to the soil surface, as in this picture (Oaxaca, Mexico).

The impacts of salinization on soil physic-chemical characteristics: The increment of sodium generates the dispersion of soil particles and, therefore, the soil structure, as soil aggregates, is disrupted, promoting the susceptibility of soil to both water and wind erosion. Additionally, the salinization increases soil pH above 8.0, promoting chemical precipitation of available soil nutrients (as HPO_4^{2-} or $H_2PO_4^-$) and, therefore, reducing soil fertility.

The salinity also decreases the capacity to exchange cations by clay, because the exchange charges are partially occupied by free cations as sodium. Furthermore, the ammonium is vulnerable to volatilization because of the high pH, increasing the N losses from the soil. As a conclusion, salinization disturbed the soil pH buffer capacity, disrupting soil nutrient dynamics.

Effects on soil microorganisms: Salinity also has adverse effect on soil microorganisms, because only some microbial *taxa* can tolerate high levels of soil pH. Therefore, soil biodiversity can decrease, due to a reduction of genes diversity; then, some functional genes can be lost, affecting soil fertility.

Effects on vegetation: The accumulation of salts affect negatively to plant growth due to the increase of the osmotic potential which decreases the availability of water to plants. Among the most important crops in Europe, wheat, barley, triticale, and rye are salt tolerant, whereas maize is sensitive.

The soil electrical conductivity (EC) has been used as an indicator of salinity. Values lower to 2,0 mS m^{-1} is considered the soil has no salinization problem, while values higher to 8,0 mS m^{-1} are considered the soil with high problems of soil salinity (Tanji, 1990; Urbano-Terrón, 1995). In Europe, around of $3.8x10^6$ ha are considered with high levels of soil salinization (EEA, 1995), mainly in the Mediterranean countries. Unfortunately, soil salinization is increasing in Spain, Greece, and Hungary (De Paz *et al*., 2004).

2.6. Erosion in managed soils

2.6.1. Extent of the erosion

Accelerated soil erosion is a major threat to soil. More than 40 $x10^9$ tons of soil is lost annually from terrestrial ecosystems by erosion (FAO, 2017; Borelli *et al*., 2017; FAO & ITPS, 2107).

In Europe, almost 90 % of this type of soil loss is the result of anthropogenic impacts. Erosion mainly takes place in agricultural lands and abandoned lands. Furthermore, erosion rates are also high in soils affected by severe wildfire, intensively managed forest plantations, and construction sites (Boardman & Poesen, 2006; Verheijen *et al*., 2009).

The arid and semiarid Mediterranean area is considered for a long time to be particularly prone to erosion because it is subject to long dry periods (which hinders plant development), followed by heavy bursts of intensive rainfall events. The precipitation mainly affects steeply sloping land, *i.e.* soil that is poor in SOM contents and scarce plant cover. For this reason, soil losses of 20-40 ton ha^{-1} are frequently recorded after heavy rainfalls in such areas (García-Ruiz, 2010; García-Ruiz & Lana-Renault, 2011).

Croplands: Arable soils, which make up as 25 % of European land as average (*https://www.indexmundi.com*) accounts for 70 % of soil erosion in Europe.

Land devoted to annual crops, such as cereals, and orchards (olives, vineyards, and almonds), is particularly vulnerable to soil erosion (Figure 8) because of the lower degree of protection offered by this type of vegetation cover. In this land uses the soil is left unprotected for a large part of the year, in annual crops coinciding with the season of heaviest rainfall in Mediterranean areas. In addition, intensive tillage reduces the amount of soil organic matter content, leading to lower soil porosity and higher runoff. Due to the relief in Mediterranean areas, cultivation is often carried out on sloping lands, involving the construction of new terraced fields.

Erosion of these areas also leads to high concentrations of suspended sediments and dissolved salts (chloride, sodium, sulphate, calcium, and magnesium) in waters. Some studies (e. g., Smith *et al*., 2011) have shown that nutrient losses may represent a significant

proportion (10-60 %) of the total amounts of nitrogen, phosphorus, and potassium applied annually. Economic losses due to nutrient loss and damage to infrastructures by soil erosion and landslides may be as much as 14 % of the annual income.

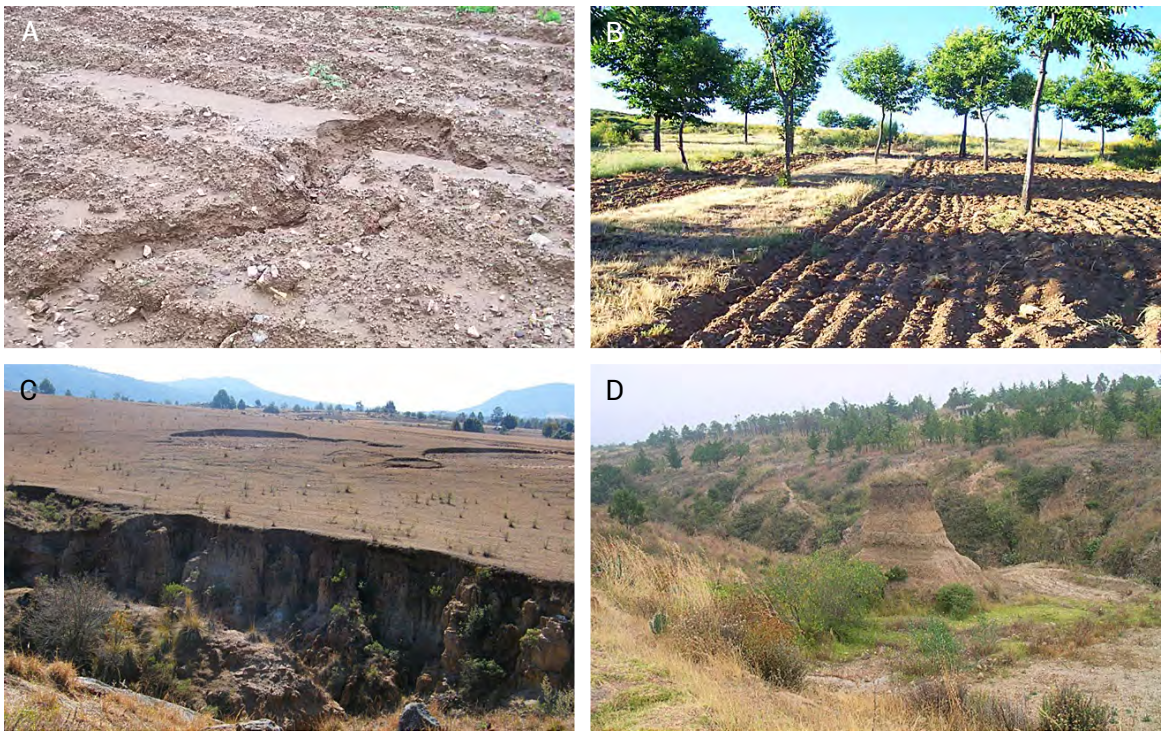


Figure 8. A) Rill formation in intensively managed arable soil, in NW Spain, B) tillage in chestnut orchard, Portugal; C) Gully formation as a consequence of livestock overgrazing (Michoacán, Mexico); and C) degraded in tepetates areas (Mexico).

Pastures: On the other hand, a total of 17 % Europe's land is covered by permanent pastures and mixed mosaics (Soukop *et al*., 2016). In semi-arid areas, rangelands are prone to soil erosion even under natural conditions. One of the most frequent problems is overgrazing, which leads to deterioration of the plant cover. In poorly managed livestock grazing, grass species are replaced by scattered shrubs. Cattle trails and ruts produced by off-road vehicles also channel run-off and favour the formation of gullies.

Land abandonment. Since the beginning of the 20th Century populations have declined in many marginal rural and mountainous areas, leading to the abandonment of thousands of hectares of land (Figure 9a).

In some areas where sufficient amounts of precipitation favours plant re-colonization of abandoned cropland, cover by shrub species improves the SOM contents and reduces runoff and erosion. However, shrubland areas are often subject to repeated fires, and thus to soil erosion processes. In some cases, the abandonment of bench terraces may also have important geomorphological and hydrological consequences. Abandonment of these stone structures (Figure 9b) usually leads to collapse of the walls and gully erosion, and extreme erosion rates are usually recorded (García-Ruiz & Lana-Renault, 2011).

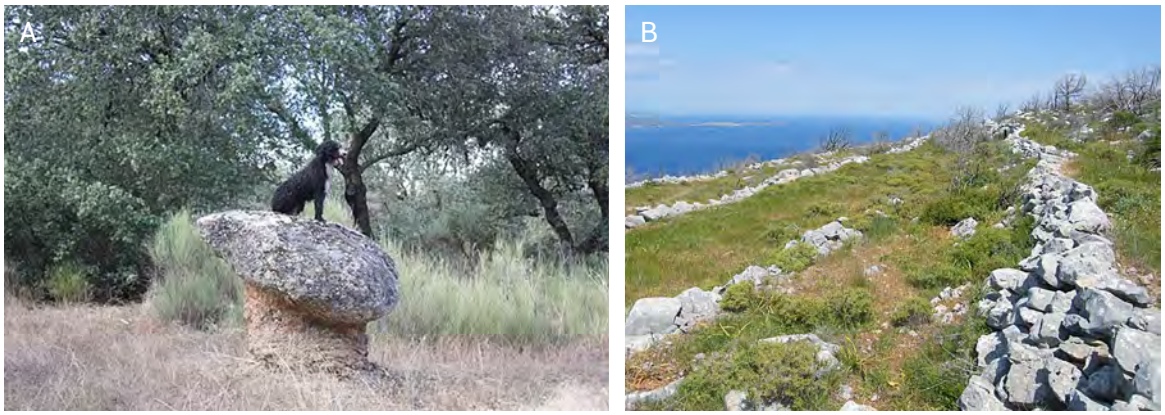


Figure 9. A) Land abandonment in Extremadura (Western Spain); and B) In semiarid region, abandoned beach terraces are occupied by shrub communities which are subjected to repeated fires, and thus to soil degradation and erosion processes (Hydra, Greece).

Forested lands, including forest plantations. A total of 34 % of Europe surface is covered by forests (EEA, 2017). In stable forests erosion soil losses are very low, generally less than 1,00 ton ha^{-1} yr^{-1}; in these ecosystems the litter layer covers the soil, providing very effective protection again the impact of raindrops and favouring infiltration. In addition, tree leaves and branches intercept the rain and moderate its effects.

However, these protective mechanisms disappear in intensively managed forest plantations, which are often are cleared for production. Harvesting activities (removal of felled logs by use of wheeled of tracked forwarders or skidders) and intensive site preparation (e. g., ploughing, windrowing) reduce surface cover and compact the soil, increasing erosion, at least temporally (Edeso *et al*., 1999).

High rates of soil erosion occur as a result of poorly designed forest roads (Figure 10). These roads collect and channel large volumes of water, leading to severe gullying. Poorly designed logging roads may cause the loss of as much as 100 Mg ha^{-1} yr^{-1} of soil by erosion.

Figure 10. Poorly designed forest roads generating large amounts of run-off (Bizkaia and Lugo, respectively, Spain).

On the other hand, afforestation schemes with flammable plantations and the encroachment of shrubs after rural depopulation have increased the frequency of wildfire, increasing erosion, which in certain cases can be extreme (see **Chapter 4**, focused on wildfire).

Construction sites. Sediments are often produced in construction sites (Pitt *et al*., 2007). Although the surface area affected is much less than in agricultural and forest land, rural buildings accounts for a substantial sediment load production (in some countries 10 % of the national sediment load). Measurements made in these drastically perturbed areas usually show erosion rates that are 3 -100 times higher than in cropland (Figure 11).

One of the most important factors affecting the accelerated erosion rates in these areas is the high erodibility of soil that is freshly disturbed by excavation. This is due to the extremely low SOM content of the subsoil deposited in the surface. In addition, the complete lack of plant protection makes these soils highly prone to erosion. The exposed material is also highly susceptible to the formation of large gullies, which can ruin pavements and foundations.

Figure 11. The low soil organic matter content of the freshly exposed soil material leads to extreme erosion rates in construction sites (Lugo, Spain).

3. Implementing agricultural and forest sustainable practices in agricultural and forest lands

It is long accepted that extensive land degradation in Mediterranean countries contributes negatively to the development, especially of the rural regions (see chapter "Land rehabilitation and rural development"). The processes of land degradation above described have implications in waters (sediment loads, contamination, flooding), air (increased greenhouse gas emissions), and food provisions.

High nature value farmlands of low intensity agriculture systems has been proposed for the European Union policy, since they are reservoirs of biodiversity and provide several ecosystem services (Gardi *et al*., 2016; see Section *High nature value farmland*). However, global estimations suggest that production would need to roughly double to keep pace with projected demands from global population growth, dietary changes (especially meat and milk consumption), and increasing bioenergy use (Tillman *et al*., 2011).

3.1. Conservation agriculture and erosion control

Proper management practices and land use policies are necessary to reduce the degradation processes described above. In recent years, many farmers are adopting **conservation practices** which combine the following basic principles of land management defined by the FAO (Figure 12):

a) Reduction of soil disturbance by using reduced-tillage or no-tillage,

b) Retention of adequate levels of living or dead cover crops, and

c) Diversified crop rotations.

These practices have been extensively assessed in a variety of soils and climates across the world, largely under Mediterranean climate. Different studies identify a large variety of benefits related to soil protection, the environment and the human health.

Conservation agriculture is focussed on the increase in the SOM contents and in the enhancement of other soil functions. These practices tend to increase water infiltration and storage in the soil, particularly required in semiarid and dryland agricultural systems of the Mediterranean region, where the SOM contents are particularly low.

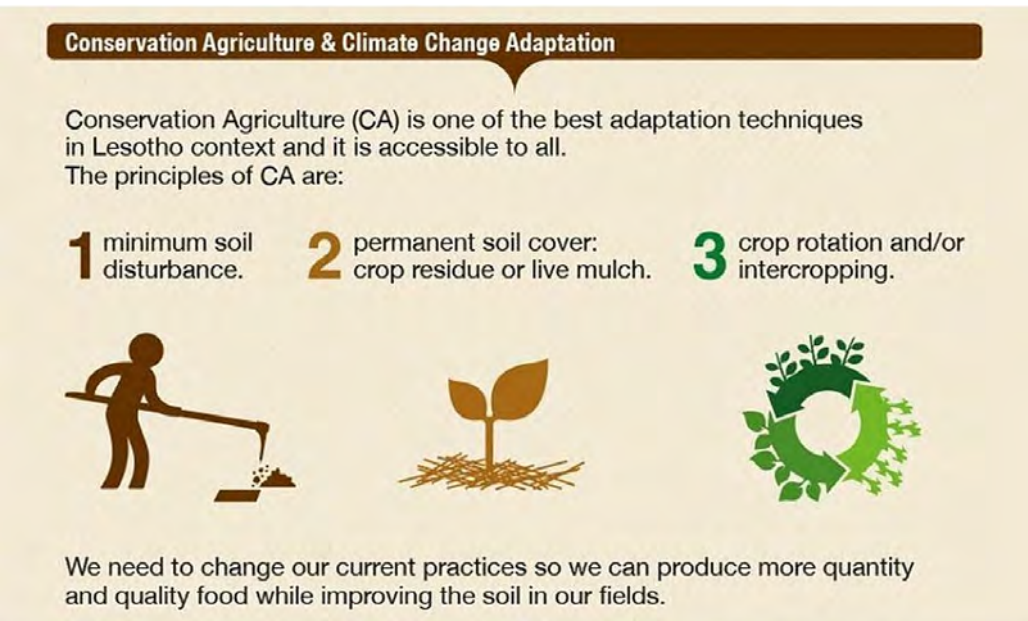


Figure 12. Principles of Agriculture of Conservation, according FAO.

Conservation practices are also implemented to reduce nitrate leaching, decreased the use of herbicides, and mineral fertilizers, and suppress weeds via cover crops competition for nutrient and water resources. Other potential benefits are the increased microbial diversity and to lower emissions of greenhouse gases, such as N_2O to the atmosphere.

In recent years, conservation tillage has become increasingly popular, due to encouragement via policies such as the Common Agricultural Policy in Europe (González-Sánchez *et al*., 2016). The organic farming (EU Regulation for Organic Food and Farming EC, Nº. 834/2007) follows the principles of organic agriculture.

The EU thematic Strategy for Soil Protection recognizes that SOM is of crucial importance to maintain soil quality and productivity and to maintain a range of ecosystem services. This policy supports are important to compensate the initial costs of implementation for farmers, when benefits of the practice may be insufficient for the first years.

3.1.1. Increase nutrient-and water-use efficiencies

Different studies have shown that only 30-50 % of the nitrogen applied to the soil is taken by crops. In the case of phosphorus, the efficiency when applied directly to the soil is near 15 %. In both cases efficiency can be incremented by incorporation of fertirrigation.

The excess of nutrients applied causes environmental problems in freshwater, coast and atmosphere, particularly in the case of N, a mobile element in the soil (see section 2.3. *Soil nutrient balance and cycles*).

Research, extension education, and soil testing have contributed to improve the nutrient use efficiency in different intensive crops. Different strategies are;

1) **Precision agriculture,** which match the application of the fertilizers with the needs of the plant and the efficiency of the fertilizer under soil type and management conditions. N fertilizers are added in small doses at sowing and more frequent during the periods of higher demand of the crop, that is not the case of P and K application, where the total amount of fertilizer is applied at sowing time;

2) **Multiple cropping systems**, in which crop rotation and intercropping (at least two crops) growing simultaneously;

3) **Fertirrigation**. i.e., fertilization via irrigation system, and

4) **Use of environmentally friendly fertilizers**, a type of fertilizers that retard and control the release of nutrients into the soil and improve the coupling between the plant demand and the soil nutrient availability. In these sorts of fertilizers, the nutrients are coated with materials, such as sodium alginate, cellulose or others, that can be degraded in the soil.

Furthermore, agroecological innovations in crop and soil management appear as a great promise for improving the resource efficiency of agriculture, maintaining the benefits of intensive agriculture and reduced harm to the environment (Chen *et al*., 2011).

Efficiency of water use

Of the particular concern is that some 70 % of global freshwater withdrawals are devoted to irrigation. Irrigation is used on about 24 % of croplands and is responsible for delivering 34% of agricultural production (Siebert & Döll, 2010).

In large portions of Mediterranean regions, where water is scarce, good water, and land management practices can increase irrigation efficiency. For example, curtailing off-field evaporative losses from water storage and transport and reducing field losses through mulching and reduced tillage will increase the value of irrigation water.

Other approaches to increase the crop yields in water-limited environments are the cultivation of crops with high water-use efficiency and the development of crops with greater drought tolerance. Biotechnology or conventional breeding are being used to achieve these goals.

3.1.2. Conservation tillage

Conservation tillage is any management that retains at least 30 % of residues on the surface and enables reduction in the intensity of tillage. They include a range of practices as follows:

1) Reduced-till: Soil undisturbed prior to direct seeding on the stubble of previous crops, which is done in a narrow seedbed, 2.5-7.5 cm wide. Weed control primarily by herbicides.
2) Ridge till: Soil undisturbed prior to planting, which is done on ridges 10 to 15 cm higher than row middles. Residues moved aside or incorporated in about one-third of soil surface. Herbicides and cultivation used to control weeds.
3) Strip till: Soil undisturbed prior to planting. Narrow and shallow tillage in row using rotary tiller, in-row chisel, and so on. Up to one-third of the soil surface is tilled at planting time. Herbicides and cultivation to control weeds.
4) Mulch till: Soil surface disturbed by tillage prior to planting, but at least 30 % of residues left on or near soil surface. Tools such as chisels, field cultivation, disks, and sweeps are used (e. g., stubble mulch). Herbicides and cultivation used to control weeds.

Farmers often use herbicides to kill weeds rather than applying conventional inversion tillage (e. g., with a mouldboard plough). The use of a chisel plough is also preferred because it stirs the soil but leaves a large proportion of the crop residues on the soil surface.

The use of conservation tillage is one of most efficient measures to enhance SOC stocks (Figure 13), increasing the stability of the aggregations of some soils and in turn it might improve the infiltration. In many cases these systems produce the same or even higher crop yields, while saving time and soil. These practices are especially advantageous in light to medium texture and with low SOM contents (Busari et al., 2015). However, the net effect of this technique depends on the type of soil (Strudley *et al*., 2008).

Figure 13 shows the effect of carbon concentration after two decades of no tillage as compare to conventional tillage.

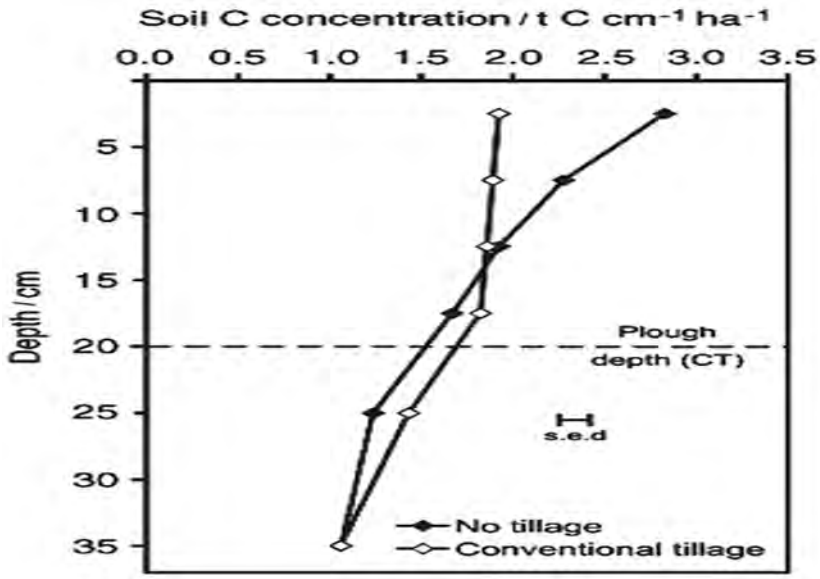


Figure 13. Trends in carbon concentration with depth after 21 years of no tillage or conventional tillage in southern Brazil (Machado et al., 2003).

Tillage to reduce or alleviate soil compaction

There are some recommendations to reduce the impact of machinery on soil compaction. Using lighter machinery is the most logical measure. However, in some cases, the use of large and heavy vehicles and big axle load is unavoidable. Other technical measures to reduce soil compaction are to limit the contact pressure of vehicles with the ground. Using wider tires or lower tire pressure are the most important factors to reduce compaction under wheels. Dual wheels and flotation tires can also minimize significantly the compaction risk.

Other useful measures are to delay the activities requiring machinery to periods when soils are dryer or froze, because they are less prone to compaction when become drier later.

A compact soil can be loosened, although the strategy will depend if the compaction is on the soil surface or below the topsoil. When it occurs only at the surface, cross-tillage usually is an effective method to correct the problem. However, if deeper soil layers are affected, compaction is less reversed, and deep loosening must be practiced. However, the effect and length of the effect of this operation can be variable, depending of the type of soil and other factors. Severe compaction in deep layers frequently requires multiple passes of loosening, which usually results costly.

The employment of rippers tends to loosen the soil in depth, although in some cases the effects are not lasting (Figure 14). In areas with low precipitation and with not irrigation, ripening for increasing the soil depth does represent advantages, since the rain does not penetrate more than few centimeters in the soil.

Figure 14. a) Ripper is usually employed in afforestation of croplands to loosen the soil in depth; and b) In the direction of the furrows is essential for avoiding soil erosion (Andean Andisols, Ecuador).

3.1.3. Mulching, cover crops and organic inputs

Mulching

Mulching is the agronomic practice of leaving mulch on the soil surface to avoid soil erosion. Mulch is referred to the material spread over the soil surface to generate a permanent or semi-permanent protective cover. Some examples of materials that can be used as mulch are vegetative residues, biological textiles or even gravel. In agricultural soils, the most frequent materials are the most readily available, such straw, grass, and chopped pruning or crop residues (maize, soybean, *etc.*). In some cases, plastic mulching is used, although this practice leaves undesirable residues.

Surface cover by crop residues plays a critical role on soil water processes. This practice reduces evaporation, increases infiltration and, therefore, reduces erosion.

Soil erosion is also reduced with increasing the amount of crop residues left on the soil surface. Establishing a dense plant cover also favors the structure activity. The soil aggregates of soil with high SOM content are much more stable. Since residue cover dissipates the energy of the raindrop, crusting is minimized (Figure 15a). However, production of more biomass requires higher addition of nitrogen to soil, a problem that increases the disruption of the global nitrogen cycle.

The quantity of residues needed to achieve maximum effects in these processes ranges 2.0-8.0 ton dry matter ha^{-1} (Ranaivoson *et al.*, 2017). These amounts depend on the type of soil constraint (in general, evaporation requires higher amounts than infiltration), but also on other factors, such as the rainfall intensity. In addition, it is important to consider the different efficiency of the types of residues. For wheat residues only 2.0 ton DM ha^{-1} if is required to reduce soil erosion by about 90 % compared to bare soil; whereas 1.5-4.5 ton DM ha^{-1} of maize, rye or rice residues are required to reduce soil loss by 50 %

Since residue decomposition releases nutrients to the soil, keeping the residues on the soil surface also contributes to the nutrient availability, at least at long term. This practice

increases soil exchangeable K. However, in the short-term (<1 year) microbial immobilization of N and P can take place when high C content residues, such as cereal straw, are used which reduces the availability of these elements in the short term.

Surface crop residues are known to maintain or increase the SOM content. The effect, however, is rather variable depending on the initial SOM content and the soil texture (usually greater in clay soils). This practice influences positively on soil meso- and macrofauna abundance (nematodes, arthropods, earthworms), improving soil biodiversity.

Another important effect is weed emergences, which is usually decreased with the quantity of residues of the soil surface. Crop residues can interfere physically impeding their emergence or through allelopathic effect. This can be very useful in Mediterranean tree-crops (e. g., almond trees, olives, vineyards, chestnuts), where traditional managements involve frequent tillages to avoid the competition between weeds and crops for water resources; a properly managed, herbage cover can reduce erosion, maintaining the crop yield (Martins *et al*., 2010).

The subsequent incorporation of organic residues into the soil prevents its compaction by the action of heavy machinery and raindrop impact. Leaving the harvest residues on the ground usually reduce the pressure of the equipment per unit contact area. In intensively managed forest soils a thick layer of organic residues has been shown to cushion the compressing action of the heavy machinery. This measure is especially useful on wet soils, which are prone to compaction.

However, in Mediterranean areas the current needs for crop and animal production have led to the intensification of soil management and overgrazing, which hinder the presence of an adequate soil vegetative cover.

In addition, in recent years another problem has been the high demand of crop residues as feedstock for energy (Figure 15b). The use of crop residues for bioenergy purposes needs to be considered with caution, especially in semidry and dryland areas (Lal, 2009); therefore, in Mediterranean areas is important to develop integrated strategies to a better stewardship of the agroecosystems.

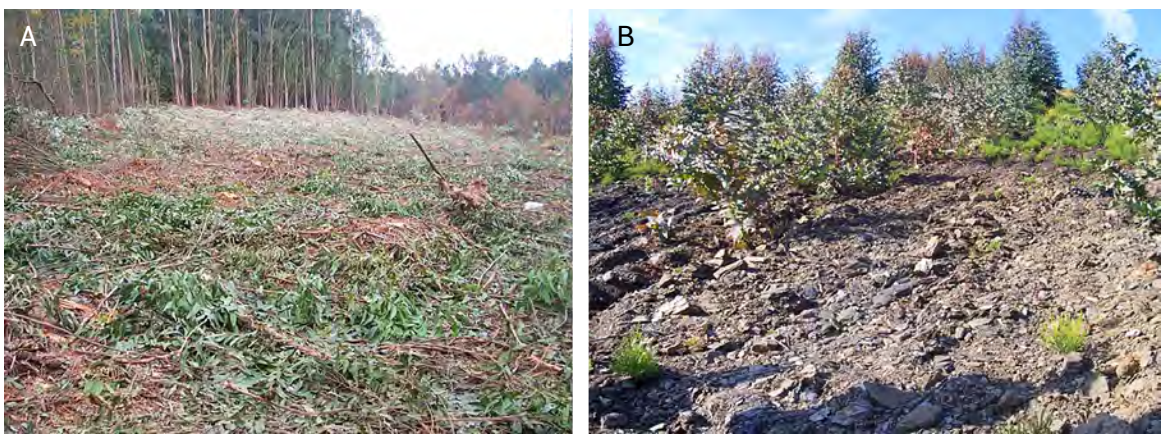


Figure 15. Forest logging residues management in intensively managed plantations. A) The logging residues are kept in the soil, preventing soil erosion and compaction by machinery; and B) The removal of branches as fuel for bioenergy in no appropriate site leads to problems of erosion.

Cover crops

Introducing a growing cover crops in some Mediterranean harvests usually produce similar benefits than mulching with respect to runoff erosion, organic matter, structure and biodiversity. In some occasions, these crops are not harvested for food. When the main crop is annual, cover crops are typically sown after the summer crops have been harvested and before cold weather. In spring, before the growing period of the main crop, they are tilled back into the soil.

In permanent crops (e. g., almond, olives, vineyards, chestnuts, *etc*.; Figure 16) they are sown between trees and vines in vineyards, and they are incorporated into the soil at the end of their growing cycle where the nutrients are released as the plants decompose.

All cover crops add organic residues to the soil and accumulate essential nutrients. In many cases, legumes, such as red clover, are established to supply the following year's nitrogen needs. Plant cover crops bring other benefits. This type of crop maintains vegetative cover once the crops have been harvested for the year, which reduce the risk of erosion. When they are planted in a dense cover crop, they prevent weeds from getting established, so there is less weeding to do come spring.

Figure 16. A) Plant cover reduces the impact of rain drops, increasing soil organic matter content and prevents soil crusting; and B) Typical conversion of chestnut orchards to chestnut coppices, stand litter and trees protecting the underlying soil.

Organic inputs

The inputs of different types of organic amendments, such as compost, manure or raw wastes, increase or maintain SOM contents with subsequent multiple benefits for soil, plants, and soil organisms.

The regular application of the organic residues to the soil helps to progressively enhance the SOM content and provides nutrients for plants and microbial communities; it also enhances the soil cation exchange capacity due to the negative charge created as a result of the decomposition (humification) of the organic residues and promotes the soil aggregate stability and porosity. The options most easily available to the farmers are adding crop residues and farm manures. In addition, some off-farm sources, such as treated sewage sludge, biochar or agro-industry wastes can be added.

There are multiple evidences that long-term application of inorganic fertilizers cannot maintain, in many cases, the optimum SOM content and crop productivity; but, also, the sole use of organic-residue amendments cannot maintain the desired agronomic productivity, due to the intrinsic low nutrient status of some soil types used for agricultural purposes. This is the case of soils that have been highly eroded or have a low pH (which usually require liming adjustment).

Addition of crop residues.- Incorporation of crop residues into the soil using appropriate machinery or retaining them on the soil surface enhances SOM contents of agricultural soils. Green manure, or cover crop, involves planting a crop that will incorporated into the soil to increase its SOM and fertility.

Animal manure (farm-yard manure and slurry manure).- Manure is a natural by-product of livestock production, containing organic carbon and it is an excellent source of nutrients for crops. Manure application is profitable on farms with a near crop land which can fully utilize all the applied nutrients. In European Union, 1.40×10^9 ton of manure is annually generated. In most cases, manure is stored and then spread out on agricultural lands. However, in occasions, manure is applied in large volumes, which generates leaching and runoff rich in nitrate, phosphorus, ammonia, and greenhouse gases emissions, leading to water and air pollution. The accumulation of Zn and Cu is also a risk for soil micro-organisms and plants (Santoja et al., 2017).

In many regions with intensive livestock manure is produced in quantities exceeding the local land capacity to receive it. Therefore, some strategies must be implemented in order to meet the environmental protection criteria. Part of the N in manure can be reduced by lowering the crude protein in the diet of the animals (poultry, pig, and dairy cattle) or by mean specific techniques applied to housing and manure storage.

Manure treatment is an alternative to the traditional direct spreading which can prevent part of these problems. The main processes for manure treatment are composting, aerobic treatment, and anaerobic digestion (Loyon *et al*., 2016).

Compost.- Application of organic composts in agricultural lands help to maintain the SOM contents, ameliorating soil physic-chemical properties, and improving crop production. Nevertheless, this practice usually fails to improve N status in the crop, and synthetic fertilizer N is recommended for better results (Gallardo, 2016).

Agricultural utilization of municipal (domestic and industrial sectors) solid wastes is a cost-effective option for managing solid wastes and for improving soil quality in croplands. Municipal waste compost is also helpful to restore ecological functions of degraded lands. It has been used for the restoration of burn areas, soils contaminated with hydrocarbons, and organic pollutants and for the remediation of saline soils. In degraded soils, compost amendment ameliorates its biological properties and enzyme activities (Srivastava *et al*., 2016).

The quality of the compost required is very variable, which depends on the type of feedstock and composting process. In spite of these advantages, the presence pathogens, pesticides, and toxic substances, like heavy metals and other organic pollutants, must be considered. The long-term application of low-quality compost may lead to their accumulation in the soil and crops. For this reason, checking the composition of the feedstocks and the resulting compost before land application is very important.

Treated sewage sludge.- Sewage sludge, a sub-product generated in waste water plants, contains useful concentrations organic residues, nitrogen, and phosphorus, and it widely employed as fertilizer in many crops. However, it also usually contains high amounts of many pollutants and pathogenic microorganisms in big industrial cities (in rural villages the contents are usually traces), which can be phytotoxic and some toxic to humans and/or animals. For this reason, it is necessary to control the concentrations in the soil of potentially toxic elements and their rate of application to the soil. Prior its application to the soil, sewage sludge should be subjected to biological, chemical or thermal treatment, long-term storage or other appropriate process designed to reduce its fermentability and health hazards resulting from its use before being applied in agriculture (Loyon *et al*. 2017).

Biochar.- In the last years use of biochar has been proposed as an option for improving agricultural degraded soils. Biochar is the charcoal produced of pyrolysis of residual biomass and it is a sub-product of some industries. Compared to fresh plant residues, compost and, especially biochar, are more slowly decomposed, which can increase soil carbon content and soil fertility in the long term (Lehmann *et al*., 2016). Another benefit of this practice is the reduction of N_2O emissions from the soil. However, they can also reduce the N efficiency of fertilization due to increased microorganism immobilization. An alternative product to biochar is the wood ash generated in certain biomass power plants, which usually contains 10-15 % of charred biomass (Merino *et al*., 2017). Both products, biochar and wood ash can be incorporated into the composting process (Agegnehu *et al*., 2017). Both products are strong adsorbent, therefore, biochar and relative sub-products have the potentially to sequester some heavy metal when applied to the soil.

3.1.4. Use of crop rotation and agronomic measures

Crop rotation.- In this practice a series of dissimilar types of crops are grown in the same area in sequential seasons. Growth of legume as a green manure in sequence with cereals and other crops allows the replenishment of soil organic matter and nitrogen. Deep rooting can also contribute to enhance the SOC stock in soil depth.

Plant diversity promoted by crop rotation is also important to favour the diversification of the root systems, leading to a greater variability of root-derived SOM compounds. Therefore, an increase in soil residues also increases the variety of soil fauna, bacteria and fungi, improving the functional diversity. Thus, a complex microbial community has the capacity of reducing the impacts of pathogens. The earthworm abundance usually results is larger and interconnected pores, increasing water infiltration. Nutrient acquisition can also be increased by arbuscular mycorrhizal fungi.

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

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

3.1.5. Special techniques for erosion control in agricultural and forest lands

Mulching also is used in reforestation plans of water-limited environments due to its effect in reducing soil erosion and water loss by evaporation. In addition, the mulch is a source of available nutrients, favouring the seedling survival and growth (Barajas-Guzmán *et al*., 2006), enhancing the soil conservation. Other techniques for erosion control are showed below.

Terraces

The construction of terraces (Figure 17) reduces the effective length and gradient of a slope. These are used on steep slopes and when properly designed allow large volumes of water to move from the soil without erosion (Stanchi *et al*., 2012).

Figure 17. Some examples of flat-channel terraces in Mediterranean environments.

Control of gully erosion.- Gullies are common in steep slopes partially covered by vegetation. They can also occur in along the course of concentrated water flows formed in arable land, as well as in poorly designed terraces and roads. In desert climates, the appearance of gullies can have devastating effects. In small gullies, grass species can be sown to create grassed waterways. In more active gullies, a series of check dams can be constructed with materials available on site (Figure 18), such as rock or logs or brush.

Figure 18. Constructions of a series of dams and other actions for preventing the gully increasing in size.

3.2. Low-intensity farming systems and ecological restoration

The most productive lands are subjected to agricultural intensification, which leads to the subsequent degradation process described above, along with loss of the biodiversity.

But, as it was explained in the Introduction section, large areas of croplands managed in traditional uses in Europe are being transformed to other uses as a consequence of the rural exodus to the cities. However, these less productive lands are being transformed to low-intensity farming systems. Some of these lands have been afforested or transformed to rangelands and grasslands. But in most marginal areas, mainly mountain landscapes, hey undergo a revegetation process as a result of abandonment. This section is focussed in the conversion of croplands to these less intensive systems that provide a richer variety of environmental services.

3.2.1. Conversion of cropland to lower intensive managements

Converting cropland to grasslands or forest (afforestation) has significant and positive effect on SOM content and the recovery of soil functions and the variety of ecosystem services. Although the most prominent effect is produced in the topsoil layer, in the long term the deeper soil horizon will also gain as a consequence of bioturbation and organic residues inputs from the fine root litter (Pérez-Cruzado *et al*. 2011; Poeplau & Don, 2013; Muñoz-Rojas *et al*., 2015). In most of the SOM gains are attributable to the labile fraction, which means that these gains are susceptible of losses as a consequence of subsequent management (Poeplau & Don, 2013). The increased grassland and forest surface in different regions of Europe has also reduced the sediment production. In mountainous areas, this has contributed to mitigate gully erosion and landslides. In riparian areas, the afforestation also favours the stabilization of eroding river banks.

Agroforestry.- Agroforestry is a practice that integrate tree or shrubs (i. e., woody vegetation) in a matrix of land use for crop and/or livestock systems combining production of food with other environmental services (Mosquera-Losada *et al*., 2009). In the Mediterranean region, some examples are intercropping chestnut and walnut trees in arable systems or integrating cover crops and/or grazed legumes (i. e., an important source of protein) in olive plantations or vineyards. In comparison with intensive agriculture, commonly monocultures, agroforestry

enhances the local biodiversity, and improve physical and chemical soil properties (e. g., porosity and SOM and nutrient contents; Torralba *et al*., 2016). Since agroforestry systems retain higher amount of carbon in above- and below-ground biomass, and in soil, the transformation of low or unproductive croplands to agroforestry has been proposed as a measure for carbon sequestration and regional carbon management (Nair *et al*., 2009).

3.2.2. Managing abandoned landscapes

The current abandoned landscapes in Southern Europe are being evolving to two different types of landscapes. In some cases, these lands are subjected to a secondary succession or plant colonisation, by native and invasive species. The naturalization of these areas towards ecosystems prior to agriculture, generates a more natural landscape with a lot of environmental benefits. It reduces soil erosion, promotes the C sequestration in both soils and biomass (and, then, to climate regulation), and allows a better regulation of the water cycle; in addition, it help to recover the biodiversity (Pereira & Navarro, 2015).

Table 3. *Examples of plant effects on soils.*

Processes / soil properties	Effects
Atmospheric deposition	Increase the deposition of hydrogen, nitrogen and sulfur in polluted landscapes by canopy trapping of atmospheric particles and gasses.
Carbon input to soil	Increase the quantity of the organic residues added to soil by leaves, branches, stems, and root death.
Nutrient availability and accrual	Increase the nutrient availability to the soil by symbiotic nitrogen fixation, mineral weathering, and stimulation of microbial activity. Increase the nutrient accumulation in the soil by addition of organic residues rich in N, P, and other macronutrients, and by favoring the physical and biochemical protection of soil organic matter in the mineral soil.
Moisture regimen	Affect the soil moisture regimen by its effect on soil physical characteristics (soil temperature, size, and distribution of pore space), and on evapotranspiration.
Soil biodiversity	Increase the diversity of functional groups and species of soil microorganisms by the addition of large quantities of organic residues of different qualities.

Two practices of restoration ecology are common in these situations, passive and active restoration (Roa-Fuentes *et al*., 2015).

Passive restoration, usually involves natural regeneration with little or without human intervention (rewilding). It is a low-cost practice, but in highly degraded soils (e. g., lack of fertility, salinization), the recovery of system' properties are slow. For this reason, this strategy is only useful when degradation is limited, and the system has the capacity for recover structure and function.

Whereas ***active restoration***, that includes interventions such as soil management and addition of amendments, planting, removal, and/or reintroduction of species, can a laborious and expensive practice but contributes to improve the capacity of the ecosystem for recover favoured by a better substrate, and the influence of plants on the soils (see Table 3).

High nature value farmland

In spite these benefits, the process of abandonment shows negative effects. In Southern Europe, the accumulation of biomass increases the fire risk, generate uniform plant communities, landscapes, and affect negatively to the sustainable development of mountain communities (Merino *et al*., 2015). In some cases, the revegetation process is managed for productive purposes (extensive livestock and leisure), to avoid environmental problems (wildfire, loss of biodiversity) and preserve cultural landscapes (e. g., Lasanta *et al*., 2015).

High-value nature farmland (HVN) is a concept develop ped by the European Commission in the 1990s as a result of the recognition of the role of traditional extensive farming systems in conserving biodiversity and soil heath. These areas are an important component of European agriculture, not only for their natural values but also for cultural heritage, quality products, and rural employment. The concept of HVN farmland refers to areas in Europe where agricultural activities support and are associated with exceptionally high biodiversity and the presence of environmentally valuable habitats and species. Such farming is usually practised in areas where natural constraints prevent intensive crop production or livestock farming. Some examples are extensively grazed uplands, mountainous meadows, *dehesas* in Spain (Figure 19B) or *montados* in Protugal (Paracchini *et al*., 2008). These systems are semi-natural agricultural habitats, made up by small crop fields intermingled mature trees and shrubs. They usually include linear features, such as field margins and hedges. As a result, the presence of species of high conservation interest that provide polinization services is important in these landscapes. The maintenance of these systems provides biodiversity and other valuable ecosystem services and host habitats which are fully or partially dependent on agriculture. Recognizing this role, different strategies to support these least favoured areas have been implemented under the Common Agricultural Policy.

Figure 19. A) Traditional land uses integrating different agricultural habitats; and B) Typical overview of a dehesa (silvopastoral stem), Extremadura, Western Spain.

4. Conclusions

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

Although each soil presents specific problems, in developing improved land use and practices in agroecosystems, solutions should focus to:

1. reduce tillage that mixes soil horizons, causing loss of soil organic matter and destabilize the soil aggregates;
2. shorten the traffic of tillage machinery and performing the necessary soil management during periods in which the soil takes optimum moisture conditions;
3. maintain the surface of the ground covered with plant residues that diminish the soil temperature and protects the aggregates destruction by rain drops impact;
4. incorporate organic amendments that contribute to maintain an adequate level of soil organic matter content, contributing to stabilize the aggregates stability and the formation of new aggregates;
5. include rotation of species, particularly meadows that include leguminous species for favoring an increment of the soil organic matter.

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6. Study cases

6.1. Study case 1: Application of charcoal for reclamation of intensively managed soils in temperate regions: linking energy production and sustainable agriculture

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Adapted from: Merino et al. (2016).

Background and Aims

Intensive management of soils from humid temperate areas usually leads to acidification and depletion of nutrients and soil organic matter, and therefore, additional sources of nutrients and organic matter, as well as the implementation of soil conservation techniques are usually required. In the last years the application of charcoal has been proposed as an alternative to conventional fertilization for crop and forest soils. As compare to other organic amendments (animal manures, crop residues, municipal waste, biosolids), which can be easily degraded in the soils, the pyrogenic carbon shows long turnover time. This makes to prolong the benefits of the organic amendment. In addition, it also enhances the carbon sink capacity of agroecosystems. Another potential benefit of applying wood ash to degraded soils is favoring the microbial activity and increase its diversity. Proper charcoal to be used as organic amendment is produced by the pyrolysis of biomass, which is known as biochar. However, the greater source of charcoal is the wood ash generated as a by-product in these boilers in conventional grate-fired combustion boilers, known as mixed wood ash (MWA). Because the increased demand for biomass energy, MWA is becoming a major source of pyrogenic organic matter in Europe and other parts of the world.

Strategies

A degraded soil, acidic and poorly drained, having low content in soil organic matter (SOM), was treated with two doses (16 and 32 ton/ha) of MWA (pH = 11,0 and charcoal contain 32 %). After the application of MWA the soil was rotovated (-20 cm depth) and barley (*Hordeum vulgare* L.) was sown.

Highlights: Evolution of soil properties after the application of wood ash

In this example, the study soil was poor in soil organic matter (less than 1 % of SOM) and very acid (pH = 5.0), which is frequent in intensively managed soils of humid temperate areas.

Application of ashes led to increases in OM content (Figure 1). This is due to the direct application of organic residues contained in the MWA to the soil. However, another reason was also the increased crop production. The higher amount of crop residues which were incorporated in the soil after harvesting, also contributed to the higher SOM content. The analysis of soil organic properties revealed a higher content of recalcitrant SOM (Figure 2a).

The study findings show that the MWA application increases in microbial activity and biodiversity (Figure 2a), probably by the rise in the soil pH (Figure 1b), the nutrient availability, and the availability of dissolved organic carbon (DOC). Since the soil showed a clay texture, some improvements on soil physical properties should also be considered. In addition, the biochar shows good binding capacity for certain metals and possibly reduced the level of active aluminum in the soil and therefore its toxicity.

Specific studies have revealed that the increased in microbial diversity are due to the development of fungi and bacteria capable of utilizing the pyrogenic SOM, rich in aromatic compounds, contained in the ash.

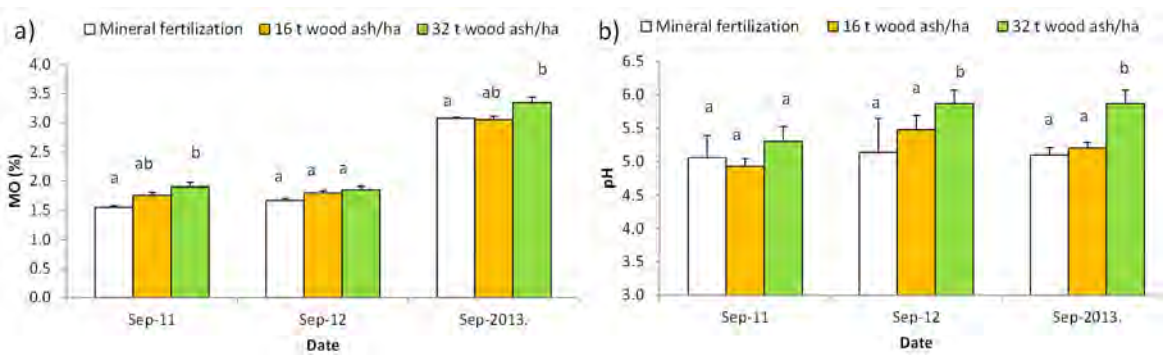

Figure 1. The initial soils showed poor conditions for agriculture (low soil organic matter content and acid pH), due to the mismanagement for years. a) Annual application of 16 and 32 t/ha of wood ash and crop residues led to an increase of OM contents. This effect is partially attributed to direct input of the organic residues contained in the wood ash (15 %), but also to the increased input of crop residues. And b) In humid temperate areas, natural acidification is another important limitation for crop production and, therefore, the soils required the application of liming agents. Wood ash is highly alkaline (pH 8.0-10.0) and contains high contents of K, Mg, and Ca, which are found as carbonates and oxides.

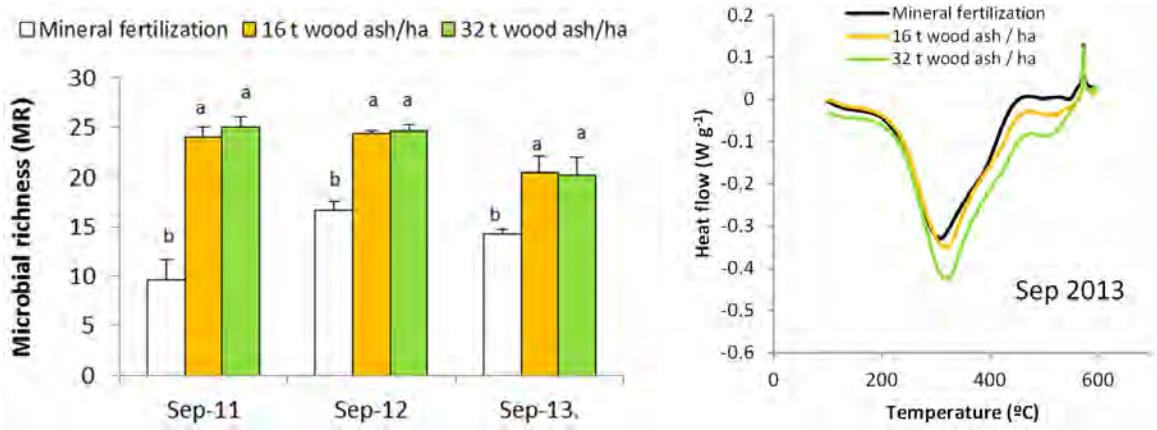

Figure 2. The supply of soil organic matter by wood ash led to a higher variety of soil organic compounds and, therefore, to a greater microbial diversity. a) The calorimetry analysis (Differential Scanning Calorimetry) revealed significant increases in carbohydrates and aromatic compounds in the treated soils. And b) This effect, along with the higher litter inputs and root exudates due to higher crop yield, created conditions that foster eco-diversity and stable microbial communities (in this case, this parameter was determined with Biolog Ecoplates®).

Highlights: Effects of crop production

These changes in soil fertility led to significant increases in barley yields, measured as total biomass of crop (Figure 3).

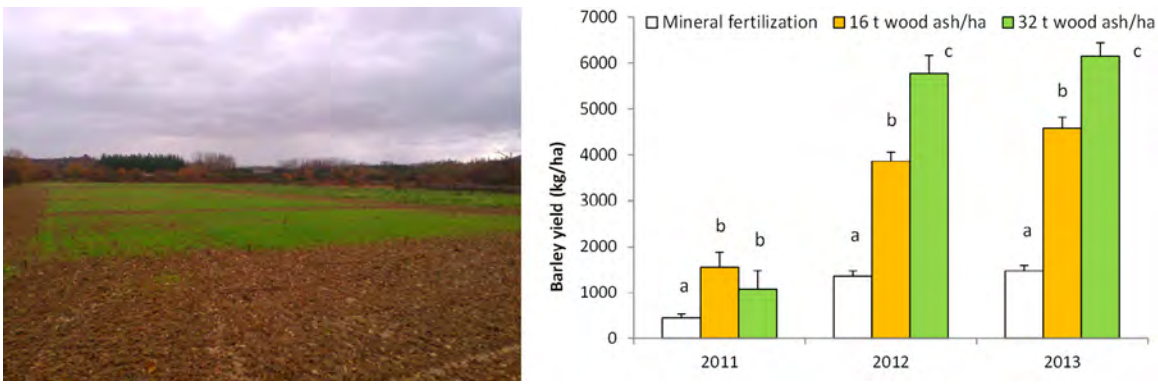

Figure 3. The better soil conditions (better soil aeration, soil pH, and nutrient availability) led to a significant increase in barley yield. Because the great number of roots, the gramineous is a proper crop for reclamation of soils with low soil organic matter contents.

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Merino, A.; Omil B., Fonturbel M.T.; Vega J.A.; Balboa M.A. 2016. Reclamation of intensively managed soils in temperate regions by addition of wood bottom ash containing charcoal: SOM composition and microbial functional diversity. Applied Soil Ecology, 100, 195-206.

6.2. Study case 2: Land use: A case study regarding the effect of agriculture on land degradation in Greece due to increase in soil salinity

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Background and Aims

Since the middle of the XIX Century, major socio-economic factors, including urbanization, intensification and mechanization of agriculture, rapid expansion of tourism, and the development of fast transportation have altered the lad use in the Greece. Currently the agricultural land accounts for almost 2/3 of the total lands surface available, forests for less than 1/3 and approximately 6 % for other uses, e. g., urban networks (Figure 1; *CIA World Factbook*, 2011). The 'green revolution' and lately the accession to the European Union in 1981, led to the intensification of agricultural production and the maximization of fodder and cash crop production (maize, cotton, sugar-beet, *etc.*), which resulted in intensive arable cropping on all fertile, irrigable lands. Demands for water consumption have increased and affected water availability and quality. Irrigation using water with high salt concentrations has increased the salinity of the soil, rendering land unproductive, abandoned and desertified, particularly in the plains located along the coast (LEDD).

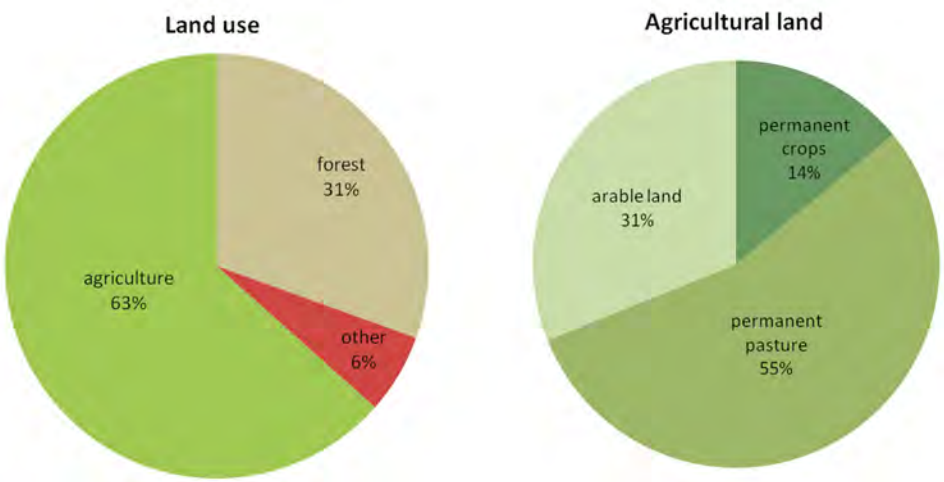

Figure 1. a) Land utilization per sector; and b) Main sectors of agricultural land in Greece in 2011 (CIA World Factbook).

Soil salinization

Salinization is an important process of land degradation and desertification in Greece and other countries of the Mediterranean region including Spain and Italy (Bathrellos *et al*., 2008; Lentini *et al*., 2009; Zanchi & Cecchi, 2010). According to expert assessments, about 15 % of the present irrigated lands face salinity/alkalinity problems (Yassoglou & Kollias, 1989). Soil salinity affects mainly irrigated lowlands with poor drainage conditions and is promoted mainly by irrigation with low quality water, poor drainage, and dry climatic conditions, e. g., Attica, Corinthia or Crete (Figure 2), where a negative water balance is observed (Bathrellos *et al*., 2008; Voudouris *et al*., 2000; Panagea *et al*., 2016). Salinity also occurs under conditions of soils being formed from igneous rocks containing minerals high in sodium, like plagioclases. After irrigation with low quality water, the water added to the soil is absorbed by the crop or evaporates directly from the soil. The salts, however, remain and accumulate in the soil causing salinization (Figure 3a). Salty groundwater may also contribute to the process. In lowlands, where no proper drainage is possible, after irrigation, the water table rises and the saline groundwater reaches the upper soil layers supplying the rootzone with salts (Figure 3b).

Soil salinity is defined as the condition where salt concentration in the water extracted from a saturated soil is high or very high. To extract the water is necessary to fill all the porous with water and let the solid phase to equilibrate with the liquid phase. The extracted solution is called saturation extract. The amount of salts is measured in grams of salt per liter of water (g salt L^{-1}). Soils with concentrations higher than 12 g salt L^{-1} are considered highly saline.

Increased salt concentrations inhibit normal plant growth because it reduces the rate and amount of water the roots can uptake from the soil. Moreover, some salts such as Cl^-, Na^+, and B are toxic to plants when present in high concentration. Hence, crops grown in salt affected soils may suffer from drought stress, ion toxicity, and mineral deficiency leading to reduced growth and productivity. Indicatively, in 2008 a heat wave in the Corinthian Prefecture, combined with irrigation with low quality water, resulted in quantitatively and qualitatively damages in grape production that accounted for more than 14.5 million € (ELGA).

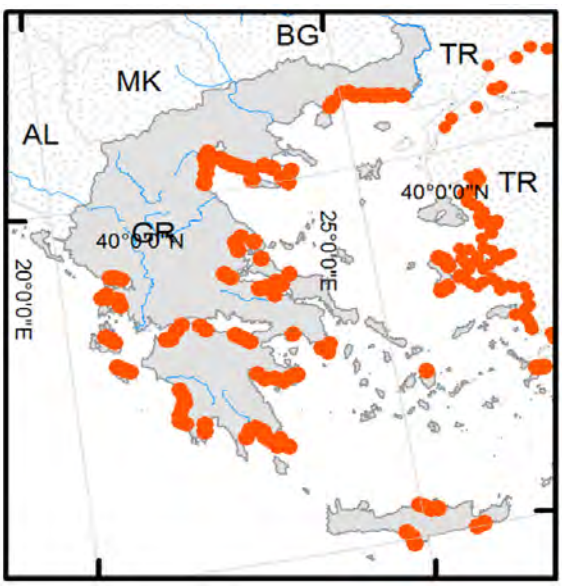

Figure 2. Areas of seawater intrusion in Greece. Adapted from Daskalaki & Voudouris (2008).

The increased frequency of drought spells and heat waves due to climate changes, expansion of irrigated lands, and shifting to higher-yielding, more demanding crops are expected to intensify the already existing problems (LEDD).

The aim of the Project is the assessment of the status of surface and ground water in order to support a more rational exploitation of the water resources and to mitigate problems such as soil salinity, alkalization, and desertification (EEA, 2015).

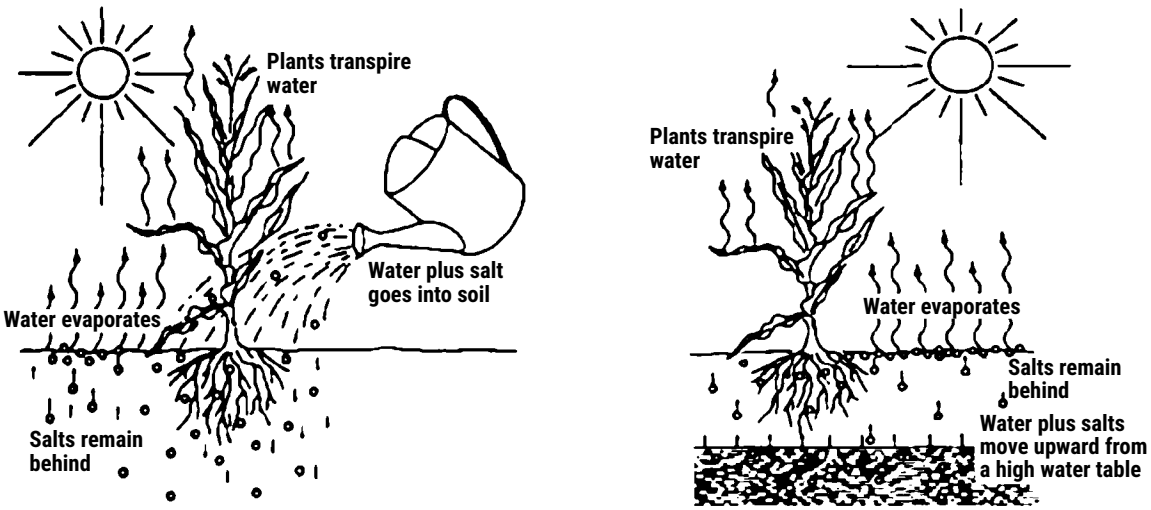

Figure 3. Salinization caused by: a) saline irrigation water; and b) saline groundwater rise (FAO).

Strategies

Preventive practices to control soil salinity include the minimizing irrigation with saline water and improving drainage to favor salts leaching.

In the semi-arid regions of Greece surface drainage is mainly achieved by the construction of surface ditches. These drainage ditches facilitate the disposal of excess irrigation and rainfall water as well as the leaching of undesirable salts from the soil. Another preventive practice aiming to improve ground-water quality and to avoid soil salinization is the recharging of the ground water. For example, on the plain of Argolis, where severe problems of saline water intrusion and soil salinization have been monitored, recharge is achieved by the supplying of the aquifers with good quality spring water through wells during the winter period (Yassoglou & Kollias, 1989).

Several other mitigating measures have been suggested including the growing of halophytes and the employing good soil/water management such as drip irrigation, irrigation scheduling, seedbed placement of applying organic residues and others sub-products (Ravindran *et al*., 2007; Ali, 2011; Yazdanpanah *et al*., 2016).

Panagea *et al*. (2016) suggested that the application of biological agents such as the fungus *Trichoderma ssp.* and various types of symbiotic associations of mycorrhizae improve existing soil properties and increase crop resistance to salinity. Netondo *et al*. (2004) tested the effect of crop rotation for green manuring for the mitigation of soil salinity

Finally, in Greece as part of the River Basin Management Plans, a national monitoring program has been implemented since 2012 including more than 2000 monitoring points across the catchment.

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6.3. Study case 3: Plant cover by use of almond tree (*Prunus dulcis* DA Webb) and organic farming to ameliorate soil quality in southern Italy

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Adapted from: Macci *et al.* (2016).

Background and Aims

The Mediterranean region, characterized by long-dry periods followed by a heavy burst of rainfall, is particularly prone to intense soil degradation processes, which are generally get worse by the presence of rugged terrain. The main effects of soil degradation are the decline of productivity and quality due to the increasing of compaction and runoff and removal of plant nutrients and losses of soil organic matter (SOM) content. Therefore, the development of efficient techniques is required to reduce or stop soil degradation. It is widely known that the vegetation plays a key role in soil protection, both directly reducing physical factors (such as the impact of raindrops) and mechanically stabilizing the soil with roots and indirectly through the incorporation of organic residues. The incorporation of humified organic matter from vegetal origin, in conjunction with the support by external organic materials, could represent a good strategy for soil rehabilitation. This strategy is based mainly on the combined action between plants, SOM, and microorganisms. In fact, the addition of organic residues as amendment, can be useful for the incorporation of different microorganisms and for the stimulation of autochthonous soil microbial population. The resulting effect of this strategy is the increase of microbial number and the stimulation of microbial metabolisms, which, in turn, is the main responsible of soil agronomic fertility, thus sustaining plant growth and development. The use of exogenous organic residues has already proved suitable for recovering degraded soils also in arid and semi-arid areas. Soil quality depends on many physical, chemical, and biological properties, and its characterization requires the selection of indicators that are particularly sensitive to changes in management practices. In general, in soils treated in this way the physical and physic-chemical parameters are less sensitive to soil use and management practices and to the degradation process, as they change slowly and are evident only when the soil undergoes drastic changes. On the contrary, biochemical and biological parameters are considered to be more sensitive indicators, even of slight modifications occurring in soil because they are dependent on microbial biomass activity and are strictly related to active nutrient pools in soil.

In this experiment , conventional chemical parameters usually related to soil fertility (total and available forms of C, N, P, and K) along with more sensitive biochemical indicators of soil

quality and functionality (dehydrogenase and hydrolytic enzymes), as well as yield parameters were studied in order to evaluate the rehabilitation efficiency of almond-tree plantations under organic and mineral fertilization in fields with three different slopes (0, 2, and 6 %).

Strategy

The trial was set up in the Metapontino area (Matera, southern Italy; 40° 23′ N, 16° 46′ E). The area has a typical Mediterranean climate, characterized by cold winters and dry-hot summers, with a mean annual temperature and rainfall of 16.6 °C and 550 mm yr^{-1}, respectively. The soil was a sandy-clay loam (USDA texture classification). with a quite low concentration in SOM (1.07 %).

Three fields (85 m × 35 m, about 3,000 m^2) with three different slopes (0, 2, and 6 %) were selected on the lands of the Pantanello farm company (Figure 1). Each field was divided into two sub-fields, where organic (O) or mineral (M) fertilizers were applied.

For each slope, a plot (about 3,000 m^2) close to each field, without trees and fertilization, was used as control (control soil, C).

The Metapontino area is suited for almond production. The local almond cultivar *Tuono* was chosen in order to recover a local plant genetic resources for protecting soil from degradation. In addition, two different rootstocks (*GF677* and *Franco*) were chosen for their greater adaptability to soil and climatic condition. The *Franco* rootstock (*Prunus amygdalus*) was selected because of its great diffusion in the past decades in the Metapontino area; it is characterized by large, long-lived and deep roots, and adapted to well-drained soil. On the other hand, the *GF677* (*Prunus persica* L. × P. *amygdalus*) was chosen as one of the most common rootstocks for almond production nowadays, being characterized by high-fruit production, drought tolerance, high vigor, deep rooting, and exceptional anchorage.

The trees were planted in a 4 x 5 m^2 arrangement. The fields contained 160 plants, 80 for O and 80 for M, that means 40 plants for each rootstock and treatment. The organic fields were fertilized using 1.50 Mg ha^{-1} per year of commercial composted cow manure (pellet). This O amendment (25 % organic C, 3 % organic N, and 3 % organic phosphorus oxide), usually applied in this area, was incorporated into the first -15 cm of soil, whereas the M fertilizer (15 % ammonium nitrate, 7.5 % phosphorus oxide, and 20 % potassium oxide) was spread on soil surface without incorporation. The spreading of the M fertilizer (total amount of 0.3 Mg ha^{-1}) was conducted three times per year during spring-summer period. In addition, the soil was harrowed at 10–15 cm depth in inter-row soil every two months. The plant residues and weeds were left on the soil.

The soil sampling and the plant yield measurements were made after one year from the beginning of the experiment. In each slope (0, 2, and 6 %), three soil samples were collected for all the experimental cases (fertilization: O and M, rootstocks: *Franco* and *GF677*, and control). Each sample, consisting of five subsamples randomly collected, was taken from the top -15 cm of the soil (150 cm^3 soil cores). These subsamples were mixed, homogenized, sieved (through 2 mm), and stored at room temperature until chemical and biochemical analysis.

Figure 1. Three fields with three different slopes (0, 2, and 6 %) set up in southern Italy.

Highlights

This study has demonstrated that almond tree in semi-arid environments was effective in improving soil quality in terms of increasing the soil organic carbon (SOC) and total nitrogen (STN) contents, and biochemical activities in only 2 years. A greater amelioration of soil properties was reached when almonds were sustained by organic fertilizer, whereas the mineral fertilizer led to the higher release of water-soluble C and N (DOC and DON, respectively) compounds, which could represent an ecological problem because they are considered potential contaminants for surface groundwater.

Among the rootstocks, the *Franco* one had the greatest positive effects on soil properties, resulting in higher SOC and STN contents and dehydrogenase activity (Figure 2), and generally, in all the hydrolytic enzyme activities. On the other hand, the *GF677* rootstock, resulted in a higher growth and yield (Figure 3).

By considering the slope, the 6 % slope has a high risk of soil erosion, showing the lowest content of SOM, nutrients, and biological activity, thus reflecting the lower plant growth and yield. However, even though soil metabolism was lower in the 6 % plot, an improvement in chemical and biochemical soil properties was generally observed for both, mineral and organic treatments with respect to no treated soil (Control).

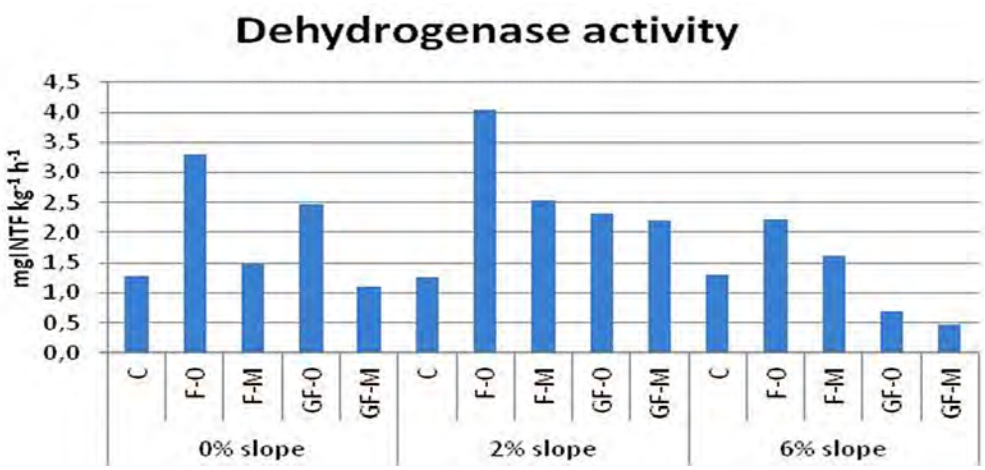

Figure 2. Dehydrogenase activity depends on the metabolic state of soil microorganisms, and it is used as an indicator of soil microbial activity. C, control; F, Franco rootstock; GF, GF677 rootstock; M, mineral fertilizer; O, organic fertilizer.

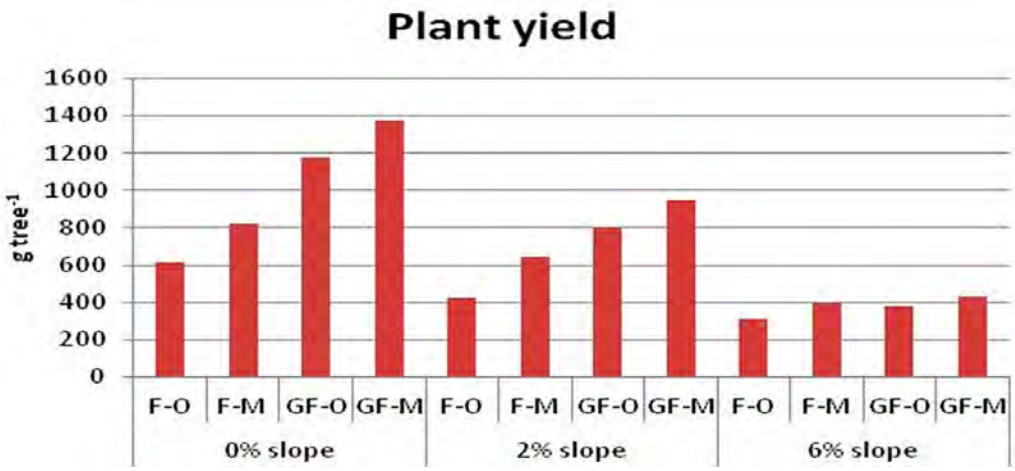

Figure 3. The measurements were carried out on 20 trees per rootstock and fertilization in each slope. F, Franco rootstock; GF, GF677 rootstock; M, mineral fertilizer; O, organic fertilizer.

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6.4. Study case 4: Improvement functions of degraded soils by application of organic amendments

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Adapted from the Final Report of the Project "Strategies for soil restoration and sustainable management of cork oak woodlands (STRAW") - PTDC/AGR-AAM/102369/2008).

Introduction

In Southern Portugal, oak woodlands presently cover more than 10^6 ha and play a important role on the rural economy, as are source of cork, high quality fruits (acorns), pastures, aromatic and medicinal plants, habitat of a great deal of game, firewood and charcoal. Until the seventies of the last century, oak woodlands were mostly used for cereal crops, which led to soils with a low organic matter and nutrient status. Nowadays, in most of the oak woodlands occur natural pastures, which have been extensively or intensively used for livestock rearing. The sustainability of this system has been questioned, by climate changes and by soil degradation. Soil degradation may enhance decline of productivity, loss of organic matter, removal of nutrients and, therefore soil quality, which is associated with outstanding threats such as failure of tree natural regeneration, die-back, and lost of biodiversity. Therefore, the development of management systems and practices are needed to recover or improve functions of degraded soils in the oak woodlands. It is indispensable to quantify the magnitude and direction of soil quality improvement in order to provide the policy makers and land managers with needed indexes of soil quality improvement.

The improvement of soil organic carbon and fertility status of soils is a key issue regarding the oak woodlands sustainable management, considering soil productivity, organic carbon sequestration and resistance to degradation. Nevertheless, soil resources and productivity and sustainability of the system are strongly dependent on the management. The installation of improved pastures has been considered one promising management system as they increase productivity and may contribute to increase soil organic matter content. Meanwhile, it has also reported that they can lead to slow and low increment in soil organic matter, being therefore necessary to develop systems and methods to promote a fast increase in soil organic matter and enhancement of its fertility status. In fact, systems regarding organic amendment practices aiming to promote stable organic C accumulation and soil quality have not been examined. For instance, it is interesting to analyse how organic residues of

understory (mainly shrubs) resulting from fire risk control, and residues resulting from oak prunning (prunning ramial wood) may enhance the level of soil organic carbon sequestration and productivity; also, organic amendments with charcoal (biochar) is also very interesting given its role on enhancement of productivity and carbon sequestration reported for tropical regions, as it is considered recalcitrant and with a long residence time in the soil, and with an enormous potencial to sequester organic carbon into the soil.

In this context, a long term experiment was developed to evaluate whether the application of different organic amendments, in a Mediterranean oak woodland, can improve soil quality and productivity, using different plant tests. To disentangle effects of fertilizers and organic amendments a treatment of fertilizers application was compared with others with fertilizer application plus organic amendments. It was was hypothesized that (i) organic residues will improve plant produtivity and soil fertility; and (ii) biochar has greaterer potencial for a faster and stable organic carbon sequestration. Measurements of plant biomass and soil characteristics, such as organic carbon and nitrogen content, soil reaction, exchangeable cations and extractable phosphorous an potassum, were carried out from 2011. It is expected that information obtained would be useful for future elaboration of managements plans of oak woodland estates, as well as for landowners and respective associations, aiming a soil sustainable management in these systems.

Experimental design

The field experiment was installed in 2011 at "Herdade da Mitra", "Universidade de Évora", Southern Portugal (38°31'38.47"N, 8°1'19.82"W), in an area representative of holm oak woodlands. The landscape is gently undulating with slopes varying from 3 to 8%, and made of a geological substrate consisting of granites and gneisses. It was formely used for cereal crops and grazing; nowdays soils are mostly covered by natural pastures and native srubs (e.g. *Cistus landanifer*, *Cistus salvifolius*). The climate is of Mediterranean type with hot and dry summers and mild wet winters; mean annual rainfall is 665 mm, and mean annual air temperature is 15.4 °C. Soils are mostly dystric Regosols associated with dystric Arenosols (WRB, 2014) developed on gneissic rocks. They are coarse textured (sandy loam; according to the Atterberg scale), acidic (pH in water: 5.4) and show low organic carbon concentration (9 g C kg^{-1}) and low fertility status, concentrations of extractable P and K (by the Egnér-Riehm test) being 2.3 and 50.3 mg kg^{-1}, respectively, in the 0-20 cm soil layer).

The study was installed in the field by using 2 m x 2 m plots and two herbaceous systems (oats and improved pasture) were used (Figure 1). Four treatments were applied for each herbaceous cover, and each treatment was replicated four times. The treatments included three organic amendments - biochar (BI), native shrub residues (SR), and branches and twigs residues resulting from Holm oak prunning (HO) - and a control (CT) where no organic amendments were applied; fertilizers were applied in all treatments according to the nature of the herbaceous cover; in addition, reference plots (SR), with natural vegetation and absence of organic residues and fertilizer application, were considered to assess changes in soil quality status. For each herbaceous cover, similar fertilizar application was adopted in treatments BI, SR, PR and CT. In plots wit oats were applied 80 kg/ha N (one half as basal dressing and the other as topdressing) and 80 kg/ha P_2O_5 (35 kg/ha P). In plots with improved pasture a dose of 140 kg/ha P_2O_5 (61 kg/ha P) was applied; these fertilizer doses were applied in 2011 and 2013.

Figure 1. Field experiment with oats (left, 22 March 2012) and improved pasture (right, 24 April 2012) as plant test.

An amount of organic carbon (11.7 Mg ha^{-1}) was annually incorporated in the 0-10 cm soil layer (that is in 2011 and 2012) in order to approximately double the organic C concentration in the top soil layer. Such carbon amount corresponded to 14.6 Mg ha^{-1} of biochar, and 23.4 Mg ha^{-1} of shrub residues and prunning residues. For calculations was considered a soil bulk density of 1,3 Mg m^{-3} and a concentration of 800 g C kg^{-1} for the *biochar* and 500 g C kg^{-1} for the shrub and prunning residues (dry mass). Concentrations of Ca, K, Mg and P in the biochar were 329.0, 318.4, 43.7 and 31.9 mg kg^{-1}, respectively; and concentrations of B, Co, Cu, Fe, Mn, Mo and Zn were 0.73, 0.39, 0.50, 3.17, 1.32, 0.60 and 0.79 mg kg^{-1}, respectively. After collection, shrub and prunning residues were smashed to bits less than one cm; Concentrations of N, P, Ca, K and Mg in shrub residues were 5.80, 0.48, 3.70, 14.2, and 2.2 g kg^{-1}, respectively; in the pruning material the concentrations were 4.50, 0.32, 2.70, 11.1 and 1.90 g kg^{-1}.

Before plot delimitation, soil was harrowed up to 15 cm depth. Afterwards treatments were randomly distributed by plots, and organic amendments and fertilizers were incorporated (January 2011) into the soil by using a motor cultivator; the sowing was manual and took place in the begining of February 2011. Biomas was evaluated in middle May and soil was sampled in September of each year.

Results

Soil properties

After 54 months of treatment application, significant differences were observed between treatments in both herbaceous systems regarding the concentration of organic carbon in the 0-5 cm top soil layer (Figure 2). Soil amended with biochar showed, in the case of improved pasture, the highest concentration (26.4 g kg^{-1}) which was significantly higher than that measured in the other treatments (12.9-14. 8 g kg^{-1}) and in the reference soil (11.6 g kg^{-1}). Similar pattern was obserd for plots with oats, but significant differences only occurred between the biochar treatment and the reference soil plots. Differences regarding the concentration of organic carbon in the particulate fraction were less pronounced. In contrast to concentration of total organic carbon, no significant differences were observed for the N concentration. It is noteworthy that organic carbon concentration in soil fertilized was similar to that observed in

the soil of reference plots. Our results suggest that the application o large amounts of organic residues does not necessarily lead to strong increase of soil organic matter which may be related to their decomposition dynamicsl. However, the application of biochar led to a high and stable concentrationt, being a good option for carbon sequestration into the soil.

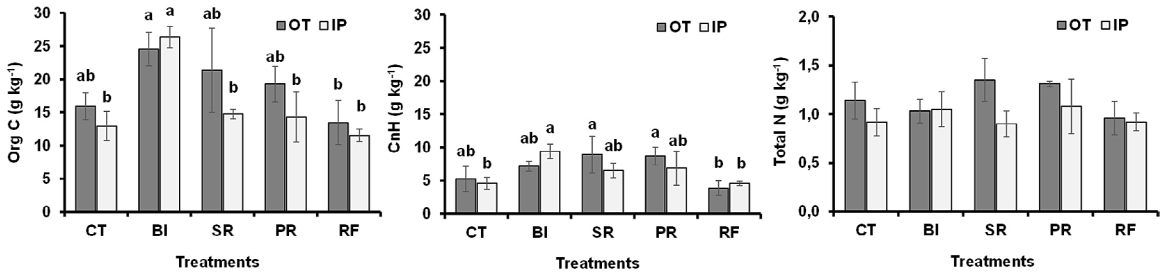

Figure 2. Concentration of organic C and pH values in the 0-10 top soil layer of experimental plots with oats (OT) and improved pastures (IP), 54 months of tratment application. Different letters for each herbaceous cover indicate significant differences (p<0.05) by the Tuky test.

Soil reaction did dot show strong differences between treatments and the reference soil. The lowest concentration of exchangeable Ca^{2+} in both herbaceous systems was measured in the reference soil. In the case of improved pasture, values were significanly lower than those measured in the soil amended with biochar, and in the soil with prunning material; in case of oats significant differences were only observed between the soil treated with prunning residues. Similarly, the sum of non-acid cations (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) showed the lowest values in the reference soil (no application of organic amendments and fertilizars) and differences between treatments were similar to those observed for exchangeable Ca^{2+}. These results undoubtedly show that increasing soil exchangeable cations might be driven by application of fertilizers and organic residues. As the difference betweem soil reference and control treatment is much smaller than with other treatments, the role of organic residues may be dominant. However, it is noteworthy that biochar was responsable for a cation increment despite its low concentrationin in non-acid cations, which may associated with its role on the retention of cations. Present results show that the increment of catión retention by the soil is a result of organic residues application.

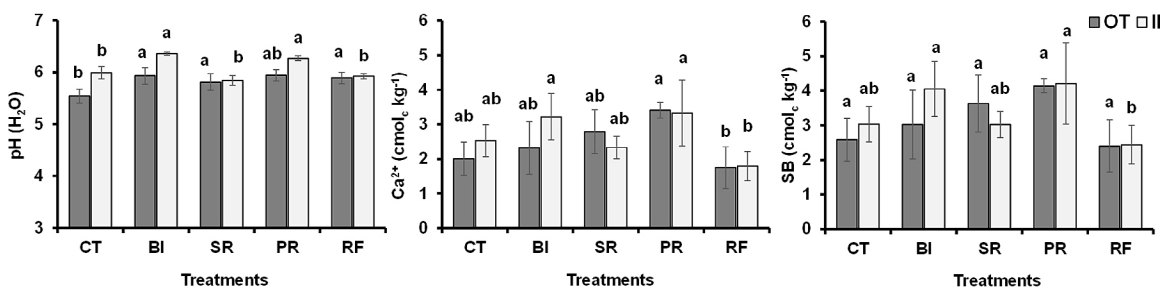

Figure 3. Values of pH (in H_2O), concentrations of exchangeable Ca^{2+}, and sum of non-acid cations in the 0-10 cm top soil layer of experimental plots with oats (OT) and improved pastures (IP), 54 months after treatment application. Different letters for each herbaceous system indicate significant differences (p<0.05) by the Tuky test.

Plant biomass

Results shown in the Figure 4 indicate that, three years after the treatment application, biomass of improved pastures was the highest (8.70 Mg DM ha^{-1}) in the soil amended with biochar, being significantly higher than that measured in treatments with prunning residues (5.42 Mg DM ha^{-1}) and without organic amendments (control; 4.39 Mg DM ha^{-1}). Similar trend was observed for plots with oats, but significant differences were only observed between plots with biochar (13.64 Mg DM ha^{-1}) and plots with prunning residues 11.25 Mg DM ha-1); it is noteworthy that oats biomass was much higher than that observed for improved pasture in all treatments.

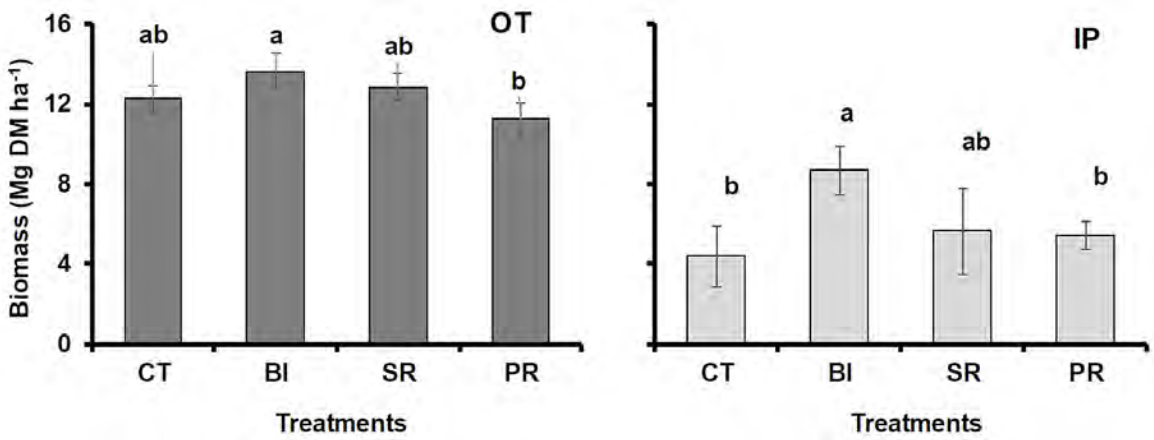

Figure 4. Biomass (mean and standard deviation, n=4) of oats (OT) and improved pastures (IP) in experimental plots, 36 months after treatment application. Different letters for the same herbaceous system indicate significant differences (p<0,05) by the de Tukey test.

Our results undoubtedly indicates that application of biochar to the soil may enhance biomass productivity as compared with other organic residues. However, such a effect may depend on the nature of herbaceous vegetation, as the effect for oats was much smaller than for improved pasture. That is, it may depend on the nutrient requirement and on the adaptation of herbaceous species to drought. The pattern for improved pastures was observed for both grasses and other species (data not ahown). Results from the control indicate that application of fertilizers should be accopmpanied by organic residues application to improve both biomass production and soil quality, the biochar showing to be the best option.

II Degradation and rehabilitation of freshwater wetlands

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1. Wetlands: definition and types

There are multiple approaches to characterize and classify wetlands. For the purposes of the following chapter, we follow the definitions of wetlands adopted in by the Ramsar Convention (1971). Ramsar is the oldest of the modern global intergovernmental environmental agreements. The treaty was negotiated through the 1960s by countries and non-governmental organizations concerned about the increasing loss and degradation of wetland habitat for migratory waterbirds. The Ramsar Convention uses a broad definition of wetlands. It includes *all lakes and rivers, underground aquifers, swamps and marshes, wet grasslands, peatlands, oases, estuaries, deltas and tidal flats, mangroves and other coastal areas, coral reefs, and all human-made sites such as fish ponds, rice paddies, reservoirs and salt pans* (Ramsar 2009). This includes aquatic environments in the coastal area, yet, this chapter addresses only freshwater inland wetlands, and it is specifically focused on open (rivers, lakes) and non-open inland freshwater wetlands. Inland wetland types according the Ramsar Convention Classification system (Ramsar 2009) are presented in Table 1.

Not all the inland freshwater wetlands consist in permanent or temporal open water bodies. Many types of wetlands in this group are ecosystems with the ground water lodged permanently or at least during significant periods of the year and that could roughly be classified as different types of “marshes” or “swamps” (see Table 1; Ramsar, 2009). Among them, they can be discriminated depending on the substrate characteristics, more precisely depending whether the soil is highly organic (peaty) or not. This is a key feature that defines somehow the ecological function of the wetland ecosystems and therefore the strategies for their conservation, management and restoration. A third group can be discriminated, grouping freshwater geothermal and subterranean wetlands, e.g. the waters flowing through karst systems.

Table 1. *Inland wetland types according the Ramsar Convention Classification system. In bold the most significant types of non-open water inland freshwater wetlands. Source: Ramsar, 2009.*

Freshwater	Flowing water	Permanent	Rivers, streams, creeks
			Deltas
			Springs, oases
		Seasonal/intermittent	Rivers, streams, creeks
	Lakes and pools	Permanent	> 8 ha
			< 8 ha
		Seasonal/intermittent	> 8 ha
			< 8 ha
	Marshes on inorganic soils	Permanent	Herb-dominated
		Permanent/ Seasonal/ intermittent	Shrub-dominated
			Tree-dominated
		Seasonal/intermittent	Herb-dominated
	Marshes on peat soils	Permanent	Non-forested
			Forested
	Marshes on inorganic or peat soils	High altitude (alpine)	
		Tundra	
Saline, brackish or alkaline water	Lakes	Permanent	
		Seasonal/intermittent	
	Marshes & pools	Permanent	
		Seasonal/intermittent	
Fresh, saline, brackish or alkaline water	Geothermal		
	Subterranean		

In general, the marshes on peat soils or peatland ecosystems (frequently termed as "mires" in European literature) are characterized by the accumulation of organic matter in soil (more precisely peat, understood as dead and partially decomposed plant remains that have accumulated *in situ* under waterlogged conditions), due to an unbalance between the net primary production and decomposition rates in the ecosystem. Hence in the peatlands the environmental constraints slow down the rate of organic matter decomposition, allowing the accumulation of peat. The presence of vegetation capable of forming peat is the key characteristic of "functional" peatlands, in other words, an active peatland ("mire") is a peatland on which peat is currently forming and accumulating due to the accumulation of plant remains of mire vegetation. The structure of the peatland ecosystem vegetation might vary from herbaceous to woody formations (either shrubs or woodland) (Ramsar, 2002).

Peatlands cover 3% of the earth surface and hold 10% of all the freshwater resources (Joosten & Clark, 2000) and they can be found in all biomes, particularly the boreal, temperate and tropical areas of the planet. Peatlands have become increasingly recognized as valuable ecosystems worldwide, supplying key services as contributing to biological diversity, the global water cycle, global carbon storage — relevant to climate change — and other wetland functions valuable to human communities. They also act as valuable archives of paleo-environmental changes from which to reconstruct past landscape change and previous climates, and determine human impact upon the environment (Ramsar, 2002, Simil et al., 2014).

Peatland are also fragile ecosystem relying on delicate climatic, hydrologic, edaphic and biotic balances. Alteration of these features, due to perturbations like peat cutting, drainage or the removal or alteration of vegetation cover causes frequently the collapse of the system. Degraded peatlands suffer from soil erosion, decreased their capability of regulation of water cycle and are prone to turn they role from carbon sinks to carbon sources (Simil et al., 2014).

The other group of inorganic marsh wetlands includes a wide (and somehow heterogeneous) group of ecosystems with several vegetation cover and structure, ranging from grasslands to forest. Their key defining feature is the periodic or permanent flooded or waterlogged conditions that eventually determines their structure and functions, and in contrast with the peatlands, the rate of decomposition of vegetation remains is high enough to prevent the accumulation of peat. They occur frequently in water course floodplains or as transitions or mosaics with different types of water bodies, shaping diverse and valuable landscapes from the biodiversity point of view. Other important group are the wet heathlands, occurring frequently on soils with a high content in organic matter and with transitional properties respect the peatlands. As in the case of peatland wetlands, they are fragile and threatened ecosystems, where perturbations in their hydrological regime, water quality, substrate or vegetation coverage might compromise their conservation.

2. Importance and causes of wetland degradation

2.1. Importance and challenges

Freshwater ecosystems are among the most valuable ecosystems on the planet. In the 5th Edition of their referenced textbook, Mitsch and Gosselink described the importance of wetlands as follows: “Although the value of wetlands for fish and wildlife protection has been known for a century, some of the other benefits have been identified more recently. Wetlands are sometimes described as kidneys of the landscape because they function as the downstream receivers of water and waste from both natural and human sources. They stabilize water supplies, thus mitigating both floods and drought. They have been found to cleanse polluted waters, protect shorelines, and recharge groundwater aquifers. Wetlands also have been called nature's supermarkets because of the extensive food chain and rich biodiversity that they support. They play major roles in the landscape by providing unique habitats for a wide variety of flora and fauna. Now that we have become concerned about the

health of our entire planet, wetlands are being described by some as important carbon sinks and climate stabilizers on a global scale" (Mitsch & Gosselink 2015).

Wetlands provide an extensive list of benefits for humans that are designed Ecosystem Services (ES). This is an anthropocentric concept, as opposed to the eco-centric concepts of ecosystem processes and ecosystem functions. ES are the direct and indirect contributions of ecosystems to human well-being and they support directly or indirectly our survival and quality of life. Ecosystem processes involve the interactions (events, reactions or operations) among biotic and abiotic elements of ecosystems which underlie ecosystem functions. Examples of ecosystem processes include photosynthesis and nutrient uptake. Ecosystem functions are intrinsic ecosystem characteristics related to the set of conditions and processes whereby an ecosystem maintains its integrity. Examples of ecosystem functions include primary productivity and biogeochemical cycles.

The Millennium Ecosystem Assessment (MA) (2005) provided a classification of ES that is globally recognised and where the ecosystem services are categorized as being provisioning, regulating, cultural, and supporting (Table 2). The latter category includes basic ecosystem processes. Later the publication of the Common International Classification of Ecosystem Services (CICES) classification (Maes et al. 2013) established a hierarchy of services which focuses on the services dimension of ecosystems, considered as a natural capital. In the Common International Classification of Ecosystem Services (CICES), services are either provided by living organisms (biota) or by a combination of living organisms and abiotic processes.

The more utilitarian approach followed by the Economics of Ecosystems and Biodiversity initiative (TEEB, http://www.teebweb.org/), classifies the environmental services in four groups like the MA 2005 approach, namely provisioning, regulating, habitat or supporting and cultural services. Provisioning services are basically material outputs from ecosystems, including food, water and other resources. Regulating services are those acting as regulators, e.g. regulating the quality of air and soil or by providing flood and disease control. Habitat or supporting services underpin almost all other services. Hence, ecosystems provide living spaces for plants or animals and in general contribute to the biodiversity of a territory. Cultural services include the non-material benefits people obtain from contact with ecosystems, including aesthetic, spiritual and psychological benefits. A recent monography focused on the application of the TEEB approach on water and wetlands (Russi et al., 2013) concludes that wetlands are a fundamental part of local and global water cycles and act as a nexus between water, food and energy, providing numerous ecosystem services to humankind. However, wetlands continue to be degraded or lost and, in many cases, policies and decisions do not sufficiently take into account these interconnections and interdependencies, making mandatory a more effective acknowledgment and integration of the wetlands into decision-making in order to meet our future social, economic and environmental needs.

Table 2. *Summary of the Ecosystem services (ES) of wetlands. Sources (Maes et al. 2013) and (Mitsch et al. 2015) based on the Millenium Ecosystem Assessment (MA, 2005) categories:*

Category of ES	Description of ES	Example of service provided by wetlands (Mitsch et al. 2015) based in MA
Provisioning services (MA and CICES)	All material and biota-dependent energy output from ecosystems, i.e. tangible things that can be exchanged, traded, consumed or used directly by people. Provision of freshwater is considered ES because its amount and quality is at least partly steered by ecosystem functioning (CICES)	Freshwater Fisheries support Peat production for fuel and horticulture Animal harvesting Timber production Direct food production
Regulating (and Maintenance) services (CICES)	All the ways in which ecosystems control or modify biotic or abiotic parameters that define the environment of people, i.e. outputs that are not consumed but affect the performance of individuals, communities and populations and their activities (CICES)	Water quality improvement River flooding mitigation Protection of coastlines from tsunamis, cyclones, and other coastal storm surges Carbon sequestration Habitat for rare and endangered species
Regulating and supporting services (MA)	Basic ecosystem processes of nutrient cycling and primary productivity that may, in turn, lead to the other three services above (MA)	Wetland functions such as hydric soil development, primary productivity, serving as chemical sources, sinks, and transformers, and water storage
Cultural services (MA and CICES)	All non-material ecosystem outputs that have symbolic, cultural of intellectual significance (CICES)	Landscape aesthetics Sites for human relaxation Ecology education Sustenance of human cultures Ecotourism, bird-watching

2.2. Causes of wetland degradation

Freshwater ecosystems have suffered intense and long-lasting human pressures since water is one of the most essential natural resources. Until the 1970's, the drainage and destruction of wetlands were accepted practices around the world and even encouraged by specific government policies (Mitsch & Gosselink 2015). The extent of the world's wetlands is now thought to be from 7 to 10 million km2, or about 5 to 8 percent of the land surface of Earth. The loss of wetlands in the world is difficult to determine, but recent estimates suggest that we have lost more than half of the world's wetlands, with much of that occurring in the twentieth century (Mitsch & Gosselink 2015). In Europe, more than 80% of wetlands area has been lost (Verhoeven 2014).

Freshwater ecosystems are transformed through widespread land cover change, urbanization, industrialization and engineering schemes such as reservoirs, water diversion, groundwater pumping, irrigation and inter-basin transfers that maximize human access to water (Vörösmarty et al. 2010). In the 20th century, >70% of largest rivers in the Northern hemisphere have been strongly or moderately affected by regulation, water diversion or irrigation (Dynesius & Nilsson 1994) and this situation is aggravating worldwide. These multiple changes have modified natural processes and fragmented the aquatic landscape, increasing the ecosystem vulnerability to the additional stresses associated with climate change.

In Mediterranean regions, human activities have impacted wetlands more than their counterparts in more humid or arid regions because of the severe competition for water that occurs in Mediterranean-climate regions (Gasith & Resh 1999). Under climate change and increasing water demand, the Mediterranean region is expected to experience the greatest proportional impact compared to other biomes (Sala et al. 2000). This makes freshwater and riparian ecosystems particularly fragile in the Mediterranean Region. As a result, conservation, sustainable management and restoration of freshwater and riparian ecosystems is a major challenge for this century (Naiman et al. 2005).

2.3. Protection, Rehabilitation and Restoration

As the society became aware of the pervasive environmental and societal consequences of the degradation of freshwater and riparian ecosystems, their conservation and recovery has become an increasingly important target. Yet, many attempts to restore functional ecosystems fail causing great ecological and economic issues (Boudell et al. 2015). For example, there is growing interest in applying river restoration techniques to solve environmental problems, yet little agreement exists on what constitutes a successful river restoration effort (Palmer et al. 2005). Among the sources of un-success in rivers, it has been pointed out that the underlying causes have not been addressed and ameliorated, when practices are focused narrowly on re-creation of certain elements, when hydrologic regime and connectivity has not been re-established (Boudell et al. 2015).

One important issue in the recovery of freshwater and riparian ecosystems is the frequent lack of agreement in the perception and expectations of different stakeholders about what is restoration, when it is needed/beneficial and for which goals, and the different approaches to execute it. The first challenge is managing to use terms with the same meaning among policy-makers, managers and researchers. The Society of Ecological Restoration (SER, 2004), defines **Ecological restoration** as *the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed.* In fluvial ecosystems, this would mean res-establishing the natural processes, structure, function, territory and dynamics and entails that true ecological restoration should be passive or self-restoration, through the elimination of impacts and pressures until the sustainable functioning of the ecosystem (Ollero 2015). In many cases the level of ecosystem degradation or the conflict with human uses prevent the true ecological restoration of fluvial ecosystems. Face to the multiple difficulties to undertake a complete ecological restoration in fluvial systems, the concept of Rehabilitation has gained acceptance. **Rehabilitation** involves the recovery of a more natural functioning, re-establishing some elements, processes or important functions (Ollero 2015). It is a broad concept that includes different degrees of improvements in the structure and/or functioning

of ecosystems, as illustrated by Williams et al (1997) (Figure 1). Yet, ecological restoration can be still considered the target upon certain circumstances.

One of the cornerstones of Rehabilitation is the definition of the reference state that theoretically represents the ecosystem state without human pressures and that should guide restoration targets. This has been represented sometimes as an historical pristine natural ecosystem yet, science has shown that some dynamic ecosystems, such as rivers, follow complex trajectories frequently making it impossible to return to a previous state. Therefore, restoration goals are moving away from explicitly defining a reference state because of the difficulty of attaining that reference state (Dufour & Piégay 2009). In addition to the difficulties, it is increasingly recognized the need that rehabilitated ecosystems play multifunctional ecologic, social and economic roles. Thus reference-based strategy is being replaced by an objective-based strategy that reflects the practical limitations of developing sustainable landscapes and the emerging importance of accounting for human services of the target ecosystem (Dufour & Piégay 2009).

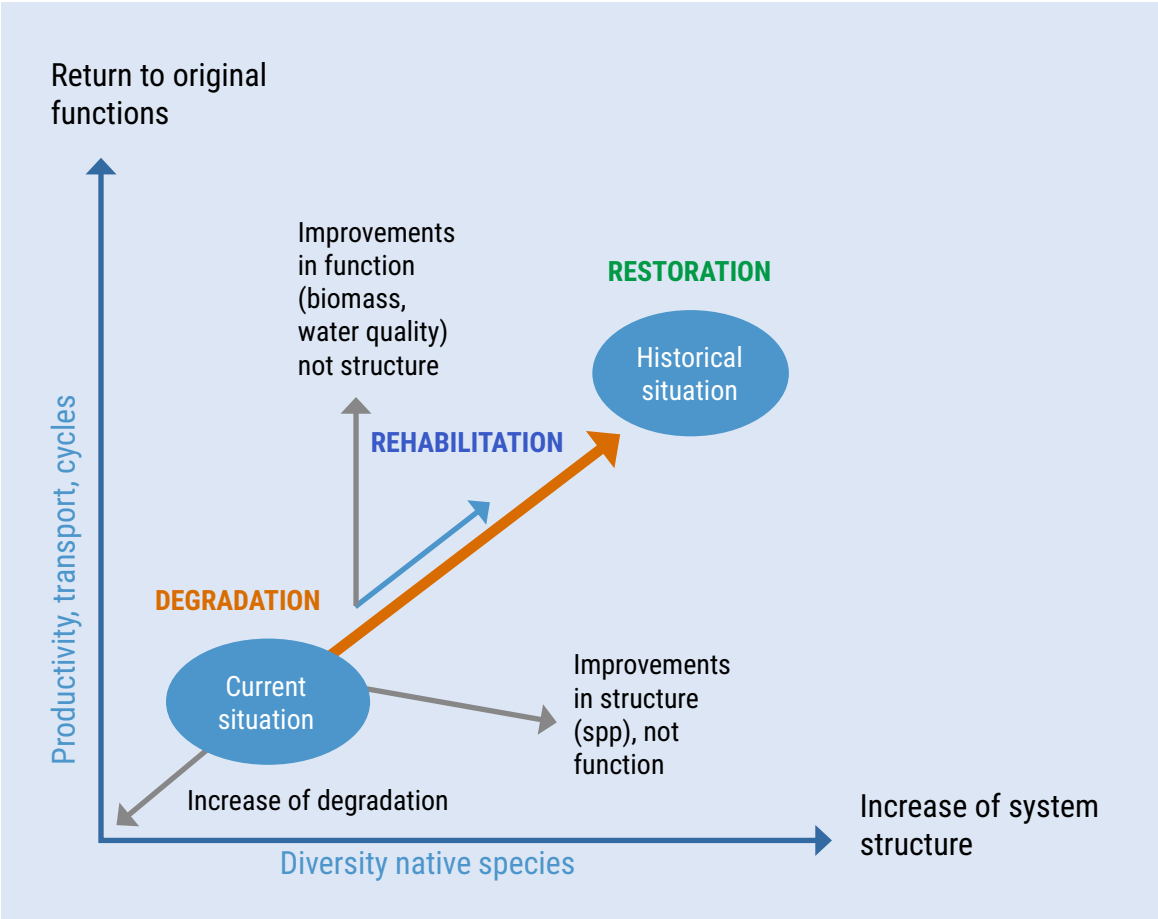

Figure 1. Trajectories of Restoration-Rehabilitation. (Williams et al., 1997, adapted from Cortes 2004).

3. Strategies for rehabilitation of major wetland types

3.1. River rehabilitation

3.1.1. Understanding rivers functioning as a basis for rehabilitation

River ecosystems are organized in a hierarchical structure within hydrographic basins. Hydrographic basins display predictable patterns of geomorphology and hydrology based on their age, form, size, tectonic setting, lithology, and climate regime. From the headwaters to the mouth, there is a continuous gradient of physical conditions (e.g. depth, flow magnitude, channel width, temperature) which creates a dynamic equilibrium in the river morphology and hydraulics (Curry, 1972). The dynamics of a river ecosystem occurs at specific spatial and temporal scales defining the structure and function of their biological communities which have evolved in adaptation to them (Lytle and Poff, 2004). The recognition of this physical gradient emerged by observing the patterns of occurrence and assemblages of freshwater species along the river continuum (Vannote et al., 1980). This concept proposes that freshwater communities would become established structurally and functionally in the river continuum, according to that longitudinal physical gradient (Vannote et al., 1980). The river continuum accommodates different levels of spatial organization, from microhabitat to the basin scale, each of which playing structural and functional roles within the hydrographic basin (Frissell et al., 1986) (Figure 2). The processes that occur within and between these levels of spatial organization are reflected by the interactions between four dimensions: longitudinal, lateral, vertical and temporal (Ward, 1989 of the so-called hydrosystems (Amoros and Petts, 1993). In hydrosystems, the first three spatial dimensions interact mutually according to the temporal dimension, from instant to an evolutionary time scale (Figure 2).

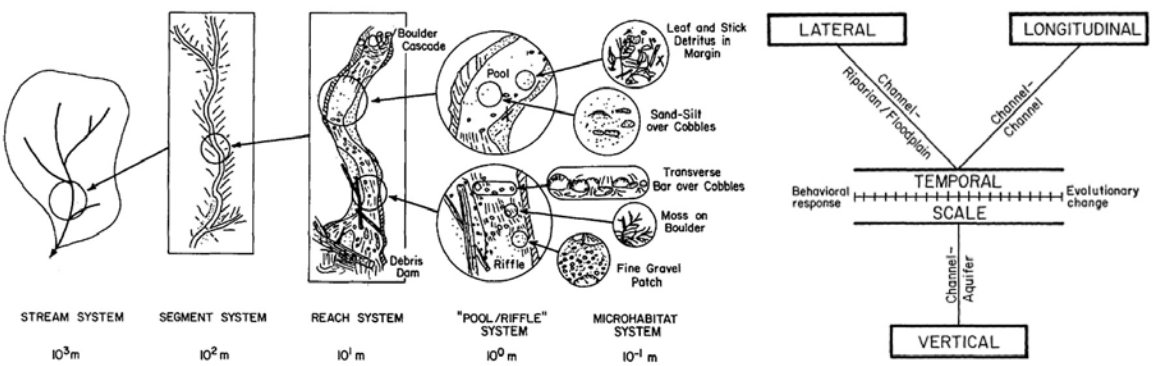

Figure 2. Hierarchical organization of a stream system and its subsystems (Left), and a conceptualization of the four dimensional nature of lotic ecosystems (Right). Adapted from (Frissell et al., 1986; Ward, 1989).

These ecological paradigms depict the functioning of intact river ecosystems. Anthropogenic pressures were changing the river continuum and forcing the adaptation of freshwater communities (Vannote et al., 1980; Ward, 1989). A continuous view of rivers is essential for effective research, conservation and management of these key ecosystems and their biotic communities, i.e. a view not just of disjunct reaches but of the entire spatially heterogeneous

scene of the river environment: the riverscape (Fausch et al 2002). River managers and users need to understand these basic principles driving the nature of river ecosystems to create a balanced trade-off between societal needs and ecosystem health when planning and implementing restoration measures.

Indeed, the understanding that the nature of river ecosystems is hierarchical, and that the processes occurring at each spatial component will affect the whole system are decisive to outline guidelines of restoration. Particularly, the flow regime plays a determinant role for ecological process in rivers, its natural variability functions as a cue for life-cycle events, and its long-term temporal character determined evolutionary traits (Bunn and Arthington, 2002). The natural flow regime, firstly described by Poff et al. (1997), was introduced to argue its critical role in sustaining the ecological integrity of the river ecosystem. The natural flow regime framework postulates that the temporal variation of flow regime shapes the mosaic of the physical habitat and regulates the structure and function of riparian and aquatic communities (Bunn and Arthington, 2002; Poff et al., 1997). The primary components of flow regime are: magnitude (or flow ratio), frequency, duration, timing and rate of change These act at different spatial and temporal scales, regulating short- and long-term river ecosystem processes (Lytle and Poff, 2004). The natural flow concept gains particular relevance in Mediterranean regions where streams are physically, chemically and biologically shaped by seasonal events of flooding and drying over an annual cycle (Gasith and Resh, 1999). In sum, river and riverscape rehabilitation from for the long term requires, on one hand, attention to ecological functioning river systems, particularly reversible processes, as well as connectivity, and thresholds (Naiman et al 2005).

3.1.2. River and riverscape restoration in Europe

River systems have been transformed through widespread land cover change, urbanization, industrialization and engineering schemes like reservoirs, irrigation and interbasin transfers that maximize human access to water (Vörösmarty et al 2010). In this context, different societal efforts have developed pursuing to reverse the degradation of river systems. In European countries, directives such as the Water Framework Directive (WFD) (2000/60/EC 23 October 2000), the Flood Directive (2007/60/EC 23 October 2007) and the Pesticide Directive (2009/128/EC 21 October 2009) explicitly require the Member States to produce integrated river basin management plans (RBMPs) which shall include programs of restoration measures to prevent further deterioration and ameliorate the ecological status of their water bodies and river ecosystems.

The rehabilitation measures can be categorized in several types based on the spatial scale they operate within the hierarchical organization of river basins:

1. Channel: margin, bank and substrate rehabilitation
2. Channel: restoring connectivity along the river
3. Riparian zone: conservation, management and recovery of riparian vegetation
4. Floodplain: restoring natural connectivity and keeping natural flooded areas
5. Sub-basin: effective management of waste and treatment systems

6. Sub-basin: ecological flow management
7. Sub-basin: best agricultural practices
8. Sub-basin: changes in land uses.

Other European legal instruments such as the Habitats Directive (92/43/EEC), promotes the conservation of river systems based on the presence of certain communities and features considered priority for conservation at the European level. Nonetheless, there are indeed many synergies as the implementation of measures under the WFD will generally benefit the objectives of the Nature directives and vice versa. Therefore, there is a need for coordination between the responsible authorities for nature conservation and water management (ETC/IM, 2015) (Table 3).

Table 3. *Key measures with joint benefits for water management and nature protection. Adapted from ETC/IM (2015)*

Measures	Pressures	Benefits for water management	Benefits for nature protection
Removing barriers Fish passes	Interruption of longitudinal continuity Barriers (dams, reservoirs, weirs)	Restore longitudinal continuity Enhancement of fish populations	Improvement of habitat connectivity between natural areas Improvement of conservation status for migratory fish
Room for rivers (removing dykes)	Interruption of lateral connectivity Physical alteration of channel Channelization/ straightening Bank reinforcement	Improvement of hydromorphological status, habitat heterogeneity Flood protection	Provision room for key habitats (wetlands, floodplains)
Establish ecological flow	Flow regulation Barriers (dams, reservoirs, locks, weirs) Abstractions Hydropeaking	Improvement of the flow regime, support on river continuum, improvement of habitats depending on flow	Improvement of habitat connectivity, return to more natural state of habitats adversely affected by changed flow regime
Floodplain reactivation and riparian zone rehabilitation, re-meandering, reconnection of backwaters	Interruption of lateral continuity Physical alteration of channel Channelization/ straightening Bank reinforcements Land infrastructures Agricultural enhancement	Improvement of the link between the river and its floodplain Provision of cover and shallow areas for freshwater communities	More natural conditions which produce better quality and more numerous habitats for aquatic and terrestrial species. Large part of floodplains are designed Natura 2000 sites

Instream habitat restoration	Physical alteration of channel Channelization/ straightening Bank reinforcements Sediment extraction or input	Improvement of hydromorphological status (self-dynamic development of structural elements)	Can have positive local effects on specific protected habitats and species in the aquatic environment
Buffer strips Establishment of buffer strips.	Diffuse pollution Agricultural enhancement Bank reinforcements	Water quality improvement (Filtration of nutrients and reducing run-off) Improvement of hydromorphological status of riparian zone	(Re-)Creation of natural habitats in areas strongly developed for agriculture Provision of favourable conditions for certain terrestrial and semi-aquatic species
Sediment-related measures, e.g. remove (fine or contaminated) sediment, add sediment, activate sediment connectivity	Sediment extraction (dredging) or input Interruption of longitudinal continuity Barriers	Increase of the diversity of riverine environment and enhancement of habitats, especially for fish and benthic invertebrates Improvement of bed (hydromorphology)	In combination with other measures, sediment-related measures can enhance the status of certain aquatic habitats and species
Prevention measures for the spread of invasive species into new areas Control measures (e.g. herbicides for plants)	Invasive non-native species	Achieve good ecological status, ensure non-deterioration Protection of affected structures (e.g. flood defense works)	Contribute to favourable conservation status of Natura 2000 sites
Increasing soil and landscape water retention and groundwater recharge	Soil compaction due to intensive agriculture Land infrastructure Deforestation	Reduction of floods Reduction of droughts Achievement of GES	Contribute to the improvement of aquatic habitats
Restoration of wetlands and forests	Interruption of lateral continuity (dikes) Physical alteration of channel Channelization/ straightening Bank reinforcements Land infrastructure Agricultural enhancement	Improvement of hydromorphological status Achievement of GES	Improvement of habitat connectivity between natural areas (counteracts habitat fragmentation)

3.2. Lake rehabilitation

3.2.1. Lakes and eutrophication

Lakes are ecological hotspots of multiple ecosystems services, including food, water, nutrient recycling, primary production, recreational and aesthetics values. Man-made water reservoirs perform other societal functions, in general related to flood/drought regulation, energy production or crop irrigation. The Water Framework Directive (WFD), directive 2000/60/EC of the European Parliament and the Council of October 23, was issued in 2000 with the purpose of establishing the protection of inland surface waters, transitional waters, coastal waters and groundwater. However, after two updates of River Basin Management Plans (RBMP), namely in 1999 and 2015, country reports showed that a substantial proportion of Europe's freshwaters did not achieve the 'good status', as already forecasted by Gibbs and Özkundakci, (EEA, 2010; Gibbs and Özkundakci, 2011). Certainly, water eutrophication is a major responsible for such failure.

Figure 3. Macrophytes in Sete Cidades lake (Azores, Portugal).

Figure 4. Algae blooms in Sete Cidades (left) and Furnas (rigth) lakes (Azores, Portugal).

The natural eutrophication process of water bodies reflects the progressive aging of reservoirs (ponds, lakes and artificial reservoirs), resulting in the accumulation of sediments, minerals and organic substances carried out from the watershed. This evolution of the water bodies takes place very slowly towards a wetland formation, which at the end of the succession achieves similar characteristics to the terrestrial ecosystem. Indeed, the natural eutrophication

process is the (slow) evolution for a climax state of ecosystems. The problem occurs when the process is accelerated in one or two decades leading to significant adverse impacts in all ecosystem functions and services. The increased nutrient load triggers an unusual growth in phytoplankton and/or macro algae, with adverse effects on biodiversity values. This process has additional negative impacts on water quality, with increased turbidity, color change and the increasing presence of cyanobacteria toxins, with consequences on most water uses for human consumption.

Therefore, this chapter addresses the eutrophication phenomenon, its causes, main consequences and drivers for action. Because of the increasing importance, a special attention is paid to the phosphorus (P) framework issue. External and internal phosphorus in lakes/reservoirs has been a major concern from a management perspective and measures aiming at lake protection and rehabilitation are presented also.

3.2.2. Eutrophication: causes and challenges

Eutrophication is mainly caused by human activities that provide increasing inputs of nutrients. The leaching of surplus fertilizers and slurries from watershed and the discharge of municipal wastewaters are among the most usual anthropogenic pressures on river basins. The 'phosphorus factor' is the most important one in lake' eutrophication processes. It should be stressed the role of phosphorus, other nutrients are much less important (nitrogen, for instance). The significance of phosphorus has been supported by numerous case histories and large-scale experiments (Vollenweider, 1970; Ruttenberg, 2003; Schindler, 2012). A phosphorus balance in a lake is depicted in Figure 5.

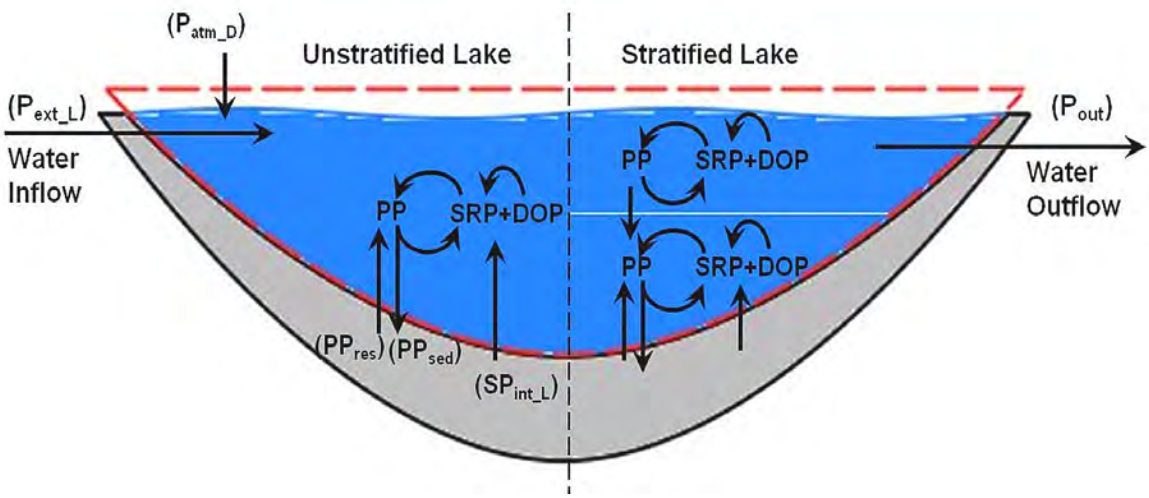

Figure 5. Phosphorus budget in a lake (Source: Ribeiro, 2008).

The reason why phosphorus is so widespread used is due to his role for sustaining human life and because is used as fertilizer (90%) in order to support the increasing need of food and other emergent industries (Elser, 2012). The consumption of phosphate fertilizers in 2007 was estimated at about 18 million tonnes, supporting the present food requirements of 7.0 billion people and the 9.3 billion forecasted for 2050. The industrial consumption is

also rising (not in detergents, but in storage batteries, pharmaceutical or food supplements according to IFA, 2012). The phosphorus losses along the chain value are a burden in terms of economy and a very significant source of negative environmental and health related impacts. Indeed, it is estimated that around 10 Mton/year of phosphorus are dissipated in the hydrosphere.

Eutrophication is probably the most widespread water quality problem in freshwaters and a major trigger regarding the occurrence of coastal anoxic dead zones. The eutrophication of waters bodies is noticed since the 1950s, yet today the phenomenon is far from being solved. Several studies in the 90´s reported that eutrophication was the major water quality problem in the United States, affecting approximately half of the lakes, being also a known problem in the estuaries (Smith et al., 1999). In addition, all 217 lakes included in a survey of the International Lake Environment Committee (ILEC) showed an increase in the level of eutrophication over the past 50 years (UNEP, 2012). Undoubtedly, the eutrophication processes activated by human activities are, in most cases, far from a significant reversal, contrary to existing expectations when this phenomenon was detected in the 50-60 decades (Sondergaard et al., 2007, Schindler, 2012).

In Europe, a report under the Nitrates Directive showed that in 33% of the surface water monitoring stations indicate a eutrophic or hypertrophic status. Indeed, more than 53% of the lakes are eutrophic and this is the main pressure responsible for the failure of the aim of 'good status' by 2015 prescribed by EU Water Framework Directive (WFD), directive 2000/60/EC (EEA, 2010a; Gibbs and Özkundakci, 2011). Derogations of the good water status prescribed by WFD have been claimed when eutrophication is the main pressure.

Even so, phosphorus inputs to many culturally eutrophied lakes were reduced in paste decades and there are now some case histories showing successful reversals of lake eutrophication (EC, 2010). Despite the phosphorus input reductions, eutrophication remains a major problem, either because external loading has not been dropped sufficiently or because internal lake mechanisms, chemical or biological, prevent or delay lake recovery (Søndergaard et al., 2003; Søndergaard et al., 2007). Actually, even when external phosphorus sources are eliminated, a slow rehabilitation process is observed because of the re-dissolution of phosphorus accumulated in sediments [3, 4].

Eutrophication is not caused by current yearly fertilization practices only (the agriculture activity will not end hopefully!) but by a long history of phosphorus accumulation in aquatic sediments (UNEP, 2012). Due to mobility of phosphorus between the sediments and the water column, eutrophication processes can persist for a long time. An examination of long-term data collected from 35 lakes in Europe and North America led to the conclusion that P release from the lake sediments continues typically for 10-15 years after reduction of P loading to the lakes (Jeppesen et al. 2005), but in some lakes may last for more than 20 years (Søndergaard et al. 2003).

Eutrophication processes triggers significant public health problem. The outbreak of cyanobacteria may release secondary metabolites generically called cyanotoxins, with toxic effects to the liver and at neurological and dermal levels . The health risks are caused by ingestion of the biotoxins presented in water, by direct contact in recreational uses with or by their presence in water for other uses, such as in hemodialysis systems. Among the worldwide cases of health related problems associated with toxins in lakes were episodes

in Sweden, USA, Australia, UK, Brazil (in this case, indicating about 50 deaths in 1996) and chronic problems before non-lethal doses . The presence of cyanotoxins in water treatment systems raises an additional problem because chlorine oxidation — the most widespread water disinfection method for human consumption — trigger the formation of halogenated organo-compounds (Brito et al., 2010).

The estimated annual cost of eutrophication in the United States of America was as high as 1.7 billion euros in 2009 (Dodds et al. 2009). The estimate damage costs of freshwater eutrophication in the United Kingdom are around 85-130 million euros per year plus 61 million euros per year of costs incurred in responding to eutrophication (Pretty et al., 2003). Before the 80-90´s, the reason for eutrophication was considered to be the phosphorus rich municipal wastewater discharges into surface waters. Therefore, billions of euros have been invested in improved wastewater treatment and other pollution-combating measures leading to the wishful think that eutrophication problem was solved (Søndergaard et al., 2007). Unfortunately, the reality fast showed the opposite (Conley et al., 2009).

3.2.3. Characterization and diagnosis

Lake and watershed planning

A programme of measures for lake protection and ecosystem restoration begins with an extensive characterisation and data collection regarding structural and functional elements of lake ecosystem and whole watershed. This is the starting point towards a characterisation and a good diagnosis. Only with this information an action plan for eutrophication control can be designed. Lake management actions should be carried out only after nutrient load studies (namely the nutrient balance in the lake) have been conducted to identify the problem from a river basin perspective. Having such information, diffuse phosphorus external loads data can be entered into statistical models associated in cause-effects relationships, relating nutrient inputs and ecosystem response, allowing a first assessment of the problem and eutrophication severity (Rast et al., 1983).

A comprehensive and robust strategy can be set only through the knowledge of the phosphorus mass balance in the aquatic ecosystem and the use of mathematical models to forecast future states of water quality due to different scenarios of environmental pressures in the watershed. Several models have been developed in past years for the modelling of river basins or to forecast different restoration measures. Watershed models can be of SWAT type (Soil and Water Assessment Tool), an open model of public domain. The SWAT model is able to predict the effect of management decisions on water, sediment, nutrient and pesticide yields with reasonable accuracy on large river basins. In aquatic systems AQUASIM (Reichert, 1994), was used by Martins et al., (2008) to model lake Furnas and Sete Cidades (Azores). The water model was based on horizontal average changes in nutrient concentrations and organism concentration that are transported in the water column by vertical mixing, sedimentation, in- and out-flows and on biogeochemical conversion processes in the water column and in the adjacent sediment layers.

It should be stressed that the river basin approach is the first step in order to minimize nutrients, organics or sediment loading into the lake, from further endangering the water quality. A lake and/or watershed management plan should identify the goals and action for

the purpose of protecting and/or maintaining the water quality conditions in a lake. Each lake management plan is different, depending on the river basin characteristics and stakeholders involved. A lake management plan is site-specific but there are issues that most should address, namely lake information (depth, size, etc) and watershed data, nutrient budgeting, water quality issues, aquatic species, including wildlife/fishery management and aquatic invasive species management/control, point and diffuse pollution control. A ten-steps lake management plan may be depicted as follows:

Step 1: Identify or prepare suitable maps and assess available GIS tools.

Step 2: Identify potential water users (abstractions, discharges, stakeholders and interests). This will give the required information about any necessary abatement measures, applications, permits, zoning issues, etc.

Step 3: Gather information from various sources when gathering data to characterize the water body and the watershed. Assess phosphorus inputs, ecosystems services, etc.

Step 4: Start to build-up decision support systems based on mathematical models, from lake models to watershed models.

Step 5: Identify the problems using a Pressure-State-Response assessment model or other similar. Therefore, after problem identification, look at its driving force and pressures, impact, location and timing.

Step 6: Prioritize the problems. All problems can't be solved at same time, prioritize the problems and solutions. In order to gain momentum, keep the public informed and seek non-governmental input.

Step 7: Look for actions and goals related to the prioritized issues. More than one action might be required, each targeted to a particular problem. Design a matrix of measures-problems.

Step 8: Estimate the economic costs and check all possibilities of funding applications.

Step 9: Prepare a SMART-type programme of measures, as well as a monitoring plan and a timely revision process.

Step 10: Allocate administrative responsibilities and define non-governmental organizations and/or users associations' roles.

Phosphorus mobility and water-sediment interactions

Even when external nutrient sources are controlled/minimized after years of P discharge in the lake, phosphorus internal desorption from the sediments to the water column may attenuate strongly, if not eliminate, any visible results derived from the external P control (Jeppesen et al., 2005; Sondergaard et al., 2007). For that reason, understanding the phosphorus mobility in sediment-water interface is fundamental to achieve its control or phosphorus recovering. The simple knowledge of the total phosphorus concentration in sediments is not sufficient to anticipate phosphorus solubilization rate and there are several factors that control, such

as temperature, pH, dissolved oxygen concentration, redox potential, nitrates, sulfates and bacterial activity, as well as the chemical structure and physical state in which phosphorus is present in the sediments (Jin et al., 2006; Gonsiorczyk et al., 1998). In order to obtain this knowledge and overcome the limited information that total phosphorus analysis in sediments provides, fractional methods of analysis should be applied to assess the phosphorus release capacity depending on environmental conditions (Ribeiro et al., 2008). The determination of the maximum phosphorus solubilization potential contributes to assess the environmental risk that the ecosystem is subject (Ribeiro et al., 2008).

Figure 6. Lake sediments sampling for fractional analysis. (Source: courtesy of D. Ribeiro).

The internal P release has been much related mainly with redox potential change in sediments due to aerobic/anoxic cycles in the hipolimnium. The concept that dissolved oxygen is the key factor to P release was proposed by Einsele (1936) and later demonstrated by Mortimer (1941, 1942). The basis behind this theory is related with the high capacity for iron (Fe) oxides and hydroxides to adsorb phosphorus and that this iron-phosphorus complex can be dissolved when the sediments are under reduced conditions due to low oxygen concentration. It is observed that hypolimnetic P concentration rises when there is oxygen depletion, giving an apparent correlation between P release and low oxygen concentration.

Efforts to preserve the Fe-P complex intact by aeration of the hipolimnium and/or destratification of the water column were applied for more than 30 years, which resulted in high operation costs, with no real improvement of the ecological status in several cases (Jeppesen et al. 2005). Indeed, some lakes that were anoxic in some period of the year released an extremely low amount of P. On the contrary, some other evidence shows that P-rich sediments may not release P during anoxia. These deviations from the theoretical statements raised doubts against the validity that “oxygen concentration is the key factor to control P release from sediments”, and that other key factors are in the role.

Focusing on the oxygen depletion there are at least two possible approaches for P release from sediments. Oxygen consumption by microorganisms during mineralization will consequently release P, while low oxygen concentration will cause inorganic P release that was bounded to Fe minerals. As both reactions occur, it is difficult to determine whether oxygen depletion is the consequence or the cause of P release. Other studies showed that the presence of sulphate and its chemical reaction with Fe can reduce the P retention capacity of the sediments. The sediments mineral composition must be taken into account as it has a strong relationship with its retention capacity. Thus, there are several other variables that can influence the P binding to the sediments, demonstrating that this is a complex problem that cannot be simplified to the statement that oxygen controls P release, leaving it valid only in some cases.

In summary, internal P load is not controlled by a single key factor, like hypolimnetic oxygen concentration. There are several other factors that influence P release from the sediments. Redox potential, pH, temperature and microbial activity may influence the direction of the P mobility either as sink or a source to the water column. Ribeiro (2008), presented a conceptual model explaining some biogeochemical interactions. In oxic conditions, Fe(III) minerals are available and SRP from OM mineralization will be adsorbed on these. Al and Ca minerals will also adsorb P if their adsorption capacity is not exhausted. In this case, as the retention capacity is high, there is a low diffusion of soluble P across the SWI. When anoxic conditions are set in the hypolimnium, Fe(III) is reduced to Fe(II) decreasing the availability of Fe-minerals and releasing P. The P that desorbs from Fe-minerals is re-adsorbed by the remaining capacity of Al and Ca mineral. While the overall retention capacity is not exhausted, SRP diffusion across the SWI will not raise despite of the anoxic period. However, as mineralization of OM continues to occur and the metallic minerals reach their maximum adsorption capacity, retention capacity of all metallic minerals will be exhausted for a certain moment. This will lead to a high net P internal load. In this case, both anoxic or oxic conditions may be represented, including lakes that are in oxic state or that were exposed to aeration and continue to have high net internal loads of P. The interaction of sulphide with Fe (that would happen in deeper anoxic layers) shows that Fe minerals availability can be reduced by this reaction.

The sequential analysis of the sediments provides very useful information regarding the P binding capacity by Al/Fe or pH impact. For P speciation, a modified scheme from Psenner and Pucsko (1988) may be used, as proposed by Romero-Gonzalez et al. (2001). This procedure allows the separation of the phosphorus Fe- and Al-bound fractions from the Ca-bound fraction. The extraction process comprises five steps and allows the fractionation of labile P (using NH4Cl as solvent at room temperature) redox-sensitive P (using bicarbonate-dithionite as solvent at 40 °C), metal oxide bound P and organically bound P (using NaOH as solvent at room temperature), Ca-bound P (using HCl as solvent at room temperature) and refractory/residual P (using NaOH as solvent at 85 °C), as shown in Table 4.

Table 4. *Sequential extraction and P-fractions.*

Solvent	Fraction	P Fractions
NH_4-CL	SRP/NRP	P labile. Phosphorus in interstitial water, unstable to surfaces adsorbed phosphates, phosphates available to algae
Bicarbonate-dithionite (BD)	SRP	Soluble phosphates under reducing conditions, phosphates bound to Fe-Hydroxides and Mn-compounds
	NRP	Organic fraction
NaOH (25 ºC)	SRP	Phosphates bound to metal-oxides (Al, Fe …) surfaces, exchangeable against OH-ions, phosphates soluble in alkaline solutions
	NRP	P in microorganisms, phosphates bound to humic material
HCL	SRP	Carbonic fractions and apatite-P
	NRP	Organic P acid sensitive
Re-P (NaOH, 85º C)	TP	Refractory organic P

Finally, it should be considered that sediment bacteria also play an active role in phosphorus mobility processes, although of less significance than geochemical processes. The iron-reducing bacteria, particularly the Geobacteraceae and Shewanelanaceae families, may reduce the iron oxides, with the consequent release phosphate into the interstitial water under anoxic conditions (Martins et al., 2009). An example of phosphorus release biomechanism (mainly organic) is the mineralization of organic matter, either through direct action of aerobic and/or anaerobic bacteria, or by the phosphatase action secreted by some denitrifying bacteria, and accumulation/release organic polyphosphates by accumulating organisms have also been mentioned as a possible match release means (Hupfer et al, 2008). However, the processes of bacterial storage of P as poly-P in oxic conditions and the release of it in anoxic conditions represents a small fraction of the P fluxes compared with those produced by the geochemical reactions.

3.2.4. Eutrophication mitigation methods in lakes

Watershed measures

There is not a fit-to-all purpose technique to reverse cultural eutrophication, a process that is called oligotrophication. The phosphorus is transported to aquatic systems by runoff, erosion and leaching from fertilized agricultural land. Therefore, the reduction of organic and nutrient loadings to water bodies can be accomplished by different policies and actions in the river basin, but all of them convergent and persistent. Oligotrophication requires time to succeed.

Removal of nutrients discharges from urban or industrial wastewater treatment plants should be mandatory if there is an eutrophication risk. Total phosphorus concentration municipal wastewaters ranges from 15 to 25 mg/L in untreated discharges. After secondary treatment provided by conventional biological processes (that is, without nutrient removal), the phosphorus concentration drop is not very significant, down to 10-15 mg/L. Only with specific P removal processes in wastewater treatment plants, a target of 1-2 mg/L may be attained. Besides, products modifications to contain lower amounts of nutrients, mainly phosphorous (detergents) have been applied in European legislation. This decision contributed to lower P levels in urban wastewaters.

Rural cropland export may vary significantly, from 2 up to 70 kg.ha¬-1year-1 in terms of total phosphorus) and in livestock excrement (in animal feeding areas total phosphorus load can be as high as 1500 kg.ha¬-1year-1). A River Basin Management Plan (RBMP) and/or EU directives can prescribe land use regulations that provide restrictive zoning in watershed to minimize intensive agriculture and fertilizers leaching to the water bodies. Public administration may also recommend restrictions on shoreline vegetation removal or new riparian vegetated borders to reduce phosphorus runoff, the use of grassy swales instead of curb and gutter drainage and strip cropping to reduce erosion are good practices.

Buffer width is an important factor in the removal of pollutants, namely phosphorus The results indicate that the design of riparian buffer zones should be conservative (Junior et al., 2015). The use of slow release fertilizers, timing and frequency of fertilizer applications, their amounts and types used, control in situ transformation of fertilizers to soluble forms, crop rotation with legumes, storage of manure during low temperature season are advices that should be provided by competent agriculture authorities. All those nutrient control strategies can (should) be applied in close cooperation with farmers and farmers associations.

When not soluble, the leaching of phosphorus is mostly connected to the transportation of matter (erosion) in the agriculture land. Therefore, a main control target is runoff and peak flow control with flow speed reduction provided by topographic pattern changes in order to provide increase water infiltration into the soil and thus less surface leaching of nutrients to the water bodies (Figure 7).

Figure 7. Land slope modification in order to promote water retention and phosphorus infiltration (Lake Furnas watershed, Azores). (Source: courtesy of J. Gouveia).

Nutrient interception in retention ponds along small rivers create artificial wetlands that are very effective to reduce solid transport. These simple civil works are made with gabions "gabiões", in portuguese that are permeable to the water flow, while providing suspended solids retention (Figure 8).

Figure 8. Retention basins in temporary rivers (Lake Furnas watershed, Azores).

The construction of a water and nutrients by-pass out of the lake's water body is also an interesting technique when land uses and fertilizer practices can´t be modified significantly (Figure 9). This approach was very valuable to minimize nutrient load input in Sete Cidades lake.

Figure 9. Water and nutrients by-pass with a retention basin in Sete Cidades lake, Azores (Source: courtesy of D. Pacheco).

3.2.5. Internal measures

The reduction of the external load down to ecosystem carrying capacity is the key point and river basin management plans (RBMP) and/or lake watershed plans (LWP) plays the decisive role, as already mentioned. However, the pattern in the lake may have been so dramatic changed that turning off the nutrients and organic loadings may be not sufficient to restore water quality and ecosystem structure in a specific time. In that case, internal lake restoration techniques should be, or must be, used also.

Several methods were developed to reduce eutrophication processes ranging from dilution, artificial destratification, hypolimnetic aeration, dredging or coagulation/flocculation, as depicted in Table 5. Among other techniques that were also applied are biomanipulation and

weed control or precipitation with aluminium sulphate, sediment capping with clay, aluminium oxide and lanthanum oxide based materials ((Søndergaard et al., 2003; Akhurst et al., 2004). With the exception of the biological processes, most measures always have the objective of trapping P in the sediments, aiming at leaving it (almost) permanently inaccessible.

***Table 5**. Advantages and drawback of selected internal measures for (internal) eutrophication mitigation*

Method	Advantages	Drawbacks
Artificial recirculation and hypolimnetic aeration	Disrupts or prevents stratification Increases aerobic habitat Maintains oxygen in hypolimnion Increases habitat and food supply	Costly (investment and maintenance) Does not decrease algal biomass May increase water fertilization May decrease water clarity Adverse impact on cold-water fish No effect on macrophytes Difficult to supply adequate oxygen
Biomanipulation	Encourages zooplankton growth	Not effective if blue-green algae dominate. Only feasible when phosphorus levels are rather low Insufficient fish removal
Chemical biocides (copper sulphate)	Reduce suspended biomass	Not specific Cyanobacteria release toxins Toxic to fish and other species
Coagulation/ flocculation and precipitation (aluminium sulphate)	Lowers lake phosphorus content Lowers suspended solids and increases water column transparency	Temporary measure Potential toxic impacts (aluminium dissolved in abstracted waters) Increased macrophyte growth due to water clarity
Dilution with low nutrient water	Reduces nutrient levels Washes out surface algae	Only feasible in small ponds Requires large volumes of water Does not eliminate sources of phosphorus from sediments or watershed
Dredging	Reduces P in sediments Increases lake volume May improve water quality	Only in artificial ponds Temporary resuspension of sediments Temporary destruction of habitat Disposal concerns High cost
Hypolimnetic discharge	Reduce P load near the sediment zone	Requires good P monitoring and control for optimization
Macrophytes harvesting	Remove infestation	Can destroy fish nurseries
Passive or active sediment capping agents	Hampers phosphorus release from lake sediments	(Most) has no phosphorus recover potential No P specific (in some cases) Time dependent

Some of those internal processes to restore water quality and to prevent eutrophication processes are further described below:

3.2.6. Destratification by air injection

A thermal stratified water body is characterized by horizontal, iso-density strata. In general, these lake strata are thickest in the epilimnion and hypolimnion since these zones are almost uniform in temperature. The metalimnion is the zone of rapid change in density and is characterized by narrow iso-density strata. Several techniques are available for artificially destratifying lakes. Most lake techniques apply compressed air injection (diffuse air) near the reservoir deepest points. Air injection should be applied in order to optimize hydrodynamics, favouring water body mixing and homogenization. The oxygenated water recirculation can reduce the phosphorus internal load that may result from the P solubilisation from sediment iron metal bounds, as mentioned before. In addition, algae have less available light during daily time, which is detrimental to their growth. Further, buoyant algae are subject to rapid changes of hydrostatic pressures, conditions that are known to disrupt cyanobacteria cells.

Aerating the hypolimnion without affecting stratification appears to possess the beneficial qualities of destratification systems without some of the more deleterious (the nutrient fertilization of the whole lake). Such lakes have suitable hypolimnion water temperatures, but lack adequate oxygen concentrations. Ejectors are used to suck air from atmosphere and inject an enriched air/water mixture into the water. The ejectors are driven by a submersible pump and spread the air water mixture inside the input pipe of the plant. A discharge of water enriched in phosphorous from the hypolimnion is an option also. The bottom discharge may have a pipe in the lake and the return to the channel. Flow by gravity (if necessary with the help of a pump to prime the hydraulic circuit.) This measure may be related to the installation of a hypolimnion aerator which helps to prevent any odours in the discharge.

Chemical coagulation/flocculation and precipitation

The phosphorus inactivation agent is typically aluminium sulphate (alum) which in neutral to alkaline water produces a gelatinous precipitate of aluminium hydroxide, Al(OH)3, which is chemically relatively stable, even under the low redox states under anoxic conditions. However, chemical precipitation with aluminium, iron and calcium salts added to the water column are adequate to remove P but only when concentration is high. Therefore, chemical precipitation techniques should be supported by jar-test to determine the most appropriate precipitant dose, using chemical constituents already mentioned (aluminium salts). External phosphorus precipitation is also possible in small lakes. It should work is a closed loop system and a phosphorus precipitation process using calcium hydroxide can be designed, integrating rapid mixing, decantation, filtration and pH correction, followed by the water return to the lake.

Active sediment capping agents

Inhibition of P release from the sediment can be promoted by an active barrier system containing a reactive component immobilizing P. Calcium compounds, composed largely of calcium carbonate in the form of calcite, present a phosphate removal capacity that is, in general, lower than those of aluminium and iron compounds. Carbonate and phosphate ions compete for calcium, which inhibits the precipitation of calcium phosphate. Clay minerals have a tendency to disperse into very fine particles when applied to the water, which may increase turbidity for long periods. Activated alumina has a very high surface area, is insoluble in water and is selective for P in the presence of chloride and sulphate ions.

Biomanipulation

The biomanipulation is the deliberate modification of the trophic chain and corresponding food chain in order to achieve a desirable or planned environmental change. A large number of practical experience over the decades shows that carps are detrimental to water quality. Their benthic mode disturbs and resuspends the fine sediment and affects water transparency. The fish excretory processes favour nutrients recirculation. The growth of macrophytes is impaired by this reduction in transparency and also the excavations that carp do to feed and reproduce. For these reasons, the elimination or drastic reduction of fish density may lead to a significant water quality improvement, but a period of 5 years is advisable to evaluate the results (Hansson et al., 1998). The lake ecosystem biomanipulation has been widely practiced in many countries as a measure of rehabilitation. However, experiments in European lakes demonstrate that the P values from which the biomanipulation is useful should be less than 50-100 micrograms per litre of total phosphorus.

Biocides

The phytoplankton blooms abatement with copper sulphate algicide is generally successful, but only for short periods of time. It is foreseeable that of large quantities of algal cells deposition over the sediment and subsequent decomposing will foster oxygen depletion, which will enhance phosphorus release from the sediments and boost the formation a new bloom algal. The dosing of algicides can be 10 to 100 times over the lethal dose required for other non-target organisms (Cooke et al , 1993). In short experiments performed in the Azores, it was found that the phytoplankton group most affected by the biocide was Cyanophyta, namely Microcystis sp. However, according to Cooke et al (1993) the treatment with copper sulphate is not advisable in blooms removal caused by Microcystis aeruginosa because intracellular neurotoxins are released into the water.

Macrophyte harvesting

Macrophyte harvesting is provided by floating mechanical equipment. The biological material should be deposited on local banks and subsequently collected, avoiding decomposition and any nutrients return to the water body. The dispersion of algae fragments should be avoided, because of biological re-infestation (Figure 9). The weeds removal is not without negative impacts because it can promote riparian erosion and have negative consequences on communities' structure.

Figure 9. Macrophytes removal in Sete Cidades lake (Source: courtesy of D. Pacheco).

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6. Study cases

6.1. Study case 1: Restoration of an agricultural drained peatland: the case study of the Massaciuccoli Lake Basin in Tuscany (Italy)

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Background and aims

From last century, the deep drainage of peatlands due to the increasing land demand for agriculture and forestry as well as the need to improve sanitary conditions (e.g. malaria) has led to remarkable changes in the structure and the equilibrium of those ecosystems.

The dehydration of peat body has determined a series of consequences on the physic-chemical properties of peat (Litaor et al.,2008) such as (i) acceleration of organic-matter oxidation, with a consequent increase in greenhouse gases (GHG) emissions into the atmosphere of up to 25 t CO_2 equivalent $ha^{-1}y^{-1}$(Wichtmann and Wichmann, 2011); (ii) increase in NO_3^-concentrations in porewater due to higher oxygen availability and the consequent mineralization and nitrification of organic N (Tiemeyer et al., 2007) and (iii) mineralization of organic P compounds and increase of absorbed and Fe-bound P pools (Zak et al., 2004). The continual recurrence of these phenomena has negatively affected the land, for example, progressively lowering the soil level (subsidence), increasing nutrient loads delivered to receiving water bodies (eutrophication) and decreasing ecosystem biodiversity and functionality (loss of ecological stability), especially in land reclamation districts (Tiemeyer et al., 2007; Wichtmann and Joosten, 2007; Verhoeven and Setter, 2010).

For these reasons, rewetting of drained peatlands has been identified since the mid-1990s as an important mitigation strategy to reverse this self-perpetuating process, which is definitely unsustainable (Erwin, 2009).

Our case study is the Massaciuccoli Lake floodplain, located in the Natural Park of San Rossore, Migliarino and Massaciuccoli, which is one of the most important residual coastal marshy areas of the Tuscany (Italy).

The basin of Massaciuccoli Lake (central-western Tuscany, Italy; 43° 49' 59.5' North latitude; 10° 19' 50.7' longitude East) extends over an area of 114 km^2, delimited to the north by the river Camaiore, to the east by the reliefs of d'Oltre Serchio Mountains, to the south by the river Serchio and to the west by the Ligurian Sea (Figure 1).

Figure 1. The hydro-graphic basin of Lake Massaciuccoli (in yellow).

The lake extends over about 13 km^2 in a large depression of a depth varying between 1.0 and 2.5 m (Amos et al., 2004) and contains a water volume of about 14 Mm^3 (Autorità di Bacino del Fiume Serchio, 2007), fed by modest tributaries characterized by a pluvial regime and descending from the hills in the east, and by water from neighbouring soils, drained over centuries and now largely used for agriculture.

From an administrative viewpoint, the basin lands extend over the provinces of Lucca (the municipalities of Massarosa, Viareggio, small parts of Lucca and Camaiore, up to and including the hamlets of Quiesa, Bozzano, Massaciuccoli, Piano del Quercione, Piano di Mommio, Montramito and Torre del Lago) and Pisa (municipality of Vecchiano, including the hamlets of Vecchiano, Nodica and Migliarino). The residential population is estimated to be around 46,700 (Pagni et al., 2004).

Two water depuration plants are located in the hydro-graphic basin, closed to the most built-up areas of two local municipalities (i.e. Migliarino, ca. 4250 AE, and Vecchiano, ca. 8500 AE), which discharge their effluents to the lake. Regarding businesses within the district, since the 1960s, there has been an important industrial area, predominantly manufacturing, which is currently expanding. The basin is also crossed by a nationally important network of roads and railways, such as the Genoa-Rosignano highway, Florence-mare highway, Lucca-Viareggio highway, the Aurelia, Genoa-Pisa and Lucca-Viareggio railways, and other secondary and local roads.

Since the 1930s, a large part of the Massaciuccoli floodplain has been drained for agricultural purposes. To ensure a water table depth suitable for cultivation, a complex network of artificial drains and pumping stations has been used to drain the superficial aquifer and rainwater. In the drained areas, cultivated peat soils (autri-sapric and endo-salic histosoils), with values of organic matter reaching up to 50% in some cases, are present (Pistocchi et al. 2012). Agricultural activities cover 40% of the district, thus testifying the importance of such activities for the territory today. Professional farms, located nearby the lake, are mainly of an average-

large size (50-70 ha) and aimed at cereals and industrial productions, maize being the central crop, followed by winter cereals and sunflower. Horticulture and olive trees are also significant, although only in limited areas. Non-professional farmer-managed farms are of a small size (3-5 ha), and located in parcels near town centres. The main activities include family-run gardens and orchards (mixed crops), and small leguminous cultivations (grain and/or forage crops), or winter-cycle horticultural crops (spinach, cauliflower, etc.) (Figure 2) (Silvestri et al. 2012).

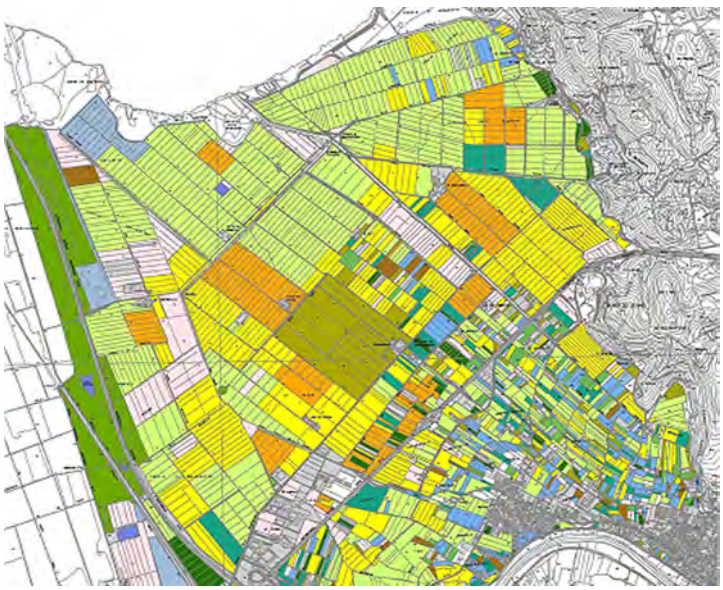

Figure 2. Year-2008-land-use in a section of the southern side of the hydrographic basin of the lake.

As a consequence of the agricultural land use and the drainage-based management, several environmental concerns arose in the last 50 years. The most important issues are those related to:

I. eutrophication of the lake due to nutrient enrichment (N, P) in the surface and groundwater. Indeed, from the 1970s, the lake, from an initial oligotrophic status, progressively turned into an eutrophic/hypereutrophic system;

II. the subsidence rate (2-3 m in 70 years) due to compaction and increased mineralization of peat. This process, started since land reclamation, left the lake perched above the drained area, which is now 0 to 4 m below the sea level (Rossetto et al. 2010).

Our project, started in 2011, aimed to evaluate the suitability of coupling rewetting with phyto-treatment systems as the solution for improving water quality, and slowing down soil organic matter (SOM) mineralisation rate, and, therefore, a method to restore the former ecological functions of this site. A pilot experimental field of 15 ha was set-up and three different management systems, with increasing anthropogenic impact, has been tested (Figure 3): constructed wetland system (CWS), paludiculture system (PCS) and natural wetland system (NWS). This implies a gradient in regulation of water regime (from a strongly controlled system to an almost-natural rewetting), plant communities (from cultivated to native communities) and harvesting strategy. The soil-plant continuum systems are expected to reduce nutrient loads.

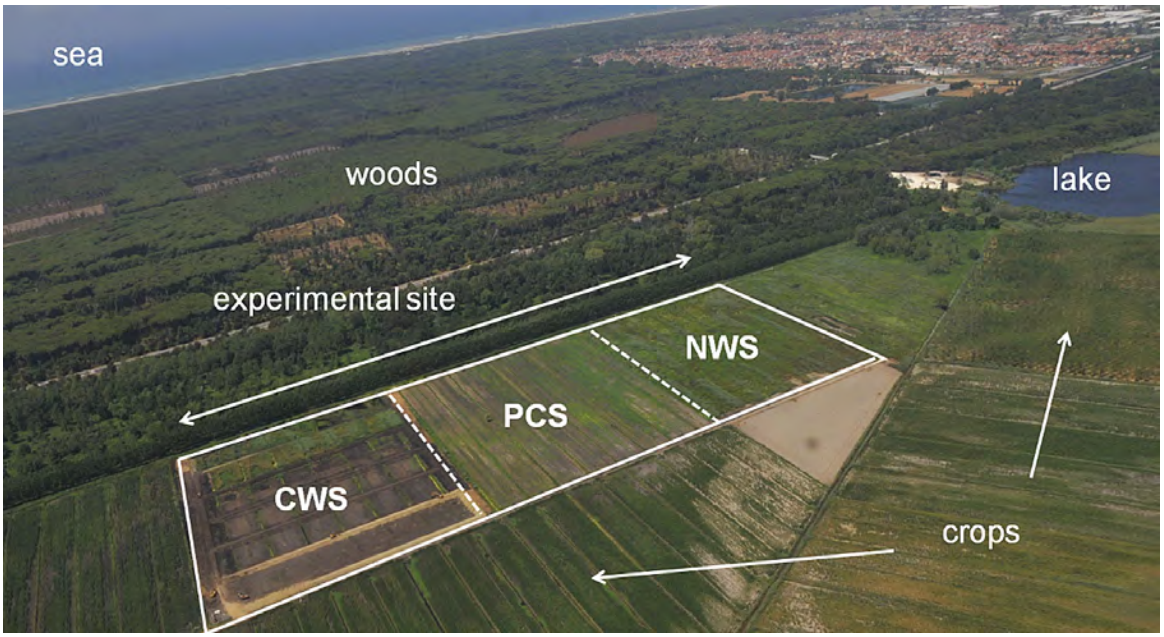

Figure 3. Aerial view of the experimental pilot field represented by three different management systems: constructed wetland system (CWS), paludiculture system (PCS) and natural wetland system (NWS). The conventionally drained area cultivated with annual crops is near the pilot field.

Strategies

A 15 ha experimental area located in Vecchiano, within the boundaries of the Natural Park of Migliarino, San Rossore e Massaciuccoli was used to compare the efficiency of three different approaches for the peatland rewetting and agricultural drainage water phtyto-treatment.

i) The CWS is composed by a series of 5 treatment cells linked each other among which the drainage water is treated flowing in a serpentine path. Each treatment cell, with a parallelogram shape, has a surface of about 6200 m^2 and a depth of ~ 40 cm.

The CWS was sized to guarantee a retention time of 1.0-1.5 day. At the end of the construction works lasted two years (up to 2012), the CWS started to operate and the spontaneous vegetation started to develop within it.

The settlement of the spontaneous vegetation was possible because the area was unvegetated before the construction works, thus some rhizomes, stolons and seeds were still present in the treatment cells.

ii) The PCS was based on growing different no-food crops and harvesting their biomass periodically to ensure nutrient removal from the fields. The system was not dammed and was crossed by a dense network of small channels (about 8 m apart) that supplied both drainage (in autumn and winter) and irrigation (in spring and summer) for the crops through lateral infiltration. The main difference between the PCS and the surrounding areas concerns the water table level. The water table in the watershed is artificially lowered to allow the farmers to cultivate, with noticeable fluctuations during the year (from −0.10 to −0.60 m) depending also on cultivation practices. In the rewetted PCS, the water table depth is kept higher because of the continuous

supply of water to be treated and thanks to the weirs, which are not moved except for management needs (e.g., harvest or maintenance of drainage ditches). Then, in the experimental fields, the water table depth is only dependent on the meteorological conditions (e.g., rainfalls or dry periods), ranging from 0.00 to −0.05 m during the winter and from −0.10 to −0.25 m during the summer (Giannini et al., 2017).

In this system three perennial rhizomatous species (*Arundo donax* L.; *Phragmites australis* (Cav.) Trin ex Steud.; *Miscantus* x *giganteus* Greef et Deuter) and two woody crops managed as short rotation coppice (*Salix alba* 'Dimitrios', *Populus* × *canadensis* 'Oudenberg') with a biennial turn have been planted (Figure 4).

iii) The NWS was set up as natural rewetted area with a surface area of 2.7 ha and surrounded by small embankments built with the top soil (~10 cm) removed long the area's borders. Natural elevation changes within the NWS helped in creating zones with a different bottom height in order to promote the colonization from a large variety of plant species.

The effectiveness of the different systems was determined following a multidisciplinary approach evaluating the status and changes of water, soil and plant communities' biodiversity (Figure 5).

About water, a complete hydrological monitoring has been performed starting with acquisition of meteorological data and measuring inflow and outflow for each phyto-system, as well as exploring the relationships between the superficial and deep aquifer (information on infiltration and/or exfiltration processes) and evapotranspiration. These data have been collected in order to have spatially-distributed value to be used in the unsaturated/saturated zone hydrological model. Since the system is a coastal one, the presence and influence of saline water on functionalities has been also evaluated. Three sampling schemes have been followed: i) continuous monitoring with probes placed *in situ*; ii) a composite sampling with automatic samplers proportional to the flow pattern; iii) an instant discrete sampling for groundwater. The monitored parameters are the following:

- continuous: pH, oxygen, temperature, electrical conductivity (EC);
- composite: EC, total suspended solids (TSS), total phosphorus (TP), total nitrogen (TN, Kjeldahl), nitrates (NO_3^-), ammonium (NH_4^+), dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), anions, cations.

About soil: a set of the soil physic-chemical properties (e.g. pH; EC; TN and available nitrogen (N_{avail}); NO_3^- and NH_4^+; TP, available and organic phosphorus (P_{avail} and P_{org}); C:N ratio; SOM and soil texture) has been measured in order to assess the soil status before rewetting and once per season in each of the three systems.

About the cropped species (in PCS) , these have been monitored and harvested for recording biomass production, above-ground nutrients uptakes and determination of the technological parameters for energetic purposes (combustion, bio-methane production).

In CWS and NWS, to follow the restoration process over time (2012-2016), a mixed approach merging phytosociological surveys with ortophotos taken by an Unmanned Aerial Vehicle (UAV) was used. During the last year of observation (2016), a destructive sampling on the

most widespread plant communities in the areas (*Phragmites australis, Typha angustifolia* and *Myriophyllum aquaticum* community) was performed in order to quantify the biomass production and the uptake of nitrogen and phosphorus (Giannini et al., 2018).

Highlights and outcomes of restoration practice

Although the above described experience has a mainly experimental value, the realization of the phyto-treatment plant proved the suitability of this solution and the data on the nutrient abatement showed the efficacy of this approach. All three systems can reduce significantly the nitrogen and phosphorous content in the treated waters before these are delivered to the lake. The obtained results are very changeable over time in relation to the meteorological conditions, the hydraulic loading rate, the nutrient loads in waters. For this reason, the observed abetment levels showed a large range of variability, from 20% up to 60%.

However, the choice of the best strategy is strongly affected by other features in addition to the depuration efficacy, such as the operational capacity of each system (that is the water volume treated per the time and per area unit), the possibility to derive an income from the phyto-treatment activity (for example through the energetic conversion of the biomass production), the costs and the impact that a large adoption of one the system tested could imply on farmers and on the current land management. All these aspects should be take into account and it is likely that the best solution may not be the same for the different areas of the Massaciuccoli Lake basin.

Naturally, no action, not even phyto-treatment, can alone solve the problems of such complicated area. Improvements in the organization of agricultural activities in the context could help to reach better ecological conditions in the surrounding areas, thus improving their level of sustainability. Rational use of fertilizers, adoption of minimum- or no-tillage techniques, more diversifies and long rotations including intercropping (cover crops, catch crops, etc.), creation of grass or woody buffer areas at the edge of the fields are just examples of how farmers can contribute in achieving important results throughout the territory, without the need for too costly actions. Agriculture could play an important role in reducing the impact of phosphorus pollution in the Lake Massaciuccoli. The definition and subsequent introduction of agricultural models to contain the load of nutrients coming from cropped fields could be integrated by acting to reduce the concentrations of phosphorus detected in drainage network before that water is delivered to the Lake (use of floating plants, the gentle maintenance of the drainage network, etc.).

The chances that these solutions are effective in solving Lake's problems depend on their level of coherence and complementarity as well as the involvement of all stakeholders present in the area. The proposed solutions have to be accepted and shared by the communities involved. Only a frank and constructive dialogue will make possible identify and solve technical/financial difficulties defining. A wide expansion of such actions, which currently go against current farming practices in the district, may require appropriate financial support in order for such actions to be adopted and managed efficiently.

Figure 4. Willows in the PaludiCulture System.

Figure 5. The Natural Wetland System.

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6.2. Study case 2: An eutrophication case study of a lake in Azores (Portugal)

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Public concern over lake eutrophication started in Azores as early as 1985. Since then, several measures and programmes were developed in order to control eutrophication processes in Sao Miguel, which is the largest (759 km^2) and most densely populated (140,000 inhabitants) of the archipelago of Azores (Portugal). São Miguel Island is also called ''The Green Island'' which is its dominant colour, due to the fact that the area is occupied by highly fertilized grass fields crossed by small rivers of temporary flow. Their flows depend mostly on the rainfall, which is often very intense leading to significant soil erosion and to fertilizers leaching into the lakes. Besides the nutrient input due to agriculture, secondary manifestations of active volcanism may also contribute to water composition in the studied lakes, through seepage of thermal waters or gaseous compounds.

These lakes correspond to the major active trachytic central volcanoes of Fogo, Sete Cidades and Furnas, linked by rift zones. During the last 5000 years the activity of these three active central volcanoes is shown by 57 volcanic eruptions. For the whole island of Sao Miguel it is possible to estimate an overall erupted volume of 400 km^3 in the past four million years. The majority of these eruptions presented were explosive, resulting in pumice deposits, of acid character, that cover the volcanoes flanks. Therefore, sediments in the bottom of the lakes present a close relation with these pumice deposits.

The eutrophic shallow lake Furnas is located in East side of the island. The Sete Cidades lake is located in the Western part of Sao Miguel and it is composed by two interconnected lakes, the so-called Azul and Verde lakes. The Azul lake is at meso-eutrophic state but the Verde lake is already classified as eutrophic (Figure 1). Fogo lake is meso-oligotrophic and is located at a high altitude, in the center of Sao Miguel.

Both lakes shown on Figure 9 are in a state of eutrophication due to the nutrient load from intensive fertilization and livestock manure that reaches the lake from the watershed. Consequently, algae blooms and the release of phosphorus from sediments during summer are recurrent events (Santos et al., 2004; Medeiros et al., 2004). To reduce the external input of nutrients, the Regional Government has designated the lake watershed as "vulnerable area" complying with Nitrates Directive 91/676/EEC and a watershed management plan was approved in 2005. The morphometric and geochemical characteristics of the lakes are presented in Table 1.

Figure 1. Lake Furnas and Lake Sete Cidades (Azul and Verde).

***Table 1**. Lakes Sete Cidades, Furnas and Fogo characterization*

	Lake Furnas	Lake Sete Cidades	Lake Fogo
Physical description[b]			
Surface Area (Km^2)	1.9	4.35	1.48
Max. Depth (m)	12	33	30
Volume (m^3)	9 212 500	47 760 500	18 040 800
Volcanic setting	Caldera	Caldera	Caldera
Chemical and trophic parameters[a]			
pH[b]	7.36	7.46	6,68
EC ($\mu S\ cm^{-1}$)[b]	142.78	113.67	47.50
HCO_3 ($mg\ L^{-1}$)[b]	49.01	28.28	6.20
Ca ($mg\ L^{-1}$)[b]	2.88	1.43	0.48
Mg ($mg\ L^{-1}$)[b]	2.27	1.32	0.80
Dissolved CO_2 ($mg\ L^{-1}$)[b]	10.33	5.98	2.97
NO_3^- ($mg\ L^{-1}$)	1.04	1.33	0.42
SO_4^2 ($mg\ L^{-1}$)	22.7	10.2	12.1
Soluble reactive phosphorus ($\mu g\ P\ L^{-1}$)[c]	11	3	–
Total phosphorus ($\mu g\ L^{-1}$)[c]	45	21	–
Chla ($mg\ m^{-3}$)[c]	17.85	10.43	2.73
Secchi disc transparency (m)[3]	0.8	2.3	–
Trophic classification	Hypertrophic	Meso-Eutrophic	Ologotrophic

[a] Variation can occur by seasonal changes.

[b] Data from Cruz and França (2006).

[c] Data from water monitoring network from regional government.

Among those lakes, the current eutrophication status in Sete Cidades is in between Furnas and Fogo ones. Temperature, dissolved O and PO_4^3 concentration profiles along the water column indicate that this Lake undergoes seasonal thermal stratification that extends between May and November, otherwise the lake is completely mixed. The surface water temperature varies from 14 °C in February to 22 °C in July. During stratification, the temperature difference between the surface and bottom of the lake is around 4 °C in May and 7 °C in July. The dissolved O_2 profiles show that O2 concentration at the lake bottom was very low in July due to stratification. At the lake surface the lowest value of O_2 concentration (7.8 mg/L^{-1}) was recorded in November with the highest one (9.7 mg/L^{-1}) in February. During summer, both temperature and dissolved O_2 values simulated at the lake bottom (approximately 15 °C and less than 1 mg/L^{-1}) were lower than those at the surface (approximately 22 °C and 9 mg/L^{-1}), consistent with the stratification conditions occurring during this period. Under anoxic conditions at the lake bottom, P was released from sediments into the hypolimnion and accumulated there due to mass transfer limitations between the epilimnion and hypolimnion. Phosphate concentration at the lake bottom was 88 µ/L^{-1} P in November compared to 7 µg/L^{-1} P in May. In general, species belonging to the diatom and cyanobacteria groups dominate the Lake Verde phytoplankton. The community was mostly composed of *Asterionella formosa*, *Fragilaria delicatissima*, *Fragilaria crotonensis* as diatom species and by *Woronichinea naegeliana*, *Microcystis flosaquae* and *Aphanizomenon flosaquae* as cyanobacteria species.

As an example of a decision support systems, prospective scenarios for water and ecological quality assessment of Lake Verde of Sete Cidades during the next 10 years were developed: the scenario control, the scenario watershed management, the scenario sediment management with P-immobilization and the combined scenario watershed plus sediment management. The studies that were carried out shown that a calibrated mathematical model can be used to support the decision-making processes in aquatic restoration programmes. The water quality model indicated that watershed and sediment management was the only responses that could achieve a future lake restoration. In addition, the P-fractionation shows that in Lake Furnas and Lake Sete Cidades, the NaOH extracted P is the dominant fraction, which contribute with more than 50% to total sedimentary phosphorus. The Phosphorus Maximum Solubilization Potential indicates that for the more bio-available P-fractions in Furnas and Sete Cidades lakes (that is, P from the first and second extraction step), are very high and the eutrophication risk is very significant. Therefore, external and internal measures are mandatory for water protection in Azorean lakes and the regional water administration is facing the eutrophication challenge with external and internal protection and restoration measures.

6.3. Study case 3: Morphological stabilization of Estuarine Banks after dredging (river Lima, Portugal)

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Background and aims

The purpose of this study is to analyze the erosion phenomena in the estuary of the Lima river, as a result of decades of unregulated gravel extraction, causing profound effects on bank morphology and the destruction of riparian vegetation, either due to tidal action or in situations of peak flow. However, the attempts to stop the progressive bank cutting have been fruitless, leading to substantial marsh losses, affecting also recreational activities. The present work shows the implementation of a restoration project, combining civil engineering with soil engineering procedures and the biophysical recovery processes along two distinct segments in the right bank (nearby Cardielos and Portuzelo villages), both in the estuarine zone of the Lima river. We must emphasize that the downstream part of this river, including the estuary, are included in a protection area of Nature 2000, associated to the preservation of wetlands and riparian layers, therefore this work is crucial to define the procedures for a more extensive action. We hypothesize that, by increasing the hydraulic roughness along the banks, we may increase the sedimentation rate along the banks and favor the recolonization process.

Lima river catchment is shared between Portugal and Spain, and the run-off average flowing into the estuary is 3298 hm^3, whereas 1598 hm3 corresponds to the Portuguese part (which includes near 35 kms length, with an average slope of 0.1%). The downstream part of this river represents a transition between a narrow and steep valley towards a progressive gentle slope (0.024%) along a shallow-vee valley form and finally into a large floodplain. The high rainfall favors the occurrence of frequent floods in the downstream areas of the main catchment. Impacts on water quality are relatively low, since we observe a dominant land use of forest stands (eucalyptus and pine trees) and in the lower parts of the valley an extensive agriculture characterized by small patches of vineyards, orchards and grasslands with cattle breeding. The areas prioritized for the rehabilitation projects Cardielos and Portuzelo, both on the right bank, were a consequence of the demanding of the municipalities, due to the loss of recreational grounds and the increasing pressure on multiple infrastructures (marginal roads, sports and leisure equipment, etc.), but also to protect a layer of marshes in the neighborhood.

Figure 1. Aerial photographs of a part of the R. Lima estuary with the localization (yellow and red) of the two considered priority segments for intervention: Cardielos and Portuzelo. Left and right are respectively images from1965 and 2010, and comparing both images it is visible the consequences of gravel mining in the morphometry.

The intense dredging related to gravel extraction along near 3 decades in the lower segment of Lima river led to a complete change in the morphological character of the river mouth, with the main current flow and talveg relocated towards the right margin derived from the intense withdrawal of sediments. Moreover, the sedimentary supply dramatically decreased after 1992 due to river regulation (two tail-race dams where built upstream: Alto Lindoso and Touvedo, The 1st one is the second most important hydropower system in Portugal, with a dam 110 m height and a reservoir capacity with 347.8 hm3, with a maximum area of 1072 ha, whereas Touvedo dam, 7 km below, has 43 m height and the reservoir covers 172 ha. Bank instability is the direct consequence of the deepening of the river channel and the exceedance of the critical height of the river banks, which lead to its subsequent collapse. We may notice in Figure 1, that compares the studied area (upper part of the estuary), between 1965 and 2010, that shows the intense sediment loss in this period and the transformation of a braided channel into a progressive linearization of the river banks, resulting a river with a significant higher energetic power. Damming is known to affect the entire downstream segments by trapping sediments and reducing the sediment transport capacity because of the strong reduction of peak flows. Downstream geomorphic and ecological effects of dams are largely determined by the relative changes in the sediment transport regime with consequences on channel incision and bank instability (Schmidt and Wilcock, 2008; Tena et al. 2011; González del Tanágo, 2015).

Strategies

The survey carried out in situ and by aerial photography led us to take into account that interventions with long-term purposes could not be limited to the consolidation of banks, but that they also should modify hydrodynamics because of the continued process of excavation of the lower layers of the bank. These ones display a complex structure, with a less cohesive layer at the base, affecting the stability of the entire bank. These aspects are similar, both in Cardielos (Figure 2), where the progressive erosion lead further to the collapse of the riparian layer.

Figure 2. Cardielos (left panel) and Portuzelo (right panel) sections in 2010, previous to the bank reinforcement, where is apparent bank collapse.

There were a multiple purposes in this project besides bank protection. We are aware that some of the civil engineering techniques used may have negative aesthetic implications, so there was a concern about visual mitigation; besides, the rehabilitation should improve the riverine habitats, allowing a revegetation process towards a riparian gallery. Therefore, we integrated in this rehabilitation different concept of bank protection and hydrodynamics processes through the selection of convenient engineering techniques, with the additional purpose to stop the irretrievable loss of an area of high biodiversity (wetlands/marshes) and marginal leisure infrastructures. The promotion of hydraulic roughness to increase progressively accretion (and, indirectly, salt marshes recover) was inherent to the project conception.

Cardielos section

The project was developed along two temporal phases carried out between 2011 and 2013. It was defined a 1st set of six triangular groynes, which were located in the most eroded segment. Afterwards it was built a 2nd layer of groynes, closer to the water level, of smaller dimension, which were further disposed along the same segment to complement and strengthen the 1st layer. This field of 2 lines of groynes, implanted along approximately 1km, is then a group of structures that act together with the objective of causing the water to flow some distance from the riverbank and to increase hydraulic roughness. A groyne roughness the bank on which it is constructed and in doing so, creates a zone of lower flow velocity where the tendency for erosion is less and deposition greater. Typically eddy currents form in the pools between groynes where the water flows upstream along the bank (Alauddin and Tsujimoto, 2012). They are wall-like structures, perpendicular to the flow direction and pointed towards the edges where the nose of the groyne is gently sloping.

Both set of layers were built with rip-rap material, whereas the 2nd line vanishes their visual impact since it is below the waterline in high tide. This group of structures (Figure 3) includes granite rocks 0.5-0.8 m diameter packed in a layer thickness between 1.5-1.9 m and originates structures ranging in length from 13-29 m, depending on the topographical

conditions were are implanted. The 2nd set (closer to the water) was composed by material with similar diameter, but packed around an axis of material with small grain size (20-30 cm) and disposed over synthetic matt. Because this set was placed in a plane more exposed to tidal and river flow energy, it was planned in order that the foundation of the structure was disposed at a level close to the depth of expected scour, where this level was indicated by a careful observation along this section of the river. The defined layout (straight in plan and perpendicular orientation) allows then that the set of groynes may trap moderate amount of sediment upstream and downstream, keeping current more or less parallel to bank and they offer a medium potential for scour at head. Both sets of groynes were rooted successively in a set of structures, which are mentioned successively from the water level: a) rip-rap between 3-5 m long; b) gabion mattress (placed on a gravel layer after shaping the bank), which is characterized by wire basket filled with rock (covered with 20 cm of soil for planting and wire mesh to decrease tidal washing); c) vegetation roll and willow fascine; d) gravel layer with soil (40 cm) covered by a tridimensional geomats, after bank reprofiling (Figure 3).

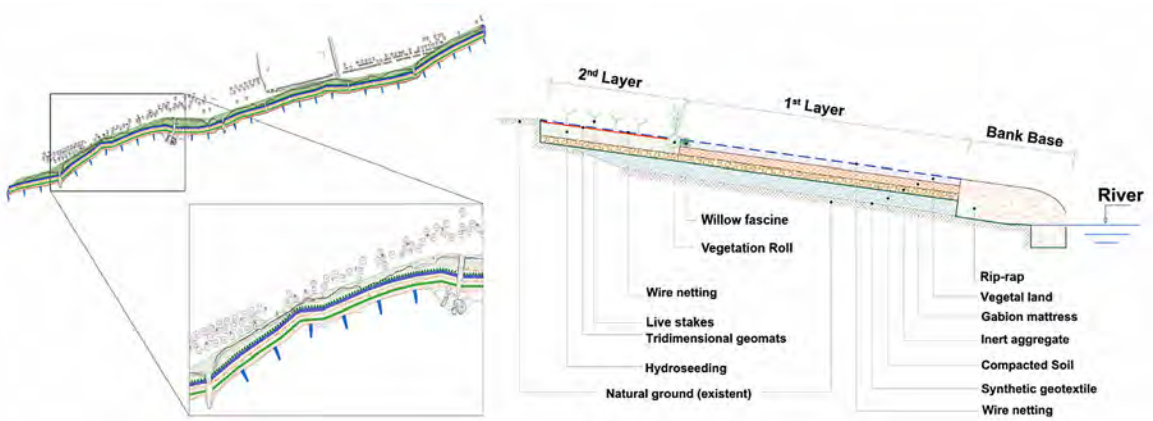

Figure 3. General view of the intervention area at Cardielos site (left, view from above, which highlights the row of deflectors and groynes and the bands dividing each intervention; right, transversal section exhibiting in detail the techniques used in each band.

Finally, the described structures were vegetated with autochthonous hygrophytic and salinity resistant herbaceous and woody species, like reeds and rushes combined with semi-halophyls macrophytes and salinity-resistant shrubs (*Juncus maritimus*, *J. acutus* or J. *effusus, Typha angustifolia, Phalaris arundinacea, Agrostis stolonifera, Scirpus maritimus, Festuca arundinacea, Phragmites* spp, *Tamaryx tamaryx, Carex* sp. or *Najas* spp). In the layer described in c), over the geomats, it was also conducted an hydroseeding since it is a space intensely used by visitors.

Portuzelo section

This part the estuary, downstream of the previous section, displays a higher vulnerable condition, which lays on the fact that the lowering of the estuarine bed caused by dredging reached a considerable higher depth close to the banks (channel deepening reached here 4-6 m). The techniques designed and implemented for this area, with approximately of 150 m length, are schematized in the profile shown in Figure 4. Again, the objectives, besides the bank stabilization, were to allow for the settlement of vegetation and to increase the

roughness on the submerged bank in order to trap sediments and to dissipate the energy from river flow and tidal dynamics, contributing for long term sustainability. Besides, this bank constitutes a barrier that protects a large salt marsh, therefore it is a crucial the permanence of this defense for the preservation of this sensitive environment. As Figure 4 shows, from the base to the top of the bank different layers were successively considered: a) rip-rap with large boulders (0.6-0.8 m) in a foundation frame of wood piles, with stakes driven into riverbed (since the river depth was higher when compared to the previous section) in order to promote roughness (groynes were not considered because of water depth); b) bank reprofiling to smooth the slope, which was covered with a layer of geogrids, filled in the lower part with gravel (for adequate infiltration) followed by soil and further vegetated, where reeds were disposed near the base and woody vegetation (mainly *Tamaryx* sp.) was planted in the upper layer, and finally the disposal of a wire mesh to decrease the potential tidal washing; c) top lining and plantation with willows species as well as of a row with broadleaf trees to improve the landscape attractively for visitors and to increase the overall bank consolidation. Besides, it was installed drains to allow the water to flow between the estuary and marshland.

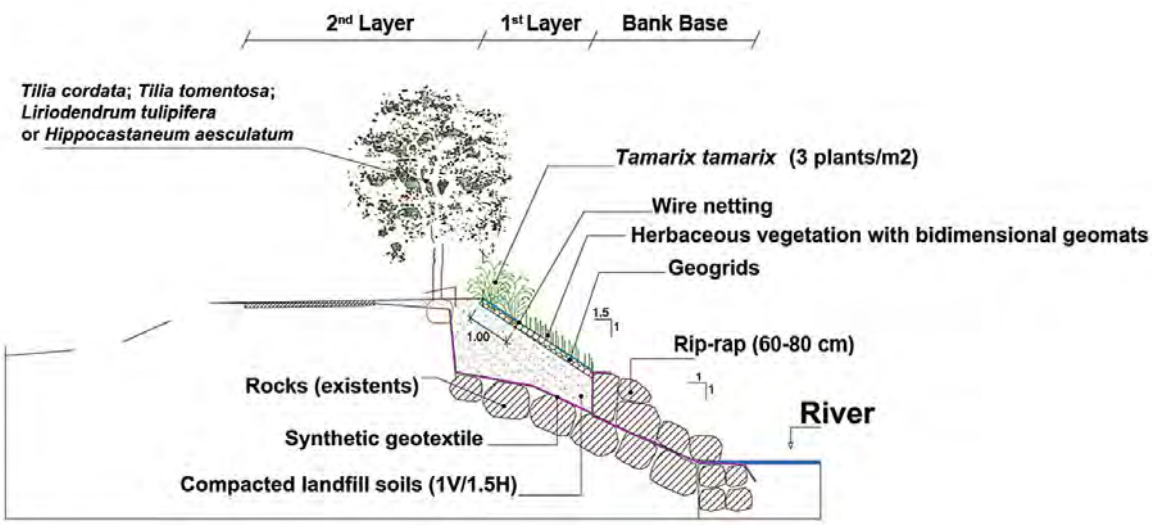

Figure 4. Planned techniques and vegetation species along the bank profile designed for Portuzelo.

Following a hydraulic study was conducted (using the HEC-RAS model) to compare the hydraulic conditions of the bank before and after the projected intervention, in order to estimate the energy dissipation of the flow energy. The hydraulic modelling enabled to simulate various scenarios, in particular the effect of increasing the hydraulic roughness to provide greater sedimentation of sediments and, consequently, to indirectly appraise the conditions for the resettlement of marginal vegetation derived from the succession of the

Highlights

This study includes local mitigation actions applied to solve the most dramatic erosion problems in specific estuarine areas sections. Of course it would be more convenient a global management plan for the restoration of the entire estuary considering the stressing agents. More holistic approaches take profit of conceptual models like the one presented by

Bergh et al. (2005), where the dysfunctional patterns in habitat and community structures were traced back to anthropogenic changes in the physical and chemical processes, with the identification of key parameters and distinct rehabilitation proposals.

In these specific areas, we adopted soft engineering solutions to coastal flooding, namely by incorporating planting of marsh vegetation in the intertidal zone for the purpose of promoting sedimentation and dissipating wave energy. We followed also the principles of Morris (2007), for whom a successful design would employ plant species that have varying degrees of tolerance to flooding, maximum drag, broad vertical ranges within the intertidal zone and that form a successional series. However, each rehabilitation method has to be observed under their specific conditions: if we provided the conditions for accretion because of a sediment deficit, other situations may require an inverse approach. This is the case of Garcia-Novo et al. (2007) that projected a hydraulic scheme favoring sand deposition upriver, avoiding its transfer to the Donāna marshes (South Spain) in order to prevent excess of silting during flood events, which caused unstable substrate with a lack of vegetation.

To estimate then the hydraulic differences in order analyze the ability to dissipate the energy created by the introduced structures, it was computed the shear stress and current velocity for different recurrence periods, between the initial situation and considering the disposed sets of groynes (Table 3). We may conclude for a significative reduction of shear stress, which reaches about 65%, corresponding also to estimate lower current velocities, as a consequence of increasing the resistance to flow (displayed by Manning coefs.), which may act also as sediment trap, protecting the base of the bank.

***Table 3**. Simulation with HEC-RAS model for comparing hydraulic parameters before and after the intervention.*

		Before		After	
Cross-section	Return period (years)	Shear stress (N/m^2)	Current velocity (m/s)	Shear stress (N/m^2)	Current velocity (m/s)
(Cardielos)	2.33	6.13	0.50	4.34	0.58
	5	7.56	0.57	5.45	0.66
	100	14.58	0.84	10.76	0.98
(Portuzelo)	2.33	10.66	0.68	3.93	0.61
	5	14.05	0.79	5.19	0.72
	100	25.51	1.10	10.39	1.1

Outcomes of Restoration practice

Following the appropriate post-appraisal of the implemented project in the target areas of Cardielos and Portuzelo we may draw some conclusions and recommendations. In the first 2 years after the project conclusion we could observe that in Cardielos all the structures showed a convenient resistance to critical environmental conditions. This is the case of the two rows of groynes, as well as the rip-rap or the gabion mattress (Figure 5), representing therefore a

convenient solution since additionally erosion impact decreased substantially and created the required barrier for bank protection preserving the built leisure structures. Besides, the subsequent field surveys allowed to observe that no more obvious scour holes were formed around the groyne layers. However, we have also to accept that not much sediment deposition was observed between these structures, contrarily to our expectations, which retarded the natural re-vegetation process. The most disappointing conditions were observed in the layer affected by to the tidal movement, where we notice a low success of the woody vegetation development, since the stakes rooting was deficient, probably because of the small size of this biological material (less than 30 cm length). Another cause was the lack of protection in relation to trampling (people and animals). In the case of the layer in the upper bank, outside the influence of the tides, we could observe better results, with higher plant survival and floristic diversity. With regard to the area of Portuzelo, it was evident the robustness of the rock base protection, as well as the stability of the plateau following the installation of the geogrid wall. However, the planting success was only relative, such as the natural colonization by macrophytes or herbs, but the viability rate was more intense with the plantations of shrubs based on tamarisk. At low tide, it was possible to check for the proper functioning of the installed drains which allowed the water to flow between the estuary and marshland, keeping a constant water level in this ecosystem, which is essential for its sustainability (Figure 5).

Figure 5. View of Cardielos (images above) and Portuzelo (below) just after the intervention in 2014.

Of course this action was focused in a specific part of an overall degraded estuarine environment. Immediately upstream and downstream of the rehabilitated sections there is still a constant progression of the pressure on the banks and the consequent set back of the bank line, which is reflected in Figure 8.

Finally, we must stress that this is only a mitigation action, even if integrative, but it will require in the future for a more complete study at a larger regional scale, including proper actions in all the estuary and even at the catchment level, in order to include the appropriate management that may contribute to overcome the deficit of sediments in the estuarine area.

Acknowledgements

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6.4. Study case 4: Mitigation in agricultural streams of Alqueva Multi Purpose Project

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Background

The Alqueva Muti Purpose Project located in Alentejo region of Portugal, involves abduction and water storage infrastructures, as well as infrastructures to improve irrigation areas (water transportation systems, pumping stations, roads and drainage networks), with the purpose of increasing the practice of irrigated agriculture (at least 120 000 ha). The implementation of some infrastructures involves streams impacts.

The streams mitigation impacts is a recurring theme related to the Environmental Impact Assessment procedures, namely through the legal obligation referenced in different Environmental Impact Declarations issued for the Alqueva Projects, due to the inevitability of environmental impacts, namely in surface water resources, such as streams. Based on a sustainable environmental policy the Alqueva Multi Purpose Project promotes the Environmental Impact Studies development in which it identifies and evaluates all the impacts resulting from this implementation projects in the streams ecosystems.

Given the relevance of this issue, which is a concern not only for Alqueva Project but also other entities, Streams Rehabilitation Projects have been implemented in order to improve and rehabilitate riparian systems representative of the different river basins. This action strategy has been followed by the Environmental and Patrimonial Department Impacts of Alqueva Project, since the design phase of the project to the implementation phase, and has involved different intervention typologies.

The Streams Rehabilitation Project in the Ardila Irrigation Subsystem (left margin of Guadiana River) is one of several projects of this type, promoted by Alqueva Project, as a way to mitigate and compensate environmental impacts.

The project under consideration results from the compilation of compensation and minimization measures of impacts defined in 3 Environmental Compliance Reports related to 7 Execution Projects (4 irrigation block irrigation projects and 3 dam construction projects). The impacts identified in studies, are mainly due to the improvement and use of the drainage networks of the irrigation perimeters, as well as the dams construction, filling of the respective reservoirs and variation of storage levels.

Aims

The case study implementation: Streams Rehabilitation Project in the Ardila Irrigation Subsystem (left margin of Guadiana River) focused on 15 streams with 30 km of extension and in 3 dams. It comprised several types of intervention, specifically: control of invasive vegetation, maintenance of riparian vegetation, planting of riparian native trees and shrubs, macrophyte planting surrounding the dams and ponds and little islet recovery.

The project implementation aimed at to promote and enhance fluvial ecosystem, contribute to the promotion of climate change adapt measures and to achieve environmental objectives (a mission enshrined in the Water Law, in line with the Water Framework Directive).

The general objectives of the case studie in question were: riparian galleries conservation and environmental valuation, in order to create favorable conditions for diversified *habitats* existence and maintenance; enhancing sustainability and biodiversity; promoting habitats ecological quality, the improvement of the natural regeneration process and consequently the restoration of the sustainability and the environmental balance of the ecological riparian corridors.

Strategies

The Stream Rehabilitation Project was implemented in the South of Portugal (Baixo Alentejo) on the left margin of Guadiana River, covered 15 streams, with a 30 km of extension, and 3 dams (Amoreira, Brinches and Serpa) and involved various intervention types. The Figure 1 shows the Rehabilitation Project area incidence.

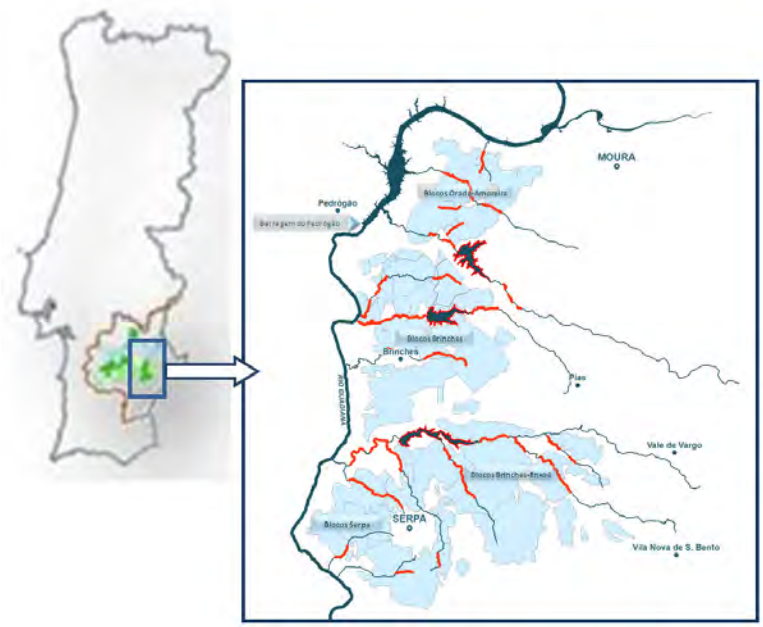

Figure 1. Rehabilitation Project Area.

The interventions associated with the Streams Rehabilitation Project involved were: invasive vegetation control, planting of riparian native trees and shrubs and recovery of ponds and an islet.

Control of invasive vegetation: Exotic and indigenous

The invasive vegetation control is a preventive intervention, with the aim to clean exotic and indigenous vegetation, with the objective to control invasive vegetation, in order to ensure habitat management through the: promotion of ecological conditions and natural regeneration of riparian vegetation strata; pruning of native trees and shrubs; reduction of the biomass to reduce fire risk.

The intervention methodology consisted in the aerial cut, with mechanical means, of exotic vegetation such as *Arundo donax*, and indigenous vegetation such as *Rubus ulmifolius*: The case of indigenous vegetation control is applied only in situations where they presented invasive behavior. In order to avoid eventual situations of slope instability, the root structure is not remove. The Figure 2 illustrates the physical control performed on exotic vegetation and native vegetation.

Figure 2. Cleaning of exotic invasive (Arundo donax) and indigenous (Rubus ulmifolius) species.

After cutting invasive vegetation, vegetal material is converted into smaller size particles through the mechanical equipment use. Figure 3 illustrate the reuse of material vegetal for organic matter valorization. When the particles presented bigger dimensions, the vegetal residues were sent to the appropriate final destination.

Figure 3. Cut of material vegetal and forwarding to proper final destination.

The natural regeneration of trees and shrubs was safeguarded with signaling tape. The Figure 4 illustrates some specimens marking.

Figure 4. Signaling tape of trees and shrubs specimens.

This intervention type require farmers authorization in plots adjacent to the streams. All these measures were accompanied by a specialized technician, to guarantee native trees and shrubs maintenance and to minimize possible fauna impacts, particularly bird nests presence. The aim is to contribute for the environmental and ecological recovery as well as for the natural *habitats* conservation and maintenance.

The Figure 5 illustrate a stream section cleaned (exotic invasive vegetation) where it is possible compare the evolution over the years and before and after the cleaning.

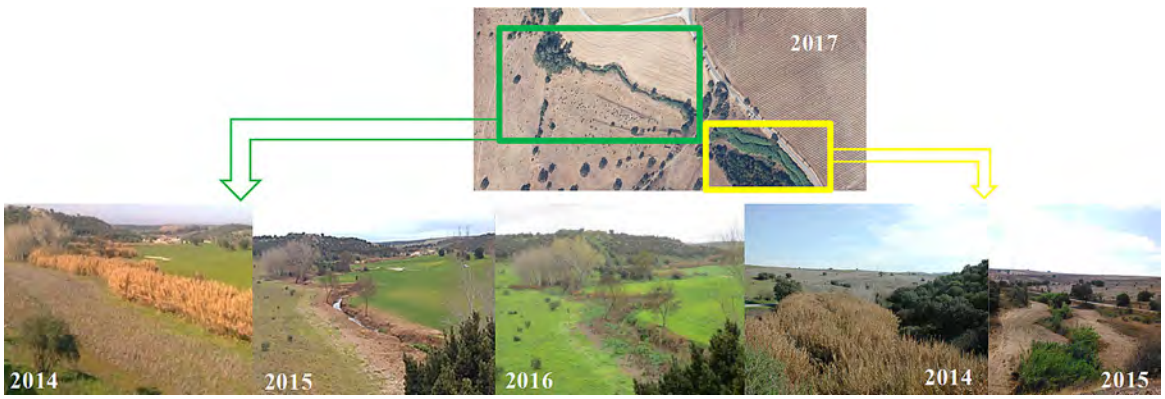

Figure 5. Streams section before and after mechanical cleaning.

Planting of riparian native trees and shrubs

Any action to clean streams will never be fully successful if it is not accompanied by replenishment and proper conduction of riparian vegetation. In this context, another measure implemented under the Rehabilitation Project was the riparian native trees and shrubs planting.

The objective of this intervention was to complete the streams tree and shrub; conserve the riparian ecosystem associated with the riparian gallery, and ensure the streams ecological and landscape functions. The plants used for this intervention comprised native trees and shrub specimens, more specifically the following species:

- Trees: *Fraxinus angustifolia; Salix atrocinerea; Salix salvifolia and Populus alba;*
- Shrubs: *Nerium oleander; Pistacea lentiscus; Pirus piraster; Crataegus monogyna; Securinega tinctoria* and *Tamarix africana*.

The intervention related to the plantation included: boiler opening; ground cover), fertilized; planting and manual placement of tutors and protective tubes in each of the specimens. The Figure 6 illustrates some work related to this intervention.

Figure 6. Boiler opening, planting and tutors placement.

Internable Bands Improvement

The aim of this intervention consists: valuation of sensitive areas from the landscape point of view; water control nutrients *input*; soil erosion risk reducing; biodiversity preservation and aquatic environments maintenance. This intervention included planting of macrophytes species at the full level storage in the Amoreira, Brinches and Serpa dams, namely: *Scirpus effusus, Typha domingensis, Scirpus maritimus L.* and *Scirpus holoschoenus*. This intervention is shown in Figure 7.

Figure 7. Macrophytes plantation.

Ponds Recovery and Islet Rehabilitation

The ponds recovery and the islet rehabilitation was aimed to strengthening the fluvial ecosystem sustainability and biodiversity, specially through the functions rehabilitation associated with this ecosystem.

The methodology implemented was based on the following actions: invasive vegetation removal; cleaning and dredging of ponds and small islet rehabilitation. A preliminary evaluation was required for the selection of sites for the rehabilitation of caught, and for the stones sinking and placement. This intervention occurred in a section located downstream of the Serpa dam. The Figure 8 shows the places associated with the rehabilitation of catch and construction of dams.

Figure 8. Ponds cleaning and dredging.

The rehabilitation ponds was relevant for ichthyofauna, since it contributes for the nesting and fixing this fauna group. The islet rehabilitation was a punctual intervention that place configuration alteration with the intended to enhance water flow and to promote streams ecosystem ecological potential. Figure 9 illustrate the intervention.

Figure 9. Islet rehabilitation.

The islet rehabilitation integrated planting of native tree species (*Salix salvifolia*) and emerging species such as macrophytes (*Typha dominguensis, Juncus effusus, Cyperus longus and Scirpus lacustris*). The macrophyte planting was carried out in the central zone of the islet, being of extreme importance, due to its colonization capacity.

For the interventions described, were used stone blocks (collected on the spot and placed stream bed) to allow improved water turbulence and oxygenation, favoring the quality and consequently contributing to the improvement of the stream ecosystem and their biodiversity.

Outcomes of Restoration Practice

The stream rehabilitation project, in the medium term, reasonably produced the results, taking into account farmers' inadequate environmental use of agricultural streams. There are, in fact, dominant streams pressures, which seem to result from: nutrient flows leading to polluting eutrophication phenomena and the development of invasive species, the silting resulting from the fine materials flow, disorderly occupation of riparian zones coupled with intensive agricultural practices resulting from the irrigated agricultural occupation crops and intensified use of fertilizers, pesticides and irrigation.

Another feature of streams located in agricultural areas is the soil mobilization up to the stream margin.

of crops that they are mobilized to the margin, the flood zone being mostly occupied by agricultural crop preventing the lateral connectivity of these watercourses.

It should be noted that the southern streams of Portugal are generally of small size (designated S1"100 km^2 according to the study of "Tipologia de Rios em Portugal Continental no âmbito da implementação da Diretiva Quadro da Agua. I. Caracterização abiótica") and located in areas with a high mean average temperature (about 16 ºC on average) and with low average annual precipitation (about 600 mm on average) in the climatic context of Portugal Territory.

In addition to these anthropogenic pressures, the climate change phenomenon has led to a reduction of precipitation quantity and period, with effects in the summer season extension, and the average temperatures elevation, conditions that affect this interventions type success.

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6.5. Study case 5: Multi-functional restoration of the Arga-Aragón River System (Navarre, Spain)

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Background and Aims

In recent years, the territory associated with the middle and lower reaches of the Arga and Aragón rivers (hereinafter, the Arga-Aragón river system), in Navarre (Northern Spain) (Figure 1), has suffered numerous managerial problems as a result of recurrent floods. Those rivers constitute the two main channels of the region, and its vertebral aquatic axis, before their confluence with the Ebro River. The river system is characterized by large and fertile floodplains, in which many intensive agricultural activities are carried out presently, and where several municipalities (12) and numerous linear infrastructures are located. At the same time, it constitutes an area that supports various natural sites protected by regional regulations and by the European network Natura2000, due to the quality of its natural habitats and the important biological diversity it hosts. In particular, most of these sites sustain riparian and alluvial forests protected by the Habitats Directive (92/43/EEC).

Since the beginning of the 21th century, the Government of Navarre and the public company Gestión Ambiental de Navarra (GAN) - with support from the Ebro Basin Agency and the Spanish Ministry of Agriculture and Fisheries, Food and Environment - have dedicated large efforts to the restoration of the Arga and Aragón rivers. The historical degradation of the Arga-Aragón river system originated from its progressive flow regulation, and more especially from the occupation of the rivers´ floodplains and the ecomorphological effects of the rectification of specific reaches, carried out in an attempt to reduce flood risks and to gain new areas for agriculture (Acín *et al.*, 2011). The restoration initiatives in the area have been conducted from a perspective of integration of European obligations in terms of water (Water Framework Directive, 2000/60/EC), flood risks (Directive 2007/60/EC) and biodiversity (Birds and Habitats Directives, 2009/147/EC and 92/43/EEC), and have been partly developed by means of different EU-funded projects (INTERREG, Life+).

Strategies

In 2008, the Government of Navarre required technical support to the Centre of Studies and Experimentation on Public Works (CEDEX) for the design of some complex restoration projects, and for the integrated and multi-functional analysis of the restoration alternatives for

the Arga-Aragón river system. The first of those works was the "Environmental improvement of the Plantío Meander in the Arga River", within the European project INTERREG IIIa GIRE-IMER (Integral Management of European Rivers). The main goal of the project was the hydraulic and ecological reconnection of the then-disconnected meander of Plantío with the main channel of the Arga River, and the reconversion of all its surface, until then used as a crop area, as a natural area, particularly for priority species, such as the endangered European mink (*Mustela lutreola*) (Figure 2). The meander had suffered a gradual disconnection and degradation, largely derived from human pressures in its environment and upstream of it. Complementarily, in the surroundings of the project area other heritage elements of interest could be found, e.g., the ruins of the Roman city of Andelos, or the Camino de Santiago, which provided an additional cultural dimension to the restoration initiative.

Figure 1. Location of the Arga-Aragón river system, within the Ebro basin in Northern Spain.

Figure 2. Restoration works on the Plantío meander (Arga River, Navarre), including different natural water retention measures, such as: reconnection of paleo-channels, creation of wetlands, plantation of forest patches with native species (local genetics) and improvement of habitats by increasing the topographical heterogeneity and the ecohydraulic complexity of the project area.

The project was developed by the regional government (2009-2010), in three phases: earthworks, reforestation, and amelioration of public uses. The project extent was around 24 ha, with a total budget of ca. 200.000 €.

The increased ability of the meander to abate ordinary and extraordinary floods and to temporarily store water and sediments was clear from the early reopening of the meander: various ordinary floods occurring at the end of 2009 and the beginning of 2010 showed that the structural and functional reconnection of the meander had been achieved (Figure 3a). Since then, the natural dynamics of the river Arga has been shaping the morphology and the ecohydrological processes of the meander, progressively increasing the naturalness of the habitats, and its capacity to retain water and to sustain numerous habitats and species of interest (Figure 3b). A monitoring programme has been carried out since the completion of the works, in order to quantify, over the years, the level of achievement of the project´s targets.

Figure 3. (a) Partial view of the meander, during the first spring after restoration (2010). The evolution of natural vegetation and the development of new hydromorphological processes throughout the meander may be observed; (b) Aerial view of the restored meander in the Arga River (Navarre), during an ordinary flood after the hydraulic, morphological and ecological reconnection developed in the project area (January 2010) (Source: Government of Navarre).

In 2009, after the completion of the aforementioned and of other similar projects in the river system, the regional government and its public land&water company presented a Life+ project to the European Commission, called "Mink Territory: Environmental Restoration of the Fluvial Territory", based on the analysis of alternatives for river restoration and flood risk amelioration developed by CEDEX. As part of that analysis, a wide range of alternatives were hydraulically and ecomorphologically simulated, on the basis on the following guiding principles:

- rehabilitation of the ecological connectivity (longitudinal, transversal and vertical) of the river system through the reconnection of the mainstems with their secondary channels, paleo-meanders and floodplains.
- restoration of priority habitats and species (with special attention to the European mink).
- improved protection of the territory against extreme hydrological events, with special attention to consolidated urban areas, and to critical areas due to the presence of infrastructures and buildings.
- improvement of hydromorphological and ecological processes in the entire river system.

Simulation were based on an exhaustive bi-dimensional hydraulic modeling associated with each block of actions, and on the analysis of their social and environmental effects (Figure 4). At the same time, the associated costs and the possibilities of implementation of each set of actions were determined.

The European Commission granted aid to the aforementioned project, with a duration of 6 years (October 2010 - September 2016), and with an estimated final cost of € 6,323,807 (EU co-funding reached 60% of total eligible costs). Within the context of this LIFE + project, CEDEX was also responsible for the external evaluation of the actions carried out in it.

Some aspects which have contributed to the successful implementation of the project were, among others (Magdaleno & Delacámara, 2015): i). the low population density in the project area (around 60 inhab./km^2); ii). the reduced size and compact character of the urban areas, despite the existence of scattered agricultural buildings along the floodplains; iii). the public property of many floodplain areas; iv). the continuous political and financial support of the Government of Navarre to the adoption of new strategies for managing floodplain areas; v). the existence of entities which have socially supported the fulfilment of those new strategies, by means of a stable and time-lasting process of dissemination and awareness, and a proactive strategy of public participation; vi). an intense inter-administrative cooperation, between the regional government, the town halls, the basin authority and the Ministry of the Environment in Madrid; vii). the intervention of local companies for the project´s execution, coordinated by very specialized personnel from the leading public entities.

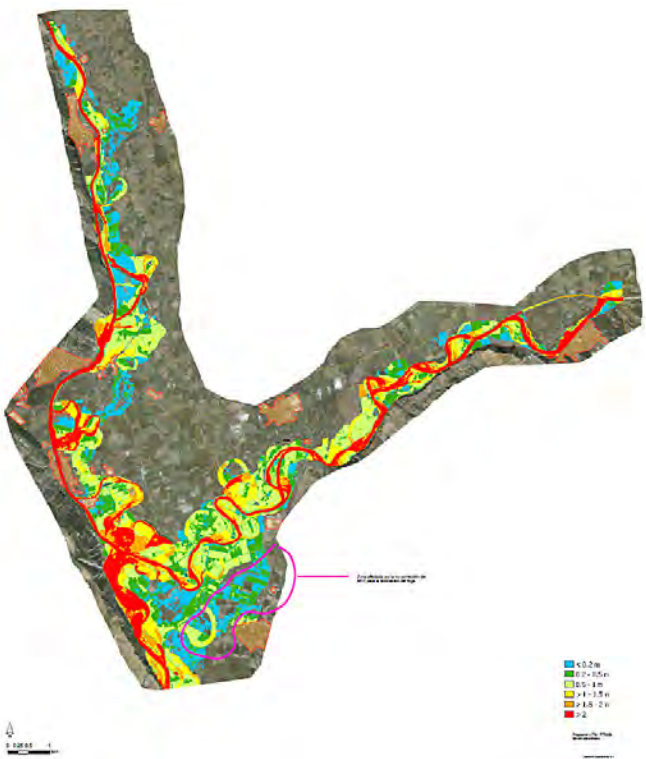

Figure 4. Example of the hydraulic, geomorphologic and ecologic modeling conducted for the design of restoration alternatives in the Arga-Aragón river system (CEDEX, 2010); in the provided image, for a return period of 10 years. Many of the restoration measures selected were aimed to increasing the water & sediments retention capacity of floodplains, and to allow a better protection of consolidated urban areas and certain critical sections.

On the contrary, the main obstacles found for the implementation of the restoration actions in the study area have been: i). the complex harmonization of the many public and private interests; ii). the obnoxious bureaucracy involved, since the horizontal approach of the project required permissions from hydraulic, forest, agriculture, biodiversity, environmental assessment, gaming and fishing, infrastructures, energy and heritage Departments, apart from those inherent to the town halls and the basin authority; iii). the adoption of shifts in the traditional managerial strategies, which are difficult to overcome in many cases and which involve reluctance among the affected population, despite the inertial model having shown inefficient for flood defence and environmental conservation; iv). the many scientific and technical uncertainties arisen during the design and implementation phases; v). the sometimes difficult interaction of the projects with the political changes gradually occurred in the town halls and in the other public institutions involved.

Despite those many obstacles, the majority of the planned actions were developed as part of the Life+ project (Figure 5). Subsequently, the achievement of the project´s objectives have been confirmed by means of the application of a range of ecological and hydromorphological indicators (CEDEX, 2014).

Figure 5. (a) Elimination of longitudinal embankments (levees) for the reconnection of channels with their floodplains and the improvement of ecohydrological processes (Aragon River, Navarra); (b) Floodplain excavation and reintroduction of sediments in the channel of the Aragón River, in order to reduce the ongoing incision of its riverbed, and to contribute to the improvement of the river system dynamics; (c) Creation of wetlands as water retention areas and refuge sites for the European mink and for different aquatic and riparian protected species; (d) Re-profiling of slopes in the old meander channels (quickly colonized by native helophytes) and creation of refuges for the European mink with the stumps of the alien poplar trees eliminated in the surrounding areas.

Highlights and Outcomes of Restoration practice

So far, and in accordance with the external evaluation that CEDEX has been carrying out and with the data collected by the own Navarre administration, the restoration measures developed have shown their effectiveness in terms of flood risk amelioration and environmental conservation, under the conditions imposed by the many anthropogenic pressures and impacts studied during the initial stages of this large process of restoration of the Arga-Aragón river system.

The multi-functional approach of the project has been successfully developed, as shown by the following indicators (Goikoetxea, pers. comm.[1]):

- removal of 7,5 km of levees and rip-raps, and relocation of many others.
- re-introduction of 105.000 m^3 of sediments in the Aragon River, which were originally part of the (removed) embankments.
- increase of 80 ha in the fluvial territory/room for the river.
- creation of 13,6 ha of wetlands, in 14 different areas.
- removal of 116 ha of cultivated poplars, and application of alternative procedures for the elimination of stumps in 7000 trees.
- reforestation of patches in degraded sites, with vegetative and non-vegetative propagules, including bio-engineering techniques in the constructed wetlands.
- habitat improvement in 128 ha, over 25 different areas.
- construction of a large number of refuges for European minks, snags for riparian birds, and artificial nests for birds and bats.
- release of many technical reports, and organization of a large array of public activities targeted to different social groups of the 12 affected municipalities.

Also thanks to the successful implementation of the restoration projects and their didactic role on authorities and people, no new extensive dredging and new defence works have been approved or implemented since the early application of the restoration actions in the study area, despite the many floodings occurred during that time. The project has enjoyed a wide national and international acknowledgement, which has also contributed to its positive social reception, and to its conversion into a paradigmatic intervention in the Ebro and in other Spanish basins.

1 Goikoetxea, M. Proyectos y acciones de restauración de ríos y conservación en el Life+ Territorio Visón. Retrieved from: https://territoriovison.eu/ (Last accessed, December 7, 2017)

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6.6. Study case 6: Louros River - Rodia Swamp restoration (1999-2003), Amvrakikos Greece

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Background

Greece has several world-renowned Ramsar Wetlands. The Amvrakikos is well known as one of the largest wetlands in Greece which unfortunately has complex management problems. Since 1990, Amvrakikos has been on Ramsar's Montreux Record of wetlands with serious degradation (i.e. officially assessed as "Ramsar sites where changes in ecological character have occurred, are occurring or are likely to occur"). In this case-study we provide a short history of restoration efforts along the lower Louros river.

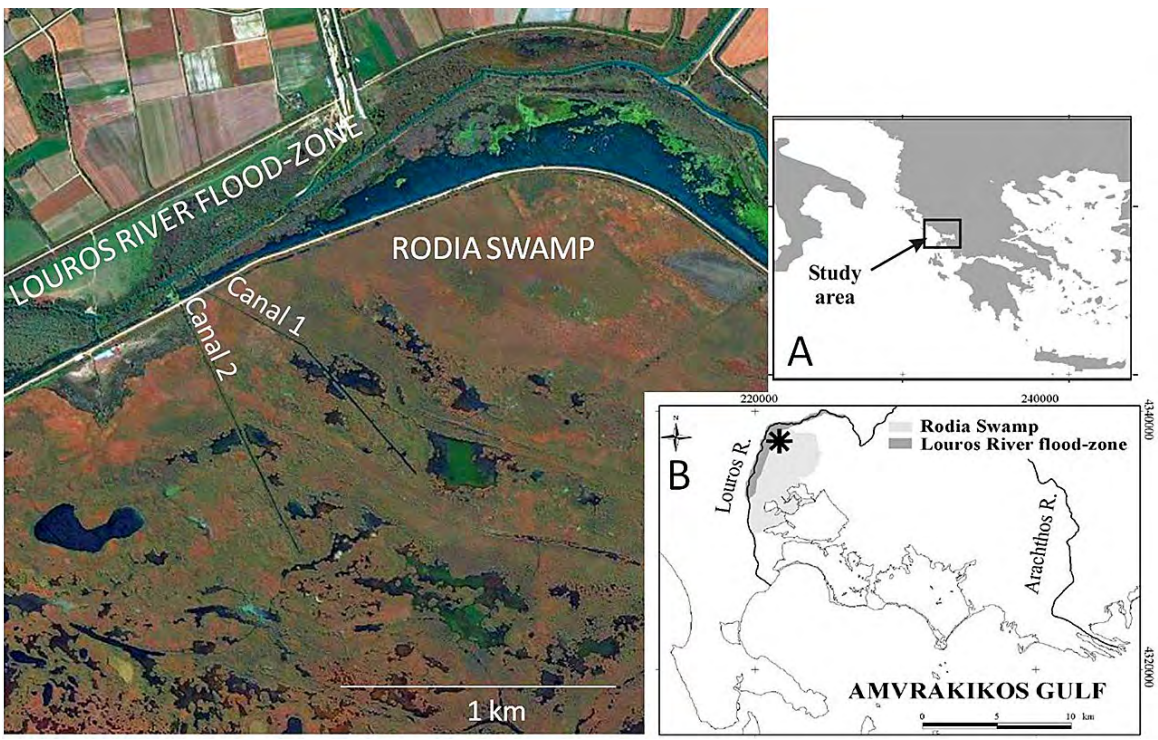

Figure 1. Google Earth image (2005) of the project site, showing the two new canals emanating from the embankment. Inset maps: A. The Amvrakikos Gulf along the west coast of the Balkans; B. Detail of the Northwestern Amvrakikos Gulf with the Rodia Swamp and the Louros River flood-zone (the asterisk shows the site of the restoration intervention where the sluice-canal system was constructed).

The Louros is one of the two major rivers entering the Amvrakikos Gulf. The river's flow is augmented by strong perennial springs after it exits its narrow upland river valley. These unique karstic springs rise in the lowland tributaries and within the delta plain creating huge wetlands with remarkable aquatic heterogeneity in the Louros-Arachtos double delta system. About 15 km before its river mouth, the Louros is flanked by an extensive reed swamp that has been artificially isolated by an embankment along the river channel since the 1960s - mainly for flood protection and land reclamation (Lawrie 2002; Zogaris et al. 2003) (Figure 1). This degraded reed swamp systems, the Rodia Swamp, covers about 27 km^2 and it connects to two extensive coastal lagoons, the Rodia-Tsoukalio coastal lagoon system. However severe degradation has taken place due to the artificial detachment of the river from its vast floodplain swamp and lagoons. During a Life Nature project (1999-2003) a restoration initiative attempted to reconnect and re-wet the river-side Rodia Swamp (OIKOS Ltd 2003). Although calls for this were made by scientists since the early 1980s the restoration action was begun and ended within this important Life Nature project.

Strategies and Aims

After much scientific planning a strategy was designed aiming at creating two sluice-controlled breaks in the artificial embankment of the Louros river in order to re-wet the former river delta swamp of Rodia in the early 2000s (described in: Theocharis et al. 2004a; Theocharis et al. 2004b). This work aimed to provide an "initiating act" for broad-scale restoration that would affect the wider Louros Delta. River water was meant to re-enter the Rodia Swamp with the following assumptions: a) the re-connection would not drain the biodiversity-rich artificially elevated Louros River flood-zone; b) the sluice-controlled flow would not cause flooding problems in the adjacent coastal lagoons that have high fishery values; and, c) the infrastructure would rejuvenate freshwater conditions in the degraded salinizing conditions of the Rodia Swamp and the adjacent stagnant lagoon conditions downstream. Although restoration of outstanding ornithological values was paramount (see Zogaris et al. 2003), many other values and services of the restoration effort were considered (OIKOS Ltd 2003). It was also hoped that after the Life Nature project the soon-to-be designated National Park authority and other government and research agencies would continue restoration and water management efforts.

During the end of the Life Nature project, in 2002-2003 the sluice-canal system was constructed in the heart of the floodplain-swamp wetlands, 13 Km upstream from the Louros river's outlet to the sea. The main infrastructure consists of two large sluices and two canals through the swamp, each 1000 meters long (6 m wide, 1,5 m deep), which can transport water directly from the river to the swamp without draining the Louros flood-zone. The flood-zone artificially elevates water levels due to the embankments creating unique riverine water meadow and flooded woodland habitats renowned as a wildlife hotspot in the region. The new canals puncture the swamp as fairly strait linear features (although the northern canal, Canal 1, is crooked). It was decided to design strait canals (i.e. not meandering) in order to promote the free movement of water away from the embankment and towards the upper central part of the swamp. In this way the river water would percolate through the extensive reed swamp affecting a larger area before it reached the saline lagoons. Former distributaries in the swamp are also linear-like features – no complex meandering is evident from the study of old air photographs and the hydromorphological imprints of the former river channels in the swamp.

The canals were constructed with the use of an "amphibious" vehicle, the IHC Beaver 300 cutter dredger (Figure 2). Dredged material (c. 14.600 m^3) was piled into low islet-like heaps

next the canals. These heaps of dredged soil were spread about so they do no artificially protrude more than 1.5 m above the canal's water surface. The line of dredged heaps is broken at very frequent intervals to allow water to pass through, and to create an islet-like feature suitable for wildlife cover (islet nesting sites, for example).

The restoration works cost € 210.000.00 in total (€ 97.000.00 for dredging works and € 113.000.00 for sluices, concrete ditches, embankment bridges, and other infrastructures). Infrastructure work took about 9 months to complete. The construction was deemed successful: even during low summer flows river water drained from the river to the swamp's new canals following the elevation gradient towards the Rodia Swamp basin, through gravitational movement (without the use of pumps). The sluice gates were opened in July 2003 and large quantities of river water entered the center of Rodia Swamp for the first time in more than 30 years.

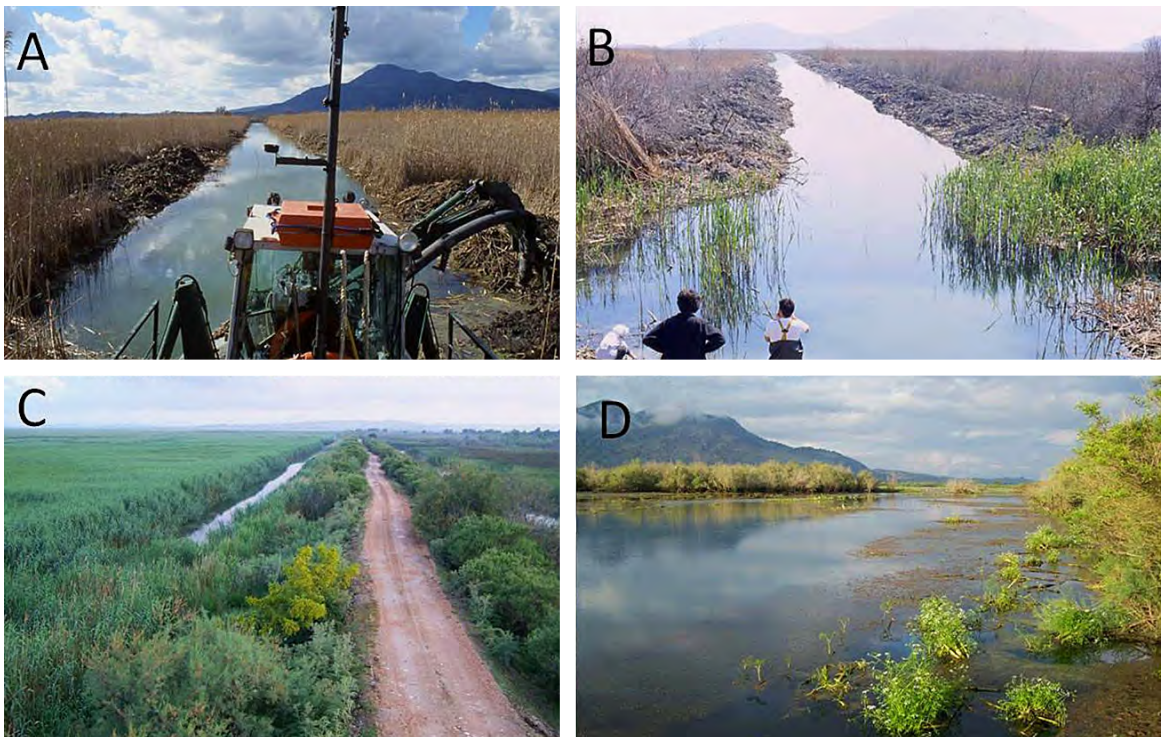

Figure 2. A. IHC Beaver 300 cutter dredger working on Canal 2 from inside of Rodia Swamp (2002); B. The completed Canal 2 from the embankment (2003). C. The road embankment on the right side of the Louros river flood-zone (at R) and the extensive reed swamp of Rodia (at L) (2008). D. Close-up of the precious water meadow and flooded woodland habitat inside the Louros river flood-zone (2005). [Photos by S.Zogaris and D. Papandropoulos].

Results

After the sluices were opened changes in the hydrology and the biodiversity of the swamp were immediately observed. The swamp's water level began to rise within a week of river water in-flow. Results of the project's initial success were evident upon a rapid assessment in the autumn of 2003 (OIKOS LTD 2003) and during a short survey in 2008 (Zogaris et al. 2009). The following aspects are indicative of this success: A) Initially, the canals could transport approximately 134 000 m^3/day of river water to the swamp. This maximum amount

(maintained with both sluices wide-open) represents about 13% of the rivers discharge during its lowest summer flows and proved that it can effectively increase water levels in nearly all parts of the swamp basin, as it did in the summer-autumn of 2003; B) Vegetation regeneration in the canals with helophytes and aquatic plants was rapid and certain freshwater-dependent species expanded into the formerly seasonally brackish *Phragmites* dominated swamp; and C) There was definitely a positive response in the freshwater wildlife of Rodia Swamp. Large numbers of water birds, including freshwater-specialized species used the canals immediately after their construction. Four years after the construction, there was a re-colonization of pygmy cormorant *Microcarbo pygmeus* in the Amvrakikos, and the breeding population chose to nest in the large heron rookery immediately next to the new canals (initially about 25 pairs than growing to about 40 pairs) (Zogaris et al. 2009). The pygmy cormorant had not nested in the Gulf's wetlands since before the mid 1960s; and incidentally it was one of the species targeted in the Life Nature project. At least 9 species of fish were recorded in the new canals, including rheophilic riverine fishes that were not expected to spread in the swamp (Economou et al. 2004). The restoration of the Rodia Swamp was also associated with other initiatives in the area, including the re-introduction of water buffalo for reed control; the herd has grown to number over 50 individuals (managed by a local monastery) (Figure 3).

Outcomes of restoration practice

The re-wetting action and the new canals in Rodia Swamp benefited many species and habitats in this heavily modified wetland ecosystem. Observational evidence supports that this was a priority endeavor that could help in restoring the ecological integrity of the wider delta system (Zogaris et al. 2009). Unfortunately after the end of the Life Nature project in September 2003 any further support of the aims for a more integrative restoration of the this complex system completely ceased. By 2005 the sluice gates had been vandalized but river water continued to enter the swamp since they were partially lodge open.

After years of study and deliberations, in 2008 the Amvrakikos was finally designated as a National Park but little changed in terms of restoration management. Greece then entered a long economic depression that plagued both the local Park management authority and government-directed conservation actions. No serious steps towards restoration have since taken place, although problems with lagoon fisheries and aquatic ecological integrity have grown and they are connected to degradation of natural river flow in the lagoons and the Gulf (Spyratos 2008, Katselis et al. 2013). The problems with water management and eutrophication have recently reached a critical stage since human-regulated river discharges are directly correlated with the development of hypoxic/anoxic conditions in Amvrakikos Gulf (Gianni and Zacharias 2016).

Improving the control of water use and abating pollution, and restoring wetland ecosystems among the interdependent wetland units are essential measures for conserving the ecological, socio-economic and cultural values of the Amvrakikos Wetlands National Park. We hypothesize that increasing river water flow through the wetlands of Rodia Swamp may provide a filtering buffer that may help heal the Amvrakikos Gulf's marine water problems. It is hoped that restoration efforts to re-wet and restore the Amvrakikos wetlands based on the existing sluice-canal infrastructure will be re-attempted in the future. The sluice-canal infrastructure remains in place and could be restored with relatively low cost if a properly designed strategy for adaptive management and scientific guidance could be put in place.

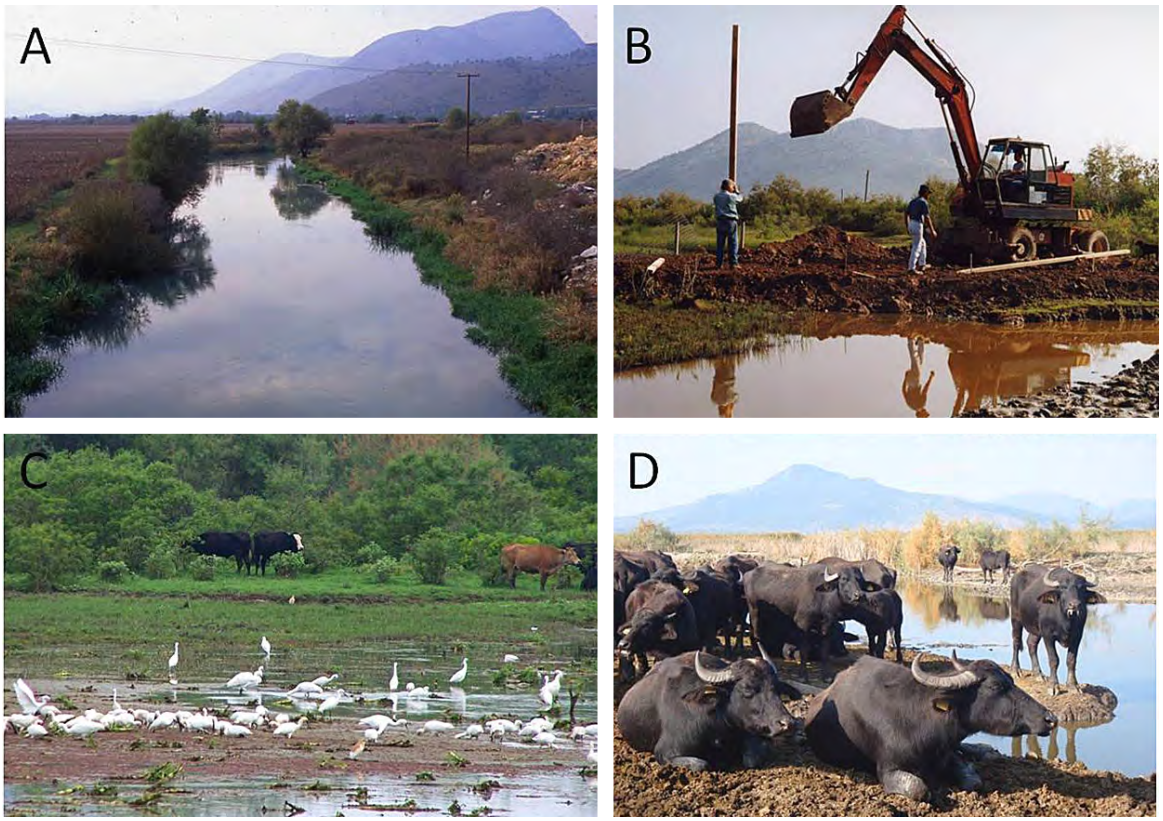

Figure 3. A. The degraded conditions of the Louros river immediately upstream of Rodia Swamp (2002). B. Life Nature works to promote fencing for Buffalo grazing inside Rodia Swamp (2001). C. Egrets, herons and spoonbills in the Louros feed in the grazed flood-zone habitats and nest in Rodia Swamp (2008). D. Water buffalos at Rodia Swamp (2008). [Photos by S. Zogaris].

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6.7. Study case 7: New tools for riparian restoration: predictive modelling of vegetation dynamics in the Odelouca river, Algarve

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Background and aims

In recent years, riparian modelling approaches have experienced notable development as cost-effective tools supporting restoration. To predict and assess the consequences of restoration measures, maintenance operations or human pressures in rivers, managers and planners may model these interactions enabling an anticipated quantification of management decisions in rivers (Solari et al. 2015). Process-based riparian modelling approaches are often based on the natural flow regime paradigm (Poff et al. 1997), which posits that the flow regime strongly influences the dynamics of riparian biological communities through hydrological and geomorphological processes (Loučková 2012). In particular, the frequency, duration and magnitude of floods are among the key processes governing riparian vegetation dynamics, plant recruitment, growth, and succession (Muñoz-Mas et al. 2017).

In free-flowing rivers, sediment deposition from floods creates nursery sites for riparian recruitment and successful establishment depends upon the subsequent moisture pattern dynamics. **Recruitment** occurs when seeds in suitable moisture habitat conditions are able to develop their root system before decoupling from the capillary fringe and avoiding their water stress decline (Stella & Battles 2010). This has been described as the "*Recruitment Box*" (Mahoney & Rood 1998) which defines the stream stage patterns that enable successful establishment of riparian seedlings (Figure 1). Conversely, flood events remove vegetation through erosive scour (Bendix & Hupp 2000; Egger et al. 2015). Therefore, successfully established seedlings usually **grow** and **succession** continues to more complex and mature vegetation stages until floods above some disturbance threshold (shear stress) cause vegetation removal and therefore **retrogression** to previous vegetation stages. One of the consequences of these complex interactions is a dynamic mosaic of vegetation, composed by patches of different age and structural features that characterize the riparian corridor and that are the object of riparian vegetation modelling.

Strategies

One of the biggest challenges of the development and implementation of a vegetation model is to overcome biogeographical differences across different climatic and landscape settings. A modelling approach based on recognition of discrete riparian vegetation patches,

characterized by their structure and function, as shaped by fluvial hydraulic processes, is potentially transferable to other geographical regions and climates. The relationships between fluvial hydraulic processes and vegetation structure could then be used to build a widely applicable scheme of succession-regression pathways rather than undertaking species-specific comparisons. Succession of taxa may differ between regions or case studies, although it is essentially driven by the same processes. The 'CASiMiR Vegetation' (Computer Aided Simulation Model for In-stream flow and Riparia; www.casimirsoftware.de), proposed originally by Benjankar et al. (2011); covers a broad geographical coverage of physical habitat factors, which characterize flood-pulse drivers and which can account for vegetation establishment, succession or resetting over a range of river conditions and climatic regions (Egger et al. 2013).

CASiMiR-vegetation approach considers a framework of succession-regression pathways across hydrologic and flow disturbance gradients**.** In general terms, **three different succession stages** are recognized (colonization, transition and mature stages) and several phases are defined within each stage, representing temporally consecutive levels of vegetation development. The differences between stages reside in the dominant development stage, habitat characteristics and the flow disturbance regime they have experienced.

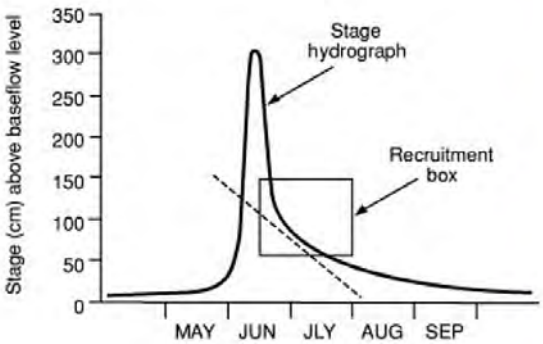

Figure 1. The Recruitment box model defines a zone in elevation and time in which riparian seedlings are likely to become successfully established if stream flow patterns are favourable. The graphs on the left represents a hydrograph that satisfies requirements for seedling establishment, source (Mahoney & Rood 1998). The picture on the right illustrates the natural recruitment of Alnus glutinosa seedlings in river Sorraia (Portugal).

Primary succession starts when seedlings succeed to colonize bare and fresh sediments. The first phase of the 'Colonization Stage', known as the '***Initial Phase***', consists of bare soil with usually close groundwater level, where few plants are present, and in some instances seedlings only become established for a short time period due to the high frequency of disturbance. The second phase of the 'Colonization Stage' is the ***'Pioneer Phase'*** and is characterized by relatively sparse vegetation, primarily species with a ruderal or stress-tolerant strategy which are adapted to frequent disturbance and strong hydrological variability (wetting and drying), typical of low water retention capacity substrates.

The 'Transition Stage' (also referred to as the 'Consolidation Stage'; (Naiman et al. 2005)) follows vegetation colonization and it is characterized by vegetation cover, biomass production and standing crop increase, while the average vegetation stand age increases and species diversity remains high. Several phases can be distinguished within this stage, some dominated by herbs or shrubs. Within a woodland vegetation series, the ***'Early Successional Woodland Phase'*** (also called the 'Stem Exclusion Phase'; (Naiman et al. 2005)) occurs when

trees are able to replace the shrubs as the dominant life forming the patch. The recruitment of dominant trees typically occurs on sand and gravel bars (during the 'Initial Phase') and, in Europe, it includes pioneer species like willows (*Salix* spp) and cottonwoods (*Populus* spp) ('softwood forest'). These species are fast growing and saplings are heliophyllous, i.e. do not develop in shady locations. The 'Transition Stage' finishes with the onset of the '***Established Forest Phase'***, also known as the 'Understorey Re-initiation Phase' (Naiman et al. 2005). In most instances, the species present during this phase are recruited to the understorey in the Early Successional Woodland Phase. They typically have slow growth rates and many may die before the onset of the next succession phase. When a portion of the old softwood forest is dying, a gap is created in the canopy and shade-tolerant ('hardwood forest') understorey species may become established. In Europe, typical species recorded during this phase are *Fraxinus* spp., *Quercus* spp., *Ulmus* spp.

The 'Mature Stage' is characterized by large standing biomass and high species diversity, though it may be lower than that recorded during the 'Transition Stage'. The '***Mature Mixed Forest Phase'*** is typical of this stage, including a combination of woody and long-lived riparian and terrestrial species, which can regenerate from understory with clearings opened by ageing overstorey trees. This is an autogenic process whereby trees regenerate and grow without the influence of external disturbances. Finally, the 'Climax Stage' should comprise terrestrial upland forest which reflects the dominant vegetation in the area beyond the influence of the flood regime of the river.

The interaction between natural disturbances and successional processes leads to the formation of a mosaic of vegetation patches on the riverscape (Ward et al. 2002) and the allocation of each one of these patches to a phase within the succession series. Disturbances are defined as 'relatively discrete events that disrupt the structure of an ecosystem, community, or population and change resource availability or the physical environment'(White 1979). Important disturbance descriptors include the event magnitude, duration and recurrence interval.

Structure of CASiMiR-vegetation

The CASiMiR-vegetation model assumes that vegetation development depends on the functional relationship between hydrology, physical processes and vegetation communities (Egger et al. 2013). Biotic factors are represented by the plant communities' succession stages. Physical factors can be represented by quantified indicator values to represent a gradient of disturbance magnitude. For floods, these indicators include hydraulic parameters such as critical bed shear stress, an indicator of morphodynamic disturbance. Shear stress is an index of fluid force per unit area (N/m^2) on the stream bed, and corresponds to the stress applied by the river flow parallel to the river bed.

The CASiMiR-vegetation model is structured in three main modules (Start, Dynamic and Visualization modules), which support users from the early creation of initial vegetation maps to the sequentially scenario-driven visualization of outputs (Figure 2). The **Start module** recreates the natural potential patch disposal of the study site, according to topography and mean annual flow. It assigns succession phases to different heights above water level and provides an initial vegetation-patch layout if needed. This module's output is a static picture of the potential riparian vegetation at a study site and generally used as a starting point for the Dynamic module, if no previous field vegetation assessment is available.

The **Dynamic module** simulates the effect of physical river processes on the survival and recruitment of riparian vegetation, and its output is a year-based temporal and spatial representation of the expected vegetation succession phases. It replicates the succession/retrogression of riparian vegetation, the shaping action of shear stress, flood duration and height over water table disturbance. Retrogression to the Initial Phase occurs where shear stress or flood duration is higher than the vegetation resistance threshold. In contrast, where vegetation is capable of resist these stresses, succession takes place with consequent vegetation ageing. Recruitment is also considered in this module, linked to water table elevation and associated to the Initial Phase, wherever suitable habitat conditions for seeds are met, based in the Recruitment Box Model (Mahoney & Rood 1998). In the Dynamic module, physical processes are modelled by zones (aquatic, bank and floodplain), which are defined in advance for inputting into the model. The aquatic zone is usually defined as the area inundated by the base river flow.

Highlights and outcomes

Predicting riparian vegetation dynamics under regulated river management in Portugal

The CASiMiR-vegetation model was applied in Portugal to analyse the riparian patch dynamics predicted for different flow regimes and to assess vegetation requirements to ensure long-term ecological maintenance and vitality of riparian structure in rivers with altered flow regimes (Rivaes et al. 2015). Furthermore, the capability of flushing flows to restore and manage riparian vegetation and the efficiency of environmental flows to satisfy riparian vegetation requirements was assessed.

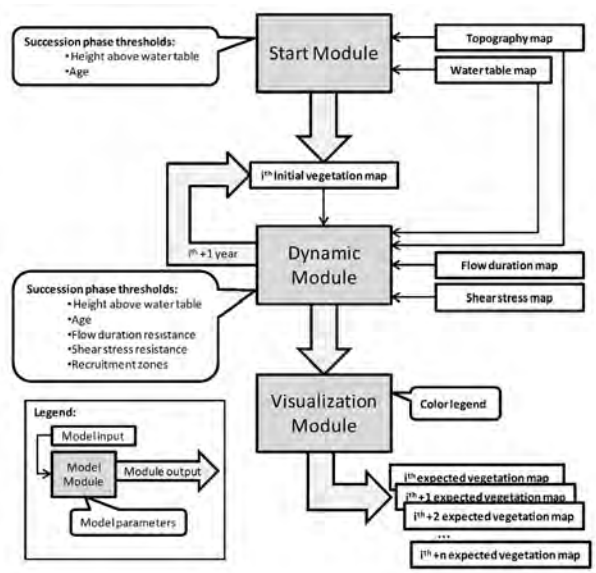

Figure 2. CASiMiR-vegetation model structure. Source (Rivaes et al 2013). Grey boxes represent model modules, white sharp edge boxes represent input and output data, and round corner callouts represent the thresholds definition driving model modules.

In the present chapter, we present results for Odelouca River where three different flow scenarios were simulated. The Odelouca field site was located in a free-flowing reach in the upper course of the Odelouca River (Lat: 37º23'05.00''N, Long: 8º18'39.46''W), Arade Basin,

in Algarve, Southern Portugal (Figure 3). The flow regime is typically Mediterranean, having two distinct periods: the winter period, with low flows that are sporadically interrupted by flash floods, and the summer period, with very low or even null flows. This river stretch was selected as the best available site with near natural conditions in terms of riparian vegetation and is representative of the downstream Odelouca River. Riparian vegetation is dominated by *Salix salviifolia, Tamarix africana* and *Fraxinus angustifolia.*

Input data

Topographic data were used to create the Digital Elevation Model (DEM) of the site, for use in data processing for hydrologic and vegetation modelling. The Digital Elevation Model (raster format, 0.5m size cell) is a graphic representation of ground surface and was created in ArcGis from the elevation data gathered in the topographic survey. **Vegetation surveys** were performed to support model calibration and provide vegetation maps for the assessment of model accuracy. **Hydrological data** were obtained from a nearby gauging station (Monte dos Pachecos 30G/01H, (http://snirh.pt), situated downstream in Odelouca river, and used to create different input flow regimes scenarios for the riparian vegetation modelling. Registered data were used to estimate site-specific maximum instantaneous discharges for different probabilities of exceedance. The annual maximum discharges at the field site were calculated considering the ratio between "Monte dos Pachecos" and the field site drainage basin areas and their mean annual precipitation. The **hydraulic data** required by the CASiMiR-vegetation model are the maximum annual discharge shear stress and the minimum annual water surface elevation. HEC-RAS 4.1.0 (Brunner, 2008) was used to determine the flow rating curve at the study sites outflow sections, and the River2D v0.93 (Steffler et al., 2002) was used to predict shear stress and water surface elevation for each of the considered discharges. In the specific case of Mediterranean rivers such as River Odelouca, given that the river stops flowing in summer, the aquatic zone was attributed to the pools that remain when flow is null. The bank and floodplain zones were defined as the areas inundated by the 1-year and 100-year return period discharges, respectively.

Figure 3. (a) Odelouca field site (summer), examples of successional phases (b) Pioneer species regeneration (*Salix salviifolia*), (c) Early successional Woodland (*Salix salviifolia*), and (d) Established forest (*Fraxinus angustifolia*).

Flow Regime Scenarios

Modelling was performed for a 10 year period, from 1985 to 1995, under three different flow regime scenarios. The **Natural scenario,** considered the actual maximum instantaneous discharges that occurred in the study site, obtained from the gauging station recordings. The other two scenarios simulated two regulated regimes, considering that a hypothetical dam could be built upstream the study site. The so-called **Dam scenario**, considered a hypothetical dam upstream with no release of environmental flow. The so-called **Flushing Flow scenario** considered the usage of flushing flows is composed of a combination of 10-year floods interspersed with 3-year floods, in order to recreate and test the effect of large an intermediate return period floods in the area (Rivaes et al. 2015).

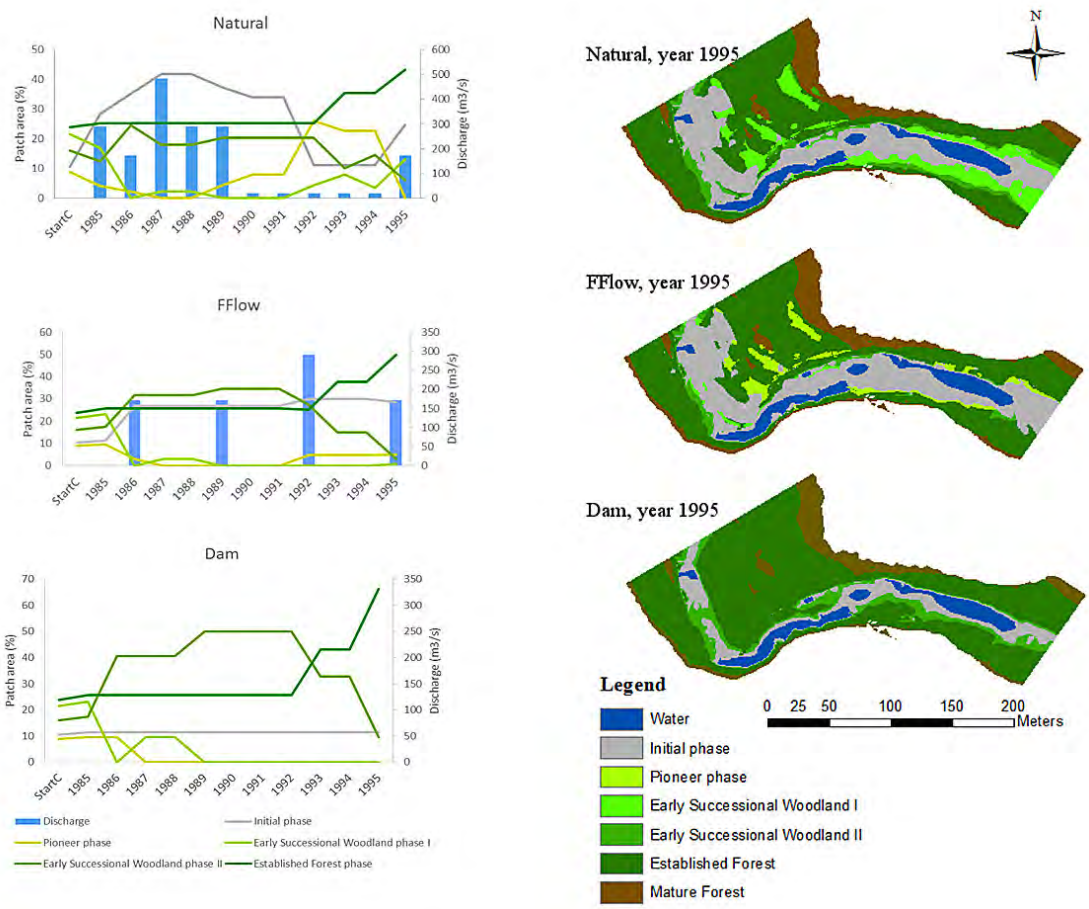

Figure 4. Riparian patch dynamics in Odelouca study site, according to the three flow regime scenarios: Natural regime (top), and two simulated regimes: Flushing flows (centre) and Dam with No Flow (bottom). Line plots (left panel) represent riparian succession phases evolution consistent with the annual maximum discharges presented in the bar plots. Maps (right panel) stand for the 1995 expected vegetation map for each flow regimes.

Discussion

One of the most striking causes of the degradation of freshwater systems is the physical habitat changes attributed to river damming. Dams disrupt connectivity and alter natural flow regime downstream with negative consequences for aquatic and riparian species life cycles. The cause-and-effect relationships between flow regime and riparian vegetation development can

be used in regulated river management, to prevent vegetation encroachment and to promote a more balanced distribution of age classes, in order to maintain a sustainable riparian landscape. The numerical model results showed that the natural flow regime brings much more spatio-temporal variability and a balanced distribution of the fluvial patch mosaic during the 10 years of modelling, than any of the other regulated regimes (Figure 4). Especially the Dam scenario showed the continuous ageing of the succession phases illustrating the homogenization and encroachment of vegetation in the river channel. This trend is appreciable before completing the whole modelling period, indeed, after 5 years with no flow release, the younger successional phases (pioneer phase, early successional forest phase) are no longer represented. Forest Encroachment is one of the sources of losing ecological integrity and impoverishment in riparian ecosystems, in addition to problems for management (Santos 2010). Additional pressures such as species invasions are more often aided by the alteration of natural disturbance regimes caused by flow regulation, this is a critical issue in rivers across the Mediterranean region (Stella et al. 2013). The Flushing Flow scenario revealed that flood events with recurrence intervals of ten and three years can modulate the distribution of succession phases, by promoting some retrogressing of vegetated patches and preventing vegetation encroachment inside the channel. This illustrates that predictive modelling of riparian vegetation dynamics could be applied to reduce the pervasive effects of flow regulation (Bejarano & Sordo-Ward 2011). Planners can use this tool to predict the outcomes of different combinations of floods, and integrating them with other target species such as fish, they can promote sustainable and more diverse riparian ecosystems. Importantly, the establishment of guidelines for environmental flow regimes considering riparian requirements should be applicable at the watershed scale (McCluney et al. 2014). Careful planning of environmental flows using a holistic perspective encompassing the ecological quality of regulated rivers, which should also include riparian requirements, must be based on a numerical modelling approach, if possible preceding dam design, so that the dam outlet structures can meet those requirements (Rivaes et al 2015).

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III Coastal rehabilitation: general strategies and examples

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1. Introduction

The coastal zone is naturally dynamic, changing due to coastal geomorphological processes (Figure 1). Worldwide, land use changes associated with coastal urban development, coupled with climate change and growing human population pressures are accelerating the alterations in the coastal zone. As a consequence, an integrated approach is required, in order to describe adequately the geomorphic-biologic dependencies, the feedbacks between processes and responses, as well as the maintenance of coastal systems, both spatially and temporally (Forbes *et al.*, 2004).

The observed climate change and the resultant sea level rise, the increase of storm surges and other extreme events that afflict the coastal zone and, also, the rapidly increasing population of the coastal zone are expected to have a significant impact not only on the resilience of the coastal systems, but also on the strategies of ecosystem services and goods (Orford and Pethick, 2006; Chapin *et al.*, 2009; Defeo *et al.*, 2009).

Coastal rehabilitation, as it emerges from the bibliography, is closely associated with ecosystem services (Barbier, 2013). The coastal zone is particularly rich in such services, which can be environmental, social, cultural and recreational. For example, sandy coasts provide several ecosystem services, such as sand and minerals for extraction, raw materials, retention and purification of ground water, plants for pharmaceutical use, food for primary and higher trophic level consumers, habitats and refuge both for plants and animals, recreation, education, scientific research, environmental and cultural heritage (Bell and Leeworthy, 1990; Peterson and Lipcius, 2003; Everard *et al.*, 2010; Martínez *et al.*, 2013).

Rehabilitation actions in the coastal zone concentrate foremost on wetlands, coastal dunes, as well as on shoreline mangrove forests. The aforementioned constitute quite dynamic

ecosystems that are located at the interface of land and sea (Craft and Bertram, 2008). However, more significant rehabilitation activities are demanded where great tourist attraction and human settlements exist, such as in sandy shorelines and coastal dunes (Martínez *et al.*, 2013).

Figure 1. Coastal dunes in the western coastal zone of Naxos island (Cyclades, Greece). Photo: N. Evelpidou, 2016.

Figure 2. Coastal dunes are often present in sandy coasts. The presence of vegetation is important for their development and stabilization. Photo: N. Evelpidou, 2015.

2. General strategies for coastal rehabilitation and examples

Generally, rehabilitation can be defined as the act of partially or, more rarely, fully replacing structural or functional characteristics of an ecosystem that have been reduced or lost. It may, also, be the substitution of alternative qualities or characteristics than those originally present with the proviso that they have more social, economic or ecological value than existed in the disturbed or degraded state (Edwards, 1998; Elliott *et al.*, 2007). Thus the rehabilitated state is not expected to be the same as the original state or as healthy but merely an improvement on the degraded state (Bradshaw, 2002).

Figure 3. Typical vegetation in coastal dunes. Photo: N. Evelpidou, 2016.

Coastal rehabilitation can be conducted through a wide variety of vegetative, hydrologic and structural methods and projects. Usually, the forms of rehabilitation are sediment-oriented. Some examples where coastal rehabilitation measurements are necessary nowadays are mentioned in the following paragraphs.

2.1. Coastal dunes and rehabilitation

Nowadays, the loss and degradation of the coastal dunes is mainly owed to human activities (e.g. Lithgow *et al.*, 2013), which typically differentiate the natural processes on the coastal zone, thus modifying the system dynamics (Figure 4). Such human activities could be generally categorized in 6 categories (Martínez *et al.*, 2013; Lithgow *et al.*, 2013): (a) housing and recreation, (b) industrial and commercial use, (c) waste disposal, (d) agriculture, (e) mining and (f) military activities.

Figure 4. Low lying dunes in the coastal zone of Agios Prokopios (Western Naxos, Cyclades, Greece). A road has been constructed between the dunes and the lagoon. Photo: N. Evelpidou, 2015.

The activities aiming to coastal dune rehabilitation include principally the reshaping of dunes and the recovery of sediment dynamics, as well as dune stabilization through the control of invasive species of plants and animals (Figure 5).

Research on coastal dunes rehabilitation has taken place in many areas around the world, including Europe, Africa, Asia and North America. For Europe, in particular, the majority of studies regarding coastal dunes restoration are found in the Netherlands (e.g. Vandenbohede *et al.*, 2010; van der Hagen *et al.*, 2008; Arens *et al.*, 2006).

In terms of rehabilitation methods, Lithgow *et al.* (2013) note, that there is no best way to restore a dune. Due to the dynamism and diversity of this ecosystem and the various geomorphological and ecological characteristics of dunes around the world, a wide range of activities may be implemented. Some examples are briefly described below.

Beach nourishment

Nowadays, many beaches suffer from unbalanced sediment budgets, which are mainly owed to human activities. Technical works on the coastal zone such as breakwaters, ports, etc. often lead to changes in the longshore currents affecting the amount of sediments reaching a particular beach. In a similar manner, the construction of dams upstream leads to reduced sediment reaching the coastal zone. All these processes significantly affect the sediment balance of the coastal zone, leading to the erosion of the coast and its landforms.

The formation of dunes could be accomplished through beach nourishment, also known as beach recharging. Through the increase of the volume of a beach or dune by nourishment, a physical buffer between the sea and inland areas is being provided, increasing that way the direct level of protection to inland areas. In addition, the added sediment intensifies the protective capacity of the entire beach system (e.g. beach, dunes and the nearshore area). The imported material may be placed on the intertidal foreshore, where it will help to protect the dunes by the increasing wave energy dispersal along the beach. Otherwise, the material can be placed directly at the dune face, in order to form an artificial foredune. The nourishment material should be as similar as possible to the indigenous sediment (Figure 6). Sediment size, grading, shell content and material should match the upper beach and dune face, the mid to lower beach and the shallow nearshore zone. The material should also be clean and free of seeds. If these conditions are not met, then the nourishment may cause unwanted changes to the beach and dune profiles, the dune ecology and the dune appearance.

Figure 5. Stabilized coastal dunes through the development of vegetation. Photo: N. Evelpidou, 2015.

There is some debate as to the most effective position to place the nourishment to achieve optimum protection. Possible locations include the upper beach and dune face, the mid to lower beach and the shallow nearshore zone. The former is most obvious as the benefits of nourishment are immediate, but, if the sediment is primarily sand, it will be rapidly redistributed alongshore or across the beach face by waves and currents to form a new equilibrium profile (coarser material tends to remain on the upper beach, but may be redistributed alongshore). Placement of sand on the mid-beach anticipates this redistribution, and provides shoreline

protection by helping to dissipate wave energy before it reaches the dunes. Placement below the water line also anticipates the redistribution and allows sand to be fed into the beach system gradually. However, maintenance is necessary for the preservation of dune integrity and time is of essence, in order for backdune species to colonize (Martínez *et al.*, 2013). Nevertheless, the restoration of vegetation and morphology can last up to ten years (Woodhouse *et al.* 1977; Maun, 2004).

Figure 6. In beach nourishment, the added material should be as similar as possible to the indigenous sediment. Photo: N. Evelpidou, 2016.

Use of vegetation

The increase or planting of vegetation can assist the stabilization and development of coastal dunes, as vegetation can trap and stabilize sediments, reduce the wind intensity and provide habitat (Figure 7) (e.g. Nordstrom, 2008). Prior to placing material along the dune face it may be prudent to remove some of the existing vegetation for later transplanting. Vegetation buried by imported shingle or wet sand to a depth of more than a few centimetres may not recover. It is, however, important which vegetation species will be used, as native species will have a higher survival rate and are easy to propagate, harvest, store, and transplant (Figure 8). In addition, exotic species can generate further ecological and economic problems. In the foredune zone, beach grass (*Ammophila sp.*) and other species can be planted to trap sand (Craft and Bertram, 2008). With time, the dunes can be increased in size, woody vegetation can colonize and soil can begin forming. If there is no dune landward of the beach nourishment project (the dune would hold the sediment and also provide sand), planting the back part of the beach with native, salt-tolerant, erosion-control vegetation with extensive root systems is highly recommended to help hold the sediments in place and, also, trap windblown sand to build up a dune as an added buffer to landward areas from flooding.

Figure 7. Vegetation can help stabilize the dunes, as it can trap sediments, reduce wind intensity and provide habitat. Photo: N. Evelpidou, 2016.

Figure 8. The use of native species for the stabilization of coastal dunes will have a higher survival rate. Photo: N. Evelpidou, 2016.

2.2. Coastal wetlands and rehabilitation

Wetlands are estimated to cover 860 million hectares or 6% of the land surface of the world (Maltby and Turner, 1983; Mitsch and Cronk, 1992). They have been described as the kidneys of the landscape for the biogeochemical and hydrologic roles they provide (Mitsch and Gosselink, 1986). They prevent floods, cleanse waters, protect shorelines, and recharge groundwater aquifers. Just as important to some, wetlands provide shelter for a wide variety

of flora and fauna and offer a unique habitat for many rare and endangered species (Figure 9). But like the species that they host, wetlands themselves have been considered by some to be endangered. Coastal wetlands are very important and valued features due to their high biodiversity and ecosystem services.

Figure 9. Coastal wetlands are a unique habitat for a large variety of flora and fauna and for many rare and endangered species. Photo: N. Evelpidou, 2017.

Despite the existence of the Ramsar Convention, the EU directives and various national legislations for their protection, coastal wetlands, in some cases, have been modified, damaged or destroyed. In addition, climate change and the subsequent sea level rise, will only add to the current loses of coastal wetlands. Estimating wetland loss worldwide is difficult, even in the more developed countries with the resources to assess the status of their wetlands (Mitsch and Gosselink, 1993) Dugan (1993), however, estimated that more than 50% of historic wetlands globally have been lost. The amount varied by country and by region. For example, in New Zealand, an estimated 90% of wetlands have been destroyed (Dugan, 1993), while in the United States more than 50% (in the contiguous 48 states) are gone nowadays (Dahl *et al.*, 1991).

Rehabilitation activities need to primarily start with the study and understanding of the hydrodynamic and salinity regimes. In the literature, most restoration projects are found in the USA (e.g. in Louisiana: Boustany, 2010; New Hampshire: Morgan and Short, 2002). A coastal restoration example from Louisiana (USA) involved a wide variety of vegetative, hydrologic and structural methods. More generally, the two main project types (sediment-oriented) are

distinguished: a) rapid reclamation of coastal land through the mechanical extraction and delivery of dredged sediments (Aust, 2006; LCPRA, 2012; Merino *et al.*, 2011) and b) large-scale river diversion projects designed to mimic the alluvial land building process (e.g. Allison and Meselhe, 2010; Nittrouer *et al.*, 2012; Simenstad *et al.*, 2006).

According to Simenstad *et al.* (2006), ecosystem recovery in wetlands can be divided in passive, active, and creative approaches. In the first case, removing barriers or stopping a degrading activity on a disturbed ecosystem can lead to its reinstatement either partly or fully (e.g. Bos *et al.*, 2002). On the other hand, more engineered activities are involved in active restoration approaches, in an attempt to recreate wetland processes and structure. These activities may involve re-establishment of tidal hydrology and/or planting vegetation to facilitate the growth of native marsh vegetation and limit/eliminate invasive species (Bos *et al.*, 2002).

Whatever the approach may be, the aim and the success of any restoration project is to "transform" a coastal wetland again to a functioning ecosystem, by reviving some natural dynamics and disturbances into the system (Middleton, 1999).

2.2.1. Wetland rehabilitation in the Mediterranean area

The Mediterranean region's wetlands are located primarily at low altitudes, while many of them are defined as coastal (Figure 10) (Hollis and Jones, 1991). According to historical references, early Mediterranean civilizations were basically based around coastal areas, as well as wetlands, and depended on them for food, water and also building materials. Consequently, the majority of the natural wetlands of the Mediterranean region have been drained (Anonymous, 1992). During the first half of the 20th century, approximately one third of the wetlands of the Mediterranean region has been drained, in order to create more space for megacities, farmland, industrial infrastructure and tourist facilities (Zalidis *et al.*, 1999). In addition, many of the remaining wetlands are threatened by modifications of their natural functions, as a result of the pollution, as well as over-fishing and hunting, excessive pumping and partial drainage (Zalidis *et al.*, 1999).

The major problem of the Mediterranean wetlands is caused arguably because of the steadily growing water demands. The primary reason is the constantly increasing population, especially in the southern coastal zones. Agricultural practices also threaten Mediterranean wetlands. The drainage of wetlands for arable land has been underway for centuries, but probably reached its peak during the last 60 years. More specifically, in the Mediterranean region 73% of water consumption is intended for irrigated agriculture.

Moreover, the wetlands of the Mediterranean region shelter a rich and diverse community of rare and threatened plants and animals and they play a dominant role in the protection, as well as the maintenance of biodiversity. More specifically, 8 out of the 29 globally threatened species of birds occurring in the Mediterranean are wetland species (Skinner and Zalewski, 1995).

The continuous alterations of the wetland functions throughout the Mediterranean region and the consequent degradation of wetland values may be addressed by adopting a conservation strategy for all Mediterranean wetlands (Zalidis *et al.*, 1999). Such a strategy includes prevention of future impacts through the regulation and correction of the effects of past negative human impacts through comprehensive management plans, which contain rehabilitation.

Figure 10. A coastal wetland in Glyfada area (Western Naxos, Cyclades, Greece) along with manmade structures in the coastal zone. Photo: N. Evelpidou, 2017.

Nowadays, attempts to rehabilitate degraded wetlands are becoming more and more ample in the Mediterranean region (Zalidis *et al.*, 1999). In recent years, several Mediterranean countries have embarked on rehabilitation and establishment of wetland areas, with Greece among them, where wild fauna and flora are given special priority (Moller, 1995).

2.2.2. Marshes and rehabilitation

Generally, marshes are among the most productive ecosystems in the world (Tiner, 1984), as they provide important life support, water quality and hydrologic functions. Both the structure and function of marshes are quite similar worldwide. Marshes serve as nurseries for many juvenile fish and shellfish species, as well as habitat for many aquatic organisms, birds and wildlife. Other valuable functions include the protection from coastal erosion, stabilization of dredged material, dampening the effects of waves and storm tides, trapping water-borne sediments, nutrient cycling and transformations, and, also, serving as nutrient reservoirs (Matthews and Minello, 1994a). Like other wetlands, tidal marshes have often been regarded as a hindrance to more productive land use.

Losses of marshes have derived from dredging, filling, diking and embankment construction for such purposes, as for example navigation channels, urban development, agricultural production, oil and gas exploration and construction projects, for instance highways, airports,

pipelines and port facilities. Marshes have also been impacted by pollutants, such as oil or chemical spills, as well as natural losses because of land subsidence, sea level rise and erosion that constitute continuous processes. In recent years, the method of "marsh creation" is widely used; coastal land is reclaimed rapidly by the mechanical extraction and delivery of dredged sediments (Aust, 2006; LCPRA, 2012; Merino *et al.*, 2011). The ultimate goal of marsh creation and rehabilitation is the establishment of self-sustaining ecosystems, which resemble in structure and function to the natural systems that they are designed to emulate or one that will become like the natural system through succession of the flora and fauna (Broome, 1990; Zedler, 1992). More specifically, marsh rehabilitation refers to partially returning an area from a disturbed or totally altered condition to a previously existing condition through human activity. In contrast, creation means conversion of a non-wetland area into a wetland. The rehabilitation and creation of marshes are often used for mitigation to compensate for wetland loss (Lewis, 1990; Kentula *et al.*, 1992). Nowadays, technology has been developed and successfully applied in order to create and rehabilitate marshes (Matthews and Minello, 1994b). Rehabilitation efforts may be accomplished through the recovery of the initial topography and elevation. Nevertheless, a decent amount of rehabilitation projects can be accelerated by means of seeding or transplanting the dominant vegetation (Zedler, 1992; Seneca and Broome, 1992; Broome and Craft, 2000).

Finally, another method that is used exclusively in the downstream areas of fluvial systems is the one of large-scale river diversion, aimed at imitating the alluvial land building process (Allison and Meselhe, 2010; DeLaune *et al.*, 2003; LCPRA, 2012; Nittrouer *et al.*, 2012; Simenstad *et al.*, 2006).

3. Study cases

3.1. Study case 1: Restoration of Pobra do Caramiñal saltmarsh (NW Iberian península)

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1. Background

The saltmarsh of Pobra do Caramiñal is located in the Ria de Arousa, on the SW coast of the province of A Coruña (Galicia, NW Iberian Peninsula). It is a small saltmarsh complex with a total area of 14 ha, located in an increasingly touristic area. In 2000, the Spanish Ministry of Environment launched a nationwide program to restore degraded and irregularly occupied coastal areas. Within this context, a plan to restore the salt marshes of Pobra do Caramiñal started to develop in 2000 and was finally executed 14 years later.

Alterations were first assessed by photographic interpretation of a 1954 flight in which the saltmarsh showed an acceptable conservation status. Subsequent aerial photographs (years 1968, 1976, etc.) showed that a significant portion of the area was already occupied, both for the construction of public infrastructures (roads and a water treatment plant) and for privately owned filled areas under irregular concessions. Occupied areas represented more than 50% of the saltmarsh complex in 1999, mostly corresponding to a private filled area (Figure 1).

Additionally, wastewater discharges, presence of invasive species (*Acacia melanoxylon, Arundo donax, Cortaderia selloana),* riparian forest degradation, and strong alterations of the hydrologic system are also worth mentioning (Figure 2).

The project carried out a detailed analysis of water, soil, and sediment quality; a description of the animal and plant biocenosis of the saltmarsh and its environment; a detailed study of the occupation of the saltmarsh complex by continued filling activities during the 1956-99 period, and an inventory of habitats of community interest as defined by Directive 92/43/CEE (Habitats Directive). This project was made by Eurcoin, CIISA, Tragsa companies in collaboration with Santiago de Compostela University. Finally, a number of interventions aimed at recovering the main habitats and improving water, soil, and sediment quality were proposed.

Figure 1. Aerial view of the Pobra do Caramiñal saltmarsh in 2014 (A), before the start of the interventions, and in 2017 (B), after the interventions. In the 2014 aerial photograph, sections (1), (2), and (3) indicate areas where fillings occupying more than 50% of the saltmarsh area were removed. Section (4) corresponds to the area where invasive species were removed and channels and riparian forests were restored. In the 2017 aerial photograph, sections (1), (2), and (3) correspond to newly restored saltmarsh areas.

Figure 2. Summary of activities carried out for the restoration of the environmental quality of the Pobra do Caramiñal saltmarsh. A) Overview of the old wastewater treatment plant built in the saltmarsh (filled area 3; Figure 1). This plant was removed and part of the saltmarsh was restored (Figure 1 B); B) Removal of filled area 2 (Figure 1); C) Dredging of polluted sediment; D) Protection barriers for containing suspended materials during sediment dredging; E) Hydrological restoration. Recovery of secondary channels within the saltmarsh; F) Removal of urban solid waste; G) Removal of invasive plant species and restoration of the riparian forest.

2. Previous study

A study on the quality of the system was carried out, focusing particularly on the concentration of toxic (Pb, Cu, Ni, Zn,…) and eutrophicating (P, N) elements and on the conservation status of its habitats.

Water, soil, and sediment quality

Water quality in the saltmarsh showed severe impacts in the vicinity of wastewater discharge points, where high concentrations of ammonium (1.5-14 mg L^{-1}), nitrite (0.1-0.7 mg L^{1}), orthophosphate (0.2-11 mg l^{-1}), and DBO5 (50-200 mg O_2 L^{-1}) were reached. Likewise, surface enrichment of certain toxic elements such as Pb (0.25-0.5 mg L^{-1}), Co (0.15-0.22 mg L^{-1}), Ni (0.20-0.32 mg L^{1}), and Cu (0.03-0.04 mg L^{-1}) was also detected. The situation improved substantially at high tide due to the diluting effect of the tidal flow.

Soils and sediments showed high surface concentrations of organic matter, nitrogen, and phosphorus. Concentrations of toxic metals were not high, consistently with the urban origin of waste and with the lack of intense industrial activities in the area. Nevertheless, enrichment in metals such as Pb, Zn, Cu, and Ni was clearly demonstrated in areas close to sewage discharge.

Highly reduced (redox potential <100 mV), black soils with a strong smell of hydrogen sulfide (H_2S) were also associated with sewage discharge areas. These are indicators of oxidation-reduction conditions characteristic of sulfate-reducing environments, where microorganisms oxidize organic matter using marine sulfate and generating H_2S, which is highly toxic for plants and fauna.

Environmental value of habitats

The habitats of community interest that are present in the area, according to Directive 92/43/CEE, are: riparian forest (residual alluvial forests of *Alnion glutinoso-incanae 91E0**), salt meadows (*Glauco-Puccinelietalis 1330*), and estuaries (code 1130). The first one is an arboreal strip around rivers and wetlands, which in the past may have completely occupied the saltmarsh and river banks. However, nowadays it constitutes a residual formation, present only in the back of the saltmarsh, while in the rest of banks it is highly degraded and has been replaced by replanted eucalyptus and acacias (Figure 1, 2).

The second of the habitats (*Glauco-Puccinelietalis* salt meadows) includes coastal saltmarsh vegetation. This habitat includes several vegetal associations whose distribution along the saltmarsh is determined by salinity and degree of soil reduction (see Sánchez *et al.*, 1998, Sánchez, 2010, among others), which in turn are largely determined by the physiography of the saltmarsh itself (Sánchez *et al.*, 1996).

Intertidal areas show marked gradients in terms of salinity and degree of flooding. Knowledge of the geochemical processes characterizing these environments is an essential aspect for restoration activities. For Galician saltmarshes, a clear relationship has been established between salinity and hydromorphic intensity of each plant community, which is determined by their redox potential (Eh). Higher or lower degrees of flooding of the saltmarsh determine the degree of soil aeration; therefore, coastal saltmarsh soils can show important variations in salinity, primarily depending on the relationship between tidal influence and freshwater input,

mainly from rivers, as well as variations in oxidation-reduction conditions (redox processes). Prolonged flooding leads to soil reduction. Depending on the intensity of reduction, soils can show suboxic or anoxic conditions. Suboxic conditions entail the dissolution of Fe and Mn oxides and hydroxides, releasing their reduced soluble forms (Fe^{2+} y Mn^{2+}) into interstitial water; these forms are phytotoxic, mainly for dicotyledonous species such as *Halimione portulacoides* (Figure 3). However, in permanently flooded soils, surface conditions are already anoxic, characterized by the reduction of sulfate ion to hydrogen sulfide (H_2S), which is highly phytotoxic for most saltmarsh species except for *Spartina maritima*. The ability to endure salinity and phytotoxicity of reduced forms determines the distribution of communities within the saltmarsh. Figure 3 shows the geochemical conditions that characterize each one of the main plant communities in Galician saltmarshes.

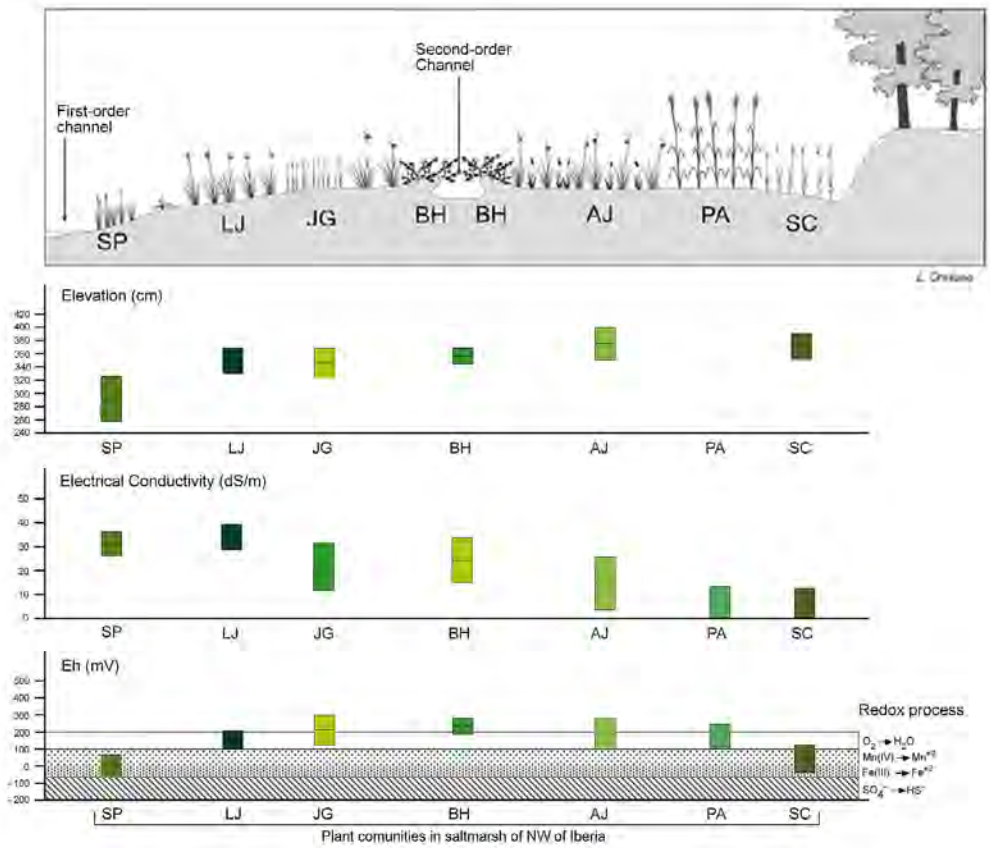

Figure 3. Schematic profile of the NW Iberian Peninsula saltmarshes. SP: Spartinetum maritimae: this community is the only one able to colonize saline, highly reduced substrates with high hydrogen sulfide concentrations in interstitial water. LJ, JG, AJ: Juncus maritimus communities: LJ: Limonio-Juncetum maritimae subass. typicum; JG: Limonio-Juncetum maritimae subass. Juncetosum gerardi, AJ: Agrostio-Juncetum maritimae. Reed beds occupy most of the higher saltmarsh, with soils that present suboxic oxidation-reduction surface conditions. The three formations are distributed within the saltmarsh according to salinity: LJ occupies the most saline environments (electric conductivity: 30-40 dS/m), while AJ occupies the least saline ones (electric conductivity: 10-25 dS/m). BH: Bostrychio-Halimionetum portulacoidis. This community appears mainly on soils with oxic surface conditions and is therefore closely associated with saltmarsh channel banks, where tidal flooding periods are short and aeration is favoured by the channel. When soils are sandy, it can expand towards the inner part of the saltmarsh. It tolerates wide variations in electric conductivity, while redox potential must be over 300 mV. SC: Scirpetum compacti. This community occupies saltmarsh banks connected to continental areas, which show strongly reduced conditions with high sulfide concentrations but, unlike in the case of Spartina maritima, flooding water must be either fresh or brackish water. PA: Phragmitetum australis community. Reed beds are also associated with saltmarsh banks flooded by circulating freshwater; therefore, redox.

Finally, the estuarine habitat corresponds to the lower portion of the saltmarsh. It does not cover a large area, but it has a high ecological interest since it represents the confluence of three different environmental systems: beach-dune, saltmarsh, and marine environment. Tidal flow and its associated fish species, such as eels, go through this area.

3. Action proposal

According to the previous diagnosis, the following interventions were proposed: 1) landscape improvement: removal of artificial elements and urban solid waste, 2) removal of introduced species and protection of exceptional specimens, 3) preparation of the saltmarsh environment for recreational use, and 4) restoration of the saltmarsh and riparian forest, the latter being the most relevant action within the project as a whole.

Restoration of the riparian forest

The first intervention corresponded to the removal of invasive species occupying a large portion of the saltmarsh banks, as well as of waste of diverse origins. The main species removed were *Acacia melanoxylon, Eucalyptus globulus, Arundo donax*, and *Cortaderia seollana*, and the riparian forest was restored by planting the main species that characterize this habitat: *Alnus glutinosa, Salix atrocinerea, Fraxinus angustifolia, Quercus robur, Corylus avellana*, and *Sambucus nigra*. Additionally, microhabitats within riparian forest areas (specifically small temporary pools, important for reproduction of amphibians) were restored (Figure 2).

Restoration of the saltmarsh system

Recovery and restoration of the saltmarsh area was the main action within the project. Using aerial photographs from 1954 as a reference, it was determined that 50% of its initial area was occupied by filled areas of different nature. The inner portion and banks of the saltmarsh (approximately 10%) were urbanized and were therefore impossible to restore. Recovery of the saltmarsh area focused on areas used for storage of materials or occupied by an old water treatment plant built in the wetland area. The area covered by these filled areas was approximately 4 Ha, and the filling was between 1 and 4 m thick. Total recovery of the saltmarsh involved the extraction of large volumes of filling material and its transportation to an authorized dumping site. Taking into account the peri-urban nature of the saltmarsh, the decision was made to remove only the filled areas in the center of the saltmarsh and part of the ones on the banks, which represented approximately a 50% increase in the saltmarsh area. The remaining filled area was transformed into an island that served as refuge and breeding area for Anatidae species, and riparian forest species were replanted along saltmarsh banks with the aim of generating a green barrier. Consequently, there was a 50% increase in shrub area as well.

The main environmental challenge was to reach full restoration of saltmarsh vegetation in the new areas created by removing filled areas. For this, a detailed land survey was required to determine the height of the newly generated saltmarsh surface so that its degree of flooding could be predicted. Likewise, a channel system surrounding the saltmarsh and the island was designed to protect aquatic birds from predators such as foxes, cats, or dogs.

Figure 4. Photographs showing the area before and after the intervention in filled area 1 (see Figure 1). A) Filled area 1 before the intervention; B) and C) show its aspect immediately after removing the filled area; D) shows the aspect of the new intertidal surface once repopulated with Juncus maritimus and Halimione portulacoides, the latter only on the channel banks (January 2015); E), G), and F) show different panoramic views of the same area 18 months later (June 2017).

Figure 4 shows different saltmarsh restoration stages. Seedlings used for restoration were obtained from the saltmarsh itself, using a hand shovel to cut small areas (400 cm^2) and directly planting them on the newly generated surfaces. For this, it was essential to know the characteristics of each environment, as indicated in Figure 3, so that the most suitable species were assigned to each environment according to its geochemical soil conditions. Thus, *Halimione portulacoides* was replanted in channel banks, *Juncus maritimus* in lower tidal plains, and *Scirpus maritimus* in higher areas with greater freshwater influence (Figure 4). Contrarily to other interventions in similar environments, the restoration of the saltmarsh area can be considered completely satisfactory in this case, to the point that only a few years later it was hard to distinguish the regenerated sections from the previously existing ones. Additionally, the new areas are being recolonized by new species such as *Salicornia ramosissima* or *Aster tripolium*.

3.2. Study case 2: Coastal degradation and dune habitats in Tuscany coasts (Italy): the study case of the Regional Park of Migliarino San Rossore Massaciuccoli

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Sandy coastal ecosystems, occupying the transition zone between marine and sedimentary terrestrial environments, are susceptible to constant changes in their morphological structure and vegetation landscape (Brown and McLaghan, 2002; Maun, 2009). At the same time, they show a great biodiversity, in terms of plant species and communities, often, where not overly altered, along a well-defined zonation (Prisco *et al.*, 2012). The beaches and dune habitats that occur there, are particularly fragile and vulnerable environments as a result of the dual threat posed by coastal erosion and human impact, both evidently ongoing phenomena (Doody, 2013). This makes them particularly worthy of attention and protection for their specific ecosystem functions (Proovost *et al.*, 2011).

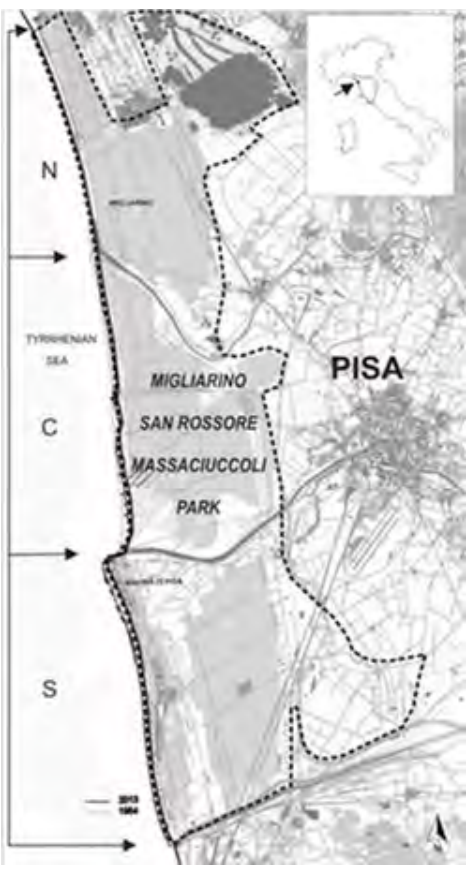

Figure 1. The study area.

Italy has a coastline of about 7,500 km, of which approximately 37 % is represented by rocky coasts and 63 % by sandy coasts, characterized by a great diversity of habitats of high natural and environmental interest (Biondi *et al.*, 2012). Over the 22% of coasts has a permanent soil anthropic consumption (ISPRA, 2016). To this picture it must be added the phenomenon of coastal erosion, which affects, to differing degrees, around 45% of sandy coasts (Valpreda and Simeoni 2003). In most cases, human pressure and coastal erosion act not only in a dominant, but often in a synergistic mode, making these habitats highly vulnerable and threatened, as is recognized by the international community itself (Carranza *et al.*, 2008; Bertacchi and Lombardi, 2014).

An example of great interest is represented by the coast of the Regional Park of Migliarino San Rossore Massaciuccoli (Tuscany, Italy) which, on a total length of about 30 km of coastline, shows some stretches of coastline in progradation (N sector), some in erosion (C sector), still others in balance (S sector), all stretches with very different issues of human pressure (Bertacchi, 2017) (Figure 1). The dune habitats (in accordance with Directive 92/43/EEC), are: 1210 Annual vegetation of drift lines; 2110 Embryonic dunes; 2120 White dunes; 2210, 2230, 2240 *Crucianellion* fixed dunes and *Malcomietalia* and *Brachypodetalia* dune grasslands; 2250 Coastal dunes with *Juniperus* spp.; 9340 *Quercus ilex* and *Quercus rotundifolia* forests and 2270 Wooded dunes with *Pinus pinea* and/or *Pinus pinaster*. However, this richness of habitat corresponds to a serious state of degradation of the same, in terms of alteration, fragmentation and disappearance, in over two thirds of the coast examined.

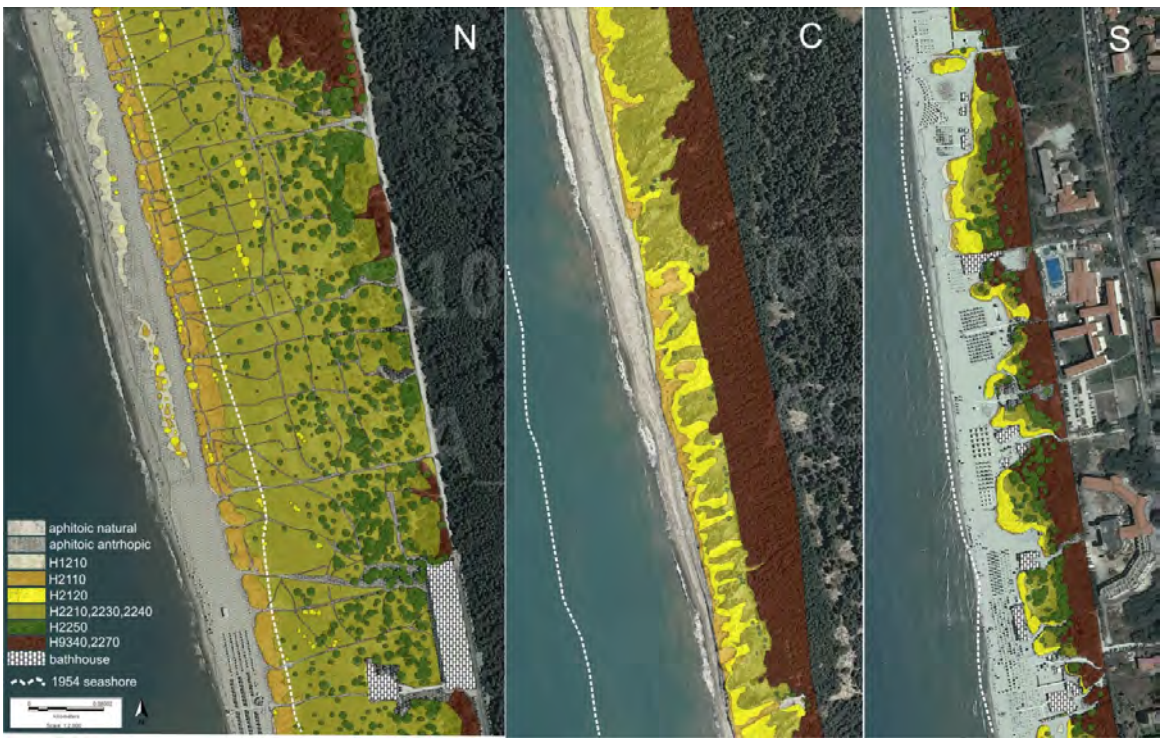

Figure 2. Excerpts from the ortho-photomap of habitats. (N, C, S see text). Aerial photos are from http://www.regione.toscana.it/-/geoscopio and licensed under Creative Commons Attribution CC-BY3.0 IT.

Figure 3. Dune erosion between 1954 and today. Aerial photos are from http://www.regione.toscana.it/-/geoscopio and licensed under Creative Commons Attribution CC-BY3.0 IT.

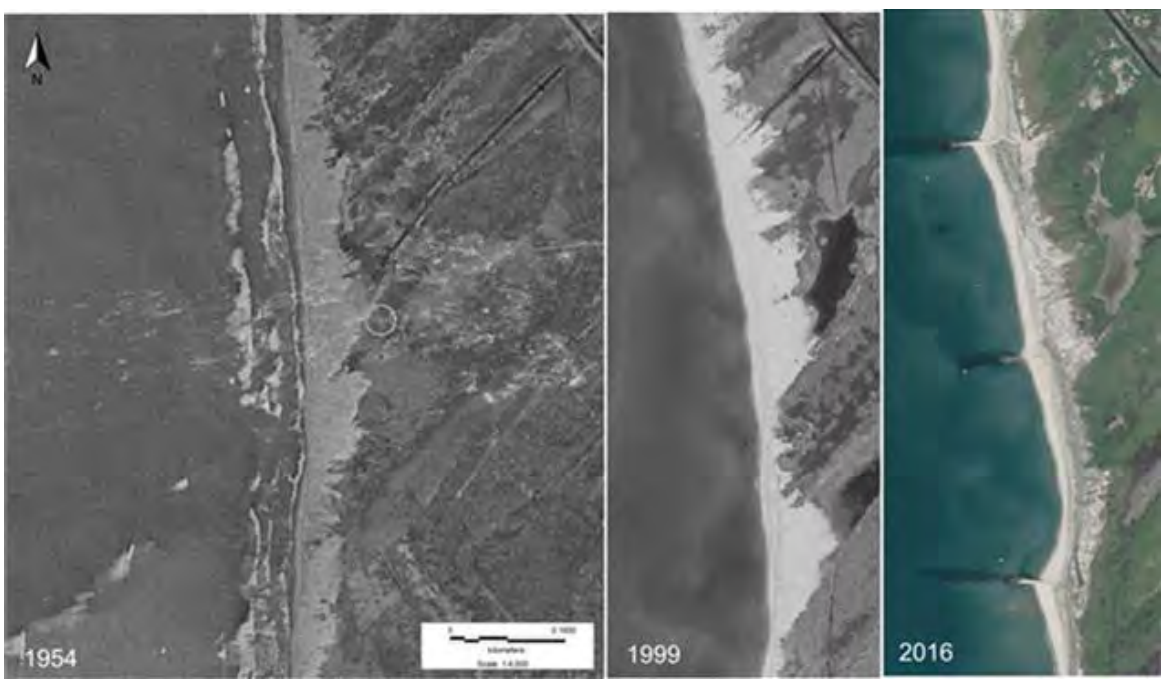

Figure 4. Effects of orthogonal breakwaters on erosion restraint. Aerial photos are from http://www.regione.toscana.it/-/geoscopio and licensed under Creative Commons Attribution CC-BY3.0 IT.

As part of the study of the degradation dynamics of coastal environments, the comparison of historical aerial photos with the current ones was very useful in the survey. In the northern sector (N) the progradation of the coastline has led to a displacement of dune vegetation to the west with a total area unchanged compared to 1954 (ca 200 ha); in the central sector (C), due to erosion, around 200 ha of sandy shore have been lost since 1954, leaving today only a thin strip of dune of about 55 ha. In the southern sector (S), although a substantially static coastal dynamics from 1954 to today, there has been a loss of about 100 ha of dunes of the approximately 145 ha present in 1954. This is due to the expansion of the bathing establishments and the permanent and temporary structures connected to them (Figure 2). These different dynamics have involved different arrangements in the dune vegetation landscape. In the northern sector, habitats related to fixed dune (2210, 2230, 2240 and especially 2250) are those in greater expansion than detectable by 1954 aerial photos, and *Juniperus oxycedrus* shows a great spread and the biggest problem in this sector is represented by the anthropic trampling. In the central sector the high erosion has determined the substantial disappearance of some habitats (1210, 2110, 2250) and the significant degradation of others (2120), with the almost disappearance of characteristic species (eg *Echinophora spinosa* and *Juniperus oxycedrus*) and the dominance of other (eg *Euphorbia paralias*). In the central sector, dramatically affected by erosion, the attempts over time to reconsolidate the dune with plantings and woody fences or the creation of orthogonal breakwaters aimed at containing the offshore transport of sand sediment, have proved to be ineffective in the first case and only partially effective in the second, as is evident from the comparison of the aerial photographs of 1954, 1999 and today (Figure 3, 4). In the high anthropized southern sector, the habitats not completely eliminated by the anthropogenic occupation are almost always extremely fragmented with a mixture of species and phytocenosis of otherwise differentiated habitats (Figure 5).

The different vulnerability of habitats to coastal degradation is clearly related to the different ways in which this is expressed and where it occurs. The marine erosion, is selective towards those species and plant communities that are not able or not fast enough to “follow” the erosion process inwards. The anthropic pressure, where is permanent (bathing facilities, buildings, car parks, etc.) clearly causes the elimination of all habitats, when it is temporary/partially, the persistence of species gives rise to a quick recovery of plant communities.

Figure 5. Three different levels of anthropic degradation (from top to bottom): very low, medium, total. Aerial photos are from http://www.regione.toscana.it/-/geoscopio and licensed under Creative Commons Attribution CC-BY3.0 IT.

3.3. Study case 3: Rehabilitation of coastal wetlands from Greece

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Lake Kerkini wetland

To start with, one of the major rehabilitation projects in Greece has been the creation of Lake Kerkini in Macedonia (Northern Greece) (Figure 1) through the damming of Strymon River in 1930. This point of view is based on the argument that the present lake was created on a location, where a much smaller natural lake existed for ages. Gradual sedimentation of the natural lake had led to a marked degradation of its flood storage and desynchronization function. The sedimentation of Lake Kerkini is caused by the high rates of sediment transport by Strymon River from the Bulgarian, Serbian and Former Yugoslav Republic of Macedonia territory drainage basin of Kerkini of 11,600 km^2. The sediment transport and the deposition volumes and rates were high during the first stages of Kerkini's operation, due to deforestations and stockbreeding activities against forested and agricultural areas in southwestern Bulgaria before World War II, which intensified erosion and deposition phenomena in Kerkini's catchment. Sedimentation was significantly reduced due to the anti-erosion hydraulic works and extended reforestations mainly in the period 1962-1977 (Psilovikos and Margoni, 2010).

There have been major changes in the local environment at the area of Kerkini most of which are directly related to the building of a new dam in 1982. This was a large project that affected the natural environment a great deal as it allowed the lake's water to reach a maximum level of 36 m above sea level, 4 meters higher than the previous dam. The annual water level range is now 5 meters. This influenced many aspects of the natural landscape. The most important effect was the damage to the riparian forest and the decrease of the aquatic vegetation including large beds of reeds and wet meadows due to the increase of the water depth and the duration of flooding.

During the first years from the new works a large bed of white water lilies grew in the lake covering an area of 325ha. However, continuous rise of the water's maximum level in the early 90's caused the water lilies to decrease. By mid-90's the area that they covered was no more than 50 to 80ha. The riparian forest was also exposed to water for more time than the trees could cope with. As a result, in the 90's the forest area had diminished by more than half, down to 325ha compared to the 700ha that it occupied in the early 80's. Scientists

admit that cormorans nesting on the forest also cause some damage to the trees. The above changes have meant large-scale damage to the birds' and fish habitats and have resulted to the reduction of various birds and fish species' populations. Logging of the forest by residents of the nearby villages has also been mentioned as one of the reasons for the forest decline.

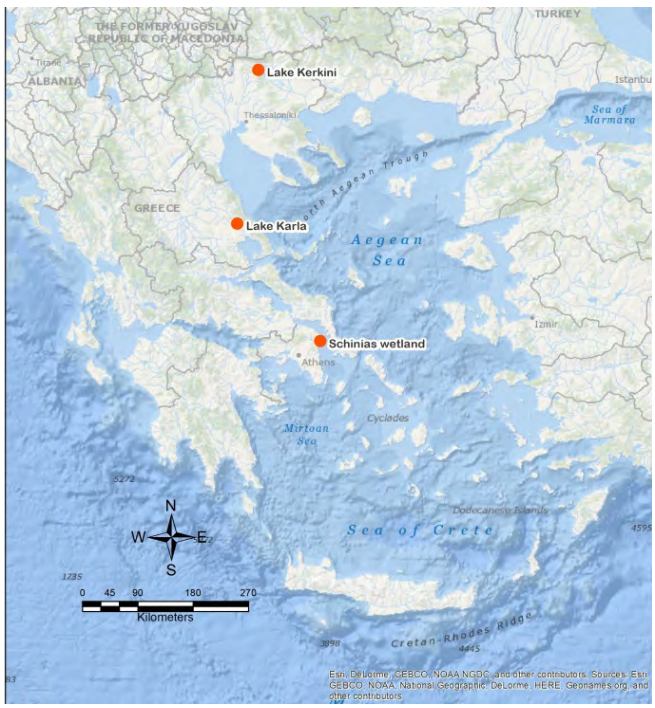

Figure 1. Location of Greek sites, with rehabilitation projects mentioned in the text.

Destruction of the lakeside vegetation is also attributed to buffaloes and cattle grazing uncontrollably. Grazing space next to the lake is not enough to support the needs of the big number of sheep and goats present at the Kerkini area, especially during the summer. The result is overgrazing that doesn't allow the re-establishment of reeds and other vegetation and is considered one of the reasons for the declining vegetation. Hunters also mention that natural habitats for animals have decreased in the mountains due to intensive logging and road building activities.

Finally, Lake Kerkini lies between the most important Greek wetlands, hosting the largest water buffalo population, many specialized aquatic organisms and waterfowls. In addition, it is extremely important for provision of irrigation water. Nowadays, the functions and values of Lake Kerkini are being threatened by increased sedimentation (Zalidis *et al.*, 1999).

Lake Karla wetland

During 1998-1999 a major rehabilitation project was conducted, both for Greece and the Mediterranean, with the support of the European Commission. It refers to Lake Karla in Thessaly (Central Greece) (Figure 1). Lake Karla (ancient Lake Voiviis) occupied the lower depression plain of Thessaly region and was one of the most important wetlands in Greece until the 60's.

The natural basin of the former Lake Karla covered an area of 1663 km^2, of which more than 600km^2 made up a southern flat plain, while the east part is surrounded by mountains and hills. But, after the construction of complimentary works for the reservoir, the drainage area of the restored Lake Karla has increased to 1171 km^2, with a perimeter of 228 km (Loukas *et al.*, 2008). Elevation ranges from 50 m to more than 2000 m and the mean elevation of the region is about 230 m. The climate of the area is typical continental, characterized by cold and wet winters and hot and dry summers. The mean annual precipitation in the watershed of the lake is about 560 mm and is distributed unevenly in space and time. The mean annual potential evapotranspiration is about 775 mm and the mean annual temperature is 14.3^{o}C (Vasiliades *et al.*, 2009; Sidiropoulos *et al.*, 2012).

The geological structure of the area consists mainly of recent grains of various sizes originating from the lake's deposits. The plain consists of aquiferous, essentially sandy intercalations, separated by layers of clay to silty-clay and is bound by schists and karstic limestones or marbles. Impermeable geological structures cover a 30.6% of the total area and are located on parts of the surrounding mountains, karstic structures cover a 14.5% and are located on the Mavromouni Mountain at the north-east part of Lake Karla watershed and finally permeable structures, which appear mainly in the plain, cover a 54.9%. The sedimentary aquifer of the lake occupies the largest part of its plain with an extent of 500 km^2 and is being over-exploited covering both the irrigation needs of the cultivated areas as well as the supply water needs of the settlements (Sidiropoulos *et al.*, 2011).

The whole basin area experienced severe, extreme and persistent droughts during the periods from mid to late 70s, from late 80s to early 90s and the first years of 2000 (Loukas *et al.*, 2007; Sidiropoulos *et al.*, 2012). The plain is one of the most productive agricultural regions of Greece, while the industrial sector, mainly based on agricultural and livestock activities, is also developed. The main crops cultivated in the plain area are cotton, wheat and maize.

Former Lake Karla occupied the lowest part of its natural basin and was considered as one of the most important wetlands in Greece until 1962. Surface runoff from the watershed and floodwaters of the Pinios River (discharging via the Asmaki ditch) supplied the lake with large quantities of freshwater. Its surface area fluctuated between 40 km^2 and 180 km^2. In 1962, complete drying of the lake took place in order to create more land for agriculture and to avoid the flooding of the low elevation lands because of its surface area fluctuations. In terms of biodiversity, the former Lake Karla endowed with a variety of habitats (pelagic, floating vegetation, shallow marshes with *Juncus sp.* and *Typha sp.,* emergent vegetation and rocks), had the ability to support a rich fish and bird fauna (Ananiadis, 1956).

The re-constructed Lake Karla occupies the lowest part of the former Lake Karla. It lies between latitude 39°26'49'' to 39°32'03' N and longitude 22°46'47'' to 23°51'50'' E and has a surface of 38 km^2. It is characterized by its shallow depth with a maximum water depth of 4.5 m and a mean depth of 2 m. It is adversely affected by both agricultural and industrial operations acting as a sink of fertilizers and agricultural effluents. Nowadays, Lake Karla is listed in the network of the Greek protected areas as it is considered a vital aquatic ecosystem, both in terms of biodiversity (it is a Natura site GR1420004, a Ramsar site and a SPA site for birds) but also as a newly re-established water resource that was drained in the 60's.

Several engineering and technical studies recommended draining the lake via the Karla tunnel and building a smaller reservoir instead of the natural lake for flood protection and for the

revelation of agricultural fields (Zalidis *et al.*, 1999; Sidiropoulos *et al.*, 2012). Although the lake was drained in 1962, the suggested reservoir was never built and only a small marsh was left with an area of 4 km^2, in order to satisfy the irrigation needs of the surround cultivations. This delay has created a series of environmental problems with anthropogenic impacts including wetland loss, significant drawdown of the aquifer's water table, subsequent soil salinization and loss of ecological and aesthetic value. The remaining small, temporary marsh was unable to support the aquatic food web, and numerous species were extirpated or emigrated. Furthermore, the remnant wetland still received discharge from a large part of the watershed, but it was too small for effective nutrient removal/transformation and sediment/toxicant trapping. These environmental problems affected directly the local economy through lower family income and higher social instability associated with reduced crop production and elimination of fisheries.

At present, the reservoir of the lake is almost refilled, while the restoration project of the wetland is still ongoing.

Schinias wetland

Another rehabilitation project that has taken place in Greece was at the coastal site of Schinias, which is located in the Marathon plain, 45 km NE of Athens (Figure 1). This area is characterized by a wide variety of natural habitats, such as coastal sand dunes covered by a pine forest consisting of *Pinus pinea* and *Pinus halepensis* (Figure 2), a rocky peninsula covered by Mediterranean maquis; the forest covers an area of about 1.20 km^2 on a sandy strip about 400 m wide, a freshwater spring (Makaria spring) an important diversity (320 species) of flora and fauna species, including threatened resident and migrating birds and a coastal wetland that covers an area of approximately 7 km^2. It is worth mentioning, that only a few coastal wetlands still exist in the coastlines of SE Greece and the Aegean Islands, being nearly dry and covered by Mediterranean vegetation, such as maquis and phrygana (Hadjibiros, 2010).

Figure 2. General view of Schinias wetland and the pine forest. Photo: N. Evelpidou, 2012.

Constant and considerable anthropogenic pressures have been forced on the area of Schinias for many years. In 1923 the wetland was partially drained, when the water of Makaria spring was rerouted to the sea. As a result, only a small percent (about 10%) of water inflow remained available for the wetland and the largest part of the area was converted in a semi-dried marsh, in which military bases, a power plant and a lap- sized civil airport were established.

However, after the Olympic Games conducted in Greece, significant improvement of the natural landscape has been achieved (Panagiotidis and Zogaris, 2009). The channeling of water deriving from Makaria spring increases the available quantity of freshwater in the wetland, providing better hydrological conditions of groundwater. The natural annual fluctuation of the water's presence in the wetland constitutes an essential restoration of the function of the ecosystem, as it increases the attractiveness of the area for several bird species. The decrease of disturbing activities, such as the disestablishment of the landing field and the removal of the industrial infrastructure, strengthens the naturalness of the existing ecosystems and has also restored the naturalness of the wetland (Hadjibiros, 2010). In addition, the freshwater fish fauna has increased. At least five species have been observed; among them, the endemic *Pelasgus marathonicus* whose presence in the water of the Rowing Centre has been ascertained. Bird diversity goes up spectacularly: 117 species were recorded in Schinias before 1997; 236 species have recently been recorded. This number includes at least 52 species that reproduce regularly in the area. The number of aquatic birds that winter there is now greater than any other count before 2004 (Panagiotidis and Zogaris 2009; Hadjibiros and Sifakaki 2009).

3.4. Study case 4: Dune rehabilitation example from the Netherlands

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In the framework of LIFE, ZENO (Zwin dunes Ecological Nature Optimisation) project, by the Flemish government's Agency for Nature and Forests, focused on the area Zwin Dunes and Zwin Polders Flemish nature reserve at Knokke-Heist with a restoration project. After an extensive study of biotic and abiotic factors by Zwaenepoel *et al.* (2007), a management plan was approved, which included a number of ecological engineering interventions such as the removal of vegetation (shrub and exotic tree species), the remodeling or creation of ponds and the removal of old infrastructure (war remnants, old horse jumping etc.), hydrogeological interventions, together with a restoration of the micro-topography in the Kleyne Vlakte dune ecosystem (advised by Zwaenepoel *et al.* (2007)). What these interventions aimed for was the rewetting of the Kleyne Vlakte with water stored within the nature reserve so that it would no longer drain towards the polder area. The project was completed in 2010, and according to the report the first results were optimistic.

Van der Hagen *et al.* (2008) investigated three restoration projects of coastal dunes in the Netherlands. The three projects were based on the conceptual Model of Dunes (Bakker *et al.*, 1979), with either abiotic or biotic restoration. Hydrological restoration activities included the removal of extraction wells and with small-scale sod-cutting, however pioneer communities did not return as expected. In the areas Van Limburg Stirum and Kikkervalleien, the activities included hydrological and geomorphological restoration, with blowing sand and reduction of the groundwater extraction; according to Van der Hagen *et al.* (2008), the abiotic restoration initiates a promising situation. Future plans need to take into consideration the influence of the remaining vegetation, as well as the ability of seeds of the target species to reach the restored area (Van der Hagen *et al.*, 2008).

Conclusions

It becomes clear that the basic “guide” for a successful rehabilitation project is the fact that any remediated ecosystem should be self-sustainable and resilient to disturbances. However, in order to understand whether the coastal ecosystem is self-sustainable and resilient in the long term or not, a long term monitoring is required (Lithgow *et al.*, 2013), which is usually quite difficult and expensive to achieve.

The complexity, as well as the several parameters that affect the coastal zone, make rehabilitation projects extremely difficult to achieve their goals. According to Hopfensperger *et al.* (2007), chances are increased, if concurrent science-based and socially acceptable decisions are taken into account. Fundamental focus of each rehabilitation activity should be the recording and understanding of both the structure and the function of the coastal ecosystem, its state of degradation, as well as the possibilities of minimizing the factors contributing to degradation.

For the selection of coastal sites for rehabilitation, multiple criteria must be taken into account (Lithgow *et al.*, 2013), deriving from geomorphological, ecological processes and from human activities. As a result, a multidisciplinary approach, with geomorphologists, ecologists and other scientists co-operating, becomes more and more valuable and necessary, in order to redevelop functions in degraded coastal landforms and habitats (Jackson *et al.*, 2013).

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IV Soil Degradation and Soil Rehabilitation. Treatments after wildfire

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1. Wildfires in the Mediterranean region

The climate and vegetation in Mediterranean-type regions (e.g. the Mediterranean shrublands and woodlands on the western coasts of Australia, South Africa, California, Chile, Portugal, Spain and other regions of Europe and Africa) naturally promote intense, fast-spreading wildfires. These regions are characterized by mild wet winters and dry summers because the subtropical highs that occur at latitudes of around 30° in both N and S hemispheres cause cool polar air to move towards the equator on the west coasts of the continents, creating dry Mediterranean climates close to these latitudes. These climatic conditions favour the accumulation of fuel loads during autumn to spring. The fuel loads then dry out in summer, when the high net radiation and temperatures and low air humidity promote fire ignition.

Wildfires have historically been recorded in the Mediterranean and neighbouring Atlantic coastal regions, but a drastic increase in fire activity has taken place since the 1960s. Human activity is a major driver of wildfire, and afforestation schemes with flammable plantations, the encroachment of shrubs after rural depopulation and urbanization close to forests have been identified as main causes. In addition, there is evidence that the decreased rainfall and increased warmth associated with global warming favour the occurrence and increased intensity of fires (Kovats et al. 2014). Megafires, triggered by extreme climate events, have caused record maxima of burnt areas in some Mediterranean countries during the last few decades (San-Miguel-Ayanz et al. 2012; 2013).

Erosion tends to occur on steep slopes with shallow soils where drought-adapted deciduous and evergreen shrubs and woodlands grow. Although erosion rates are low in unperturbed

conditions in Mediterranean-type ecosystems (1-5 mm per century; Selby 1993), the climatic characteristics also make these regions prone to soil erosion and land degradation after fire (Figure 2). Thus, most of the annual rainfall occurs between autumn and spring. Episodes of particularly intense rainfall (50 mm in less than 1 hr) are rather common and cause devastating erosion. The synergistic effects of fires and high erosion rates contribute to the increased risk of desertification in these regions.

Figure 1. Large areas of Mediterranean regions are seriously degraded as a consequence of recurrent wildfires. Loss of some soil by erosion, such those developed on limestone (left) or granite (right), may be particularly severe after fire. Photos by Agustin Merino.

General strategies to prevent forest fires

In many areas of Mediterranean regions forest landscapes are subjected to a process of homogenization. In general, more diverse landscapes are less vulnerable to wildfires. Landscapes divided in smaller units (different land uses, riparian vegetation, firebreaks and others) can also provide greater diversification in terms of economic activity.

Figure 2. Prescribed burning conducted in a shrubland to reduce the fuel load and decrease the risk of occurrence a large wildfire. Photo by Agustín Merino.

The risk of forest fires can be reduced by means of vegetation management. This includes the selection of appropriate forest species, as well as planning of firebreaks and roads to facilitate access for fire fighters. Prescribed burning (intentional burning under controlled conditions for the purposes of forest management, farming, prairie restoration and fuel load reduction) is a preventive technique used to control the fuel in shrubland and understorey vegetation in forests. In some countries this technique is now used as a major preventive measure. Although prescribed burning has a much lower impact than wildfire, it must be conducted following specific protocols to reduce the risk of soil degradation.

2. Effects of fire on soil properties and soil conservation

Fire intensity can be defined as "the energy output from fire", whereas *fire severity or burn severity* refers to the "degree of organic matter loss or decomposition both aboveground and belowground" (Keeley, 2009). In this section, we will address those aspects involving degradation of soil after wildfire, referred to as soil burn severity.

2.1. Soil Burn Severity

Wildfires that cause high levels of soil burn severity can significantly perturb soil properties, thus affecting plant growth and having important implications for ecosystem recovery. Such perturbations may enhance long-term soil degradation (e.g. erosion and loss of nutrients and soil organic matter), making revegetation by colonizing vegetation more difficult. Characterization of the level of soil perturbation is essential as a basis for designing post-fire management strategies and emergency post-fire stabilization and rehabilitation treatments (Robichaud 2009).

The term *soil burn severity* is often used to describe the level of perturbation caused by fire in soil and, although there is no single definition or standardized means of measurement, most studies have emphasized the use of parameters related to the consumption of soil organic matter (SOM). Indeed, loss of SOM is very important, not only for soil chemical and biological properties, but also for physical properties that directly influence erosion and hydrological processes (Weill and Brady, 2016).

Burnt areas are usually spatially complex mosaics of burned soil patches encompassing a wide range of levels of soil burn severity, in which barely affected areas coexist with others affected by a high degree of soil degradation (Figure 3). These differences are mainly attributed to the different temperatures reached in the soil and the duration of exposure, which are influenced by fuel loading, combustion type, vegetation type, slope, soil texture, moisture, biomass in the soil organic horizon, and time since last burn.

Soil burn severity is of potential value as an indicator of the level of impact that fire has on the soil and of the influence in post-fire hydrological and erosive responses. Evaluation of the spatial distribution of soil burn severity is therefore a critical step in the decision-making process, in order to identify areas of the burnt surface where degradation is expected to be highest and to determine the different rehabilitation tasks required (Jain et al 2008; Parsons et al. 2010).

Figure 3. The effects of wildfires on an ecosystem are often not homogeneous. The intensity and severity of the fire depend on different factors, such as the local accumulation of biomass, slope and the type of wildfire. Photo by Agustín Merino.

Several visual indicators of soil burn severity have been defined on the basis of the immediate changes (observed in the field) to the forest floor (level of consumption of organic layer) and mineral soil (changes in colour or structure) and also the deposition of ash from the aboveground combustion of biomass after wildfire. Table 1 and Figure 4 summarise the soil burn severity indicator system developed for Atlantic areas (Vega et al., 2013; Arellano et al. 2017); the maximum temperatures usually reached in these levels of SBS are greater than 500ºC in the upper soil profile (Figure 5).

Table 1. *Six levels of soil burn severity classes, including an unburned state, determined from the immediate post-fire soil and duff visual characteristics (modified from Ryan and Noste 1985).*

Level of soil burn severity	Forest floor (Litter layers Oi + Oe + Oa)	Mineral soil (Ah horizon)
0	No evidence of fire	No evidence of fire
1	Oa layer (lower duff) partially or totally intact.	Undisturbed
2	Oa layer totally charred and covering mineral soil. There may be ash.	Undisturbed
3	Forest floor completely consumed (bare soil). There may be ash.	Undisturbed. Soil structure unaffected. SOM not consumed. Surface fine roots not burned.
4	Forest floor completely consumed (bare soil). There is no charred residue. Thick layer of ash.	Soil structure affected. SOM consumed in the upper layer. Surface soil colour altered (grey). Surface fine roots burned.
5	Forest floor completely consumed (bare soil). No charred residue.	Soil structure affected. SOM consumed in the upper layer. Surface soil colour altered (reddish). Surface fine roots burned.

SOM, soil organic matter

Figure 4. Different levels of soil burn severity (1, 3 and 5). In the lower levels of SBS the litter layer is still present, which prevents erosion. In the highest levels of SBS, the temperature in the mineral soil is high enough to produce loss of large quantities of soil organic matter, with the subsequent effects on soil structure and infiltration. Photos courtesy of CIF Lourizán.

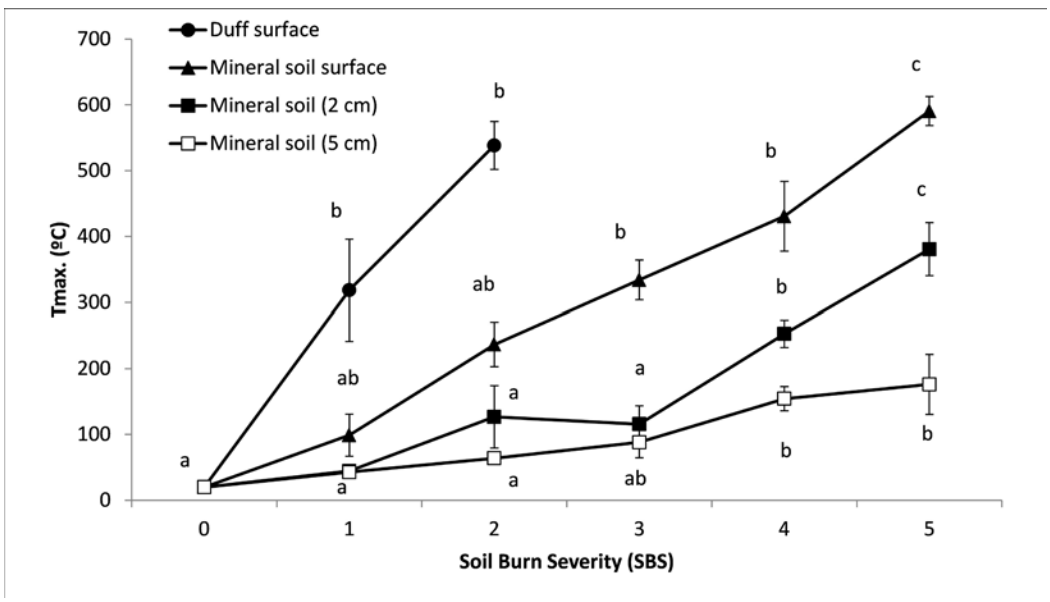

Figure 5. Temperature in the litter layer, mineral soil surface, 2 cm soil depth and 5 cm soil depth for the different levels of soil burn severity (T, level 0; B1, level 1; B2, level 2; C, level 3; D, level 4; and E, level 5). Different letters indicate significant differences ($p < 0.05$) between soil burn severity levels in each part of the soil profile. These data were recorded in an experiment using monoliths burned in a combustion wind tunnel to simulate different fire conditions. Source: Merino et al., 2018.

2.2. Changes in key soil properties related to erosion and revegetation in burnt areas

Wildfires affect soil properties that are important for soil conservation and subsequent rehabilitation.

Soil organic matter (SOM). In addition to combustion of the soil organic layer, fire reduces the SOM content of mineral soils (Figure 6). Losses higher than 60 % are usually recorded in soils affected by high levels of burn severity.

Alterations in the SOC composition may even take place during fire. The C/N ratio usually decreases after severe wildfire. This effect is attributed to the greater impact on C than on N of heating from volatilization or to the fact that wildfire typically burns forest ecosystem components with low C/N ratios. On the other hand, SOM labile fractions, such as carbohydrates, are easily lost (Figure 7). At the same time, important gains in aromatic compounds (charcoal) can be formed (Knicker et al. 2005). The changes have an important negative impact on soil colloidal properties (affecting soil aggregate, water and nutrient stores) and soil microbial composition.

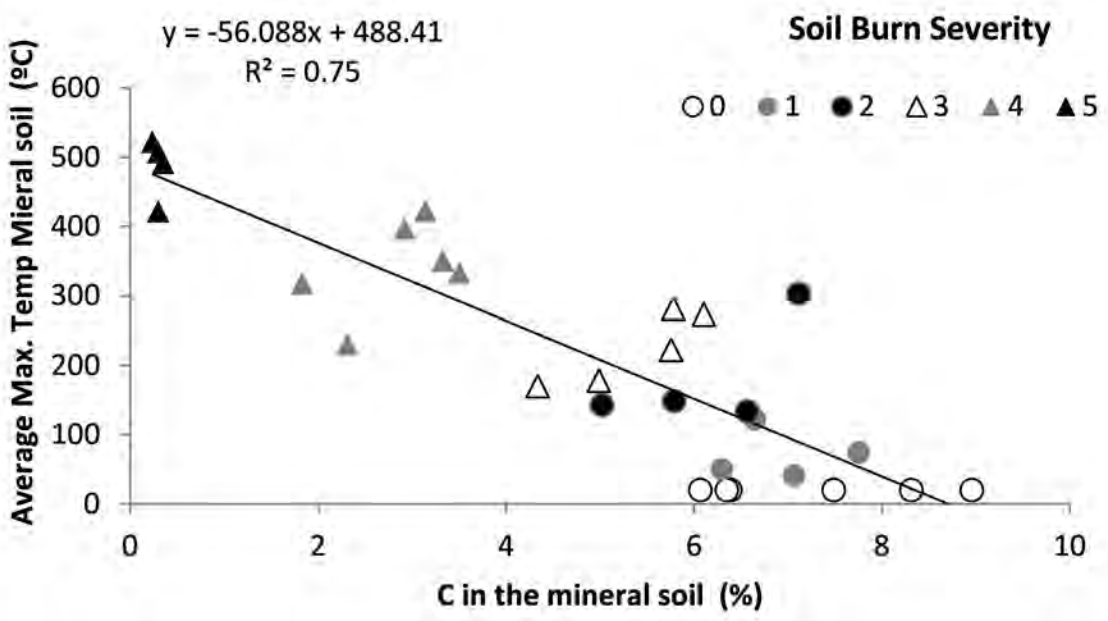

Figure 6. Relationship between the maximum temperature recorded in the mulch during the fire and the C content of the mineral soil (0-2 cm). The data were obtained in an experiment using monoliths burned in a combustion wind tunnel to simulate different fire conditions. Source, Merino et al. (2018).

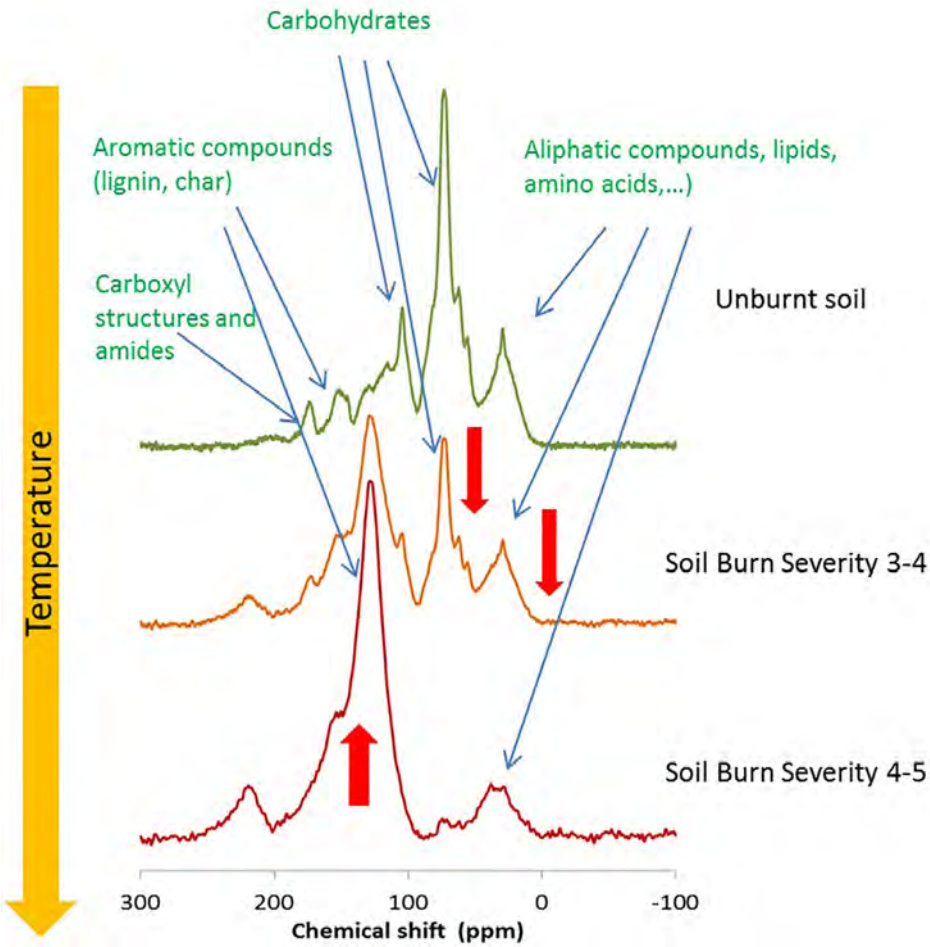

Figure 7. Solid-state 13CP MAS NMR spectra of mineral soil samples affected by different levels of soil burn severity. Soil heating usually leads to loss of carbohydrates and other C-labile forms (cutins, amino acids, and others) and thus affects soil structural stability. Burning leads to sharp increases in charcoal, aromatic C compounds, highly recalcitrant forms of C, which may persist for decades. Source, Merino et al., 2014.

Nutrients and pH. Fire events cause the release of nutrients from non-plant-available into plant-available forms in soil. Fire strongly affects soil by increasing the concentration of soluble base cations (Ca, Mg, and K), which is linked to the increased combustion of the litter and mineral soil (Vega et al., 1993).

In moderate levels of SBS (level 1 and 2 in Table 1), in which the soil organic layer has been partially combusted, this effect involves an increase in soil fertility. This process is expected in prescribed fires and is taken advantage of to improve rangelands.

However, in high-severity fires, in which litter is completed burned and the SOM of the mineral soil has been lost, release of these elements may be too high. As this process occurs when there is no vegetation, net export of elements occurs from the system in the run-off water and drainage in the rainy season. This process is critical for all the nutrients, especially P, which is replaced very slowly in the system (the natural contributions are very small). Heating leads to the depletion of the medium-term P reserves and the consequent release of mineral P as soluble orthophosphate, which can exported from the soil in the run-off and drainage.

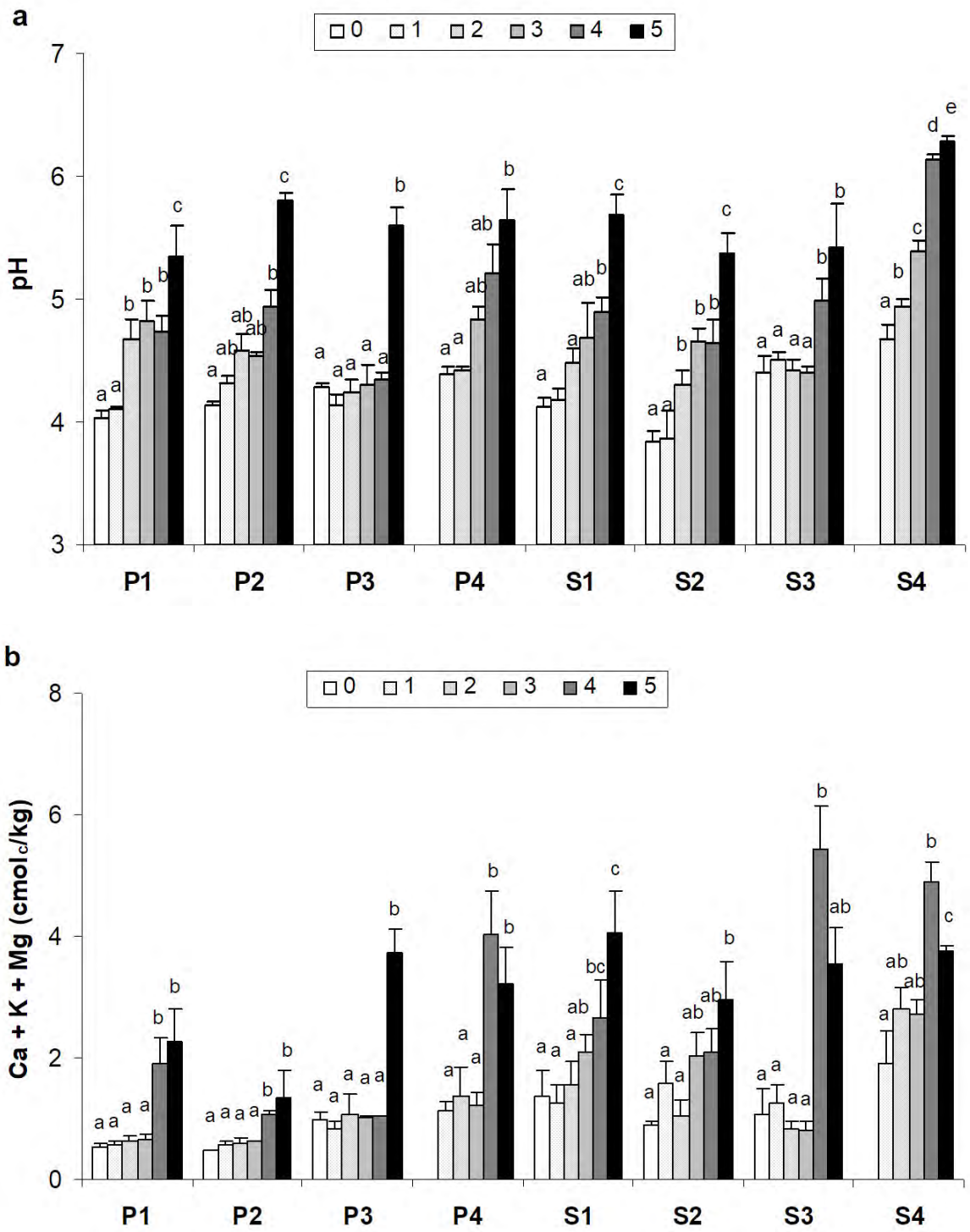

Figure 8. Extractable (soluble) soil cations (sum of Ca, K, and Mg) and pH of the mineral soil (0-5 cm depth) under different soil burn severity classes (levels 0 to 5; see Table 1) for eight study sites for eight study sites. The data correspond to 8 soils developed under pine plantations (P1-P4) and shrublands (S1-S4). Error bars represent 1 standard error (SE). Different letters indicate significant differences ($p < 0.05$) between soil burn severity levels at each study site (Vega et al. 2013a).

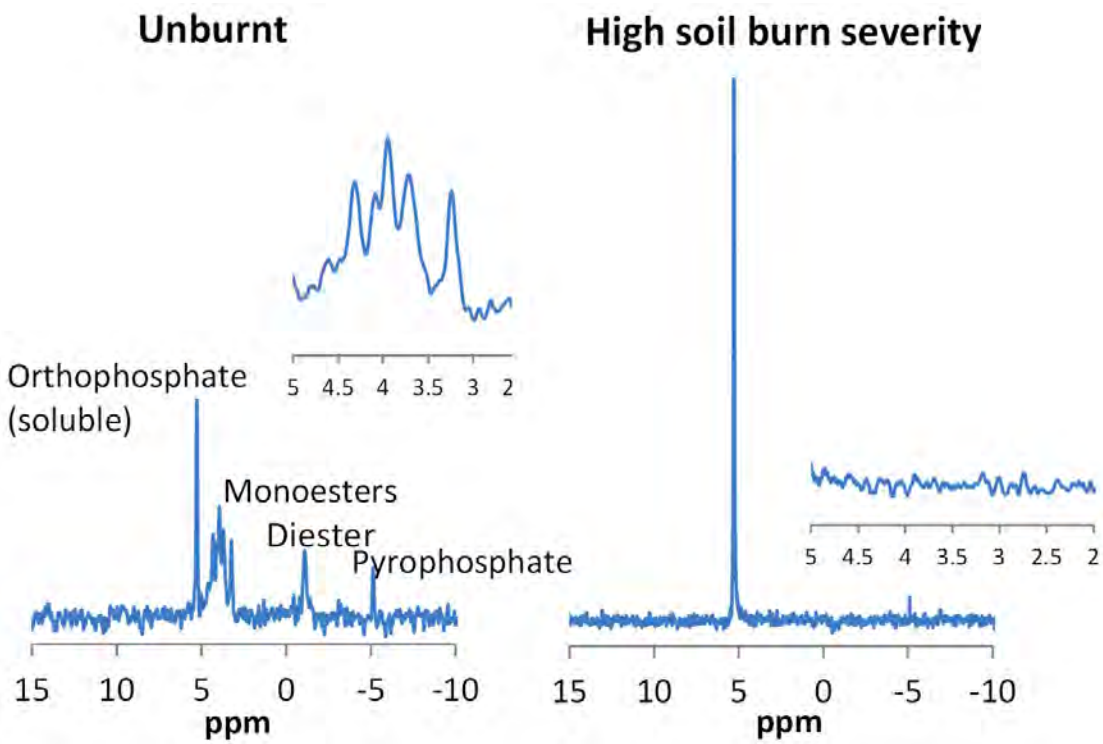

Figure 9. ^{31}P NMR spectra of the soil organic layer before and after burning. Fire consumes the organic forms (monoesters and diester), which are the main reservoirs of P in the soil, abruptly releasing inorganic soluble P (PO_4^{-3}) and pyrophosphate. Source, García-Oliva et al., 2017.

2.3. Soil hydrological properties, hydrological processes and erosion

Soil structure

At high levels of SBS (4 and 5 in Table 1 and Figure 2), the changes in SOM (content and composition) decrease the stability of the soil structure (Figure 9). As the soil lacks an organic layer, the weak soil aggregates are exposed (unprotected) to the direct impact of raindrops and are very vulnerable to destruction by rain. Under these conditions, the smallest particles of soil become detached and tend to enter the pores of the uppermost layer of the soil, generating a superficial crust. The immediate effect is collapse of the aggregates and sealing of the uppermost layer of the soil, with negative consequences on hydrological processes. Water infiltration is reduced, and surface run-off and erosion therefore increase (Figure 9). The crust also prevents germination of seeds, hampering natural revegetation.

Figure 10. The soil crust that develops after wildfire not only reduces infiltration, but also has a negative impact on plant establishment. Seedlings must break the surface crust (and usually do not succeed if the crust is hard). Photo by Agustín Merino.

Water repellency or hydrophobicity of soils. Some soils rich in SOM, especially coarse textured soils, are naturally hydrophobic, which delays infiltration of water into the dry soil matrix (Figure 10). After fires that cause low or moderate levels of SBS, this effect appears to increase up to 40% in the upper 6-8 cm and may persist for 3 and 19 months. Although it is not clear why the soil becomes hydrophobic, the process is thought to mainly involve the formation of functional hydrophobic films on the soil particles. However, neither the origin nor the composition of these compounds is well known.

Hydrophobic soils show a low infiltration capacity and easily generate run-off, greatly enhancing the erosive power of the rain after the fire. For this reason, rain events that occur after fire can have devastating effects (Shakesby and Doerr, 2006).

Figure 11. The soil surface sometimes become hydrophobic. A) Water droplets resisting infiltration into soil due to extreme water repellency. Hypodermic needle included for scale (Doerr et al. 2000). This effect is due to the formation of organic films that repel water, preventing its infiltration. B) The lack of infiltration makes burned soils particularly susceptible to erosion.

Alteration of hydrological processes. Hydrological processes are drastically affected by wildfire. The partial or complete combustion of the vegetation and litter leads to a rapid response in terms of overland flow and run-off. This is due to the lower storage capacity of water, to the decrease in transpiration and evaporation surfaces and also to the lower

obstacles to overland flow. As a consequence, shorter lags between rainfall events and peak flooding are often observed after wildfire.

Erosion. Erosion rates may be very high in burnt areas as a result of the higher soil erodibility and enhancement of hydrological processes. Erosion rates above 50 Mg ha^{-1} (1 Mg = 10^6 g) are usually recorded in areas affected by high levels of SBS. Although erosion usually reaches maximal levels during the first year after wildfire and declines thereafter (Figure 15), in dry areas, where the revegetation is very slow, erosion may not occur until later, even up to three years (Figure 16). This reflects the highly variable rainfall intensity and also the slow rate of vegetation recovery.

3. Land restoration after wildfire: revegetation and erosion control in burned areas

3.1. Natural mechanisms of soil protection after wildfire: natural revegetation and litterfall in wooded areas

In environments prone to wildfires, many plant species have the ability to regenerate after fire. In thermophytes and re-sprouters (e.g. Ulex spp.), heat and / or the smoke may help break seed dormancy. In addition, these species benefit from low levels of competition from other species. In cases of low SBS, the presence of soluble ash provides readily available nutrients, which also

facilitates revegetation in acid soils. However, in high SBS, natural regeneration may be difficult as a result of strong soil degradation and low seed availability (Figure 14). In some areas the large accumulations of ash hinder germination due to high pH and high osmotic pressure.

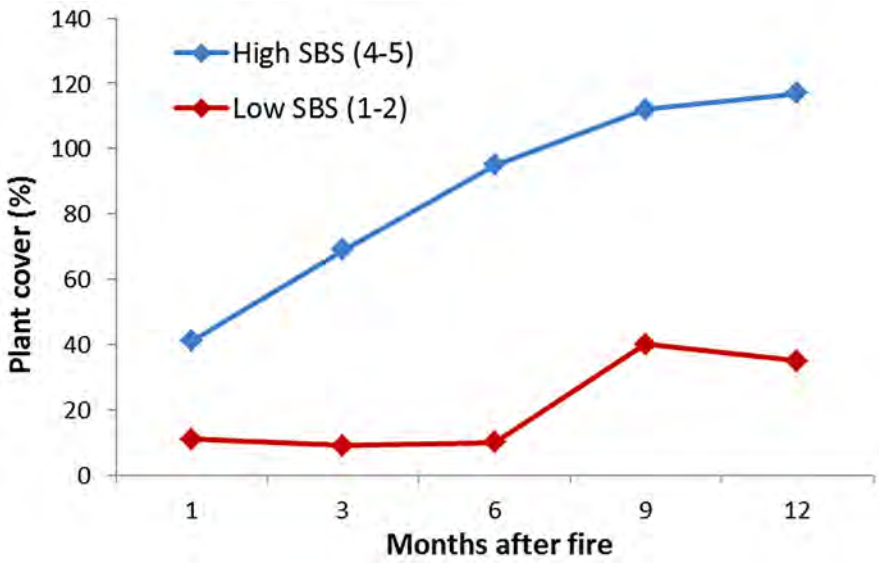

Figure 15. Differences in revegetation rates after fire in areas affected by low and high soil burn severity. Source, Vega et al., 2013.

On the other hand, in forest fires of moderate intensity, where the tree canopy has not been totally burned, the abundant litterfall covers the soil and provides very effective natural protection against crust formation and erosion (Figure 17). In addition, this natural cover is an important source of nutrients. For this reason, extraction of burned trees is only advisable after leaf-fall has occurred.

Figure 16. In low-intensity fires, the leaves/needles that fall cover the soil surface, providing valuable protection. Photo by Agustín Merino.

3.2. Emergency actions against erosion in burnt forest areas

Due to the above-described effects, soils affected by high-severity fire are prone to degradation and extreme erosion, and it is therefore advisable to carry out emergency treatments. These treatments prevent or reduce the impact of fire and favour natural revegetation. Different strategies reduce erosion by providing soil cover, trapping sediments or promoting infiltration. Detailed protocols can be found in the handbooks by Napper (2015) and Vega et al. (2013b) and also in the article by Ferreira et al. (2015). Here we briefly summarize the main emergency treatments used to avoid degradation of soils and waters.

Mulching. Mulching with straw, logging residues and other types of material provides immediate cover on the soil and protects it from erosion. This treatment is recommended in areas of moderate and high burn severity, where the erosion potential is high. This practice can reduce run-off volumes by 50% and soil erosion by 90%. It also favours revegetation and survival of plants after fire by reducing the temperature of the soil, maintaining soil moisture and providing nutrients to the soil as the mulch decomposes. Mulching also absorbs rainwater and reduces surface hydrophobicity, thus also reducing the maximum downstream flows.

The material used as mulch should be selected on the basis of availability as well as the production and transportation costs. Straw is commonly used as mulch because large areas can be treated effectively and quickly. Straw is lighter and generally more effective than logging residues (the same weight covers a larger area). Nevertheless, straw is not always readily available and transportation may be expensive. The straw is applied with helicopters (heli-mulching) to treat large areas, or manually in smaller areas. Straw can also be applied from forest tracks by using blowers. The straw can be applied in strips.

Figure 17. (a) Application of straw by heli-mulching; (b) straw broadcast from a helicopter rapidly provides soil cover; (c) straw applied in contour strips; and (d) crew applying straw by hand. Photos by Agustin Merino.

Straw is generally applied at a rate of 2.5 t ha^{-1}, which covers approximately 70% of the surface. Loss of cover due to decomposition is offset by natural vegetation. Although straw can move through run-off, it forms small dams that accumulate sediments. The most common reason for the failure of straw mulching is movement by the wind, which can remove the straw or cause it to accumulate in some areas, thus suppressing vegetation growth. Straw may contain seeds of invasive plants. In some cases, straw application is combined with sowing. The straw enhances germination of the seeds, and the new plants help to stabilize and maintain the straw on site.

In small burned areas (in hectares), chopped logging residues from unburned vegetation from adjoining areas can be applied to the soil surface. This practice also serves to reduce the fuel load and the risk of wildfire. Scrub and small trees can be chopped to produce material that can be applied in quantities of between 8 and 10 t ha^{-1}. However, the material is expensive to produce and is less effective than straw.

Figure 18. (a) Clearing the burnt area with slash spread uniformly; (b) close up view of the slash generated with heavy equipment. Photos courtesy of CIF Lourizan.

3.3. Erosion barriers

Erosion barriers consisting of logs, prefabricated fibre rolls or fences are placed on steep slopes to reduce water flow and trap sediments. The barriers also promote infiltration and reduce run-off. Tree trunks are usually available in burned areas, and installation (log erosion barriers) is usually relatively easy. However, the ability to trap sediments depends on the degree of contact between the trunk and the soil surface. The space between the trunk and the ground surface should be covered to prevent water flowing below the trunk.

Other types of materials, such as rolls of fibre or sandbags can also be used. Rolls of fibre, usually of diameter 0.25 m, are prefabricated from rice straw or jute nets. In comparison to tree trunks, these types of barriers have the advantage of being malleable and therefore adaptable to the ground surface.

Fences constructed from burnt branches and stems are designed to reduce the effective length of the slope. This type of barrier is often expensive and may not be very effective. However, silt fences (made of geotextile fabric) are inexpensive and easy to install.

This type of barrier is used in the following situations: (i) slopes affected by high-severity fire; (ii) steep slopes, between 25 and 60%; (iii) in cases of water repellent soil; (iv) in basins with high risk of sediment transfer to the river bed. The barriers are most effective during low intensity rain events (less than 30 mm h-1) and much less effective during intense rainfall.

Figure 19. a) Contour felled log erosion barriers with trapped sediment; b) rolls of fibre placed across the hillslope; c) fence made from branches and stems; and d) silt fence. Photos courtesy of Agustín Merino and TRAGSA, CIF Lourizan.

3.4. Channel treatments

Dams can be constructed from bales of straw, stones, sticks or even sandbags to stabilize channels and retain sediments (Figure 21). This system is effective for accumulating sediment and ash in temporary channels during the first rain event after the wildfire. Channelling treatments can be combined with other strategies to mitigate run-off and erosion on hillslopes. They can be used to prevent reservoirs filling up with sediments after wildfires. In the event of heavy rainfall, the channels may overflow or be destroyed.

Dams made from bales of straw are most suitable for gently sloping areas. This type of dam becomes less effective over time. Dams made from tree trunks are more resistant than those made from straw. They are used to prevent gully erosion and to stabilize the channel gradient. Vertical logs are transversely anchored by stakes. Stones can also be used to build barriers to reduce bank erosion and to accumulate sediments. They are useful to prevent formation of gullies.

Figure 20. Small dry dams installed to retain sediments, flooding mitigation and slope stabilization. Photos courtesy of CIF Lourizan (upper) and by Agustín Merino (lower)

3.5. Seeding

Direct seeding can be carried out to accelerate the establishment of fast-growing vegetation species that will provide soil cover until the native vegetation recovers (Figure 22). Sowing can be conducted aerially or manually. The seed mixtures used vary from one region to another. The most common species in Mediterranean regions include ryegrass and other commercial mixtures of seed legumes, such as *Lolium multiflorum* (Lam.) Husnot, *Lolium perene* L. and *Agrostic capillaris* L., among others. In some cases, the seeds are applied by hydroseeding (the seeds are added to a fixative and fertilizer mixture).

However, the effectiveness of sowing is often reduced by the effects of rainfall or the lack of adaptation of the plant species to soil conditions. In addition, seeded grasses usually compete with and may displace the native vegetation. Direct seeding is therefore usually advisable.

Figure 21. When properly managed, seeding can reduce the accelerated erosion during the first year. However due to problems such as seeds being washed away by the rain, lack of adaptation of the plant species to soil and climate conditions, competition with native vegetation, this technique is not usually advisable. Photo by Agustín Merino.

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5. Study cases

5.1. Study case 1: Assessing the effectiveness of different emergency post-fire rehabilitation treatments for reducing soil erosion in NW Spain

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Adapted from

Vega, J. A., Fernández, C., Fonturbel, T., González-Prieto, S., & Jiménez, E. (2014). Testing the effects of straw mulching and herb seeding on soil erosion after fire in a gorse shrubland. Geoderma, 223, 79-87.

Fernández, C., Vega, J. A., Jiménez, E., Vieira, D. C. S., Merino, A., Ferreiro, A., & Fonturbel, T. (2012). Seeding and mulching + seeding effects on post-fire runoff, soil erosion and species diversity in Galicia (NW Spain). Land Degradation & Development, 23(2), 150-156.

Background and Aims

One of the most serious consequences of forest fires is the increased soil erosion. The application of emergency post-fire rehabilitation treatments is generally proposed in severely burned areas, where the soil is prone to erosion and natural revegetation is very low (Figure CS1.1). Hillslope treatments are considered beneficial in these cases because they are expected to prevent soil loss and sediment delivery to downstream water bodies. The study was conducted in Galicia (NW Spain), an area that is particularly prone to wildfires. A system of identifying soil burn severity based on visual indicators has been developed in this region and is used for designing soil rehabilitation tasks (Vega et al., 2013a, 2013b).

Strategies

To assess the efficiency of the different treatments on soil erosion, two experiments were carried out in shrublands (*Ulex europaeus* L., *Erica cinerea* L. and *Daboecia cantabrica*) located in Galicia (NW Spain). In the first experiment, the treatments (Figure CS1) tested were (a) straw mulch: manually spread at a rate of 2.5 Mg ha^{-1}, (b) wood-chip mulch: 0.5 cm thick and 2 cm wide, manually spread at a rate of 4 Mg ha^{-1}, (c) cut-shrub barriers: four permeable cut-shrub barriers (made from shrubs cut in unburned areas) located in each plot with this treatment,

and a spacing of every 10 m along the longest dimension and (d) control: no treatment. Treatments were applied four weeks after fire, 10 days after plot installation. Erosion was determined in experimental macro-plots (30 x 10 m) by using fences made from geotextile fabric to trap sediment (Figure CS1 d).

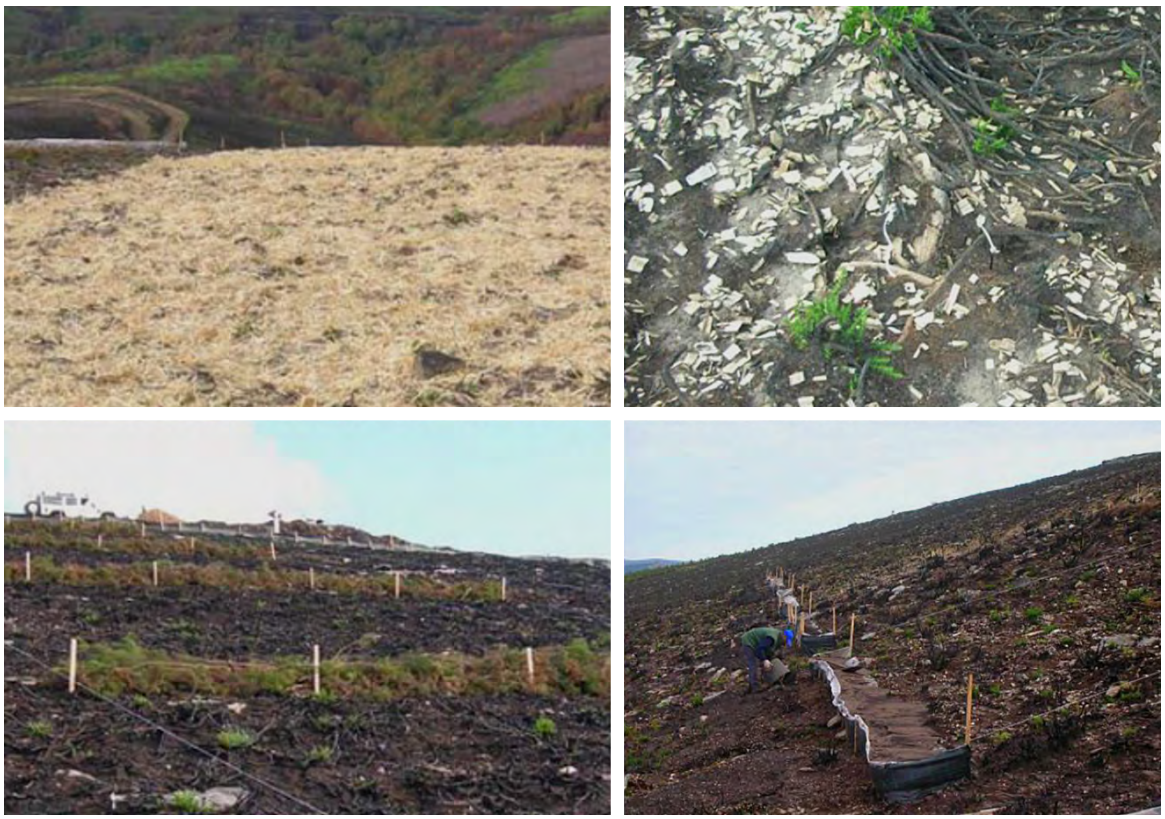

Figure CS1. Post-wildfire treatments assessed in the first experiment.

In the second experiment, the effect of seeding was evaluated by similar methods. The seed mixture comprised *Lolium multiflorum, Dactylis glomerata, Festuca arundinacea, Festuca rubra, Agrostis tennuis and Trifolium repens*. The seeds were applied at a rate of 4 g m^{-2} immediately after the fire.

Highlights

The first experiment identified straw mulching as the most efficient treatment, due to the smaller area of bare soil in this treatment relative to the other treatments (Figure CS3b). The straw prevented the impact of the rain drops and erosion in this soil, which was very sensitive to erosion as a consequence of the fire-induced degradation (SOM loss and low structural stability). The wood chip mulch was much less efficient and did not prevent erosion.

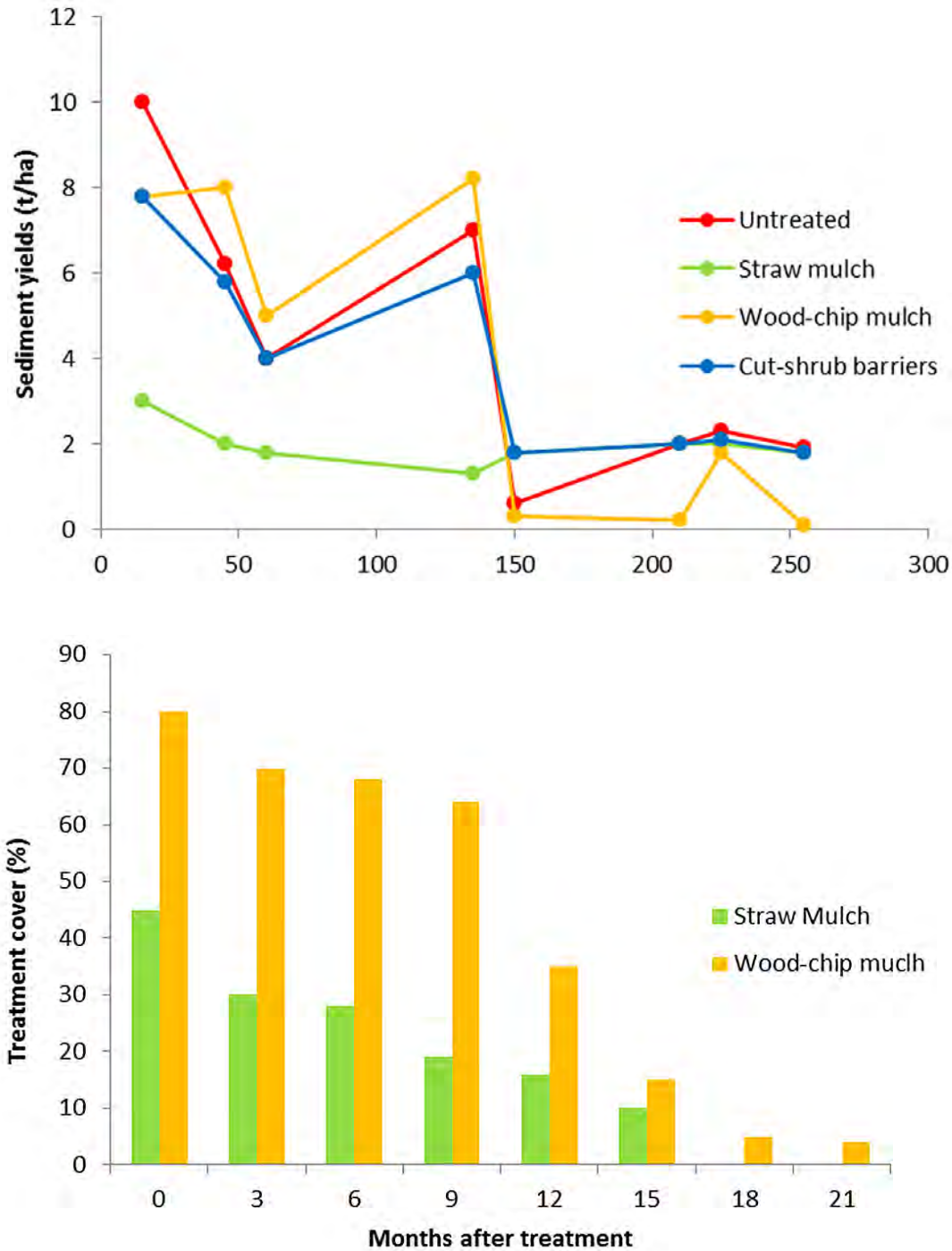

Figure CS2. a) Soil loss in a severely burnt area after the application of three different treatments: straw mulching, wood-chip mulching and cut-shrub barriers. B) The greater efficiency of the straw relative to wood-chip mulching is due to the higher cover provided by the straw.

The second experiment revealed that seeding was not efficient in protecting the soil against erosion. This may have been because the low temperatures in autumn and winter may have limited vegetation growth of both native and seeded plants.

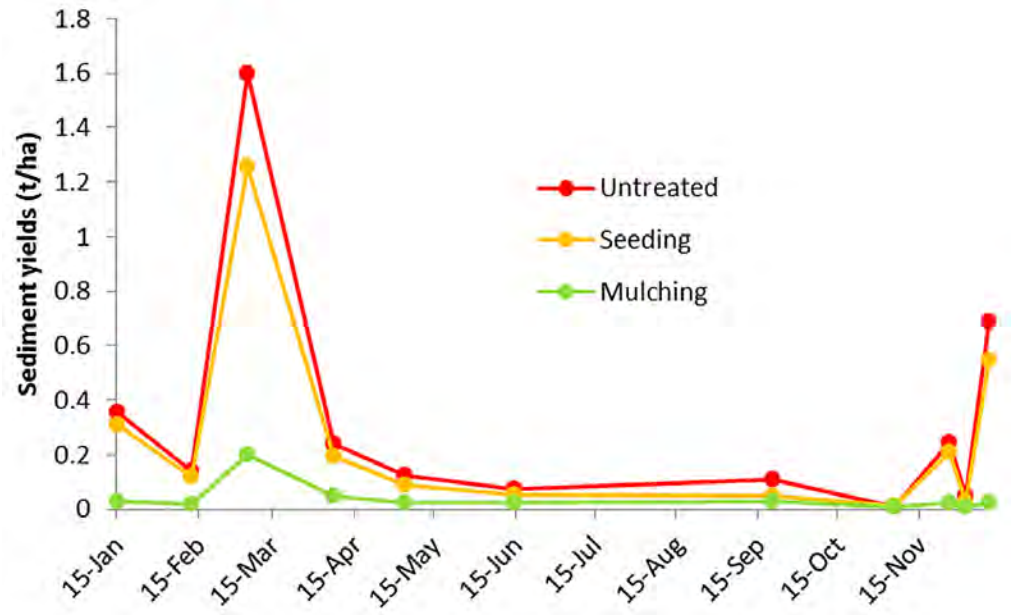

Figure CS3. When seeding is carried out in autumn or winter, the low temperatures prevent germination and growth, reducing the potential effect of the treatment.

Outcomes of restoration

In NW Spain, the application of straw mulch to zones affected by the highest levels of soil burn severity was the best treatment for controlling soil erosion, yielding a reduction of around 66% compared to control plots. Retention of large amounts of sediment was favoured by adherence of straw to soil particles after the first rainfall events, whereas most wood chips did not adhere to the soil and substantial amounts were removed along with the sediments. The potential protection provided by seeding is lower than that provided by mulching.

5.2. Study case 2: Comparing the survival and size of resprouted and planted trees for post-fire forest restoration in central Portugal

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Background and Aims:

Active restoration techniques (planting and seeding) are usually applied after fire, as there is strong political and social pressure to reforest burned areas. However, taking advantage of expected natural post-fire regeneration (passive restoration), mainly in regions where plants use strategies to survive fire (e.g. regeneration from seeds, resprouting), may represent a much more cost-effective approach to restoring burned areas, enabling treatment of much larger areas at the same cost. In the case study reported here, post-fire monitoring of tree survival and post-fire responses (recovery mode and growth) began in the winter following a wildfire that occurred in the summer of 2003 in central Portugal (Tapada de Mafra). At the same time, the forest managers decided to reforest part of the area by planting some species, including *Fraxinus angustifolia* (ash) and *Quercus faginea* (oak), which were also being monitored to evaluate their post-fire resprouting capacity. This provided an opportunity to compare survival and growth of planted versus resprouted trees. The aim of the study was to compare the survival and growth of trees brought from nurseries and planted in a burned area with that of naturally regenerating trees.

Strategies

After the fire, 19 burned ash trees and 56 burned oak trees were monitored. These were adult trees about 5-15m tall, assumed to have been affected by a high-severity fire (charring height was on average 80-95% of total tree height, and the whole canopy was consumed by fire). Charred stems were logged after fire and remaining stumps (part of the trunk protruding from the ground after the tree has been felled) were protected from herbivores. The stumps were assessed 21 months after fire. The status of each individual (alive or dead, i.e. resprouting or not resprouting) was noted, and for live trees, maximum shoot height and maximum shoot basal diameter were measured. Seedlings were obtained from nurseries and planted in 2004 and 2005. The seedlings were protected from herbivores with individual protective sleeves similar to those used on resprouted trees. A sample of 122 trees obtained from nurseries (30 ash and 30 oaks in 2004, and 29 ash and 33 oaks in 2005) and that had been planted in the vicinity of the monitored trees, were also assessed 20-22 months after planting. For each individual tree, the status (dead or alive) was noted, and the basal diameter and plant height were measured. Survival, height and diameter of resprouted and planted trees were then compared.

Highlights

Plant survival 20-22 months after fire/planting was higher in resprouted than in planted trees, in both oaks (98.2% against 76.7% for 2004 and 66.7% for 2005), and ash (100% against 86.7% in 2004 and 96.6% in 2005). The resprouted ash trees were also significantly taller (median = 208.0 cm) than the trees planted in 2004 (median = 53.0 cm) and 2005 (median = 108.5 cm) (Figure CS2). A similar pattern was found for oak, with resprouted trees (median = 180.0 cm) being much taller than individuals planted in 2004 (median = 37.0 cm) and 2005 (median = 47.0 cm). A similar pattern was also observed for the diameter of resprouted trees.

Figure CS1. Ash plant (*Fraxinus angustifolia*) obtained from a nursery (left) and a resprouted ash tree from cut burned stem (right), 2 years after planting/fire.

Outcomes for restoration practice

Mediterranean-type ecosystems are often dominated by shrub and tree species with the ability to resprout after fire. This characteristic could be used in post-fire restoration, mainly by assisting natural regeneration, e.g. via thinning, selection of shoots in coppices, control of unwanted vegetation or protection from grazing animals. The study findings showed that taking advantage of basal resprouting may be an appropriate alternative strategy to replanting ash and Portuguese oak in central Portugal, as both the survival and size of resprouted plants were higher, at least at the early stages of tree growth. This may be true for most Mediterranean areas and tree species, as long as they are have the ability to resprout

(i.e. most broadleaf species). There are several advantages of using resprouted plants for restoration: the costs are lower, as less heavy equipment or no such equipment is needed, and site preparation is often not required. Soil mobilization and subsequent erosion risk are also minimised. Plant survival and growth are enhanced as the resprouted trees have a well-established root system, and vegetation cover is achieved more quickly, again with important implications for preventing soil erosion.

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5.3. Study case 3: Multitemporal methods of detecting burnt areas based on images acquired after the fire event

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Background and Aims

Forest fire is a major threat to the biodiversity of protected heath landscapes. Planning and management of post-fire actions (reclamation and restoration interventions) in burnt forest areas requires accurate detection and assessment of the area of interest, the size and spatial distribution of the damaged zones and the behaviour of the fire event.

In this context, the capacity to locate and detect destroyed surfaces is important to support post-fire management, in particular to test the ability of the ecosystem to recover naturally as well as to assess the dynamics of recovery.

Remote sensing is an important means of obtaining information that enables discrimination between burned and unburned areas with different types of vegetation, by analysis of spectral indices.

Italy is affected every year by numerous fires of small dimensions, and detection and mapping of the burned areas is a very important task for developing environmental policies. Satellite data have been found to be useful for observing fire scars (Chuvieco & Congalton, 1988 (1); Koutsias & Karteris, 1998 (4); Pereira and Setzer, 1993 (6); Epting et al., 2005 (2), although until now very little attention has been paid to the recognition and analysis over large areas affected by small fires affecting only a few hectares of land (Martin et al., 2006 (5).

Strategy

The data analysed were acquired with two different sensors: the IRS-LISS3, a multispectral push broom sensor with four bands (Table 1) and a resolution of 6000 pixel/line corresponding to a ground pixel size of 25 m; and the SPOT4-HRVIR, a multiband (Table 1) sensor with a ground resolution of 20 m.

Table 1. *Comparison of corresponding spectral bands of IRS-LISS3 and SPOT4-HRVIR*

	IRS-LISS 3		SPOT-HRVSS 3	
Band	**Central Wavelength**	**Spectral Interval**	**Central Wavelength**	**Spectral Interval**
1 Green	0.555 µm	0.52-0.59 µm	0.545 µm	0.50-0.59 µm
2 Red	0.650 µm	0.62-0.68 µm	0.640 µm	0.60-0.68 µm
3 NIR	0.815 µm	0.77-0.86 µm	0.855 µm	0.79-0.9 µm
4 SWIR	1.625 µm	1.55-1.70 µm	1.625 µm	1.55-1.70 µm

Two pairs of multispectral images were selected (Sardinia region 15/05/2006 and 02/07/2006) and a window of 434.434 pixels (about 16,291 ha) was extracted, with a burn scar of 320 pixels (fire of 12 ha on 14/04/2006).

Figure 1 shows the two IRS windows of the test area, whereas Figure 2 shows the related fire scar.

The second pair of multispectral images processed was acquired by SPOT4-HRVIR (Sardinia region 01/07/2006 and 01/09/2006) (Figure 3) with a test area corresponding to a window of 133.560 pixels (about 1958 ha), characterized by a burnt scar of 1.637 pixels (fire of 24 ha on 20/06/2006).

Table 2*. The data set analysed*

Image	Date of acquisition of first image	Date of acquisition of second image	Dimension of subset	Dimension of fire scar	Percentage of burnt area	Vegetation cover in burned area	Date of related fire
IRS	15/05/06	02/07/2006	434434 pixel 16291 ha	320 pixel 12 h.	0.074 %	100% wooded	14/04/06
SPOT	01/07/06	01/09/2006	133560 pixel 1958 ha	1637 pixel 24 h.	1.23 %	50 % wooded	20/06/06

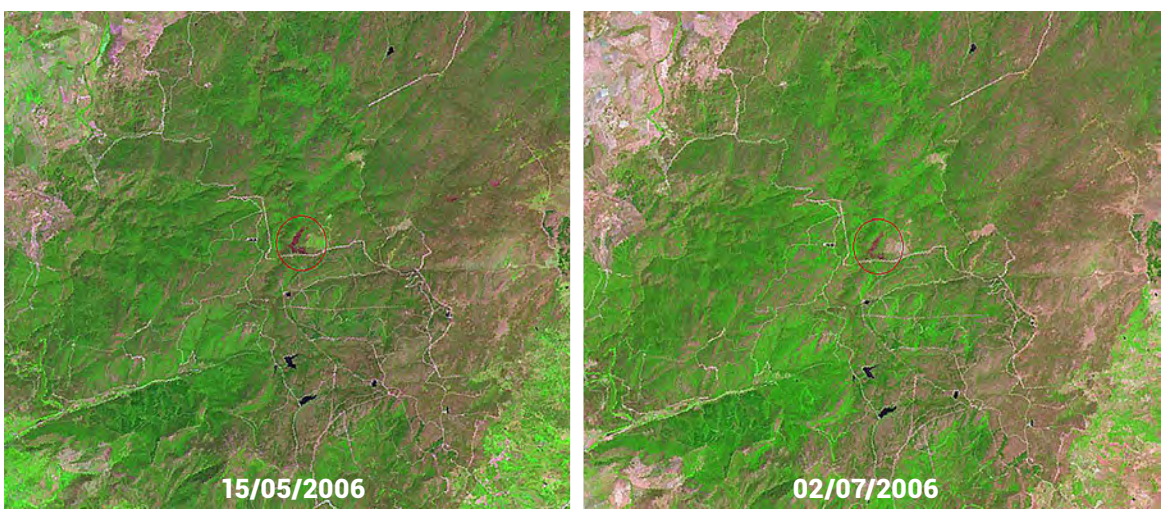

Figure 1. Window of the IRSS-LISS3 multispectral images representing the test area. Band combination: R=4,.G=3, B=2.

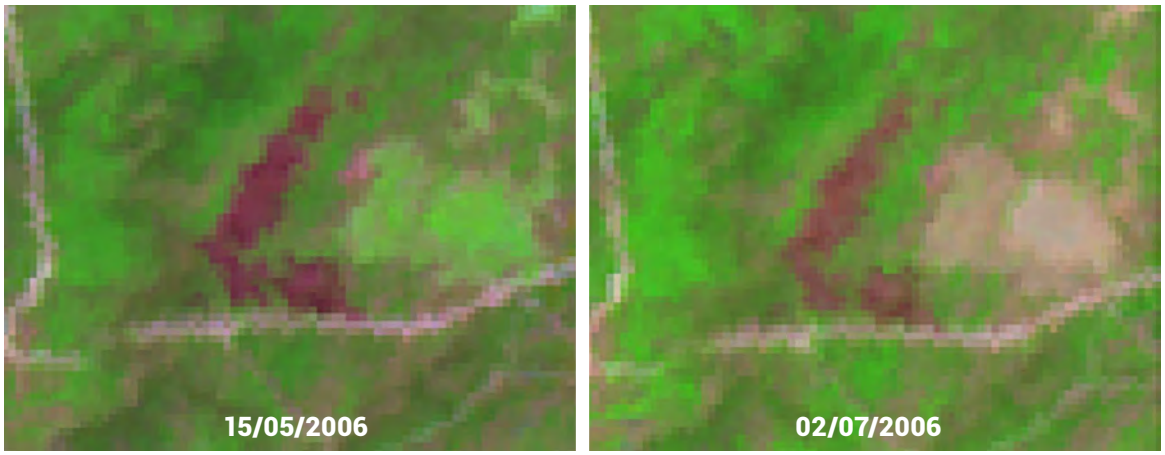

Figure 2. Details of fire scar on the IRSS-LISS3 multispectral images from Figure 1. Band combination: R=4, G=3, B=2.

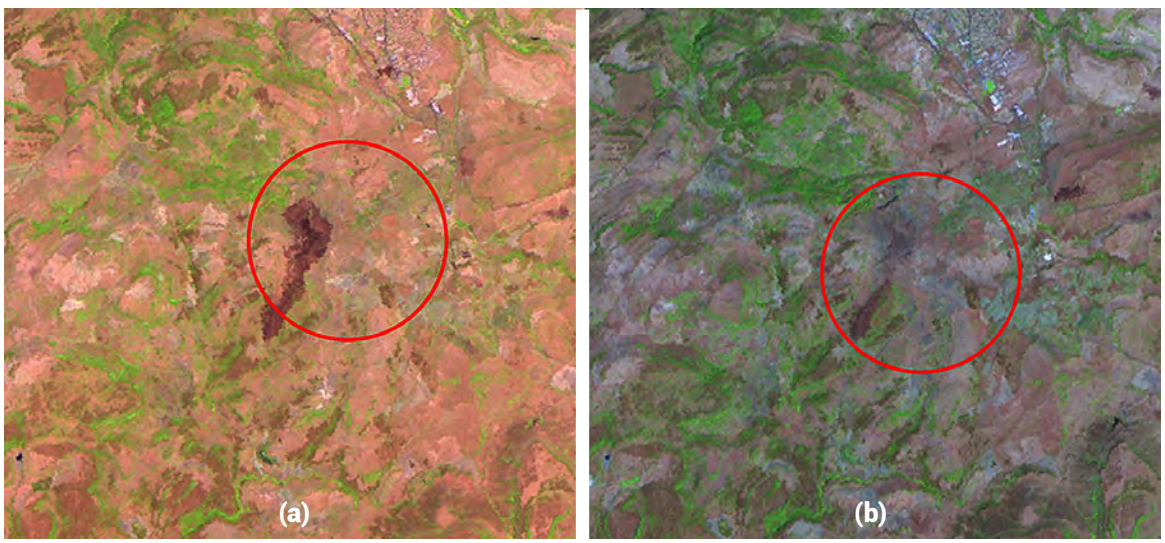

Figure 3. The selected subset of the SPOT images acquired on 01/07/2006 (a) and 01/09/2006 (b). Band composition: R=4, G=3, B=2. The fire scar considered is inside the red circle.

Methodological Approach

Among the methods proposed by the scientific community for detecting and mapping fire scars, some (i.e. the slicing threshold methods) are based on selection of a suitable threshold (Ki) applied to multispectral indexes evaluated on only one image acquired after the fire season.

Among the multispectral indexes defined for fire detection, the NBR (Noise Burn Ratio) [Koutsias and Karteris 19984], the NDII (Normalized Difference of Infrared Index) [Pereira and Setzer, (1993)6] and the NDVI (Normalized Differential Vegetation Index), with their corresponding differential forms for the multi-temporal approach (dNBR, dNDII and dNDVI respectively), have been widely tested and used. All such indices are based on a normalized difference between two spectral bands whose value varies according to particular environmental characteristics and features (Table 3).

***Table 3**. Definitions of the NBR, the NDII and the NDVI for SPOT and IRS data*

NBR	NDII	NDVI
$\frac{NIR - SWIR_{2.2}}{NIR + SWIR_{2.2}}$	$\frac{NIR - SWIR_{1.6}}{NIR + SWIR_{1.6}}$	$\frac{NIR - RED}{NIR + RED}$

NIR and RED represent the near infrared and the red bands respectively, and the SWIRj variables represents the short wave infrared bands centred at 2.2 μm and 1.6 μm. The multitemporal forms dNBR, dNDII and dNDVI of the corresponding indexes are simply defined as the difference between the index determined in the first image and the index determined in the second image.

dNBR = NBRbefore - NBRafter

dNDII = NDIIbefore - NDIIafter

dNDVI = NDVIbefore - NDVIafter

In this study, only the dNDII and dNDVI multitemporal methods are considered, because of lack of the MIR band in SPOT and IRS multispectral images. As in most multitemporal methods, those considered also rely on pairs of images - one acquired before and another acquired after the fire season. However, in this study we consider a new multitemporal approach based on the analysis of pairs of satellite images, both acquired in a suitable period after the fire season (Figure 4), in order to evaluate the sensitivity of the indexes to highlight the different temporal evolution of the spectral behaviour of burnt areas from those not affected by fire. This approach is based on the assumption that a burnt area presents new growth of vegetation that is generally different from the usual evolution of a vegetation in an area not affected by fire.

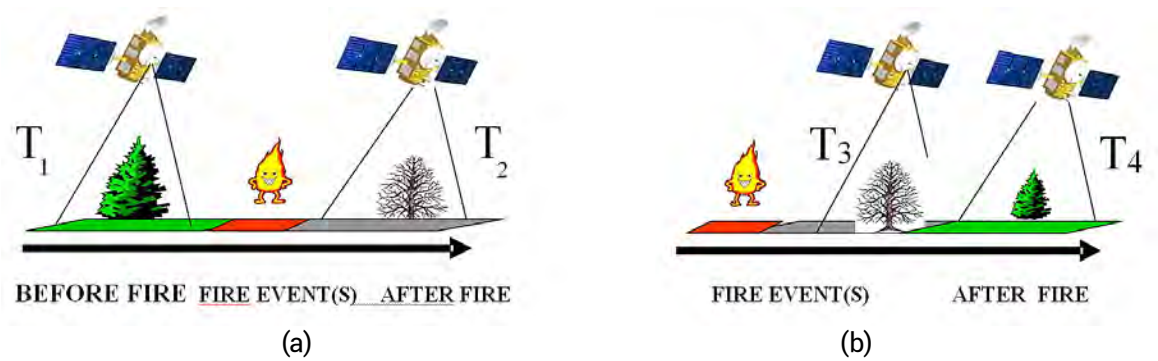

Figure 4. General scheme (a) of the multitemporal approach based on pairs of images acquired before (T1) and after the fire season (T2) respectively, and (b) the proposed multitemporal approach based on pairs of images both acquired after the fire event (T3, T4).

Highlights

The performance of the method for detecting and mapping burned areas was assessed in terms of true positive (pixels correctly classified as burned area) and false positive (pixels not affected by fire but wrongly classified as burned area) results. The performance was slightly better when the number of true positive was higher and the number of false positive lower. Unfortunately, these values are correlated and variation in the threshold selection yielded a greater number of true positive results as well as of false positive results. A trade-off must be therefore defined according to the user's requirements. The performance of the proposed method was assessed by varying the threshold selection of a fixed sampling step, in order to identify the most suitable threshold. A photointerpretation step was first performed on the selected image windows, in order to discriminate in each image the subset of burnt pixels from those of the pixels not affected by fire (Figure 5). The NDII and NDVI images were then for both the pairs of IRS and SPOT images (Figures 6 and 7).

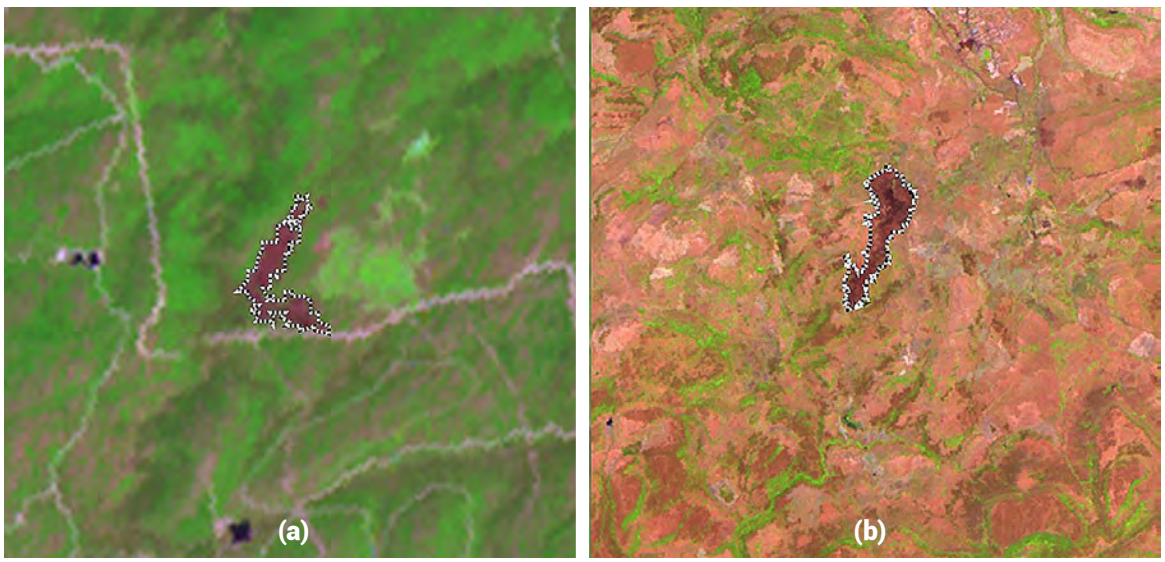

Figure 5. Interactive definition of fire scars on (a) the IRS and (b) the SPOT images. Band combination: R=4, G=3, B=2.

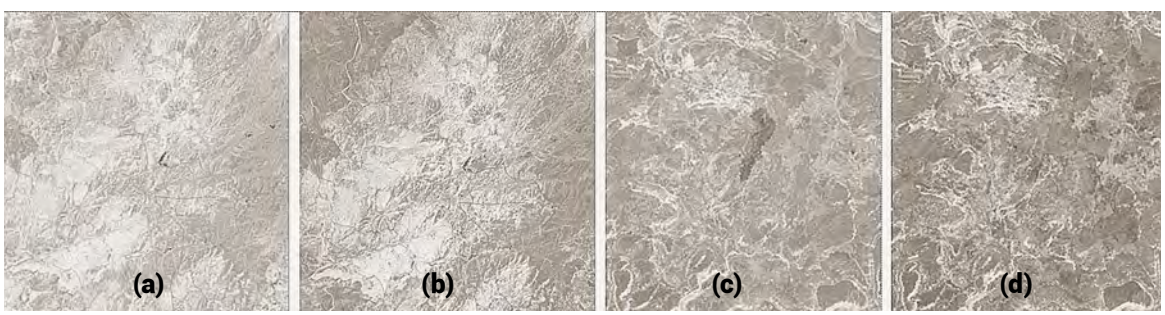

Figure 6. NDII of the IRS images from (a) 15 May and (b) 02 July and of the SPOT images from (c) 01 July and (d) 01 September.

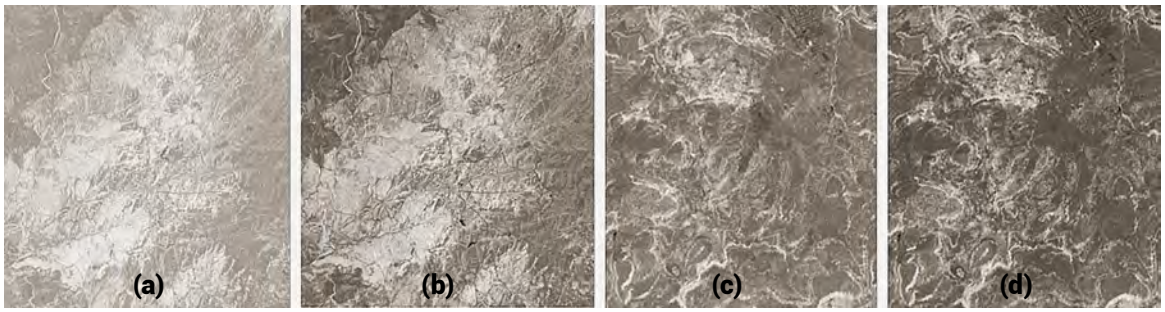

Figure 7. NDVI of the IRS images from (a) 15 May and (b) 02 July and of the SPOT images from (c) 01 July and (d) 01 September.

The dNDII and dNDVI images were then produced for both satellites.

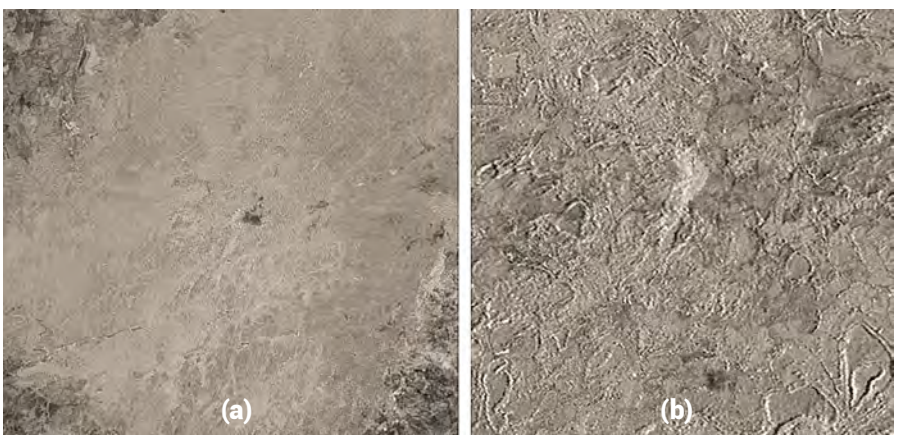

Figure 8. dNDII images evaluated with (a) IRS and (b) SPOT pair of images

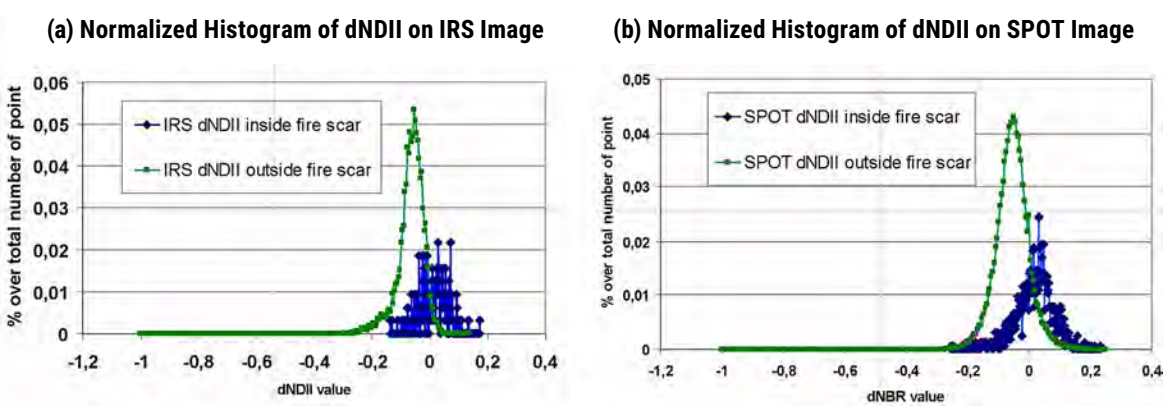

Figure 9. Normalized histograms of dNDII evaluated with the IRS (a) and SPOT (b) data

The dNDII images obtained from (a) IRS and (b) SPOT data are shown in Figure 8, whereas the corresponding histograms are shown in Figure 9, in which the occurrence was normalized to the total number of points. The result obtained using the NDVI index are shown in Figure 10, whereas the histograms of dNDVI images for both the IRS and SPOT satellites are reported in Figure 11. In this case, the histogram values are also normalized to the total number of points.

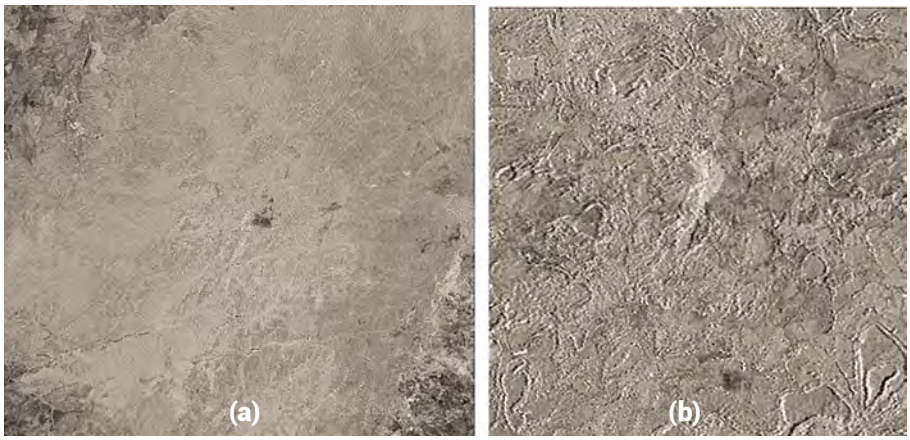

Figure 10. dNDVI images evaluated on (a) IRS and (b) SPOT pair of images.

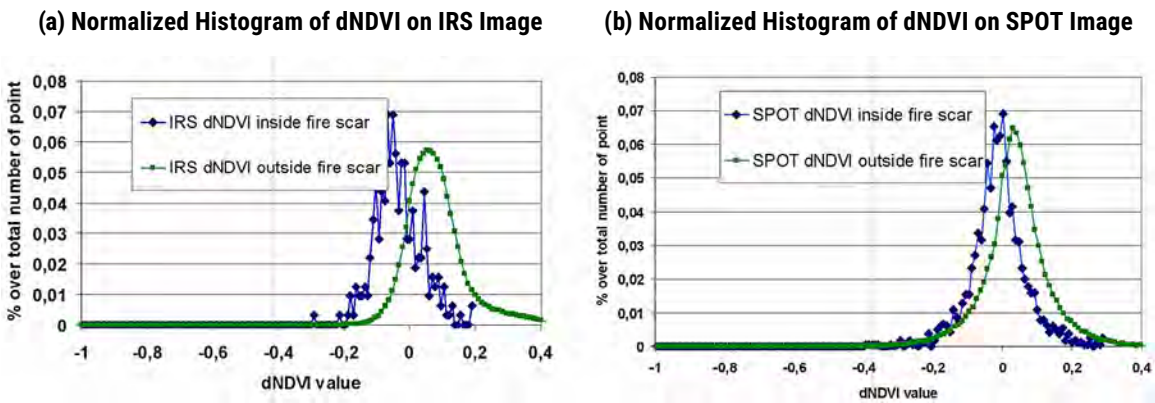

Figure 11. Normalized histograms of dNDVI evaluated on (a) IRS and (b) SPOT data

Analysis of the histograms in Figure 9 and Figure 11 reveals the partial overlapping of the histogram relative to the burnt area with that of the pixels not affected by fire, for both the dNDII and dNDVI indexes. The overlapping makes it impossible to completely discriminate the burned area from the unburned area by using a simple threshold. As a consequence, for any threshold value, some of the pixels corresponding to the fire scar are correctly classified as burned area, while others are wrongly classified as unburned. At the same time, some of the pixels outside the fire scar will be erroneously classified as burned area. By varying the threshold, the performance of the two multitemporal methods dNDII and dNDVI were thus assessed in terms of the percentage of true positives (i.e. the ratio between the true positives and the total number of pixels corresponding to the fire scar) and the percentage of false positives (i.e. the ratio between the number of false positives and the total number of pixels composing the image outside the fire scar).

The performance of both multitemporal methods was very similar when the multispectral images used and the range of time in which the considered images are acquired were varied. In fact, the IRS images were acquired at the beginning of the fire season, whereas the SPOT images were acquired during the last part of the same fire season. Both methods therefore seem to be quite robust. It is, however, important to point out that neither of the methods

are efficient enough to be used in an automatic burn scar detection procedure over large areas, because of the high number of false positives. In fact, although the percentage of false positives is quite low, this value was determined over a very large number of unburned points relative to the burned points, and therefore the resulting number of false positives is very high relative to the number of true positives.

Conclusions

Definition of effective standardized and replicable procedures for post fire landscape detection is increasingly important for the critical process of vegetation recovery. Detection via remote sensing techniques is most effective.

The performance of different methods of mapping fire scars was first assessed as the percentage of true and false positives on two pairs of images acquired by two different multispectral satellites, namely in a window of the images characterized by a single small burnt surface affecting a small part of the selected window. The performance of each method was compared with those of threshold methods based on a single image acquired after the fire and applied to the same NDII and NDVI indexes and evaluated with the same images used in the multitemporal method. The usual method based on examination of a single image acquired after the fire performed better with the image acquired a longer time after the fire relative to the corresponding image acquired a shorter time after the fire. This conclusion holds true for variations in the satellite acquiring the multispectral images and the range of time of acquisition. The performance of the proposed multitemporal approach was similar to that of the single image methods when the NDVI index was applied and much better when the NDII method was applied. The proposed multitemporal approach based on the NDII index was therefore identified as the best of the slicing threshold methods tested. The good performance was maintained by changing the adopted multispectral image and the period of acquisition, indicating the robustness of the procedure. Nevertheless, the NDII multitemporal method is not yet efficient enough to be used in an automatic burn scar detection procedure over large areas, because the resulting small number of false positive points are still excessive. Further refinement of the proposed approach is therefore required to improve the performance.

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5.4. Study case 4: Fighting fire with fire: does prescribed burning impact soils? Examples from Melbourne, Sydney and Perth (Australia)

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Background and Aims

Fire has been used by humans since ancient times. It has been used for cooking, illumination, warmth and rituals and also as a land-management tool to promote grass growth and halt encroachment of woody species, to prepare new cropland areas and even to hunt and control animal movements. Fire was the first tool that allowed humans to exert changes at the landscape scale, and many of these traditional uses for fire are still maintained today.

A new term for the use of fire in the landscape has emerged in the last decades: "prescribed fire". Prescribed fire, also known as prescribed burning, controlled or managed fire, is any supervised burning conducted to meet specific land management objectives. The many objectives of prescribed burning include restoration of natural fire regimes, creation of pastures, weed control and wildfire suppression operations. However, the most frequent use in many fire-prone regions is the reduction of hazardous fuel loads to decrease wildfire risk. In the USA and Australia, prescribed burning is already the most widely used wildfire-risk reduction practice; however, concerns about associated soil effects and other ecological and human-health impacts restrain its use in Europe and other parts of the world.

Fuel reduction burning is conducted during low-risk fire weather (i.e. conditions that are not very hot, dry or windy), in order to minimize ecological impacts and the chances of the fire escaping. Thus, the fires are generally of relatively low intensity and their direct effects on soils are usually negligible or very limited. The aim of the present case study was to determine the temperatures reached and their duration in the soil during three prescribed fires in Australian eucalypt forests in three contrasting locations. Prescribed burning has been used extensively for fuel reduction in Australia since the 1950s, when it was recognised that the Central European approach of 100% wildfire suppression was not sustainable, as many of the Australian forests and woodlands have very high fuel loads and are fire adapted, with natural fire intervals of less than 50 years.

Figure CSX-1. Left: Prescribed burning (monitored in three dry sclerophyll forests in Australia): top, Sydney (New South Wales, 23/09/2014); Centre, Melbourne (Victoria, 19/04/2016); Bottom) Perth (Western Australia, 04/04/2017). Right: Soil sampling after fire: Top, Sydney; Centre, Melbourne; Bottom, Perth.

Strategies

As part of a wider study focused on the effects of fire on the carbon cycle and water quality, we monitored prescribed fires in three different regions of Australia: Sydney (New South Wales) (23/09/2014), Melbourne (Victoria) (19/04/2016) and Perth (Western Australia) (04/04/2017). The type of land cover in these areas was dry sclerophyll eucalypt forest, although with somewhat different vegetation species and fuel loads. Soils ranged from sandy (Sydney) to fine-textured (Melbourne). The type of prescribed burning used was *underburning*, i.e. a surface fire that burns ground and understory fuels, but does not affect the overstory canopy (Figure CSX-1). Before each fire, three transects (30 m long and 10-30m apart) were established and 1 sampling point was located every 3 m (i.e. a total of 30 sampling points per fire) along each. Two or three thermocouples connected to data loggers (Lascar, Easylog) were installed in each sampling point to record the temperature every 10 seconds. At each location and sampling point, one thermocouple measured the temperature/duration in the litter layer (at ~0.5-1 cm depth), and the another one (or two in some cases) was placed at different levels in ground and aboveground fuels, the surface of the Oa soil horizon (only in the Sydney fire), and at 1-2 cm depth within the Ah soil horizon (in the Sydney and Perth fires).

Highlights

The behaviour of the three prescribed fires studied ranged from low intensity (<500 kW m^{-1}) in Perth, to moderate intensity in Melbourne (500-2000 kW m^{-1}), and to moderate-high intensity (2000-5000 kW m^{-1}) in the Sydney fire (Figure CSX-1). All fires were ignited in the afternoon (12h-16h), with ambient temperatures of 22-26 °C, 45-55 % relative humidity and negligible wind speeds. Fuel moisture content of the litter and fine fuels (<1 cm diameter) was ~10% for both the Perth and Sydney fires, but higher (no data) in the Melbourne fire, as heavy rainfall occurred only a few days before the fire.

The litter layer (1-3, 1-5, and 1-2 cm deep in Sydney, Melbourne, and Perth, respectively) was mostly completely burned in the Sydney and the Perth fires, but only partially consumed in the Melbourne fire, probably due to the higher moisture content at the latter. Despite the differences in fire intensity and behaviour, the maximum temperatures (Tmax) recorded in the litter layer were similar for the three prescribed fires, ranging from sampling points with T max < 100 °C to the highest Tmax recorded of 889, 887 and 744 °C, and average Tmax (± standard deviation) of 487 (±280), 547 (±321), 428 (±215), for Sydney, Melbourne and Perth respectively (n = 27, 27 & 25). Burning residence times in the litter layer were very variable, both within and between fires. Most sampling points in the litter layer recorded temperatures >300 °C for only a few minutes, less than 10 minutes in all cases except at four of the Melbourne sampling points, where residence times of 33-85 minutes were recorded. At these points, relatively high ash loads indicated unusually high fuel loads. The temperature in the Oa horizon (Figure CSX-2) was only recorded in the Sydney fire. Of the 8 points monitored, in half of them Tmax<100 °C and in the other half Tmax ranged from 297 to 781 °C.

Despite the high temperatures registered in the litter layer and in some parts of the organic soil layer, the highest Tmax recorded in the mineral soil was 79 °C. Only 4 of the 14 sensors placed in the mineral soils (4 in Sydney and 10 in Perth) indicated temperatures slightly higher than 60 °C, which is considered the threshold above which the most temperature-sensitive biological soil properties begin to be affected. Importantly, the slight increase in

soil temperature only lasted for a few minutes (Figure CSX-2). No records are available for the mineral soil in the Melbourne fire, but at most of the sampling points (22 of 30) the litter layer had not been consumed to its entire depth and therefore the direct impact on the mineral soil underneath these can be assumed to be negligible.

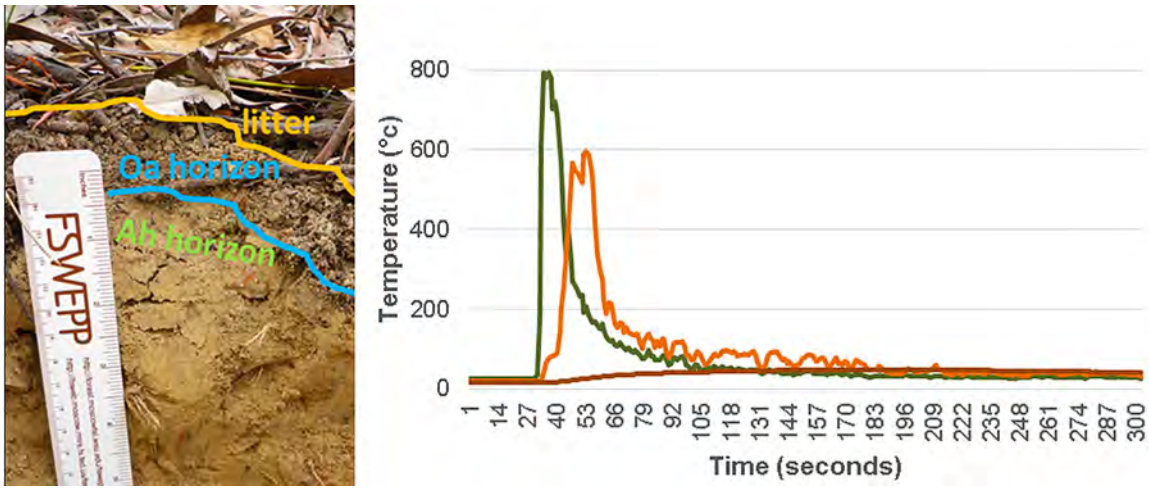

Figure CSX-2. Left: An example of a typical soil profile in the Sydney study area. Right: Temperature/duration profiles recorded in a single sampling point at the litter layer (green line), Oa horizon (orange line) and Ah horizon (brown line) during the Sydney fire.

Practical outcomes

None of the three prescribed fires studied is expected to have had substantial direct impact on the mineral soil as the recorded maximum temperatures in the first centimeters of the A horizon remained generally below 60 °C, with no measurement indicating temperatures exceeding 80 °C. Although our results must be considered with caution, as they are based on only three fires, they suggest that the prescribed burning practices applied here do not have a substantial direct impact on the mineral soil. Notwithstanding, full assessment of the impacts of prescribed fires in soils, must also consider indirect effects (e.g. inputs of ash, post-fire soil erosion, changes in vegetation, etc.). In addition, any cumulative indirect effects resulting from repeated burning should also be taken in consideration (e.g. prescribed fires in Australia are usually carried out every 5-10 years).

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V Pollution: soil contamination and soil rehabilitation treatments

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1. Introduction: Causes of soil contamination in the Mediterranean region

Soil pollution can be defined as the introduction of compounds into the soil environment at concentrations altering its functions or that are a threat to human health. Soil is, in fact, the basic natural resource for humans which are especially exposed through ingestion of food grown on polluted areas and inhalation of contaminated dusts. Pollutants in soil can be originated from several sources, especially in developing countries, which not only experience a rapid growth of population due to increasing rate of rural urban migration but also industrialization which is accompanied by air, water and soil pollution. The contaminants encountered at these sites include trace metals (such as lead, cadmium, mercury, chromium and nickel), volatile organic compounds (VOC) (such as benzene, toluene, and trichloroethylene), and semi-volatile organic compounds (SVOC) (such as total petroleum hydrocarbon (TPH), polycyclic aromatic hydrocarbons (PAHs) and polychlorinated hydrocarbons (CHCs)) (Table 1). Organic and trace metal contaminants are found to coexist at many sites (Krishna, 2010).

***Table 1**. Potential organic and inorganic contaminants of environmental concern*

Inorganic contaminants	Organic compounds
Barium	Acetone
Beryllium	Oil/fuel hydrocarbons
Cadmium	Aromatic hydrocarbons
Chromium	Benzene
Copper	Chlorophenols
Lead	Ethylbenzene
Mercury	Phenol
Nickel	Toluene
Vanadium	o-xylene
Zinc	m,p-xylene
Arsenic	Polycyclic aromatic hydrocarbons
Boron	Chloroform
Selenium	Carbon tetrachloride
Sulfur	Vinyl chloride
Cyanide (complex)	1,2-dichloroethane
Cyanide (free)	1,1,1-trichloroethane
Nitrate	Trichloroethylene
Sulfate	Tetrachloroethylene
Sulfide	Hexachlorobuta-1,3-diene
Asbestos	Hexachlorocyclohexanes
	Dieldrin
	Chlorobenzenes
	Chlorotoluenes
	Pentachlorophenol
	Polychlorinated biphenyls
	Dioxins and furans

The industrial operations which mainly contribute to trace metal and organic pollutant soil contamination are smelting, metal forging, manufacturing of alkaline storage batteries, combustion of fossil fuel and the spillage of liquids such as oil or solvents (Collins et al. 2002). Moreover, agricultural activities such as application of agrochemicals (fertilizers, pesticides and herbicides), use of sewage sludge in agricultural practices and irrigation with polluted water could also add significant amounts of organic and inorganic contaminants to the soils (Vaca-Paulìn et al. 2006; Liu et al. 2006). Among the organic contaminants, PAHs are the most widespread in soils, water and wastewater (Puglisi et al. 2007). PAHs originate mainly from combustion of fossil fuels and direct release of oil and its products (Johnsen et al. 2005). Chlorinated hydrocarbons (CHCs) are used mainly for the manufacturing of synthetic solvents and insecticides. They are environmental contaminants that bioccumulate and hence can be detected in human tissues.

There is no European-wide monitoring of contamination diffusion. Monitoring exits only on a country-by-country basis. Countries are at different levels of progress and apply different methodologies and definitions. Several countries have initiated national inventories of

contaminated sites. However, data on the number of contaminated sites based on national inventories is not currently comparable, since it is based on different national approaches. Therefore, national totals do not represent the scale of the problem, but give only an indication of the efforts made by each country.

In 2011, information available from 20 European countries revealed that the estimated number of potentially contaminated sites in Europe was 2,5 million and about 45% of them have been identified. More than 10% of the identified sites or around 127000 have been confirmed as contaminated sites. The ratio of remediated sites to contaminated sites is around 45% as more than 58000 contaminated sites have already been remediated (European Commission, 2014; Panagos et al., 2013).

In these sites, the main contaminant categories were trace metals and mineral oil contributing to around 60% in soil contamination, followed by BTEX, CHCs, PAHs and others which had similar contributions to soil contamination varying between 8 and 11% (Figure 1).

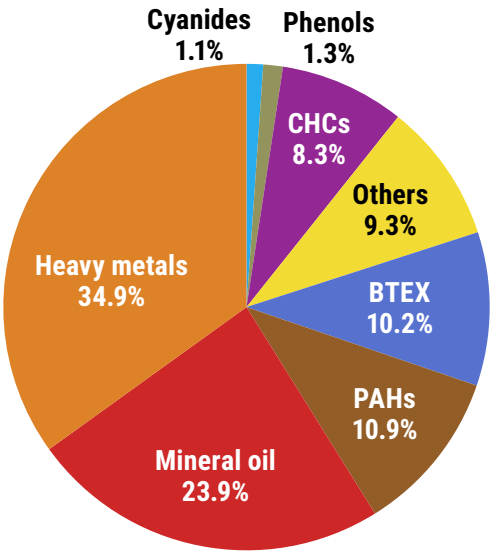


Figure 1. Distribution of contaminants affecting soil in Europe (European Commission, 2014; Panagos et al., 2013).

Land contamination usually affects areas with a high density of urban agglomeration and with a long tradition of heavy industry, or in the vicinity of former military installations.

The largest and most affected areas are located in north-west Europe, from Nord-Pasde Calais in France to the Rhein-Ruhr region in Germany, across Belgium and the Netherlands. Other areas include the Saar region in Germany; northern Italy, north of the river Po, from Milan to Padua; the region located at the corner of Poland, the Czech Republic and the Slovak Republic, with Krakow and Katowice at its centre; and the areas around all major urban agglomerations in Europe.

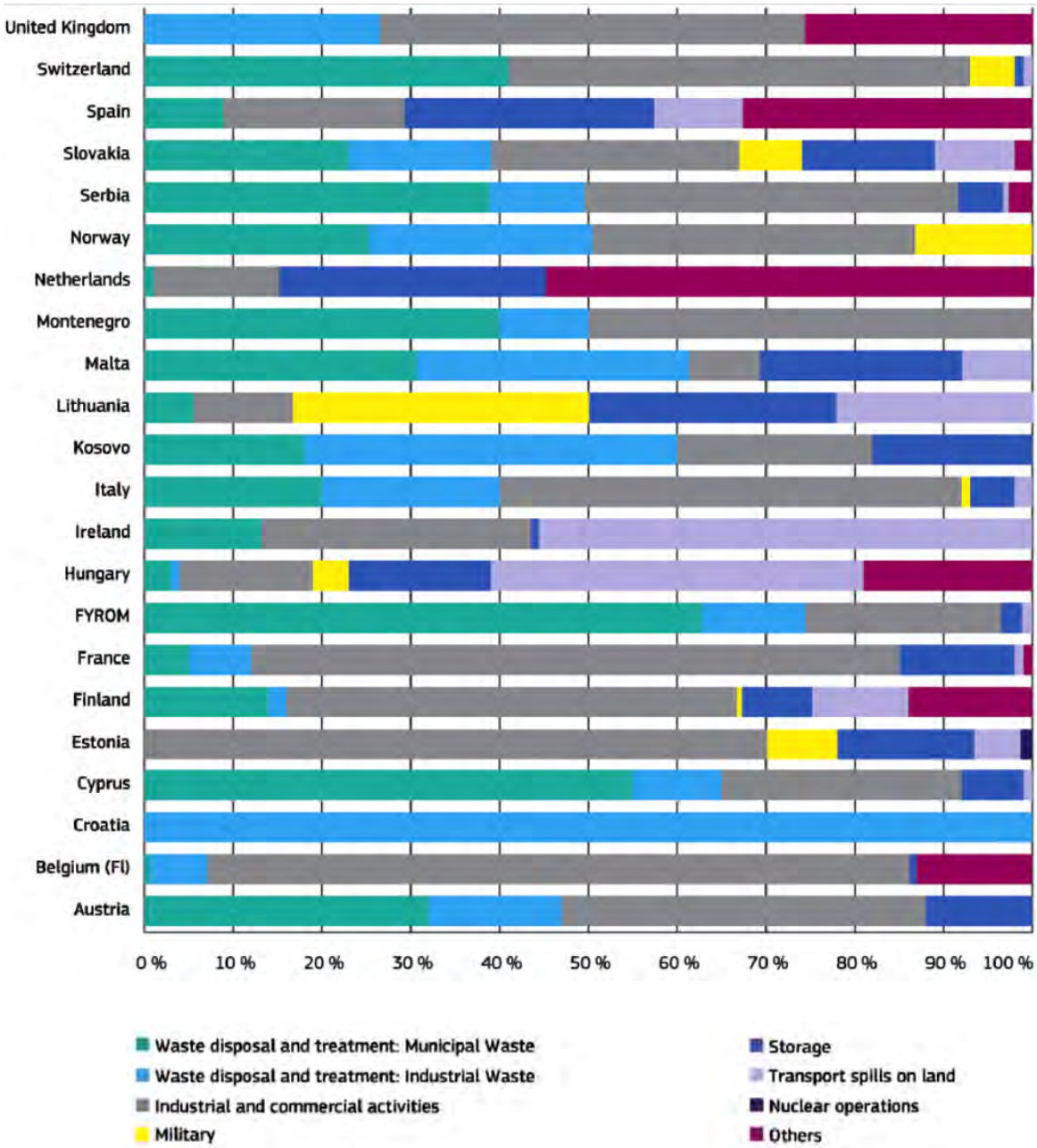


Figure 2. Breakdown of activities causing soil contamination as reported in 2011 (European Commission, 2014).

2. Soil-contaminat interaction

2.1. Key properties of contaminants

A number of contaminant properties influence the partitioning of inorganic and organic contaminants, and contaminant mixtures, between the solid, liquid and gaseous components in soil and sediment, hence should be understood when predicting the behavior of contaminants.

The key properties of contaminants include:

1) Solubility - The aqueous solubility of contaminants can be defined as the maximum concentration of a chemical in the aqueous phase when the solution is in equilibrium with the pure compound at a standard temperature and pressure. Solubility thus controls the amount of a chemical that can partition into the aqueous phase and hence be capable of transport through it (e.g. from soil pore water to groundwater).

The aqueous solubility of organic contaminants is one of the key factors determining their behavior and impact on the water environment. For pure substances, aqueous solubility is related to molecular structure and polarity. As a general rule, the more soluble organics are charged or contain oxygen or nitrogen groups that can form hydrogen bond with water. Therefore polar organic contaminants (e.g. methanol, phenol) will be soluble in water (itself a polar solvent) and non-polar organic contaminants (e.g. benzo(a)pyrene, PCBs) remain largely insoluble or hydrophobic (Brusseau and Bohn, 1996).

2) Volatility - Volatility can be defined as the tendency of a compound to partition into the gaseous phase, and is typically measured by the vapour pressure (Keith, 1988). It is a property relevant primarily to organic compounds, although it may influence the partitioning of some inorganic compounds, such as mercury or cyanide. The vapour pressure is a measure of the pressure exerted by the vapour of a compound at equilibrium with its pure condensed phase (either solid or liquid) at a standard temperature. It may range by many orders of magnitude for organic compounds. The composition of a mixture will influence the vapour pressure of any substance present in the mixture. The volatility of a compound is determined by the strength of the intermolecular forces between molecules. Therefore, solids (with strong intermolecular forces) have lower vapour pressures than liquids and gases that have weaker intermolecular forces (Brusseau and Bohn, 1996).

3) Immiscibility with water - Liquid phases that are immiscible with water in soil will tend to move through the unsaturated zone as a separate phase, a non-aqueous phase liquid (NAPL). The NAPL will partition in the soil by a combination of (Suthersan, 1997):

- moving through the pore space due to gravity and capillary forces;
- coating the solid matrix;
- dissolving in pore water;
- volatilization;
- trapped in pore spaces under capillary forces (at residual saturation).

In general, the residual saturation of a NAPL will tend to increase as the permeability of the matrix decreases, and downward flow cannot take place until the residual saturation is exceeded. The retention capacity is a function of the residual saturation and soil porosity and, for oil in the unsaturated zone ranges from about 3 to 5 lm^{-3} for highly permeable media to 30 to 50 lm^{-3} for low permeability media (Pankow and Cherry, 1996). The NAPL retained in the unsaturated zone will act as a secondary source of contamination due to solution and volatilization and the attenuating properties of the soil to other contaminants changed because of the surface coating of the matrix. The nature of organic contaminants can substantially increase the intrinsic permeability and subsequent transport properties of contaminants through soils (Kunkel and Anderson, 1999).

2.2. Effects of contaminants on soil properties

The ability of a system to withstand a disturbance (resistance) and the speed of return towards its predisturbance state or a new stable state (resilience) are the two components of ecosystem stability as described by Pimm (1984), McNaughton (1994) and Loreau et al. (2002). Specifically, resistance refers to the immediate capacity of a system to maintain its properties (e.g. community size, composition, and function) in the presence of a disturbance, while resilience describes the ability of an impacted system to recover its initial status (e.g. community structure and function) following a disturbance over time. As may be in Figure 3, resilience can recover the previous state after system disturbance (a) or reach a new stable state according to the reaction capacity of the disturbed system (b or c). Of course the strongest damage to the disturbed system is in (c) because the system looses a large part of its function.

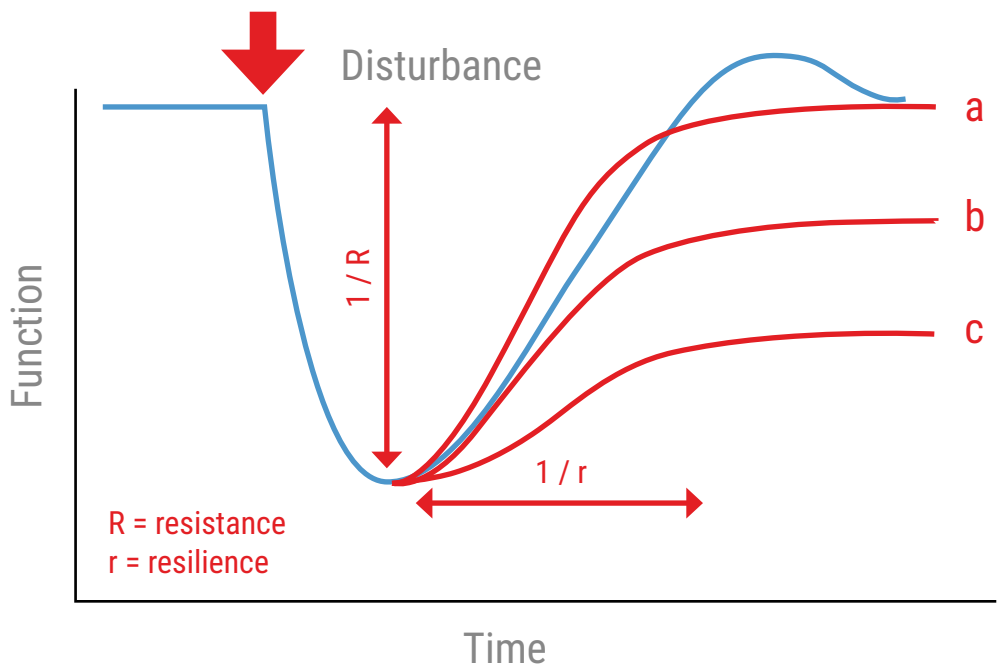


Figure 3. Schematic representation of soil response to disturbance (resistance) and then recovery over time (resilience) (Griffiths and Philippot, 2013, modified).

Soil contaminants have great impacts on ecosystem stability. Contaminants commonly reduce the microbial biomass (Shukurov et al., 2014), affect the taxonomic diversity of soil communities (Stefanowicz et al., 2008), and it may alter a variety of enzyme activities, thereby disturbing the nutrient cycling and the capacity of soil to perform key ecological functions, such as mineralization of organic compounds and synthesis of organic substances (Giller et al., 1998; Moreno et al., 2011). Undoubtedly, soil microorganisms are essential for proper functioning of ecosystem and soil fertility. The structural diversity of a bacterial community has been found to be very sensitive to environmental changes, reacting by shifts in its composition (Vivas et al. 2008; Moreno et al. 2011). On the other hand, organic contaminants are assimilated by microorganisms as a carbon source for growth and energy and an increase in microorganism quantity is regarded as an indicator of contaminant degradation (Shukor et al. 2008).

In sites co-contaminated with organic and metal pollutants, the maintenance of a phylogenetically and functionally diverse microbial community can be seriously affected by the synergistic cytotoxic effect of multiple contaminants on soil microorganisms (Lin et al. 2006). A study on methyl tertbutyl ether (MTBE) biodegradation in presence of trace metals demonstrated that the metal ions Cu^{2+} (at 1 and 10 mg l^{-1}), Cr^{3+} and Zn^{2+} (at 10 mg l^{-1}) determine an inhibitory effect on MTBE degradation by *Pseudomonas aeruginosa* strain (Chi-Wen et al. 2006). Olaniran et al. (2011) evaluated the inhibitory effect of trace metals (cadmium, mercury and lead) on the aerobic biodegradation of 1,2-Dichloroethane (1,2-DCA) by autochthonous microorganisms in soil microcosms. In the Olaniran et al. (2011) study, a dose-dependent relationship between degradation rate of 1,2-DCA and metal ion concentrations was observed for all the trace metals tested, except for Hg^{2+}. In soil, trace metals can have long-term toxic effects within ecosystems and have a negative influence on biologically mediated soil processes. It is generally accepted that accumulation of metal reduces the amount of soil microbial biomass and various enzyme activities, leading to a decrease in the functional diversity in the soil ecosystem and changes in the microbial community structure (Barea et al. 2005). However, metal exposure may also lead to the development of metal tolerant microbial populations (Giller et al. 1998). However, some of the trace metals are essential for sustaining the metabolic proccesses of living organisms. There is also the case of trace metal deficiency. For examples, Ni deficiency in plants disrupts nitrogen metabolism leading to the accumulation of toxic amounts of urea, inducing chlorosis and necrosis.

In soil, the interaction between elements it has been observed and both positive and negative effectss are reported. As an example, P, Ca, Fe and Cu react negatively with Zn and reduce its absorption by roots or/and its translocation to shoot in plants.

In addition, several studies have shown that the activities of enzymes in soil are related to the trace metal contamination. The enzymes in the soil play an important role in the process of organic matter decomposition, including the decomposition and the detoxification of contaminants. They can rapidly change in response to changes in soil caused by both natural and anthropogenic factors (Nannipieri et al. 2002; Gulser and Erdogan 2008). Chander et al. (1995) found that the activities of almost all enzymes in the soil were significantly reduced by 10 to 50 times with the increase of the concentration of trace metals.

Soil contaminats generally have also strong negative effect on plant community due to both direct contact toxicicy and indirect deleterious effects on the abiotic and microbial components of soil.

In contaminated sites, the vegetation cover has not only an aesthetic importance, it has different roles, such as stabilization of the area, prevention of wind-blown dust, and reduction of run-off and erosion problems.

2.3. Soil properties related with contaminant behaviour

Soils will react with contaminants to greater or lesser degrees, depending on the physical and chemical properties of the soil and nature of the contaminant.

The mobility and bioavailability, and hence potential toxicity of contaminants in soil depend on their concentration in soil solution, the nature of their association with other soluble species, and soil ability to release the contaminants from the solid phase (Krishnamurti et al. 2007). Of course for trace metals, the total metal concentration is of interest, but it is now accepted that understanding the environmental behavior by determining their speciation is of paramount importance. Metal speciation in soils, related to the distribution of an element among chemical forms or specie, is generally carried out with specific extractants which solubilize different phases of metals (Mulligan et al. 2001). Therefore, chemical speciation allows the estimation of the mobile and bioavailable fraction, thus indicating their potential toxicity in the natural compartment of the environment.

Figure 4 shows some typical physical and chemical properties of soils related with the mobility and bioavailability of contaminants.

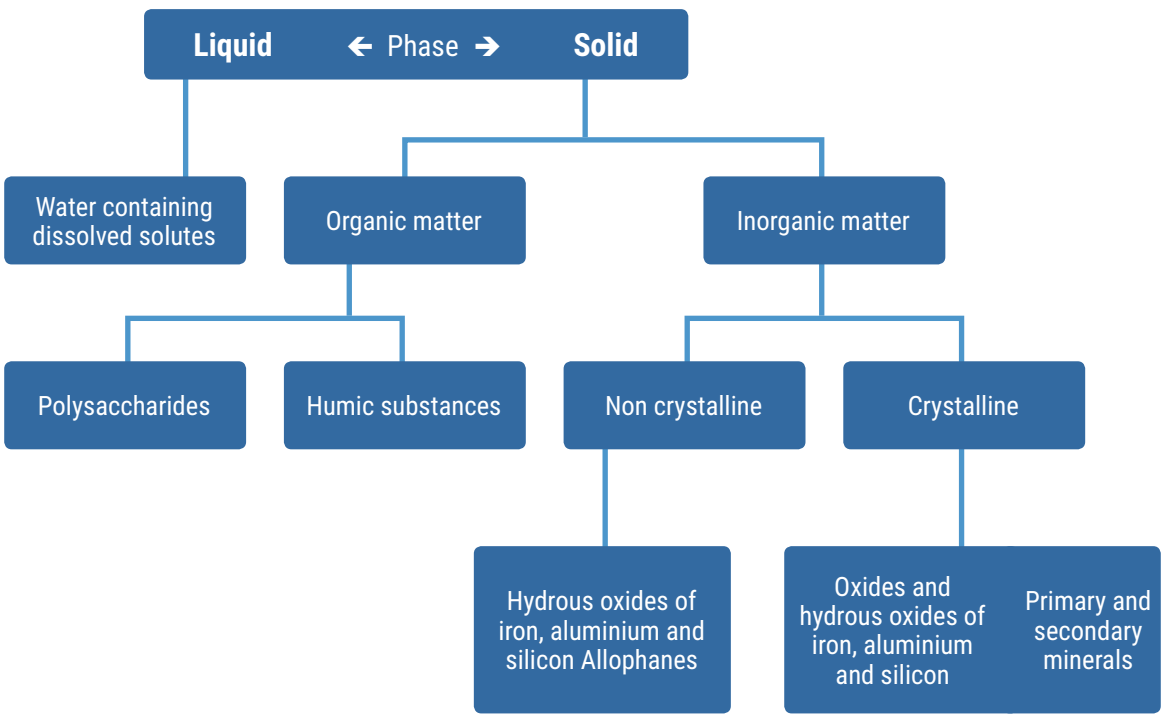


Figure 4. Physical and chemical properties of soils related with contaminants' mobility and bioavailability.

The procedure to fractionate the soil trace metals is able to differentiate the following fractions: 1. Exchangeable fraction associated with carbonated phase (Fraction 1). Metals are adsorbed on the soil components and Fe and Mn hydroxides. This is the most mobile fraction potentially toxic for plants. 2. Reducible fraction associated with Fe and Mn oxides (Fraction 2). Trace metals are strongly bound to these oxides but are thermodynamically unstable in anoxic and acidic conditions. 3. Oxidizable fraction bound to organic matter (Fraction 3). It is well known that metals may be complexed by natural organic substances. These forms become soluble when organic matter is degraded in oxidizing conditions. This fraction is not considered to be bioavailable and mobile because the metals are incorporated into stable high molecular weight humic sub-stances, which release small amounts of metals very slowly. 4. Residual fraction (Fraction 4). The residual solids mainly contain primary and secondary solids occluding the metals in their crystalline structures. It is considered to be unextractable and in an inert form.

2.3.1. Organic contaminants

Physical-chemical properties of organic pollutants (e.g. aqueous solubility, polarity, hydrophobicity, lipophilicity and molecular structure) control their fate and behaviour in soil (Reid et al. 2000).

Moreover, several environmental factors, e.g. organic matter (Puglisi et al. 2007), clay minerals (Lair et al., 2007), temperature, water content, pH, salinity, supply of oxygen and nutrients are well known to affect biodegradation of organic contaminants in soil (Kurola and Salkinoja-Salonen 2007).

Together with abiotic factors, biotic agents are of great importance in controlling the contaminant degradation in soil environment. The presence of suitable microorganisms for degrading the organic contaminants is critical for the naturally occurring biodegradation.

However, some site conditions, such as marginal environmental conditions or high concentrations of contaminants or organic vapors, can limit the microorganism growth and activity (Moreels et al. 2004).

Temperature - Generally, bacterial metabolic activity and contaminant biodegradation increase with increasing temperature up to an optimum value reported to be around 30-40 °C (Zhang et al. 2005).

Electrical conductivity - A high electrical conductivity of soil inhibit microbial activity (Luna-Guido and Dendooven 2001; Ramirez-Fuentes et al. 2002). An inhibitory effect of artificial salinity on mineralization of oil has been reported (Rhykerd et al. 1995). Mille et al. (1991) found an inhibitory effect of salinity above 0.4 mol/L NaCl (2,4% NaCl) that was greater for the biodegradation of aromatic and polar fractions than for the saturated fraction of petroleum hydrocarbons. However, different results have been obtained when investigating naturally salt containing soils, since indigenous microorganisms in such environments can be salt-adapted (Geiselbrecht et al. 1998). An interesting phenomenon is that low concentrations of salt (<1 %NaCl) slightly stimulated mineralization in some cases (Ulrich et al. 2009).

Moisture - Another very important parameter is the moisture level; the optimum moisture level for the biodegradation of petroleum hydrocarbon reported in literature is between 45 and 85 % of the soil's water holding capacity (US EPA 2006). At higher water contents, there is a risk

of the onset of anaerobic conditions arising from the slow rate of oxygen diffusion through water. At lower water contents, water availability becomes a limiting factor for microbial activity, movement and bioavailability of contaminants (Treves et al. 2003).

pH - Contaminant degradation has also been shown to be favored in slightly alkaline soils, where hydrocarbon degrading bacteria become less competitive with increasing acidic conditions (Maeir et al. 2000).

Nutrients - Bacteria require nutrient elements, such as nitrogen and phosphorus for incorporation into biomass and the synthesis of cellular components. The presence of these nutrient elements in soil is therefore critical for the biodegradation of organic contaminants (Atlas and Bartha 1992). The optimization of the C:N:P ratio is thought to be one of the most important actions enhancing the rates and extents of petroleum hydrocarbon biodegradation in soil.

Organic matter and clay minerals - Normally, as the time of contact between contaminant and soil increases there is a decrease in chemical and biological availability, a process termed 'ageing' (Hatzinger and Alexander 1995). For example, Uyttebroek et al. (2007) reported a biphasic loss of PAHs in a contaminated soil with phenanthrene and pyrene. In particular, the degradation and volatilization of PAHs was fast during the first 30 days and slow but continuous during the 140 day experimental period. Sorption to clay minerals and organic soil components are considered the dominant processes in the sequestration of organic contaminants in soil. These soil-contaminant contacts influence their bioavailability, and are responsible for the decrease in contaminant degradation (Reid et al. 2000; Semple et al. 2007). Soil aggregation is a complex biological process. Primary particles are bound together by bacterial products and fungal hyphae into stable microaggregates (2-20 mm in diameter). These are further bound into larger microaggregates by bacterial polysaccharides (20-250 mm in diameter), acting as binding agents. Microaggregates are subsequently packed into macroaggregates (>250 mm in diameter), through bacterial polysaccharides and fungi mycelia acting as interaggregate binding agents increasing the size of macroaggregates. Organic contaminants that are similar to organic matter, i.e. they have a phenolic structure, can be entrapped and/or strongly bound within soil aggregate; the resulting bounds lead to stable ''almost irreversible'' incorporation of pollutants into the soil (Gevao et al. 2000). It has been shown that organisms such as bacteria, earthworms, or plants can access these supposedly unavailable fractions by ''facilitated desorption processes'' (Park et al. 2001; Stokes et al. 2006) or diffusion back out of the micropore (Johnsen et al. 2005). A laboratory experiment on the biodegradation of phenanthrene in soil proved that fungal mycelia bridged air-filled pores and thereby provided a continuous network of water-paths that mobilized soil bacteria and facilitated the access of the bacteria to the contaminant (Wick et al. 2007).

Humic substances - Humic substances (HSs), both exogenous and endogenous, have been found to greatly strengthen aggregate formation and stability in soil (Piccolo et al. 1997), thus representing an important factor in the control of organic compound incorporation into less or inaccessible compartments. Moreover, chemical, photochemical or enzymatic catalysts can mediate the formation of covalent bonds between pollutants and soil HSs (Gevao et al. 2000). HSs showed an important role on sorption and binding of PAHs or PAH metabolites (Conte et al. 2001). The water-dissolved fraction of humic acids (HAs) can act as carriers of PAH compounds. In an experiment on phenanthrene degradation using Sphingomonas sp. and two humic acid concentrations, the increase of HAs increased the rates of phenanthrene degradation (Smith et al. 2009). This can only be interpreted by an HA-mediated transport of phenanthrene to the cells, supplementing diffusive uptake from the freely dissolved phase.

2.3.2. Inorganic contaminants

As described for the organic contaminants, trace metals can also be involved in a series of complex chemical and biological interactions. The most important factors which affect their mobility are pH (Gomes et al. 2001), sorbent nature and presence and concentration of organic and inorganic ligands (Harter and Naidu 1995), including humic and fulvic acids, root exudates and nutrients.

Organic matter and clay mineral - Organic matter has a large capacity to adsorb heavy metal nonspecifically because of its high cation exchange capacity and specifically when forming simple covalent bonds and chelates (Stevenson and Fitch 1986). The carboxylic and phenolic groups, present in large number in the structure of humic and fulvic acids, are responsible for the adsorptive capacity of organic matter (Harter and Naidu 1995; Kinniburgh et al. 1996). In a study performed by Kinniburgh et al. (1996) on metal ion binding by humic substances, a prevalence of carboxylic sites was identified at acidic pH (median value 2.98), while phenolic type prevailed at basic pH (median value 8.73). A study aimed to quantify the contribution of mineral and organic soil compounds to the heavy metal sorption capacity, clearly showed that organic compounds are the major source for metal sorption in soil. In this study the organic carbon showed a sorption capability for heavy metals 6-13 times higher than soil clay minerals (Lair et al. 2007).

pH - Acidification of soil directly influences the types of adsorption to both organic and inorganic soil particles (Sauve et al. 2000). The H^+ ions may exchange heavy metals at the cation exchange sites, thus desorbing the nonspecifically bound metals into the soil solution increasing their availability to biota (Alloway 1995).

A study on the investigation of the role of organic matter in bounding zinc in agricultural soils demonstrated that the content of organically bound Zn is related to pH and soil organic matter content (Dabkowska- Naskret 2003).

Soil pH regulates sorption competition between elements, negative charge of the exchangeable complex, dissolution of soil components, and ultimately determine the metal bioavailability and toxicity in relation to soil-respiration response (Azarbad et al., 2013). In many studies, soils with high pH values showed less respiration-variation rates, being less sensitive in producing toxicity problems.

Redox potential - Redox reactions, both biotic and abiotic, are of great importance in controlling the oxidation state and thus, the mobility and the toxicity of many elements, such as Cr, Se, Co, Pb, As, Ni and Cu. Oxidized conditions generally favor metal insolubility while reducing conditions favor metal solubility or mobility. Decreased mobile metal under oxidizing conditions can be caused by co-precipitation with Fe and Mn as amorphous oxides that scavenge metals (Baumann et al. 2002).

Under high concentrations of sulphate ions, the opposite trend (greater solubility under aerobic conditions) is observed for many metals; this is due to the fact that under the low redox potential, sulphate ions are reduced to the sulphide form that may form a complex with Cd, Ni and Zn metals and immobilize them as sulphide salts (Alloway, 1997).

3. Soil and sediment decontamination

Decontamination refers to processes or methods for treating contaminants such that they are contained, removed, degraded, or rendered less harmful (Kansas State University, 2005). Soil decontamination generally refers to processes that directly treat soil and affect the contaminant. This can either happen *in-situ* or *ex-situ* involving physical removal of soil and treatment on-site or at another location (off-site treatment).

Treatment methods are divided into chemical-physical, thermal and biological.

In Europe, contaminated soil continues to be commonly managed using "traditional" techniques, e.g. excavation and off-site disposal, which accounts for about one third of management practices. However, increasing regulatory control of landfill operations and associated rising costs, combined with the development of improved *ex-situ* and *in-situ* remediation techniques, is modifying the pattern of remediation practices.

In-situ and *ex-situ* remediation techniques for contaminated soil are applied more or less equally (Figure 5).

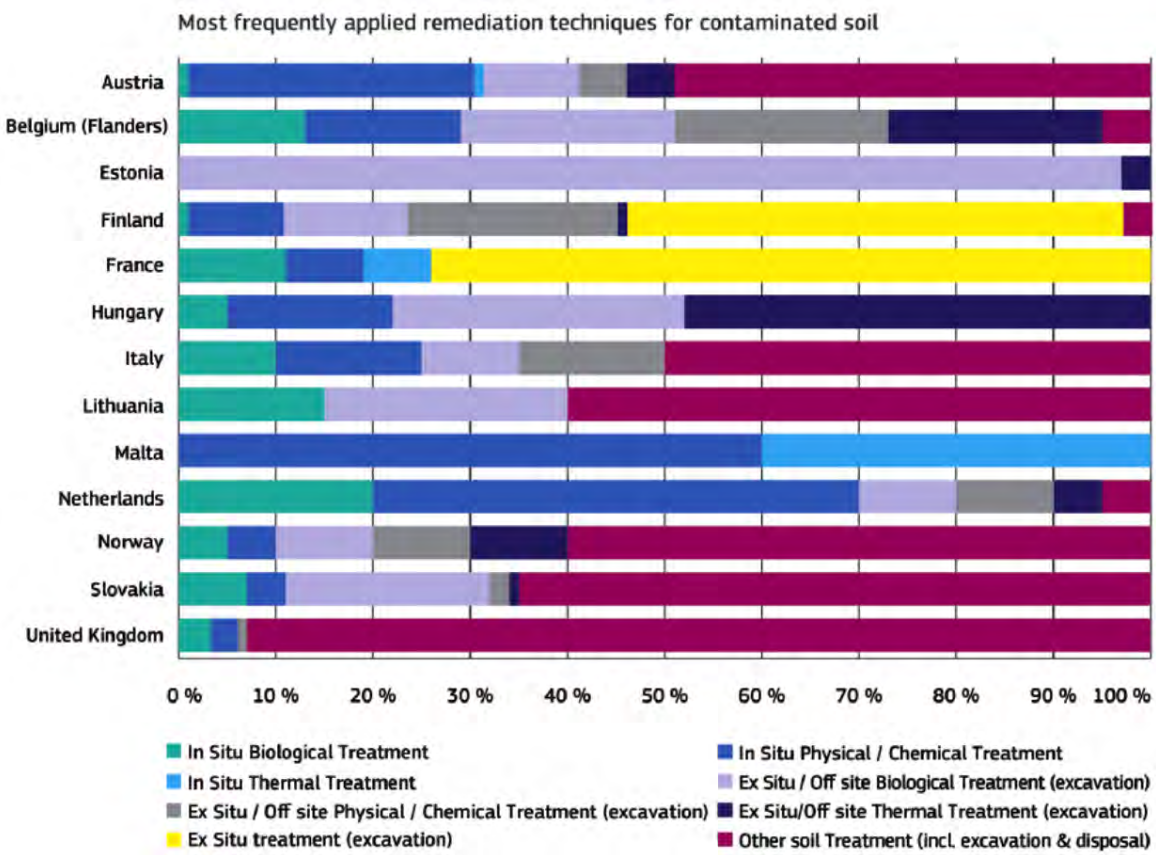


Figure 5. Dominant remediation technologies for contaminated soil reported in 2011 (European Commission, 2014).

3.1. In situ treatments

In situ: the treatment in place without excavation of contaminated soils or sediments.

3.1.1. In situ biological treatments

According to Perelo (2010), the bioremediation strategies can be the following: i) natural attenuation, which utilizes the "self-healing" natural capacity of the indigenous microbial population in combination with natural occurring physical and chemical processes; ii) biostimulation, which consists in the stimulation of indigenous microorganisms by introducing nutrients and oxygen into the contaminated matrix; iii) bioaugmentation, which involves the addition of external microbial strains (indigenous or exogenous) with specific degradation capacity.

Available *In situ* biological treatment technologies include:

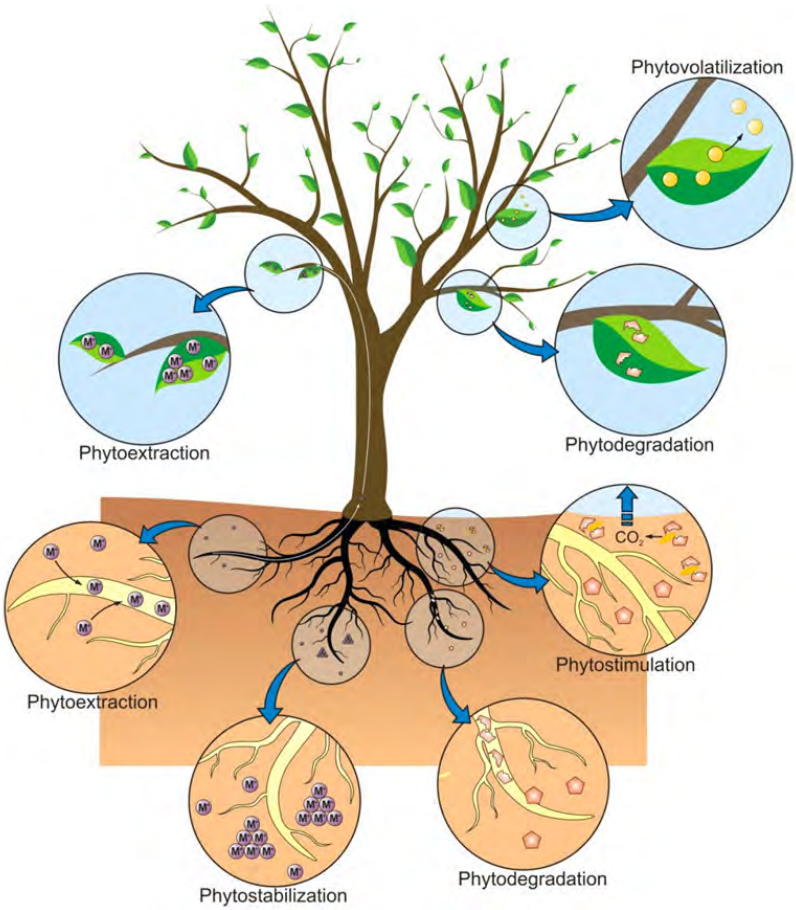
- Phytoremediation;
- Bioventing;
- Aeration.

Phytoremediation, a technology that uses plants to clean up pollutants from the environment, was defined in the '90s as a promising technology for soil remediation (Cunningham and Berti 1993; Raskin et al. 1994). The phytoremediation techniques copy the natural evolution of the ecosystems through energy capture and conservation (biomass production). This technique is based on soil/spoil contaminants stabilization (the containment processes) or on soils decontamination (removal processes). The effectiveness of this technology has been widely demonstrated in soil for many classes of pollutants, like oil hydrocarbons, polycyclic aromatic hydrocarbons, pesticides, dyes, chlorinated solvents, and heavy metals (Kagalkar et al. 2011; Nedunuri et al. 2000; Newman et al. 2001), and has also shown a strong potential for treatment of different contaminated matrices, such as sediments (Bert et al. 2009; Bianchi et al. 2010; Doni et al. 2015), sewage sludges (Peruzzi et al., 2017), waste water (Salvato et al., 2012). According to several authors (Chaney et al. 1997; Raskin et al. 1997, US EPA 2000), phytoremediation is usually classified on the basis of plant action; the well-known terms of Phytoextraction, Phytostabilization, Phytovolatization and Phytodegradation belong to this classification.

The main plant-based technologies of phytoremediation, each having a different mechanism of action generally include: (1) phytoextraction, in which plants absorb metals from soil and sediment and translocate them to harvestable shoots where they accumulate (2) phytodegradation, utilizing plants to degrade organic contaminants from soil and sediment; (3) phytostabilization, where plants stabilize, rather than remove contaminants by plant root metal retention (phytostabilization uses plants to convert soil metals/metalloids into less mobile forms, but not remove the chemical elements from the contaminated site); and (4) Phytovolatization, where plants are able of absorbing contaminats from the soil, biologically converting them into gaseous species within the plant, and releasing them into the atmospher.

Remediation of metals/metalloids contaminated substrata can be achieved by phytoextraction (a removal process) or by phytostabilization (a containment process) (Mahar et al., 2016, Mendez and Maier, 2008; Sarwar et al., 2017). According to Robinson et al. (2015), the efficiency

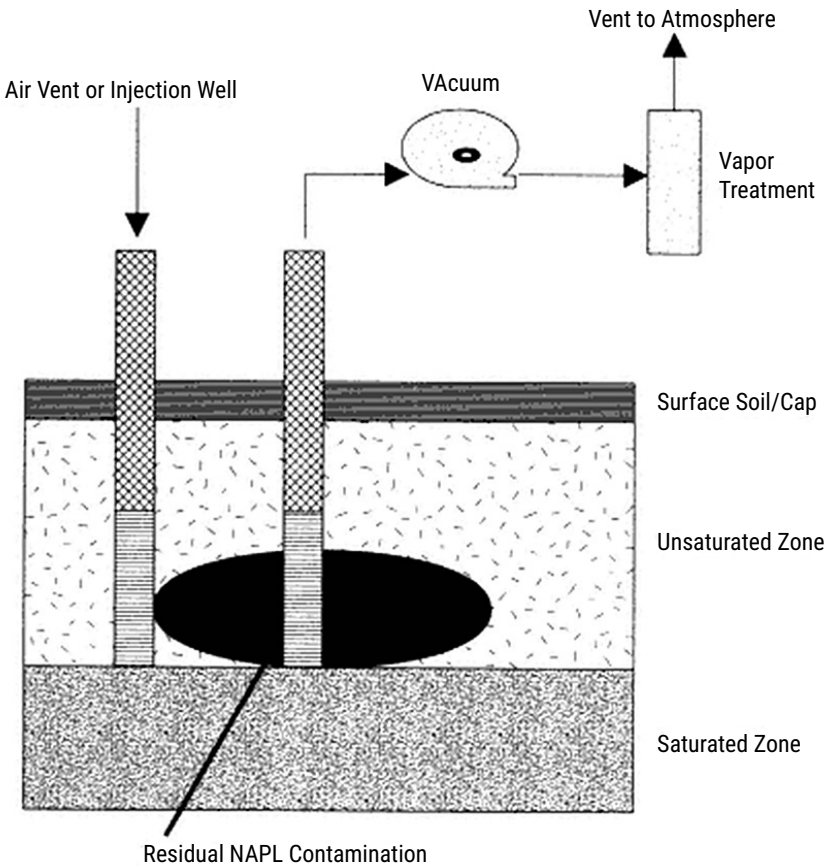
of metals/metalloids extracted per unit of land by this technique is low and more research is needed to demonstrate how phytoextraction could work at field scale. Phytostabilisation has the advantage, over phytoextraction, to eliminate the need of treating harvested shoot residues as hazardous waste. Among the *in situ* technologies, phytostabilisation is considered the most suitable for rehabilitation of different mine wastes or soils developed on mine wastes with multielemental contamination and submitted to Mediterranean conditions (Abreu and Magalhães, 2009; Mendez and Maier, 2008; Sheoran et al., 2010). In a great number of mine areas from Portugal, some of the waste materials disposed on the landscape constitute tailings with variable dimension, where incipient soils (e.g. Spolic Technosol Toxic) have been developed. These soils present vegetation cover, mainly composed of spontaneous species adapted to the edaphic conditions, as is the case of several shrubs belonging to the genera *Calluna*, *Cistus*, *Cytisus*, *Erica*, etc., or even introduced species as pine trees or eucalyptus. The spontaneous colonization of contaminated sites with well-adapted vegetation creating a vegetative cover can constitute a natural attenuation process of soil contamination by providing a long-term stabilization, contaminants containment and visual improvement. Vegetation cover reduces aeolian dispersion of soil/spoil particles, and roots prevent water erosion, immobilize potentially hazardous elements (PHE) and provide a rhizosphere environment where elements can be stabilized. However, these species cannot be accumulators of hazardous elements in shoot tissues (leaves, twigs, fruits and seeds), once they can be used/consumed by humans and/or animals and wildlife (Abreu and Magalhães, 2009 and references therein; Mendez and Maier, 2008).



Source: Favas et al., 2014.

The assisted phytoremediation is the *in situ* phytotechnology that uses inexpensive natural and/or industrial, agro-forestry by-products combined with plants to speed up the bioremediation process. Depending on the characteristics of the contaminated soils or waste materials, the addition of low cost amendments can contribute to increase organic matter content, water-holding and cation exchange capacity, pH, nutrients content as well as microbial activity and soil structure improvement. Certain amendments application together with plant physiological processes (e.g. rhizosphere modification and/or compounds released by roots) enhances key biogeochemical processes such as adsorption, precipitation and redox reactions that may decrease the metals/metalloids bioavailable fraction contributing to its immobilization in the media (Adriano et al., 2004) and increasing weathering and pedogenic processes of the waste materials (Abreu and Magalhães, 2009 and references therein; Macías et al., 2007).

Bioventing process injects air into the contaminated media at a rate designed to maximize in situ biodegradation and minimize or eliminate the off-gassing of volatilized contaminants to the atmosphere. Unlike biosparging, which involves pumping air and nutrients into the saturated zone, bioventing pumps the air only into the unsaturated or vadose zone (Mihopoulos et al., 2001). Bioventing also degrades less volatile organic contaminants and, because a reduced volume of air is required, it allows for the treatment of less permeable soils.



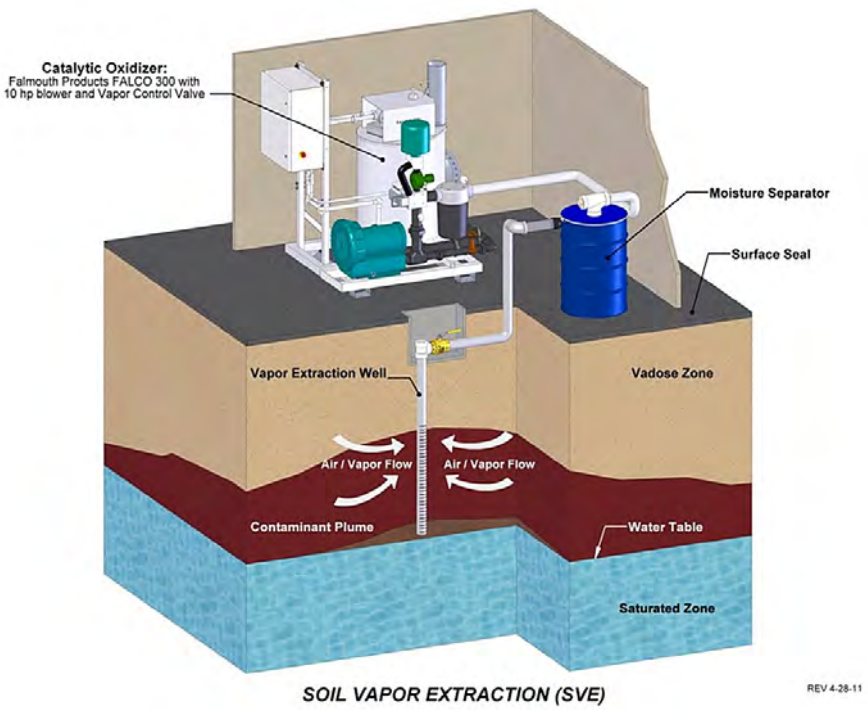
Aeration technology evaporates the volatile components of petroleum from the soil into the air. It is a well-developed process in which the area of contact between the water and the air is increased. The contaminated soil is spread thinly and tilled or turned to increase the rate of evaporation.

3.1.2. In situ physical/chemical treatments

Available *In situ* physical/chemical treatment technologies include:

- soil vapor extraction;
- electrokinetic;
- stabilization/solidification

Soil vapor extraction (SVE), also known as soil venting or vacuum extraction, is an accepted, recognized, and cost-effective technology for remediating unsaturated soils contaminated with VOCs and SVOCs (Zhan and Park, 2002; Halmemies et al., 2003). SVE involves the installation of vertical and/or horizontal wells in the area of soil contamination. Air 'blowers' are often used to aid the evaporation process. Vacuums are applied through the wells near the source of contamination to evaporate the volatile constituents of the contaminated mass which are subsequently withdrawn through an extraction well. Extracted vapors are then treated (commonly with carbon adsorption beds) before being released into the atmosphere (USEPA, 1995a). The increased airflow through the subsurface provided by SVE also stimulates the biodegradation of contaminants, especially those that are less volatile (USEPA, 1996b, Halmemies et al., 2003; Harper et al., 2003).



Source https://www.falmouthproducts.com/soil_vapor_extraction.html

SVE is generally most successful when it is applied to lighter more volatile petroleum products such as gasoline. Heavier fuels, such as diesel fuel, heating oils and kerosene, are not readily removed by SVE. The injection of heated air enhances the volatility of the heavier petroleum products, but the large energy requirements make it economically prohibitive (USEPA, 1995a, Zhan and Park, 2002).

Benzene, toluene, xylene, naphthalene, biphenyl, perchloroethylene, trichloroethylene, thrichloroethane, and gasoline are all effectively removed from contaminated soils by SVE systems (USEPA, 1996b; RAAG, 2000).

Potential Advantages: as it is an *in-situ* technology, site disturbance is minimal; can treat large volumes of soil at reasonable costs; capable of removing hydrocarbon fuels from beneath buildings and paved areas without disruptions; low labour requirements; it is effective at reducing VOCs in the vadose zone, thereby reducing the potential for further migration (USEPA, 1996b); short treatment time, ranging from a few weeks up to 2 years under optimal conditions; In-situ volatilization can be used in conjunction with various contaminant removal systems, such as bioremediation.

Potential Limitations: its applicability is limited to cases involving volatile compounds at site with a low groundwater table (Unsaturated as mentioned earlier); although the treatment time is relatively short in regards to other techniques, developing accurate models to predict the duration of this process is difficult if not impossible; concentration reductions greater than 90% are difficult to achieve (USEPA, 1996b; Zhan and Park, 2002); high moisture levels in the soil can reduce its permeability, and thus reduce the effectiveness of the technology by restricting the air flow through the soil pores; volatilisation is generally not appropriate for sites with a groundwater table located less than 0.9m below the land surface.

Electrokinetic

Electrokinetic remediation, also called electrokinetic processing of the soil, electromigration, electrokinetic decontamination or electrocorrection, can be used to extract metals and some types of organic residues, such as PAH, of saturated or unsaturated soils, sludges and sediments. This technique consists on the application of a direct current of low intensity between the electrodes located in the soil. The materials used for the construction of the electrodes can be graphite, stainless steel and platinum. Electrolysis of the water (in the disperse electrolyte) produces ions H+ in the anodes and ions OH- in the cathodes, generating a localized change of pH, which leads to the desorption of the contaminated ions. Some variations of this technique involve the direct extraction of metallic ions already in the metal form and others involve the extraction of metallic ions using a sequential process of ion exchange resins. Electrokinetic remediation can be also used to delay or prevent the migration and/or diffusion of the contaminants, directing them to specific sites and diverting them from the freatic sheets. Currently, the application of electrokinetic process has been considered promising, especially for the remediation of low permeability contaminated soils, where the electric field generated mobilizes electrically charged species, particles and ions in the soil by the processes of electromigration, electrophoresis and eletroosmosis. For the migration process in the electrodes, the contaminants can be removed by reduction in the cathode, precipitation, pumping next to the electrode, or in a more complex form with ion exchange resins. However, the electrokinetic process is limited by the solubility of the contaminant and by desorption of the contaminants in the surface of the soil. Heavy metals in their metallic state are not being sufficiently dissolved and separated from the samples of soil. The process is also not efficient when the concentration of the ions to be removed is low and the concentration of diverse ions is high. Moreover, factors such as heterogeneity and anomalies in the local surface (boulder, large quantities of iron or iron oxides, large rocks and gravel or materials such as shells) can reduce the efficiency of removal.

Stabilization/ Solidification

Stabilization generally refers to the process that reduces the risk posed by a waste by converting the contaminant into a less soluble, immobile, and less toxic form. Solidification refers to the process that encapsulates the waste materials in a monolithic solid of high structural integrity (Suthersan, 1997; Anderson and Mitchell, 2003). In situ stabilization and solidification involves three main components: (1) a means of mixing the contaminated soil in place; (2) a reagent storage, preparation, and feed system; and (3) a means to deliver the reagents to the soil mixing zone (Nyer, 1996). In situ and ex situ stabilization/solidification is usually applied to soils moderately contaminated by heavy metals and other inorganic compounds. However, stabilization of soils that contain low levels of organic constituents is feasible, even for volatile organics (Riser-Roberts, 1998; Druss, 2003; O'Day and Vlassopoulos, 2010).

Potential Advantages: raw materials are inexpensive; technology is well established and equipment is readily available; least expensive of the ex-situ technologies.

Potential Limitations: restrictions may be imposed on future land use; long term integrity of solidified materials are not well established; no test protocols; presence of high levels of organics in the soil may interfere with process.

3.1.3. In situ thermal treatments

Thermal treatment generally involves the destruction or removal of contaminants through exposure to high temperature in treatment cells, combustion chambers, or other means used to contain the contaminated media during the remediation process.

It includes:

Vitrification technology uses an electric current to melt contaminated soil at elevated temperatures (1,600 to 2,000 ºC or 2,900 to 3,650 ºF). Upon cooling, the vitrification product is a chemically stable, leach-resistant, glass and crystalline material. The high temperature component of the process destroys or removes organic materials. Radionuclides and most heavy metals are retained within the vitrified product. Vitrification can be conducted in situ or ex situ.

3.2. Ex situ treatments

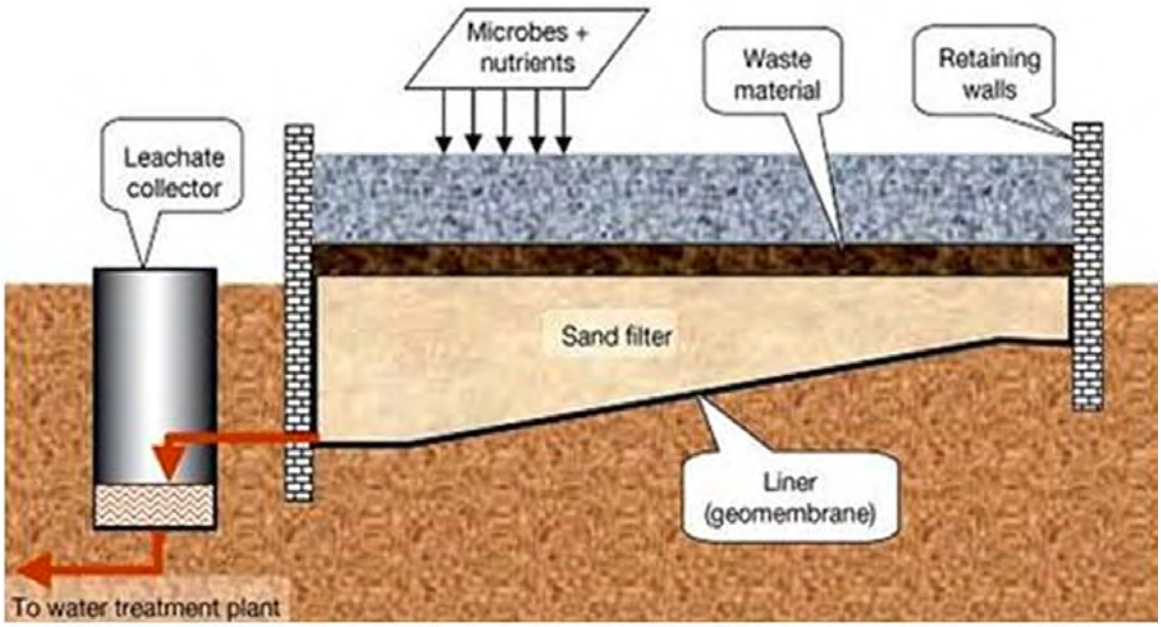
Ex situ: requires pumping of the groundwater or excavation of contaminated soil prior to remediation treatments.

3.2.1. Ex situ biological tratments

Available *Ex situ* biological treatment technologies include:

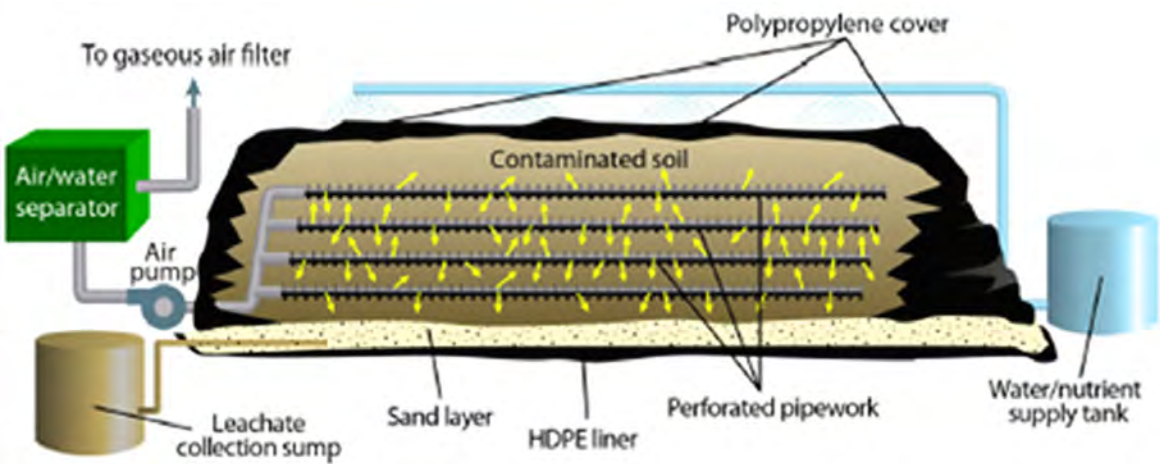
- Landfarming;
- Biopiles;
- Composting;
- Bioslurry systems.

Landfarming usually involves the spreading of excavated contaminated soils in a thin layer (no more than 1.5 m) on the ground surface of a treatment site and stimulating aerobic microbial activity within the soils through aeration and/or the addition of nutrients, minerals, and water/moisture (Hejazi, 2002). Bacteria, which have been selected for their success in breaking down hydrocarbons, are frequently added to the soil to achieve speedy degradation. The enhancement of microbial activity increases the degradation of adsorbed petroleum products (Riser-Roberts, 1998). The soil must be well mixed in order to increase the contact between the organics and microorganisms and to supply the oxygen required for aerobic biological degradation.



Falck W.E. Remediation of sites with mixed contamination of radioactive and other hazardous substances Technical Report, Juanary 2006.

Soil biopiles, also known as biocells, is a biodegradation technique used for the remediation of excavated soil contaminated with petroleum contents. This technology involves the accumulation of contaminated soil into piles and the stimulation of microbial activity either aerobically or by the addition of nutrients, minerals or moisture.

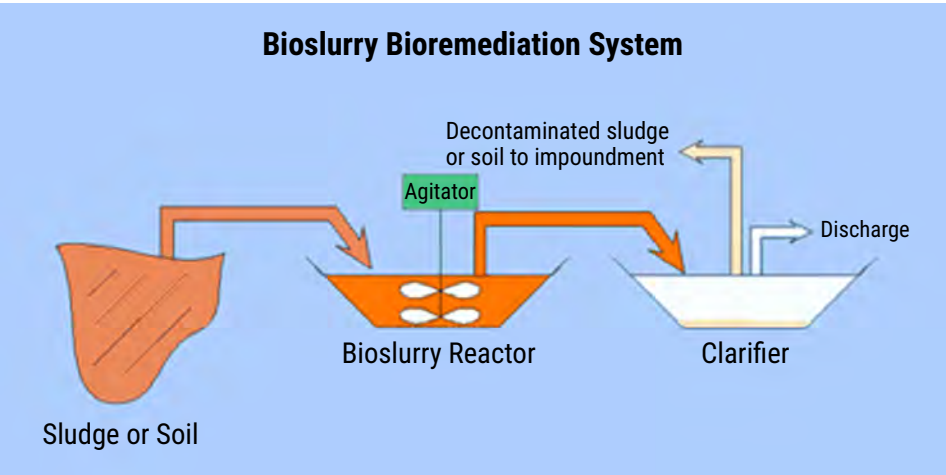


Source: UK EA 2002.

Biopiles are in a way similar to landfarms due to the fact that this technology also uses oxygen as a way to stimulate bacterial growth. However, while tilling or plowing aerates land farms, biopiles are aerated by forcing air to move by injection through perforated piping placed throughout the pile (EPA, 2003).

Composting involves mixing the contaminated soil with a bulking agent such as straw, hay, or corncobs to make it easier to deliver the optimum levels of air and water to the microorganisms. The most common designs are static pile composting, mechanically agitated composting, and window composting.

Bioslurry is an *ex situ* biological treatment that requires excavation of contaminated soil. It is accomplished by combining the excavated soil with water and other additives. In this system the bacteria selected for breaking down the contaminant is also added. The excavated soil is treated in a controlled bioreactor where the slurry is mixed to keep the solids suspended and the microorganisms in contact with the contaminants. In these reactors, biodegradation occurs at a rapid rate, with typical treatment times ranging from less than 1 month to more than 6 months (RAAG, 2000). After the process is completed, the slurry is dewatered and treated soil disposed (Zhang et al., 2001).



Schematic of a bioslurry bioremediation system. Source: Adapted from the U.S. EPA (8).

3.2.2. Ex Situ physical/chemical treatments

Available *Ex situ* **physical/chemical** treatment technologies include:

- Dehalogenation;
- Soil washing

Dehalogenation is a full-scale technology in which an alkaline polyethylene glycol (APEG) reagent is used to dehalogenate halogenated aromatic compounds in a batch reactor. Contaminated soils and the reagent are mixed and heated in a treatment vessel. In the APEG process, the reaction causes the polyethylene glycol to replace halogen molecules and render the compound nonhazardous or less toxic.

Soil washing is a cost-effective, relatively fast and useful *ex-situ* technology to split the raw material into fractions of homogeneous particle size, density or surface chemistry (Dermont et al. 2008). Solvents are selected on the basis of their ability to solubilize specific contaminants, and on their environmental and health effects. During soil washing, the specific surface area of the separated fractions, as well as their texture, affect their interactions with contaminants by sorption phenomena, and consequently the effectiveness of possible contaminant removal paths. Larger particles, due to their low area-to-volume ratio and negligible surface charge, can be readily decontaminated during soil washing itself (Weiss et al. 2010). Conversely, clay and silt, which contain high fractions of fine particles, may retain a significant amount of contaminants because of more stable soil structure and inability of the washing solution to permeate into microsites (Jeon et al. 2014). Due to this phenomenon, an effective soil or sediment washing requires the material being washed to contain a substantial percentage of sand or other coarse material, and it aims at obtaining "clean" sandy-coarse fractions after their separation from the smaller sized soil fraction. In fact, since hydrocarbon contaminants tend to bind and sorb to smaller soil particles (primarily clay and silt), separating the smaller soil particles from the larger ones reduces the volume of contaminated soil (Riser-Roberts, 1998). The smaller volume of soil, which contains the majority of clay and silt particles, can be further treated by other methods (such as incineration or bioremediation) or disposed in accordance with federal regulations. The clean, larger volume of soil is considered to be non-toxic and can be used as backfill (USEPA, 1996a; RAAG, 2000; Chu and Chan, 2003). Soil washing is often combined with other technologies.

Important observations related to soil washing performance are:

- Complex waste mixtures require a combination of solvents
- Pre-treatment is required for soils containing humic acids
- Organics adsorbed onto clay particles are difficult to remove (CPEO, 1997).
- Since soil washing does not destroy or immobilize the contaminants, the resulting soil must be disposed of carefully
- Wash water needs to be treated before its final disposal
- Soil washing is most effective for soil that does not contain a large amount of silt and clay.

Variations of Soil Washing

Chemical Extraction

Chemical extraction techniques operate much like soil washing but differ in the fact that instead of using water or water with wash- improving additives, it makes use of an extracting chemical to separate hazardous contaminants from soils.Furthermore, after initial physical separation steps are taken to grade soil into coarse and fine fractions (see: soil washing previous), chemical extraction separates the constituent contaminants into respective phase fractions: organics, water, inorganics and particulate soils.

Solvent Extraction

Solvent extraction is a common form of chemical extraction which uses an organic solvent as the extractant. Organically bound metals can be extracted along with the target organic contaminants.

Acid Extraction

Acid extraction operates in the same vein as chemical extraction, except it uses hydrochloric acid to extract heavy metal contaminants from soils.

3.2.3. Ex Situ thermal treatments

Available *Ex situ* thermal treatment technologies include:

- hot gas decontamination;
- incineration;
- pyrolysis;
- thermal desorption;
- Vitrification (*In situ*/*Ex situ*).

Hot gas decontamination involves raising the temperature of contaminated solid material or equipment to 260ºC (500ºF) for a specified period of time. The gas effluent from the material is treated in an afterburner system to destroy all volatilized contaminants. This method will permit reuse or disposal of scrap as nonhazardous material.

Inceneration. High temperatures, 870 to 1.200ºC, are used to volatilize and combust (in the presence of oxygen) halogenated and other refractory organics in hazardous wastes. The destruction and removal efficiency (DRE) for properly operated incinerators exceeds the 99.99% requirement for hazardous waste and can be operated to meet the 99.9999% requirement for PCBs and dioxins.

Pyrolysis is defined as chemical decomposition induced in organic materials by heat in the absence of oxygen. Pyrolysis typically occurs under pressure and at operating temperatures above 430ºC (800ºF). The pyrolysis gases require further treatment. The target contaminant groups for pyrolysis are SVOCs and pesticides. The process is applicable for the separation of organics from refinery wastes, coal tar wastes, wood-treating wastes, creosote-contaminated soils, hydrocarbon-contaminated soils, mixed (radioactive and hazardous) wastes, synthetic rubber processing wastes, and paint waste.

Thermal desorption involves the application of heat to excavated wastes to volatilize organic contaminants and water. Typically, a carrier gas or vacuum system transports the volatilized water and organics to a treatment system, such as a thermal oxidation or recovery unit. Based on the operating temperature of the desorber, thermal desorption processes can be categorized as either high-temperature thermal desorption (320 to 560ºC or 600 to 1,000ºF) or low-temperature thermal desorption (90 to 320ºC or 200 to 600ºF).

Vitrification technology uses an electric current to melt contaminated soil at elevated temperatures (1,600 to 2,000ºC or 2,900 to 3,650ºF). Upon cooling, the vitrification product is a chemically stable, leach-resistant, glass and crystalline material. The high temperature component of the process destroys or removes organic materials. Radionuclides and most heavy metals are retained within the vitrified product. Vitrification can be conducted in situ or ex situ.

Table 1. *Summary of the most common used decontamination technologies*

Remediation technology	Application	Pros and Cons	Effectiveness	Cost (€/ton)
Soil vapor extraction	In situ soil and sediment	Causes minimal environmental disturbances; low maintenance; short treatment time; applicable for unsaturated coarse grained soils; applicable for volatile organic contaminants.	Average removal of 75% of PCBs by mass	50-100
Electrokinetic	In situ soil, ex situ soil and sediment	A more environmentally sustainable method than those requiring combustion or those relying mainly on non-renewable resource; limited by the solubility of the contaminant; alkaline soils reduce the effectiveness of the process.	Removal of approx. 62% of weathered Aroclor from sediments within 88 days. A study saw an 83% PCB removal rate.	80-300
Bioaug-mentation	In situ soil, ex situ soil and sediment	Good potential for organic contaminant degradation; difficult to completely predict/ control; may take a long time to see effects; may require multiple treatments; performance can be limited by dissolution rates and can be a function of geochemical conditions and indigenous microbial population.	Removal of organic contaminants	60-70
Bioventing	In situ	Causes minimal environmental disturbances; applicable for unsaturated soils; the heterogeneity of soil may reduce the efficiency of remediation; as a rule, areas with a low permeability may be less flown through and the required supply with oxygen is there obstructed; high concentration of contaminations may cause a blocking of the soil pores.	Removal of organic contaminants	10-50
Phytoremediation	In situ soil, ex situ soil and sediment	Causes minimal environmental disturbances; improves the biological and ecological activity of soil; applicable for a large range of contaminants; potential for secondary wastes minimal; aesthetically pleasing; remediation usually requires more than one growing season; treatment limited to soil explored by roots (generally less than 1 m from surface); dependant on climate and hydrologic plants.	Variable Some studies found that PCBs decreased by 60% and heavy metals by 20% after 1-2 years of treatment	60-100
Chemical dehalogenation	Ex situ	Fast acting; able to treat a variety of contaminants; ineffective in fine grained soils; heavy machinery is required for mixing.		50-400
Soil washing	Ex situ	Ability to recover metals and clean a wide range of contaminants (inorganic and organic) from coarse soils/sediments; since soil washing does not destroy or immobilize the contaminants, the resulting soil must be disposed of carefully; effective for soil/ sediment that does not contain a large amount of silt and clay	Average removal rate of 92% on silt, 89% on clay	40-60

Land farming	Ex situ	Low demands on the technology and process control; improves the biological activity of soil; applicable for a large range of organic contaminants; potential for secondary wastes minimal; relatively short treatment time; high demand for area owing to the insignificant width of the layer; need to control soil conditions to optimize degradation rate; runoff collection facilities must be contrasted and monitored.	Variable Removal of mineral oil hydrocarbons; may not be effective for high contaminat concentrations of >50,000 ppm.	30-60
Biopiles	Ex situ	applicable for a large range of organic contaminants; potential for secondary wastes minimal; relatively short treatment time.	Removal of organic contaminants	60-70
Composting	Ex situ	Well controllable microbiological remediation technique; an increased process temperature effect brings about a faster degradation; low land requirement.	Removal of organic contaminants	60-70
Bioreactors	Ex situ	Suited for the treatment of fine-grained, cohesive soil or sludge (e.g. the highly loaded residual fraction of soil washing); faster degradation by a higher level of soil homogenization; as the processes are controllable adding of degradation strains is possible; high expenditure of energy for the movement of the soil and the separation of the process water and drying of soil possibly required upon termination of the treatment.	Frequently treatment of soil with significant PAH loads	60-100
Thermal treatments (e.g. incineration)	Ex situ soil, sediments, and liquids	High removal efficiency; adverse human health and environmental impacts from off gassing; limited by the concentrations and types of metals present; effective at very high heat; temperatures up to 1200º C are required to achieve 99.99% removal efficiencies.	Removal of organic contaminants; metals are not removed and end up in the flue gases or in the ashes.	250-1000

4. Conclusions

Contamination of the soil by organic pollutants and trace metals has been shown to be one of the major environmental problems that the governments and researchers must solve in the next decades. Several studies available in the literature warn about the negative effects of these substances on living organisms. In order to avoid that this problem become more serious, several remediation technologies have been elaborated and improved. Physical and chemical techniques are usually used, however bioremediation, as it is ecologically correct, has gained great prominence, both in the remediation of petroleum and heavy metals.

The physical and chemical properties of soils/sediments may significantly influence the decontamination treatment. Knowledge of the soil or matrix type and contaminant properties are important factors in the prediction of contaminant partitioning and therefore mobility.

Classification of the soil type allows some prediction of soil-contaminants interactions, which may be beneficial or detrimental to the performance of the remediation system.

The following conclusions can be made about the nature and interactions of contaminants and soils/sediments:

- the variability of soil composition and its ability to interact with contaminants is of fundamental importance;
- soils/sediments are complex systems containing primary and secondary mineral phases, water, soil, gas and organic phases and must be carefully characterized to select the best remediation strategy and optimize the recovery process;
- complex reactions between soil and contaminants include sorption, oxidation reduction reactions, precipitation, complexation, hydrolysis and biological degradation and these govern how contaminants are bound, or mobilized, within a soil;
- biotic degradation processes and the presence of biomass influence the properties and availability of both organic and trace metal contaminants.

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6. Study cases

6.1. Study case 1: Soil bioremediation system at field-scale

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Introduction

An increasingly industrialized global economy over the last century has led to dramatically elevated releases of anthropogenic chemicals into the environment. Prevalent contaminants include petroleum hydrocarbons (TPH), polycyclic aromatic hydrocarbons (PAHs), halogenated hydrocarbons, pesticides, solvents, metals, and salt. Several recent research activities have focused on the application of phytoremediation as a sustainable reclamation strategy for bringing soil polluted by organic and inorganic contaminant into productive use. Compared to existing physical and chemical methods of soil remediation, which often generate secondary waste, phytoremediation is, in fact, cost-effective and less disruptive for the environment. This technology can be employed in any geographical area that can support plant growth and it is more likely publically accepted due to it aesthetical aspect. An additional benefit of phytoremediation is the improvement of chemical soil quality through organic materials, nutrients and oxygen supply by plant and microbial metabolic processes. Plants also improve physical soil quality providing groundcover and stabilizing the soil with their roots. In recent years, studies have demonstrated the efficacy of organic matter application, in supporting phytoremediation; the organic matter addition increased microbial biodiversity and activity, nutrient availability, cation exchange capacity, porosity and water-holding capacity. All these characteristics enhance the soil health and provide a medium satisfactory for plant growth.

In the present investigation, a biological approach made up native plants and horse manure has been proposed at field-scale to bioremediate and functionally recover a soil historically contaminated by heavy metals and hydrocarbons. This approach has been proposed after the satisfactory results of a preliminary meso-scale experiment carried out on the same contaminated soil in pots exposed to the same climate as the polluted site under remediation.

Experimental layout

The soil historically contaminated by heavy metals and hydrocarbons (about 10.000 m^2), is located in an industrial area in San Giuliano Terme Municipality (Pisa, Italy).

On April 2008 the tree plantation was carried out following this scheme: *Populus nigra (var. italica)* and *Paulownia tomentosa* 2 x 2 m with interposed *Cytisus scoparius* 1 x 1 m. The horse manure was applied to the soil surface at the dose of 20 t ha^{-1} and incorporated into the soil by soft harrowing.

Figure 1. On April 2007, the polluted field was divided in five plots and in each plot the soil was removed until clay basement, the bulky wastes were removed manually and by sieving and, after mixing, the soil was replacing in the same plots.

Soil samples were periodically (every four months) collected from the experimental area at 0-30 and 30-60 cm depth. The results obtained after two years (T4) of the field experiment are reported in this work.

Chemical and biochemical processes involved in the remediation practice

After two years of field-scale experiment (T4), *Populus nigra* and *Paulownia tomentosa,* that showed a suffering state at the beginning of the process, were well adapted and in vegetative growth. Instead, *Cytisus scoparius* was in healthy from the start.

The bioremediation approach, consisting in the use of plants and organic matter, was able to promote the biological activity and maintain a good level of available nutrients for microorganisms and plants. These properties are suitable to describe the biological evolution, the metabolic capacity of soil and the establishment of a re-naturalized geo-chemical and ecological soil ecosystem.

The reduction heavy metals over time confirms the effectiveness of the technique, even if this remediation activity is more pronounced on the soil surface (0-30) (Figure 2). Obviously, greater root growth, will promote the soil biological activity and decontamination of the soil also at 30-60 cm.

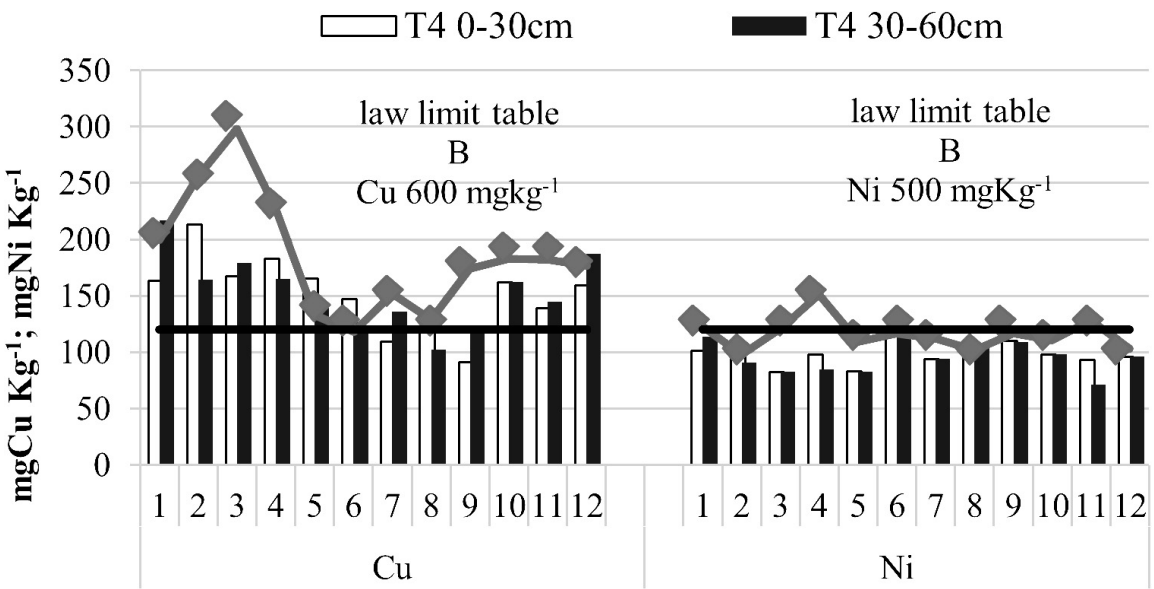

Figure 2. Trend of Cu and Ni in T0 (October 2007, 0-60 cm) and T4 (February 2010, 0-30 and 30-60 cm) soil samples compared to the law limit D.Lgs. 4/2008, Table A, urban use. Mean values (n=3; SD<10%).

The lower TPH values were generally observed at T4 compared with T0, even if, at 30-60 cm depth the decrease was less evident (Figure 3). This trend was probably due to a worse colonization of roots and microorganisms in deep soil, as suggested by the lower enzyme activities. In fact, numerous studies have confirmed that microbial degradation of TPH is enhanced in the rhizosphere zone.

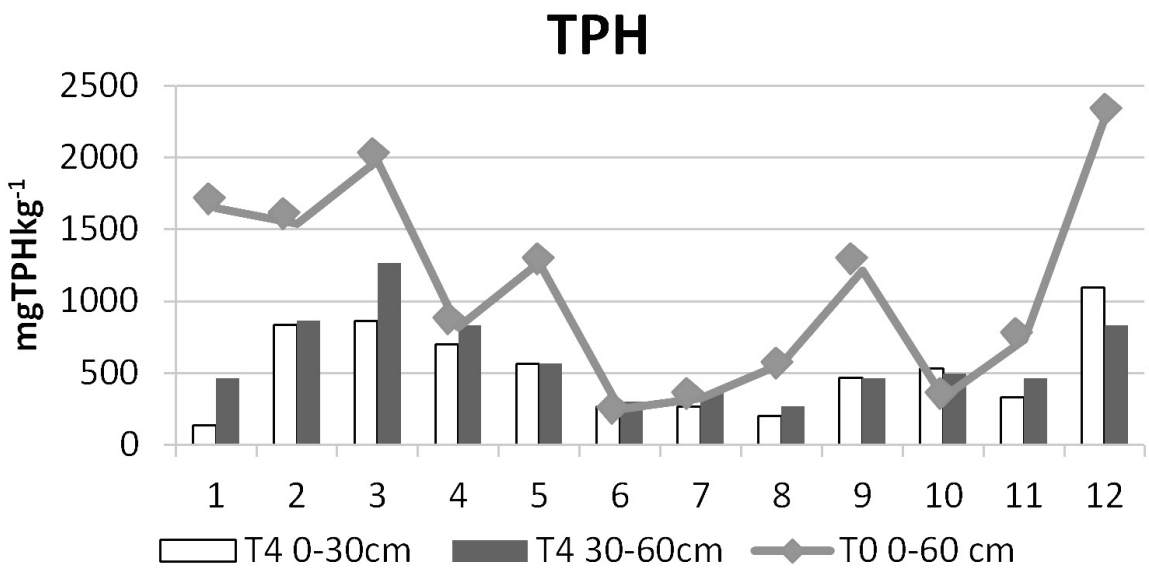

Figure 3. Trend of TPH in T0 (October 2007, 0-60 cm) and T4 (February 2010, 0-30 and 30-60 cm) soil samples.

Figure 4. T0 (image on the left) and T4 (image on the right) This biological system seems very promising to perform both decontamination and functional recovery of a polluted soil also at real-scale level.

6.2. Study case 2: Bioremediation of hydrocarbon polluted soil through ecological and chemical treatments

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Introduction

Bioremediation is a biological strategy for the recovery of polluted environment ensuring the conservation of biophysical property of ecosystems; it is based on the use of living organisms to reduce or eliminate environmental hazard resulting from accumulation of toxic chemicals and other hazardous wastes. This technology can involve the application of microorganisms with specific degradative abilities (bioaugmentation) e/o the stimulation of autochthon microorganisms (biostimulation).

The present investigation is about the effects of some treatments on the bioremediation of a polluted soil and the selection of specific parameters useful to study the evolution of biochemical processes which take place in the decontamination.

Experimental design

The soil from CSIC-CEBAS in Murcia (Spain) was polluted by oil refinery wastes.

The experiment was carried out, for three months, in laboratory microcosms under controlled temperature and humidity.

The bioremediation treatments were the following: 1) a liquid humic fraction (HF); 2) a commercial mixture of bacteria and enzymes (ME); 3) a surfactant (tween 80) with addition of mineral N-P nutrients (ND); aerated soil (A), and 5) control soil (without treatment and aeration, BN).

Chemical (total and soluble forms of C, N, P), biochemical (enzyme activities involved in nutrient cycles) and microbiological (CO_2 evolution) parameters were determined to study the soil metabolic processes involved in the degradation of hydrocarbons. Chemical and biochemical parameters were determined monthly while the metabolic activity (carbon dioxide evolution) every two days. Total residue hydrocarbons (TPH) and a test of seed germination and plant growth (IGC) were carried out at the end of the experiment to assess the efficiency of the bioremediation process.

Evolution of soil properties

All treatments showed a similar metabolic picture. In general, the results showed a decrease over time in total C suggesting a progressive degradation of organic substrates including the pollutants, without accumulation of by-products as assessed by the reduction of water soluble carbon (WSC) (Table 1).

	WSC				NO_3^-				NH_4^+			
	$T_{1\text{-}15gg}$	$T_{2\text{-}30gg}$	$T_{3\text{-}60gg}$	$T_{4\text{-}90gg}$	$T_{1\text{-}15gg}$	$T_{2\text{-}30gg}$	$T_{3\text{-}60gg}$	$T_{4\text{-}90gg}$	$T_{1\text{-}15gg}$	$T_{2\text{-}30gg}$	$T_{3\text{-}60gg}$	$T_{4\text{-}90gg}$
BN	850a	491a	326b	330c	255a	428a	488b	508b	4,97a	3,96b	4,29b	1,04a
A	663b	465b	417b	397c	188b	492a	532b	645c	6,26c	4,53c	4,59a	2,31b
ME	864b	392a	260a	300b	245b	501b	597a	694c	7,10c	3,26a	4,17a	2,52a
HF	930a	326a	425c	304b	224c	513a	601a	666a	6,07a	3,31c	5,49c	2,63b
ND	760b	469a	352a	414b	3048a	4334a	4411c	4468a	4,02a	5,18a	4,13a	2,57c

***Table 1**. Values of water soluble carbon (WSC), nitrate and ammonium during the experiment for the five different treatments (different letters indicate statistically different values, $p < 0.05$)*

The decrease of WSC was probably due to the concomitant presence of a high microbial respiration activity (high CO_2 production) (Figure 1) and an electron-acceptor NO_3 form whose formation overcame the consumption (Table 1). The activation of N cycle brought to the formation of hydrolysis products (i.e. NH_3) which rapidly were oxidised to NO_3 that increased steadily during three months incubation. This metabolic pathway was characterised by (i) decrease of hydrolytic enzymes (urease) (Figure 2) due to the depletion over time of nitrogenated substrates, (ii) activation of C cycle, as evidenced by the increase of β-glucosidase which produces easily metabolisable glucose from di-glucoside cellobiose, (iii) increase of metabolisable C respiration, as evidenced by the high rate of CO_2 formation which linearly increased during the incubation.

In spite of similar metabolic trend, the efficiency of the bioremediation process discriminated between the different treatments. In fact, total hydrocarbon degradation and index of plant germination and growth decreased in the order: ME>ND≅HF>A>BN (Figure 3).

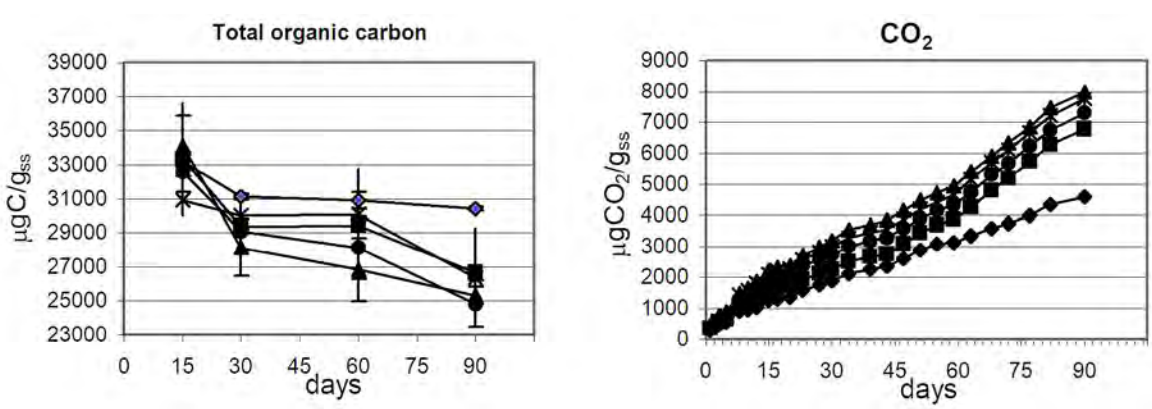

Figure 1. Evolution of total carbon and production of CO_2 during the experimental period. (Legend: ◆ BN; ■ A; ▲ ME; ✳ HF; ● ND).

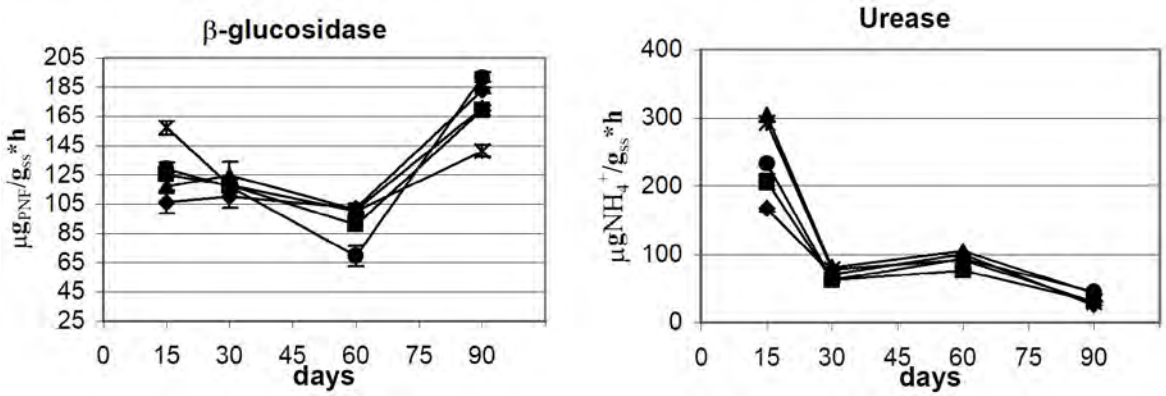

Figure 2. Changes in β-Glucosidase and Urease activities during the experimental period. (Legend: ◆ BN; ■ A; ▲ ME; ✳ HF; ● ND).

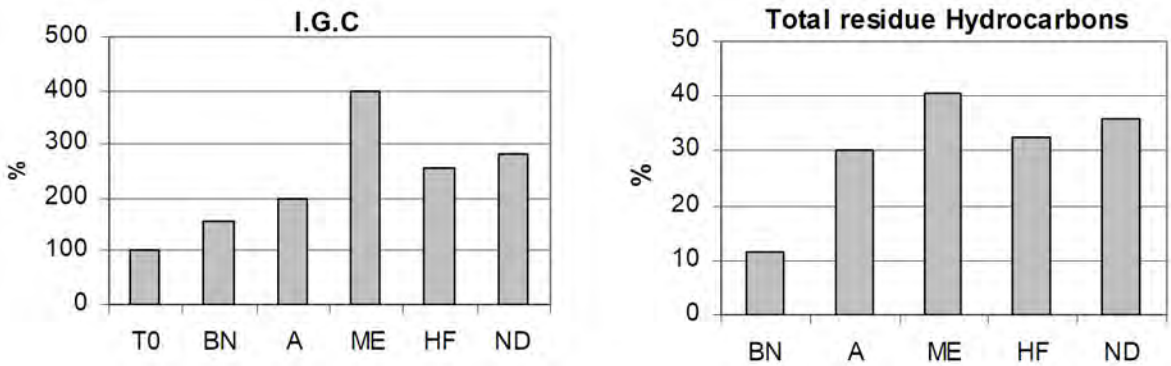

Figure 3. Growth index % of Lepidium sativum and abatement % of total residue hydrocarbons for the different treatments at the end of experiments.

In conclusion, added micro-organisms efficiently governed the metabolic process of organic compound degradation, but even the autochthonous micro-organisms, if stimulated by nutrients (in the presence of detergent) or by humic substances alone, were effective not only in sustaining the soil metabolism and recalcitrant substrate degradation, but also to preserve the soil ecological functionality. Finally, this study has clearly evidenced that the soluble humic substances behaved similarly to the ND treatments in that it should reflect both the stimulatory property of nutrients and the action of detergent. Thus, the process seemed feasible without employing costly and alien chemical substances.

6.3. Study case 3: Bioactivators as a potential strategy for dredged sediment recovery

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Adapted from

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Doni S., Macci C., Martinelli C., Iannelli R., Brugnoli P., Lampis S., Andreolli M., Vallini G., Masciandaro G. *Bioactivators as a potential strategy for dredged sediment recovery. Ecological Engineering 2018* (in press).

Introduction

A large amount of sediments is dredged every year from ports and waterways in order to maintain adequate depths for ship navigation, but the fate of these sediments is an issue worldwide recognized. Usually, dredged sediments are disposed of in specific facilities and may cause environmental problems due to their contamination. Metals and organic compounds, in fact, accumulate in sediments due to the limited hydrodynamic energy on the inside portions of harbors. The principle factor responsible for increased adsorption of contaminants is the fine fraction of sediments and the organic matter (Burton, 1991). Sand, which has a low specific surface area and a low surface charge density, is not very reactive and it has often a lower contamination than fine material. Sediment washing is a relatively simple and useful ex situ remediation technology, which is based on the separation and volume reduction processes. In view of this, sediment washing may be used to separate and concentrate the contamination into a smaller volume of fine sediment particles. This technology is usually used in combination with other technologies. The combination of sediment washing with natural techniques could represent, when suitable, the most convenient economic solution. Bioremediation technique is based on the capacity of microorganisms to degrade the organic

compounds and to reduce their toxicity or concentration. In the bioremediation approach, the natural process of organic compounds degradation is accelerated by creating optimal environmental conditions (such as temperature, pH and nutrients) for autochthonous microorganisms activity (biostimulation), or by introducing microorganisms with specific capacity of degradation (bioaugmentation).

In this work, the mechanical grain size separation of dredged sediments was carried out in order to reduce the volume of contaminated sediments to be treated and to make the material more homogeneous. After the sediment separation pre-treatment, the effectiveness of the bioremediation technology on the decontamination of the two sediment derivate granulometric fractions (sand and silt-clay), as well as of the raw sediment, was evaluated.

Experimental design

Sediment washing was conducted in a pilot separation installation. Fresh water-sediment slurry was sieved by a vibrating screen (4 mm size mesh) and conveyed in a hydrocyclone which allowed the recovery of the particles having a diameter lower than 200 µm. This fraction (sediment fraction <200 µm) containing the finer fraction (fine sand, silt and clay) was pumped in another hydrocyclone where a further mechanical separation was produced by centrifugal force. This hydrocyclone consists in a conical shell with a tangential inlet for feed (water-sediment mixture), an outlet at the top (overflow), and another outlet at the bottom (underflow). The overflow is enriched in water and fine fraction (pelite < 63 µm), whereas the underflow concentrates the remaining sandy fraction (>63 µm<200 µm). The <63 mm fraction was allowed to settle for 24 h; after which, the supernatant was removed.

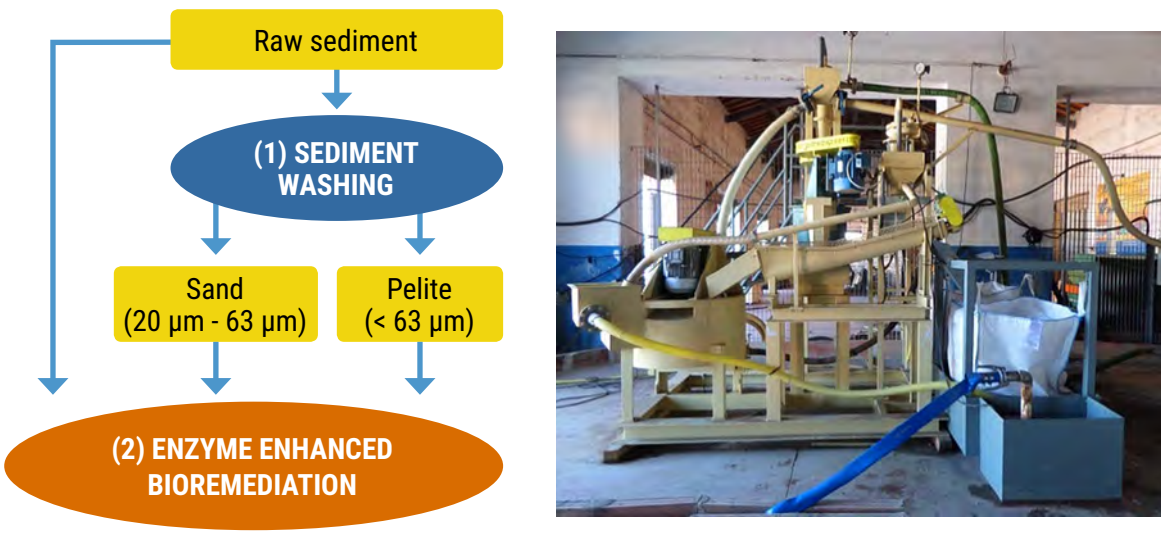

Figure 1. Pilot separation installation.

The two resulting solid fractions; sand (>63 µm<200 µm) and pelite (<63 mm), as well as the water effluent and raw sediment were analyzed (Table 1).

	Unit	Sediment	Sand	Pelite
Cu tot	mgCu Kg^{-1}	123[a]	55[c]	93[b]
Zn tot	mgZn Kg^{-1}	240[a]	104[b]	207[a]
Cd tot	mgCd Kg^{-1}	n.d.	n.d.	n.d.
Cr tot	mgCr Kg^{-1}	64,0[a]	21,1[b]	62,6[a]
Ni tot	mgNi Kg^{-1}	49,1[a]	16,3[b]	44,1[a]
Pb tot	mgPb Kg^{-1}	59,7[a]	22,3[b]	47,5[a]
Total petroleum hydrocarbon (TPH)	mgTPH Kg^{-1}	5447[a]	3973[b]	5648[a]

***Table 1**. Characterization of the three matrices after sediment washing.*

For the bioremediation, 20 kg of polluted sediments (raw sediments, sand fraction and silt-clay fraction) were placed in plastic containers (mesocosms). All containers were maintained under controlled temperature and humidity for three months. For each of the three matrices, the bioremediation treatments carried out in triplicate were the following: (1) a mixture of microorganisms-enzymes-nutrients (bioactivator treatment); (2) sediment without treatment (control sediment). In the bioactivator treatment, 100 g of a commercial product containing a mixture of microorganisms, enzymes and nutrients was added.

Chemical, biochemical and biological parameters were determined immediately after bioactivator application (t_0) and after three months (t_{90}) from the beginning of the experimentation.

Evolution of sediment properties

The sediment separation was crucial to concentrate the organic and inorganic contamination into a smaller volume of fine sediment particles (pelite). Treated and untreated sand fraction showed, in fact, significant lower values of heavy metals (HM) and organic contaminants (TPH) with respect to the silt-clay and raw sediment samples (Table 2).

	Sediment		Treated Sediment		Sand		Treated Sand		Pelite		Treated Pelite	
	t_0	t_{90}	t_0	t_{90}	t_0	t_{90}	t_0	t_{90}	t_0	t_{90}	t_0	t_{90}
HM	17,4	16,7	19,3	19,3	7,40	7,90	7,70	7,80	16,5	15,7	17,2	16,6
TPH	5114	4365	5066	2732	3898	2999	3982	1783	5597	5523	5522	5298
Mic. resp.	103	43,1	4518	1594	49,2	27,3	7065	2840	242	111	14406	6170
β-glu.	14,6	223	116	741	1,64	12,4	34,5	120,4	19,1	91	75,5	1193

***Table 2**. Chemical and biochemical parameters at the beginning (t_0) and at the end (t_{90}) of the bioremediation treatments. HM, Total heavy metals (meqHM kg^{-1}); TPH, total petroleum hydrocarbon (mgTPH kg^{1}); Mic. resp., microbial respiration (mgCO_2 kg^{-1}); β-glu, β-glucosidase activity (mmol kg^{-1} h^{-1}).*

Microbial data obtained by means of total count method evidenced that addition with the biostimulating product enhanced the initial population of both fungi and bacteria when compared with untreated trials (Figure 1). Moreover, a general increase in biomass of both bacteria and fungi were observed in all the treated samples at the end (T_{90}) respect to the beginning (T_0) of the experimentation.

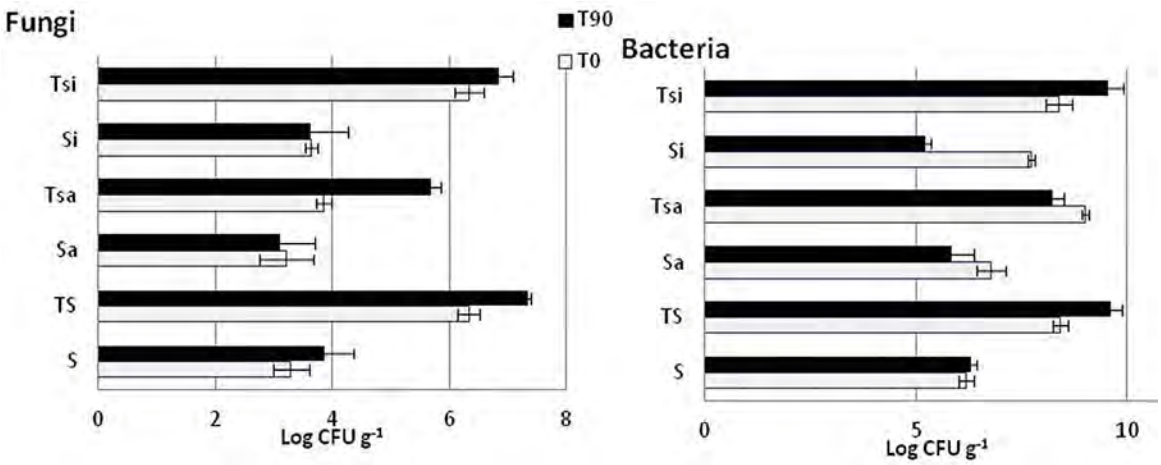

Figure 1. Microbial total count for Fungi and Bacteria obtained from Sediment (S), Sand (Sa), and Pelite (Si) samples, either treated (T) or not, at t_0 and t_{90}.

The hydrocarbon removal reached at t_{90} sampling time 46% and 55% in raw sediment and sand fraction treated with bioactivators, respectively, while natural attenuation (control sediment and sand) allowed only a 15 and 23% removal, respectively. The pelite fraction was not able to degrade significant amount of this contaminants. In conclusion, the results indicated that biactivator treatment significantly reduced the time required for the remediation of raw sediment and sand fraction, most likely because of the enhancement of microbial degradation of organic contaminants through the introduction of microorganisms and the improvement of nutrient balance.

6.4. Study case 4: Decontaminated river sediments for environmental applications

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Introduction

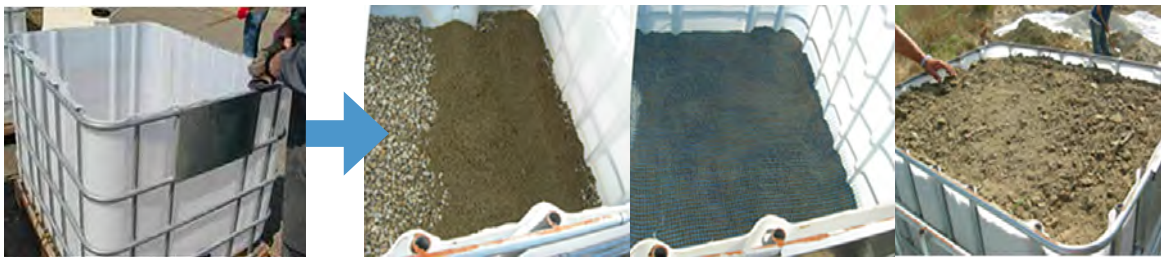
Sediment dredging is essential for maintenance and development of waterways and harbors and it is also necessary for navigation, remediation and flood protection. The continuous dredging of waterways and harbors creates large volumes of dredged material whose fate is an issue worldwide recognized. On the basis of an European Sediment Network estimation, it can be assumed that around 100-200 million m^3 of contaminated sediments are yearly dredged in Europe and need to be disposed of in specific and expensive ways. The European policy encourages treatment and valorization of dredged sediments, and this will be a technological challenge in the near future. Instead of focusing on how to deal with the dredged material as a waste problem we can search for solutions whereby sediment is an opportunity. Especially when sediments can be a low-cost resource to solve other environmental problems like the loss of soil due to plant nursing activities and road construction. Every year about five million m^3 of soil are removed from the ground due to plant nursing activities. To prevent the risk of lowering of the ground level of about 8-10 mm $year^{-1}$, plant nurseries are forced to buy soils from third-party catchments, which are often of poor quality and contribute to soil exploitation elsewhere. Similar problems are met by the road building industry, whose yearly demand for sand, gravel and aggregates for stability and draining purposes is around 30 million m^3, for an average value of € 450 million.

The CLEANSED European project (LIFE12 ENV/IT/000652) is aimed at evaluating the effectiveness of an innovative approach for the sustainable management of polluted dredged sediments in order to turn them into a matrix to be reused in the agriculture and environment.

Experimental design

Polluted sediments were dredged from the Navicelli Canal (Pisa, Italy) and transformed from a contaminated waste into a valuable material via a specific decontamination treatment, and then put to productive use in two different sectors: A) plant nursing and B) road construction.

A) For plant nursing, dredged brackish sediments were decontaminated by using the AGRIPORT technology, which is a phytoremediation technology developed in a previous European project (ECO/08/239065/S12.532262). The sediment phytoremediation was carried out at pilot scale in containers of about 1 m^3.



gravel-sand drainage plastic network

The sediments were pre-conditioned by adding an agronomic soil (30% v/v) to improve their clayey granulometric composition, and by topping the mixture with high quality compost (4 kg m^{-2}) to favour the initial adaptation of the selected vegetal species. The following plant treatments, in duplicate were tested for about two years: 1. *Nerium oleander* + *Paspalum v.*(O) 2. *Tamarix gallica* + *Paspalum v.* (T) 3. *Spartium junceum* + *Paspalum v.* (S) 4. *Phragmites australis* (Ph) 5. *Paspalum v.* (P) 6. Control (no plants) (C).

After a physical-chemical, biochemical and toxicological preliminary characterization, the AGRIPORT treated sediments were submitted to two-months landfarming technology as a post-treatment phase, in order to obtain a suitable substrate for nursing plant growth. This post-treatment was a necessary step, given the high quality of plants and flowers requested in the plant nursery sector. Three ornamental species, selected on the basis of their water needs, were the following: (1) *Photinia x fraseri* var. Red Robin, fast growth species and water exigent; (2) *Viburnum tinus* L., slow growth and xerophytic and (3) *Eleagnus macrophylla* L., in between the two species for water need and growing at medium rate. The experimentation was carried out in duplicate in containers of about 5 m^3 filled with the AGRIPORT decontaminated sediments mixed with an agronomic soil extracted from an actual plant nursing area. Three different soil-sediment mixtures were tested: CTL, (control) sediment 0% : soil 100%; T33, sediment 33% : soil 66%; T50, sediment 50% : soil 50%. Eight plants for each specie, were planted in each matrix in order to assess plant response during the two years growth (nursing plants are typically sold after two or three years growth); in the present paper, the results after six months from plantation are reported. The monitoring concerned: a) matrices physical, chemical, biological and hydraulic properties, b) plant physiological response and biomass, c) root ball cohesion and root development.

B) For road construction, fresh brackish sediments were sampled and characterized. Based on the water, organic matter and organic contamination content, the landfarming technology was selected as the most suitable treatment to have an idoneus substrate for road construction. The landfarming treatment, carried out for five months, consisted in the periodically (once-twice per week) aeration by mechanically moving the sediment and turn it over. After landfarming treatment, in order to further reduce the water content up to the level required for compaction in road construction, the sediments were mixed in laboratory with different lime percentages and the related water loss was evaluated. The optimum water content for road construction of about 20% was obtained by using a lime percentage greater than 15%.

Results

A) Plant nursing. The preliminary characterization of the AGRIPORT treated sediments showed that the phytoremediated sediments had properties quite idoneous for their reuse in plant nursery activity; however, the landfarming process was able to increase the microbial metabolism, further reducing the organic contamination, and homogenizing the substrate.

The soil-sediment matrices (T33 and T50), used as growing substrate for the cultivation of ornamental plants, showed higher content in organic carbon and in the hydrolytic enzyme activities with respect to the control soil (CTL), suggesting the establishment of better conditions for plant growth and development. This was confirmed by plant analysis; in fact, no significant differences among thesis (CTL, T33 and T50) were observed for the total biomass.

Concerning the biological indicators, the lower dehydrogenase activity in the initial sediment mixtures (T33 and T50) with respect to the agronomic soil, suggested a general lower metabolic activity in the sediment. After six months from plantation, not significant differences among matrices and plant species were observed. On the other hand, the two hydrolytic enzyme activities, β-glucosidase and phosphatase, linked to the C and P cycles respectively, were higher in the matrices with sediment (T33 and T50) with respect to the agronomic soil alone (CTL). These results were quite unexpected since in the initial characterization these enzyme activities were significantly lower in the sediment than in the soil. Probably, the association of soil and sediment creates better environmental conditions (better structure and aeration) for the microorganisms in carrying out their metabolic activity. The three selected plant species showed, during the first growing season, a great adaptation to both the soil-sediment mixtures (T33 and T50) with a biomass production, a height and a leaf area development similar to the plant species grown in the agronomic soil (CTL). Among the selected species, *Viburnum tinus* is the only one that allowed a correct root ball compaction in both soil-sediment mixtures.

B) Road construction. The specific technical goals of the road construction were:

- preliminary assessment of the optimum percentages of use of the treated sediments for the different parts of a road (foundations, intermediate layer, etc);
- construction of a site for the demonstration of the use of sediment mixtures in the construction of an actual life-size road.

The first phase of the design of the experimental road was the definition of the most appropriate mixtures of decontaminated sediments and other standard materials, based on the mechanical features induced by the use of the sediments themselves in order to evaluate the suitable materials for the different layers of the road.

The sediment organic matter content decreased greatly during the landfarming process (averagely 15%) due to the activation of the sediment microbial biomass (increase in enzyme activities), responsible also of organic contaminants degradation (near to zero at the end of landfarming). This reduction permitted to reach organic matter values near to 2%, considered the optimum value for sediment reuse in road construction as reported in the "Technical Specifications for road embankments" document. Concerning the sediment water content, a great reduction was achieved at the end of the landfarming process, reaching a value of about 40%; however, to reach the optimal content of about 15-20%, the addition of lime (15%) was necessary.

The possibility of considering the sediments as a resource rather than a waste necessarily imply the updating of environmental policies, in line with recent guidelines the European Union and the support of researches aimed at identifying innovative technologies.

6.5. Study case 5: Trace metal(oid) stabilization by raw and thermally modified geo-materials as soil amendments

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Argyraki, A., Boutsi, Z., Zotiadis, V. Towards sustainable remediation of contaminated soil by using diasporic bauxite: Laboratory experiments on soil from the sulfide mining village of Stratoni, Greece. Journal of Geochemical Exploration, 2017, 183, 214-222.

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Introduction

The stabilization of inorganic - non degradable contaminants in soil has been suggested as a sustainable remediation method aiming in breaking the pathway between the source and the receptor in the widely used 'source- pathway-receptor' risk assessment approach. Various mineral-based amendments can be used in raw or modified form for inducing immobilization of inorganic contaminants in-situ, with different modes of molecular-scale sequestration within the soil matrix (e.g., O'Day & Vlassopoulos, 2010new applications with a variety of natural and reprocessed materials are emerging. By sequestering contaminants in or on solid phases and reducing their ability to partition into water or air, amendments can reduce the risk of exposure to humans or biota. A variety of mineral types are commonly used to amend contaminated soils, with different modes of molecular-scale sequestration. Regulatory, social, and economic factors also influence decisions to employ mineral amendments as a treatment technology. In the present study, seven different mineral-based amendments in raw or thermally modified form have been tested for inducing immobilization of inorganic contaminants in soil.

Materials and Methods

The used geo-materials include different types of palygorskite-rich and saponite-rich clay as well as finely grained red diasporic bauxite. Thermal modification has been achieved by heating the geo-materials at different temperatures ranging from 105 to 450 °C. All materials have been characterized in terms of mineralogical and chemical composition. A pot experiment was set up, where the amendments were mixed at different proportions with contaminated soil from the sulfide ore mining village of Stratoni, north Greece. A detailed discussion on the soil contamination and potentially health concerns in this particular area is presented elsewhere (Argyraki, 2014). Briefly, mining operations and ore processing to produce sphalerite and galena concentrates at the Stratoni flotation plant over the last 60 years have resulted to surface soil contamination with Pb, Zn, Cd and As. A composite soil sample of 30 kg was collected from the 0-20 cm depth by mixing 5 sub-samples collected over an area of 2500 m^2, air dried and sieved to <2 mm. The soil was thoroughly homogenized by manual mixing and was used to fill 1-kg polypropylene pots. Total concentrations in untreated soil have averages of 1000 mg/kg Pb, 712 mg/kg Zn, 6 mg/kg Cd, 2900 mg/kg Mn and 296 mg/kg As. The effectiveness of soil amendments has been evaluated by comparing water leachable contaminant concentrations of treated and untreated soil after a four week period of repeated cycles of wetting and mixing.

Results and discussion

The contaminated soil sample had a sandy-silty texture with 64% sand, 34% silt and 2% clay by mass on average and relatively acidic pH of 4.8. Identified minerals with XRD analysis in soil comprise mainly (95 %) common soil inorganic phases, such as quartz, feldspar, mica and clay minerals. The absence of calcite is noted, reflecting the predominance of silicate rocks in the area. The most abundant trace element bearing phase in soil found by SEM-EDS was Fe and Mn oxides and oxyhydroxides (Figure 1a-b). Sulfide grains corresponding to the primary mineralization of the wider area were also identified including galena, sphalerite, arsenopyrite and pyrite (Figure 2).

A 95% reduction has been observed for water leachable Pb by using a mixed palygorskite-saponite clay. However, the maximum efficiency was observed by using treated bauxite at the temperature of 450 °C with negligible water leachable concentrations of Pb, Zn, Cd and As after treatment (Argyraki et al., 2017). Specific surface area (SSA) of the clay materials is the crucial parameter affecting surface charge and thus the capability of the studied elements to form inner or outer sphere complexes. Mineralogical changes in bauxite during heat treatment within the range of 350-450 °C, associated to dehydroxilation of both Fe and Al-rich phases, lead to structural changes of the minerals resulting in higher SSA and greater efficiency for contaminant retention.

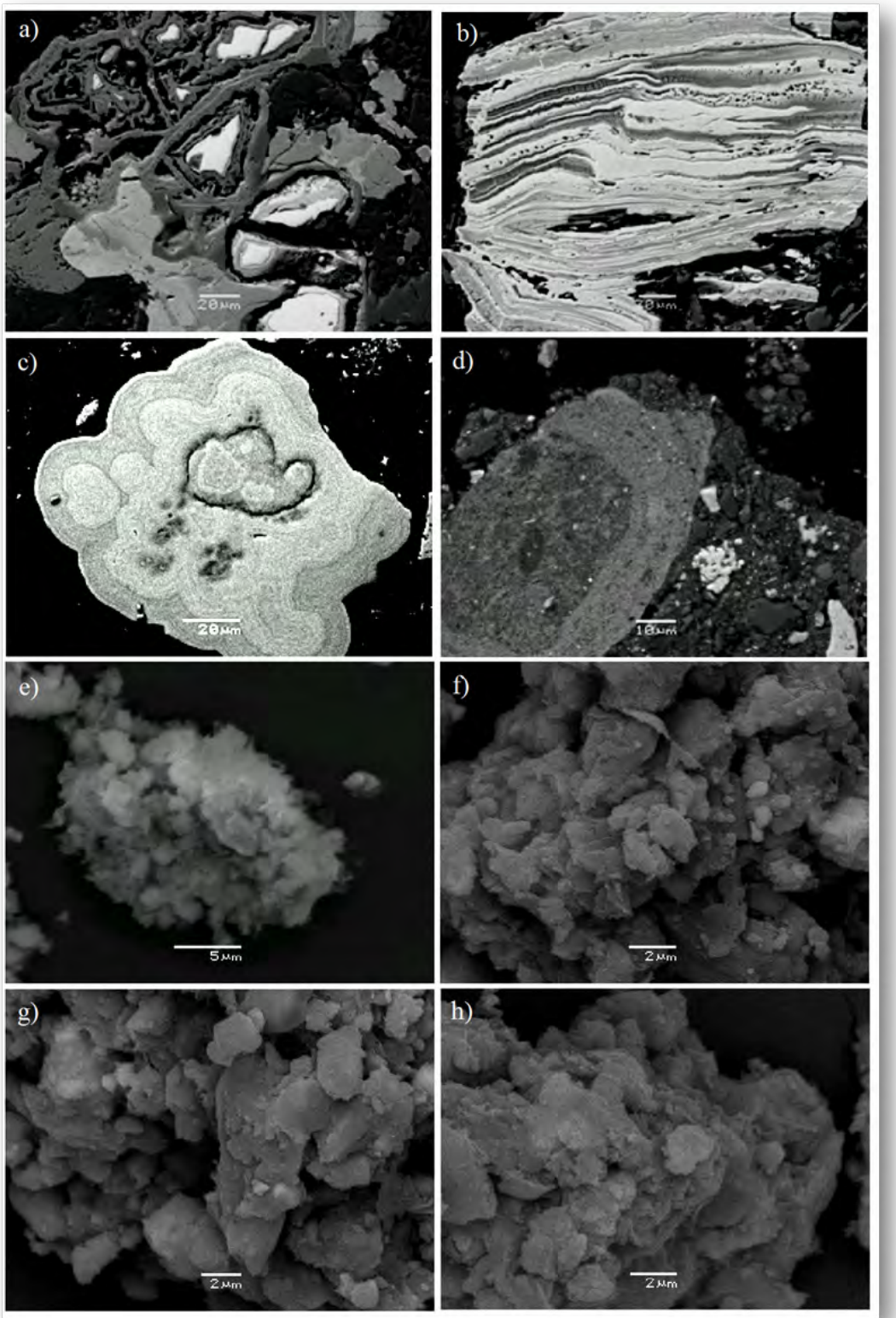

Figure 1. SEM micrographs of contaminated soil and geo-material amendments. Contaminated soil (a-b), showing bright metal(oid) enriched grains in BSE mode. Raw bauxite grains (c-d), showing zoned goethite in BSE mode. Raw and heat activated bauxite (e-h), showing changes in the micromorphology of grains. Raw palygorskite rich clay (i) free surface micromorphology. Raw saponite rich clay (j) free surface micromorphology. Raw palygorskite-saponite rich clay (k) free surface micromorphology.

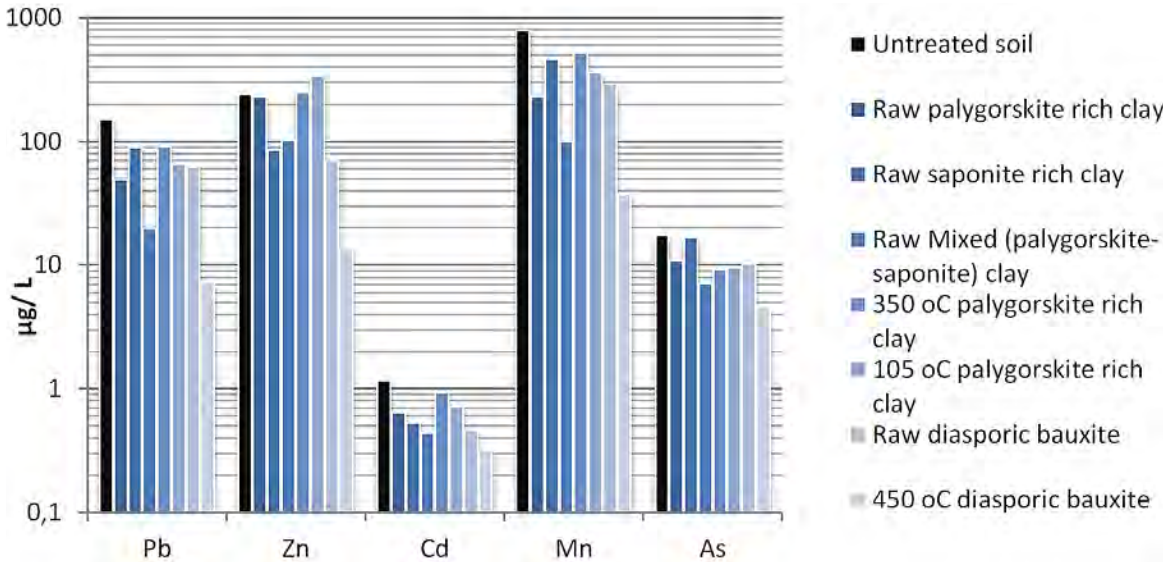

Figure 2. Comparison of water leachable concentrations of trace elements in untreated and treated soil after addition of 7% geo-material amendments.

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6.6. Study case 6: Polluted soils and sediments resulting from mining activities: a case study in the Anllóns River

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In the Anllóns River (Galicia, NW Spain), high arsenic concentrations are found in soils and sediments, which are attributed to natural geogenic arsenic enrichment exacerbated by mining activities (Devesa-Rey et al., 2008). Arsenopyrite mineralization in hydrothermal quartz veins is associated to gold ores which were exploited during the Roman Empire and then from 1895 until 1910, with intermittent withdrawals after that period. Arsenic concentrations in the rocks of the area are usually around 1% but, in mineralized zones with semi-massive arsenopyrite, they can reach up to 10 % while soil arsenic contents as high as 4,000 mg kg^{-1} are detected (Boixet., 2007).

Arsenic contamination in the Anllóns river has been studied using the methods for pollution assessment shown in the Figure 1. These include the determination of the total concentration of the contaminants, their chemical species in the solid phase, their mobility and bioavailability, and their potential toxicity, as well as the study of chemical species in solution. Martiñá-Prieto et al (2018) analysed by XRF and chemical extraction fifty soil samples of C-horizons covering an area of 50 km^2 along the river course, and found that pseudo-total arsenic ranging between 2 and 489 mg kg^{-1}, up to 8 times higher than the regional generic reference level. Arsenic has low solubility in the standard leaching tests DIN 38414-S4 and TCLP (up to a maximum of 0.25% of pseudo-total As), which is related to the predominance of low solubility As fractions (Lombi et al, 2000), mainly associated with crystalline Fe oxides. Based on these leaching tests the risk of As transfer to groundwater and superficial waters may be considered low. Nevertheless, changes in environmental conditions such as solid:liquid ratios, pH conditions, presence of phosphate and longer contact time bring about an increase in As mobilization which can reach up to 5% of pseudo-total As (Martiñá-Prieto et al, 2018). Consequently, these factors must be taken into account when assessing the transfer risk of lithogenic arsenic towards aquatic ecosystems and living beings.

In the riverbed sediments high concentrations that can reach 264 mg As kg^{-1} were detected downstream the mineralized area to the river mouth (Devesa-Rey et al., 2008; Rubinos et al., 2010). Costas et al. (2011) found even higher values (up to 308 mg kg^{-1}) at the estuary and estimated that the Anllóns River exports to its estuary 460 kg y^{-1} of dissolved arsenic annually. Most arsenic in the sediments of the Anllóns River is bound to Fe-oxides forms and in the residual fraction (Devesa-Rey et al., 2008; Rubinos et al., 2011), which are low-mobility phases. This low solubility was confirmed by the results of availability tests, addressed to estimate the leaching potential of arsenic and its effect on the survival of microorganisms (TCLP

extraction), the bioavailability to higher plants (a weak acid solution of 1 M HCl) and to superior animals (PBET extraction that simulates the conditions of the stomach of mammals). This latter extractant solubilized the highest arsenic concentrations, not exceeding 11% of the total-As (Devesa-Rey et al., 2008). Nevertheless, as in the soils, this apparent low arsenic mobility in the contaminated sediments varies with changes in the environmental conditions. It increases with the water:sediment ratio, and therefore may be higher during high-flow resuspension events (Rubinos et al., 2010) and also with increased ionic strength, as it occurs in estuarine environments (Rubinos et al., 2011). Mobility is also strongly dependent on the pH; arsenic solubilisation occurs simultaneously with that of the oxides and hydroxides of Fe and Al at acidic pH, and organic matter at alkaline pH, suggesting that arsenic is associated to these components (Rubinos et al., 2011). Mobility is also promoted by the presence of phosphate, having high concentrations in some sections of the river (Iglesias et al., 2011; Barral et al., 2012), coming from wastewaters discharged into the river course and fertilizers eroded or leached from the soils of the basin. Interestingly, although phosphate favours arsenic release from the sediments, it was shown in Microtox® bioassays that it counteracts the acute toxicity of As^{V}, although it had no effect on the toxicity of As^{III} and DMA^{V} (Rubinos et al., 2014).

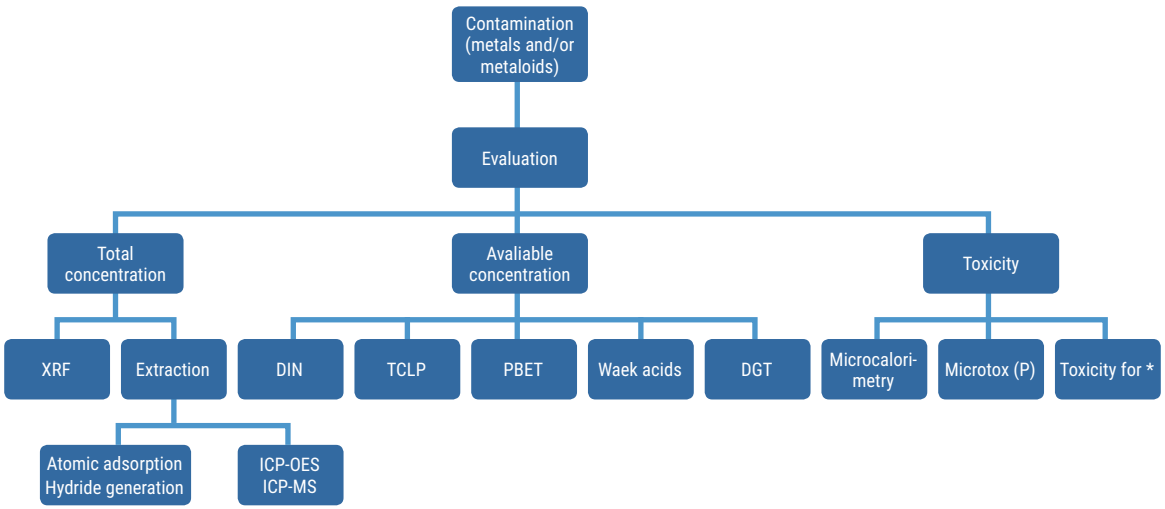

Figure 1. General scheme of procedures for the assessment of pollution in soil and sediments. DIN refers to DIN 38414-S4.

In addition to the effect of these abiotic factors, arsenic biogeochemistry and its potential toxicity are also affected by the biological status of the riverine ecosystem. The surface of the sediments of the Anllóns River is covered by epipsammic biofilms mainly constituted by algae and bacteria immersed in an exopolysaccharide matrix (Devesa-Rey et al., 2009, Prieto et al., 2016a, b). In various experiments, the mutual interaction between arsenic and benthic biofilms in this mining impacted river has been proved. The presence of biofilms increased arsenic retention by the sediments (Prieto et al, 2013). More arsenic is accumulated by biofilms growing in sites with higher As concentration in comparison with unpolluted sediments; the As-contaminated sites showed reduced biofilm growth (Barral-Fraga, *pers.com.*). Arsenic toxicity to the autotrophic component of the biofilm was confirmed by microcalorimetry (Prieto, *pers.com.*).The presence of biofilm affected the arsenic transfer between the sediment

and the aqueous phase. Thus, it reduced the release of arsenic from the sediment to the water column and avoided the reduction of As^{V} to the usually less toxic As^{III} (Prieto et al., 2016c). In turn, the presence of biofilm increased the retention of dissolved arsenic by the sediment (Prieto et al., 2013; Prieto et al., 2016d); moreover, as revealed by ICP-MS determination, the biofilm avoided the reduction of As^{V} to As^{III} in the water column and promoted the occurrence of the less toxic organic species MMA^{V} and DMA^{V} by biomethylation (Prieto el al., 2016b).

Overall, these studies have put forward the complexity of arsenic pollution in the Anllóns Basin and the need for an approach through diverse methodologies directed to study the biotic and abiotic factors involved in its biogeochemical behaviour.

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6.7. Study case 7: Potential contributions of free-living bacteria for cleaning up chronically petroleum contaminated soils

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Morales-Guzmán G, R. Ferrera-Cerrato, M.C. Rivera Cruz, L.G. Torres-Bustillos, R.I. Arteaga-Garibay, M.R. Mendoza-López, and R. Esquivel-Cote, A Alarcón. 2017. Diesel degradation by emulsifying bacteria isolated from soils polluted with weathered petroleum hydrocarbons. Applied Soil Ecology. 121: 127-134.

Introduction

In Mexico, the greatest oil industry activities take place in the states of Veracruz, Tabasco, Campeche and Chiapas, which are vulnerable to soil contamination (SEMARNAT, 2010). Between 2008 and 2013, 627 sites were subjected by environmental emergencies, and 20.3% of them were affected by the oil industry activities. These environmental emergencies are usually treated by washing soils, inducing chemical oxidation, and performing physical separations (Volke and Velasco, 2002). Bioremediation facilitates technological development at low costs by using plants, fungi, or bacteria to neutralize toxic substances, transforming them into substances less harmful to the environment (Rivera-Cruz et al. 2004). Bacterial genera like *Pseudomonas*, *Azospirillum*, *Paenibacillus*, *Bacillus*, among others, as well as those symbiotic bacteria such as rhizobia-like bacteria, are able to use some fractions of petroleum hydrocarbons as energy or carbon sources, and consequently contribute on their partial or complete degradation (López-Ortiz *et al.,* 2012; González-Paredes *et al.,* 2013).

Tabasco State is characterized by intensive oil industrial activities, but resulting in chronic oils spills that affects natural ecosystems, agricultural lands, and grasslands. Therefore, few ecological studies were focused to know the impact of chronic contamination by oil spills in soils (Gleysols and Histosols) based on performing and characterizing soil profiles from which either free-living bacteria or filamentous fungi were described and isolated. Some free-living bacteria were selected on the basis of tolerating and furthermore, degrading fractions of petroleum hydrocarbons such as polycyclic aromatic hydrocarbons (Rivera-Cruz *et al.,* 2002a, 2002b, 2002c, 2006); furthermore, these bacteria possesses other physiological activities by which they may act as like N-fixers, P-solubilizers, or lipolytics.

Experimental design

Free-living bacteria were described and isolated from chronically petroleum contaminated soils in Tabasco (Mexico): Two soils (Gleysol) contaminated with weathered petroleum (A) moderately contaminated (MC), and (B) highly contaminated (HC). Historically, these soils have been exposed for more than 50 years to chronic contamination due to accidental crude oil spills (Rodríguez-Rodríguez *et al*., 2016). In addition, a soil (Gleysol) with no prior contamination impact (with 150 mg kg^{-1} biogenic hydrocarbons) was used as control. The cultivable populations of total bacteria were quantified by performing the dilution and plate counting technique. Moreover, a selection of emulsifying bacterial strains was made by means of the diesel degradation test.

Results and discussion

A greater percentage of the total population of bacteria in the control soil was found (83.58%) in comparison to highly contaminated (58.51%) or moderately contaminated (23.67%) soils (Figure 1). The later allowed to selected free-living bacterial strains able to emulsify and degrade more than 90% of diesel; moreover, six bacteria were the most efficient strains (Table 1). The bacterial populations present in the contaminated soil samples can be attributed to inherent bacterial capabilities for tolerating and being adapted to high contamination levels (Morales-Guzmán *et al.,* 2017).

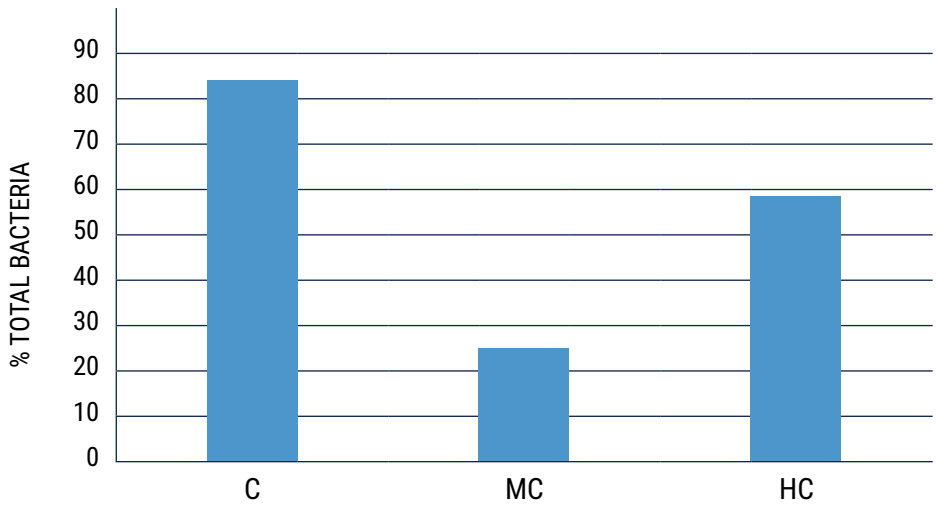

Figure 1. Percentage of the total free-living bacteria quantified from chronically contaminated soils from Tabasco State (Mexico). Abbreviations: C= Control, MC= moderately contaminated, HC= highly contaminated.

The bacterial populations present in the soil samples can be attributed to the inherent bacterial capability for resistance and adaptability to high contamination levels (Morales-Guzmán *et al, 2017*).

***Table 1**. Diesel degradation percentages by free-living bacteria isolated from chronically contaminated soils from Tabasco State (Mexico).*

Bacteria		Percentage
Serratia marcescens	C7S3A	96.2
Citrobacter freundi	CCC4D53	17
Serratia marcescens	C11S1	17
Stenotrophomonas pavanii	CCC10S1	≤ 10
Stenotrophomonas maltophilia	C5S3FN	≤ 10
Raoultella ornithinolytica	C5S3	≤ 10

(Adapted from Morales-Guzmán et al., 2017).

The described studies and results indicate that although soils from Tabasco State are continuously exposed to oil spills, they may show certain resilience against such contamination, and this may be explained in part due to the presence of beneficial microbial groups whose physiological and biochemical activities allow them to persist under such stressful conditions. Moreover, these microorganisms and their associated plants may eventually contribute on the natural degradation of certain labile fractions of petroleum hydrocarbons which are utilized as energy and carbon sources for microbial metabolism.

Furthermore, these free-living microorganisms may be propagated and inoculated in specific plants species (for instance, *Leersia hexandra*, *Echinochloa polystachya*, *Clitoria ternatea*, *Leucaena leucocephala*, among others) that also have been characterized to grow at contaminated areas, and to contribute on the degradation of petroleum hydrocarbons from either artificially or chronically contaminated soils. Then, the biotechnological utilization of both, microorganisms and plants, may potentially enhance bio-/phytoremediation strategies directed to recover and clean up extensive land areas subjected to contamination by petroleum hydrocarbons.

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6.8. Study case 8: The flux-meter: implementation of a portable integrated instrumentation for the measurement of CO_2 and CH_4 diffuse flux from landfill soil cover

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Introduction

The municipal solid waste landfills represent significant sources of atmospheric contamination due to uncontrolled emissions of landfill gas (LFG) from the landfill cover, which are present even when there is a system for the capture and combustion of LFG. LFG is mainly composed by methane and carbon dioxide — the two main gases responsible for the greenhouse effect — with a percentage of approximately 55% and 45% respectively. In particular landfill CH_4 accounts for about 22% (USEPA, 2001) to the total methane dispersed into the atmosphere. Moreover, the presence of other gaseous species in the LFG, such as volatile organic compounds (VOC) is an additional problem cause of the impact on human health. The Italian and European legislation provides that landfill emissions have to be evaluated to assess both the contribution to the greenhouse effect and the direct effects on the population. Consequently appropriate methodologies should be taken to limit them when necessary.

There are many techniques available to quantify the flux released from landfill, i.e. gas survey, eddy correlation and flow chambers, but the technology here presented, known as "accumulation chamber method", is a static not-stationary method based on the determination of concentration gradient of CO_2 and CH_4 measured inside the accumulation chamber laid down on the soil.

The collaborations between West Systems, IGG-CNR and Perugia University have allowed implementing a portable and simpler instrument (Virgili, 2008). This instrument is able to obtain flux measurements directly on the field in a very short time (each measurement takes about 2-3 minute). The main difference from the other methodologies is that soil flux measures carried out with the accumulation chamber are independent of flux regime (advective or diffusive) and of soil characteristics (porosity, diffusive coefficient, etc) (Tonani and Miele, 1991). This characteristic with the advantages of manageability, simple use and fast flux measurements of the instrument has allowed extending its application also to evaluate diffusive emissions from the cover of municipal solid waste landfills (Cossu et al., 1997; Cioni et al., 2002; 2003; Capaccioni et al., 2005; Raco et al., 2010).

Accumulation chamber - transient method

To understand the relationship between the flux and the initial slope of the gas concentration – time line, the transient method should be considered. The transient or static not-stationary method of accumulation chamber consists in measuring – during the time (t) – the gas concentration (C) under investigation within the chamber only opened on the side laid on the ground. When a diffusive flux from the soil occurs, a gradient concentration (dC/dt, expressed as ppm/sec) is measured within the chamber. Then the flux of a specific gas from the soil surface can be estimated as follows (Natale et al., 2000):

$$Flux = \frac{P \cdot V}{R \cdot T} \cdot \frac{dC}{dt} \cdot \frac{1}{A}$$

where P and T are the environment pressure and temperature respectively, R (with a value of 8.314472 m^3 Pa K^{-1} $moles^{-1}$) is the gas constant, V is the volume and A is the area of the chamber.

From the equation [E1] it follows that the flux (expressed as moles m^{-2} day^{-1}) is proportional to the gradient concentration (expressed as ppm sec^{-1}) measured (Chiodini et al., 1998), while the proportionality constant is a function of the height of the accumulation chamber (H=V/A if the chamber has a cylindrical or parallelogram shape) and of the air pressure (P_{air}) and air temperature (T_{air}). Since the shape of the accumulation chamber is fixed and P_{air} and T_{air} are measurable, the flux of the specific gas is calculated directly by the gradient dC/dt.

However, during the measure, the gas concentration (C) rises within the chamber and, consequently, the gradient diffusion of the specific gas inside the chamber decreases over time leading to a possible underestimation of the flux (Figure 1). Therefore it is important to measure the gradient in few minutes after the chamber is laid down to the soil, as demonstrated also by Chiodini et al. (1998).

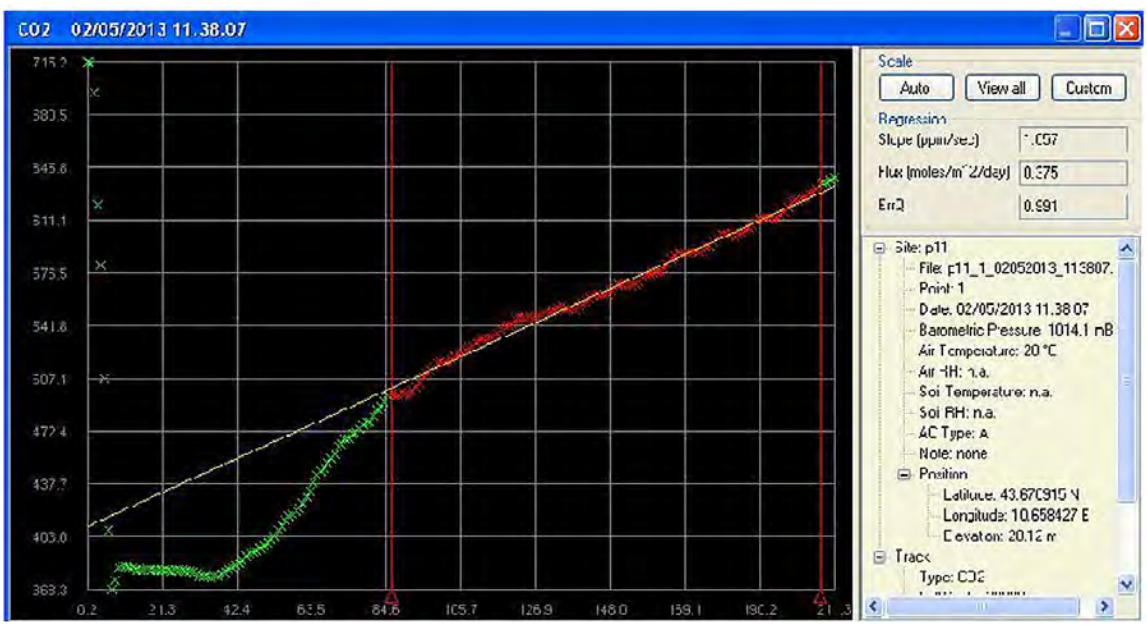

Figure 1. Graphical representation of a typical field measurement.

A pneumatic system connects the gas sampled in the accumulation chamber and the gas analyser located in the backpack. Therefore this system implies a delay between the concentration in the chamber and value measured by the analyser and could produce an artefact in the first seconds of the measurement, as showed in Figure 1. In this case the determination of the gradient dC/dt is always performed in the range highlighted in red waiting about 80 seconds.

The most important advantage of FLUX-meter is given by the integration of high performance detectors, able to measure very low CH_4, CO_2, H_2S and VOC concentrations, in a portable instrumentation characterized by manageability, simple use and fast flux measurements. All these characteristics respond to the needs of scientists and engineers engaged in the determination of gas exchange at the soil-air interface linked to the gas escape from the landfill cover, from agricultural soil, from volcanic area and from geothermal exploration sites.

Example of data processing

The integrated instrumentation here presented has been used on a municipal solid waste (MSW) landfill covering a surface area of about 140,000 m^2 by means of 226 flux measurements.

In order to quantify the total flux released from the landfill cover it is necessary to carry out a series of measurement in specific points inside the whole investigated area located in a grid as regular as possible. The size of the sampling grid is a critical point, due to the great spatial variability of gas flux values; generally a mesh size of 20x20 meters generates a reliable estimation when at least 100 measurements have been obtained.

In order to quantify a specific gas released from the investigated area and to recognize the presence of zones characterized by anomalous fluxes, statistical and geostatistical approach are adopted.

The estimation of the total amount of biogas discharged into the atmosphere has been carried out by the methodology based on partitioning the flux data by means of cumulative probability plots, formalized by Sinclair (Sinclair, 1970 and 1991). For each identified population the arithmetic mean of raw data and its 95% confidence interval has been calculated using the Sichel estimator (Sichel, 1966). An implementation of the Sichel method has been developed by West Systems. The software, which runs on personal computer, calculates the cumulative probability plot, using the data recorded by the PDA during the sampling activity and calculates the total output of LFG taking into account the extension of the investigated area (Raco *et al.*, 2010). By way of example, in Figure 2 the cumulative probability plots of CH_4 flux data is reported and each data subset is recognizable by different colours, while in Table 1 the main statistical characteristics of each population and the estimation of total amount of LFG and its 95% confidence interval are shown.

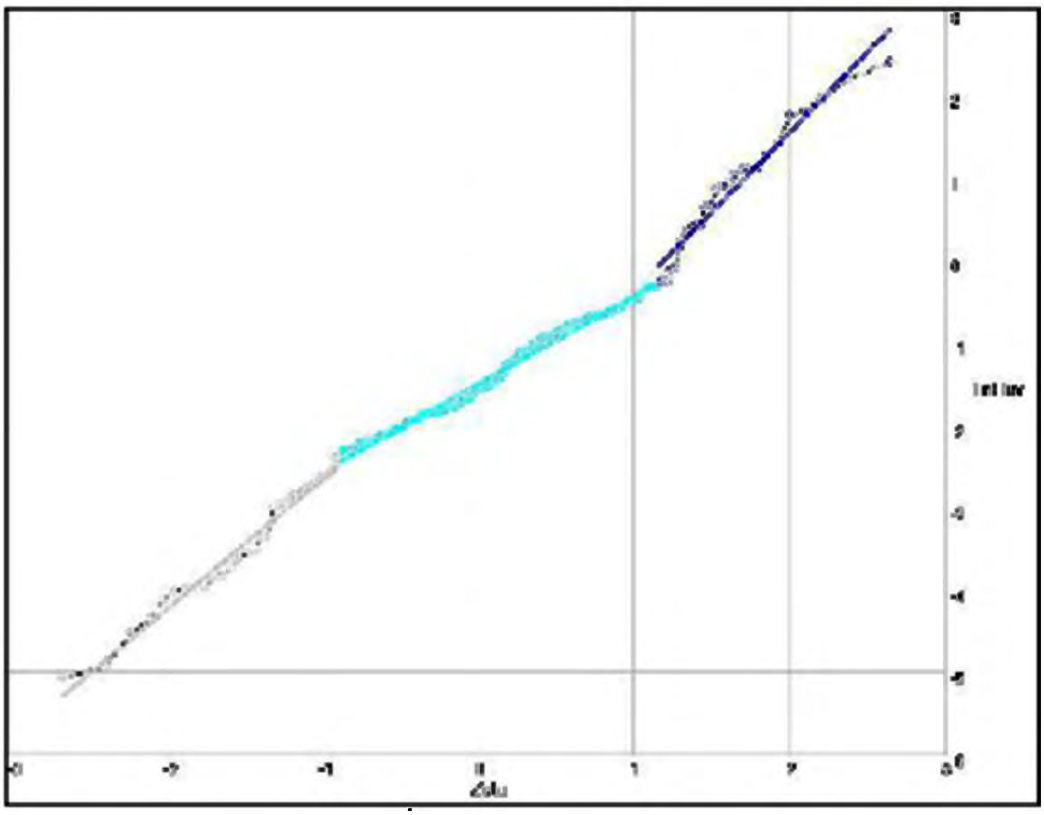

Figure 2. Cumalative probability plots of CH_4 flux.

Subset	N. measures	Mean	Variance	Sichel-V	Flux (Nm^3/h)	95% confidence interval (Nm^3/h)	
1	12	2.19	0.846	1.506	95.4	217.2	61.3
2	131	-0.964	0.889	1.557	46.0	56.4	38.9
3	45	-3.303	0.342	1.185	1.2	1.4	1.0
Tot. CH_4					142.5	275.0	101.3

***Table 1**. Main statistical parameter of each subset of CH4 flux data.*

The Geostatistical approach is used to identify anomalous fluxes from the landfill cover; the data have been mapped using kriging method starting from the study of the experimental variogram. The Geostatistical analysis produces isoflux and standard deviation maps that are presented in Figure 3.

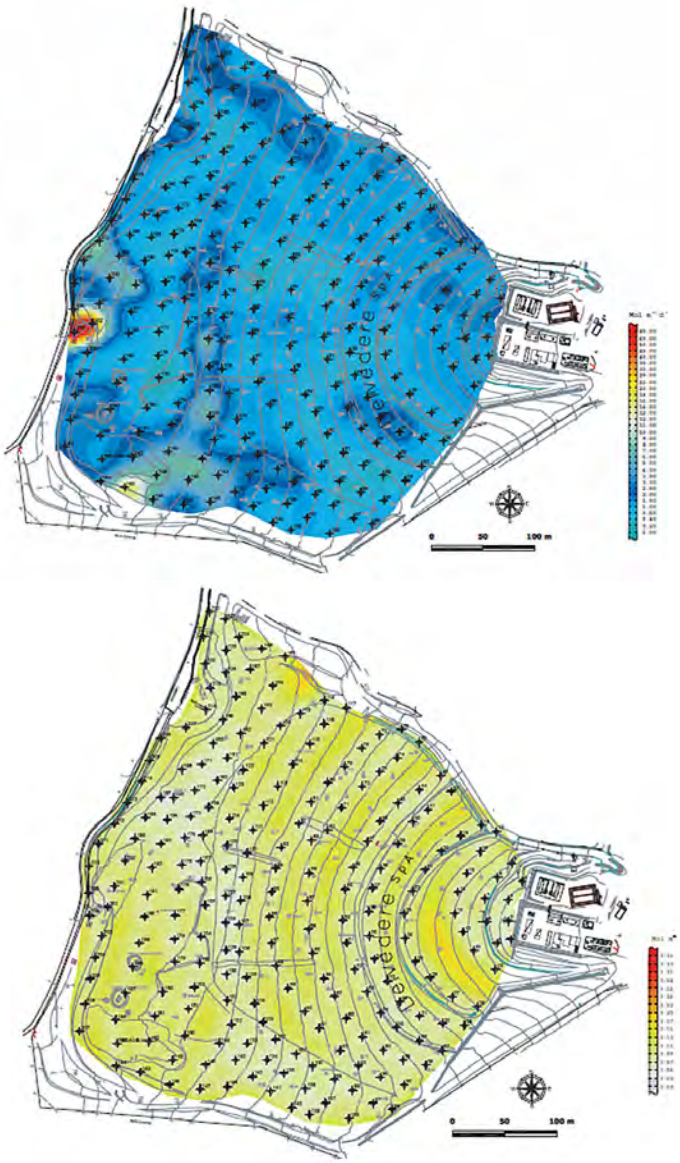

Figure 3. Isoflux (on the left) and Standard deviation map (on the right) released from the cover of the MSW landfill. The black dots represent the locations of flux measurements performed with FLUX-meter.

Conclusions

The integrated equipment set up by WEST Systems allows the measurements of the real amount of CO_2, CH_4, H_2S, and VOC discharged into the atmosphere. Moreover the portable FLUX-meter permits to carry out quick and reliable punctual measurements, which can be processed by an implementation of the Sichel method, developed by WEST Systems, in order to estimate the total amount of LFG discharged into the atmosphere and consequently lost by the capture system. Besides, the elaboration of isoflux maps allow an easy visualization of zone of high fluxes.

6.9. Study case 9: Phytostabilization of Mine Soils/Wastes: Natural Attenuation and Assisted Phytoremediation

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Adapted from

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Santos, E.S., Abreu, M.M., Macías, F., de Varennes, A., 2014. Improvement of chemical and biological properties of gossan mine wastes following application of amendments and growth of *Cistus ladanifer* L. Journal Geochemical Exploration 147, 173-181.

Introduction

Large amounts of waste materials are frequently found in abandoned or inactive mining areas, which due to their high concentrations of metals and metalloids become sources of soils and waters contamination. Often, these mine wastes owing to their adverse physical characteristics, low pH and low content of organic matter and nutrients are extreme environments inhibiting or reducing plant development, and consequently local biodiversity. Contaminated mine areas, frequently, pose a serious environmental risk and can be a threat to human health. Recovery of contaminated sites is a need and a challenge.

Remediation approaches of mine areas using current engineering techniques (Bech et al., 2013) have been proven to be economically prohibitive, and sometimes ecologically unfriendly. These remediation techniques can also, in some cases, reactivate waste materials, creating conditions to increase acid mine generation and even to improve erosion processes during recovery works. Therefore, low-cost and environmentally friendly alternatives must be applied for the recovery of mine areas.

Phytoremediation is a less expensive, non-invasive, and more acceptable technology for remediation of contaminated media (soils, sediments, water, and air) that uses plants and their associated rhizospheric microorganisms (Arthur et al., 2005; Pilon-Smits, 2005; Singh et al., 2003). It includes a variety of techniques that takes advantage of the natural ability of plants to uptake, accumulate and/or immobilize potentially hazardous elements (PHE). Phytoremediation is particularly suitable for sites with large volumes/areas of contaminated materials and is non-disruptive to the landscape and to those living near the contaminated site.

Natural Attenuation

The Panasqueira tungsten and tin mine, one of the few mines still operating in Portugal in the twenty-first century, is located in Central Portugal, Beira Interior region, at approximately 300 km north-east of Lisbon and 200 km south-east of Porto City. This mine has been operating since 1896 and the ore exploitation has generated large amounts of waste materials, constituting huge tailings. The active waste heaps steadily increasing every day are bared. However, the stabilization of old tailings (30-80 years old) as well as the relatively recent (5-10 years old) ones was already accomplished by nature. These dumps, where soils have been developed present an effective vegetation cover mainly composed of *Arbutus unedo* L., *Calluna vulgaris* (L.) Hull, *Cistus ladanifer* L., *Cytisus striatus* (Hill.) Rothm, *Erica arborea* L., *Erica australis* L., *Pinus pinater* Aiton and *Ulex* spp.. The waste materials contain high concentrations (mg/kg) of hazardous elements: As (466-12,000), Cd (2.6-87), Cu (214-3,741), W (40-12,000) and Zn (340-4,224).

***Table 1**. Chemical elements concentrations in soils, aerial part of Arbutus unedo (leaves, twigs and fruits) and arbutus brandy (liquor) (* detection limit)*

(mg/kg)	As	Cd	Cu	Pb	W	Zn
	Soils					
Geometric mean	680	2.4	233	60	138	332
Maximum	7790	79	4080	205	1450	12300
Available fraction (% of total)	<4	4-76	2-39	<0.5-26	<0.04-12	1.5-60
	***Arbutus unedo* (leaves and twigs)**					
Geometric mean	1.4	1.04	3.5	0.41	0.49	100
Maximum	5	12.9	12.9	1.3	3.4	570
	***Arbutus unedo* fruits**					
Minimum	<1*	0.02	2.4	<0.1*	<0.5*	7
Maximum	2	0.08	3.2	<0.1*	0.7	14
Liquor (mg/L)	0.01	<0.05*	3.27	0.34	<0.05*	0.1

The arbutus trees, also known as strawberry trees (*Arbutus unedo*), have a particular interest due to their abundance in the area, vegetative development and also because the mature arbutus berries (extremely sweet) are frequently used to produce sweets, jam, jellies and brandy by their fermentation and distillation. Could this species be used in phytostabilization programs and still bring added value to populations? A biogeochemical study was undertaken to evaluate the impact of mining activity on soils, mainly those developed on waste materials, as well as on *A. unedo* growing on these soils, and to access the possible risks for human health linked to the consumption of the arbutus berries and the derived brandy, and also to evaluate the potential of arbutus tree in the phytostabilization of the contaminated soils and tailings (Abreu et al., 2014). Soils, the aboveground part of *A. unedo* (roots, leaves+twigs and fruits), as well as arbutus brandy were collected and

analyzed for PHE. The Hazard Quotient (HQ) for As, Cd, Cu and Zn was calculated to assess the potential health risk (non-cancer) for the populations exposure to these elements for the fruits consuming (Abreu et al., 2014). Soils in the mining area were contaminated, mainly with As, Cu, Pb, W and Zn. The elemental concentrations in the soil available fraction, extracted with DTPA, were quite variable, but Cd and Zn were the elements that, in general, had the higher percentage of the total soil concentration, reaching a maximum of 76% for Cd and 60% for Zn (Table 1). The majority of the samples of the A. unedo shoots (leaves and twigs) showed concentrations of Al, Cu, Fe, Pb, Mn and Zn in the normal rage for plants in general; however, Cd concentrations exceeded the values considered tolerable for crops, but are lower than the toxic limit for cattle. Cadmium, W and Zn are preferentially translocated from the roots to shoots (Translocation Coefficient >1), whereas Al, As, Cu, Fe and Pb are, in the majority of the samples, accumulated in the plant roots.

Arbutus unedo is not an accumulator species for As, Cd, Cu, Pb, W and Zn (Transfer coefficient from soil to shoots <1). Concentrations of metals/metalloids in fruits were low and the calculated HQ for As, Cd, Cu and Zn in fruits was <0.1, consequently, and according to EC 466/2001 legislation, the fruits consumption do not constitute health risks (nephrotoxicity) for humans. The *A. unedo* fruits can be used to produce local liquor and the concentration of the chemical elements determined in the alcohol is within the range of the European legislation. The *A. unedo* trees fulfills its role on the natural attenuation of the contamination in the Panasqueira mine area because spontaneously colonizes the tailings, belongs to the group of plants of the first stages of vegetation development, promotes waste weathering and pedogenesis and decreases water and wind erosion. This species is not an accumulator of metals/metalloids and their concentrations in the above-ground part of the plants will not represent a threat for the biological systems.

Assisted Phytoremediation

Gossan wastes represent one of the most hazardous mine wastes in several mining areas from the Iberian Pyrite Belt as is the case of São Domingos mine located in the Southeast part of Portugal in the Baixo Alentejo ≈60 km SE from Beja. This nowadays abandoned mine was exploited from the pre-roman period until 1960 decade, both in the *gossan* and volcanogenic massive sulfide ore deposits. Considering the elements mobility, mass/volume and the bioavailable fraction of the potential contaminants, the *gossan* wastes represent the fourth most hazardous mine waste in São Domingos mine (Santos et al., 2014 and references therein). Owing to the limitations of gossan wastes for plants germination and growth the assisted phytostabilization using cost-effective amendments together with spontaneous plants could be a good and sustainable option to improve physical (e.g. structure), chemical (e.g. increase of organic matter and nutrients, immobilization of contaminants, decreased leaching) and biological (e.g. increase of microbial activity and diversity) characteristics of these wastes. In this way, a microcosmos study was undertaken to evaluate the influence of combined use of different application rates of amendments and *Cistus ladanifer* L growth on the improvement of chemical and biological properties of gossan wastes from the São Domingos mine; and assess the growth and accumulation of chemical elements in the plants during the phytostabilisation process. *Gossan* wastes (GW) with very acid pH (4.32), low extractable P and K, and high concentrations (g/kg) of several chemical elements (As-3.03; Cu-0.23; Fe-129; Pb-9.21; Mn-0.062; S-13.7; Zn-0.036) were mixed with a mixture of organic and inorganic amendments (rockwool, agriculture wastes and wastes from fruit spirit

distillation) at different levels (zero (control), 30, 75 and 150 Mg/ha) and putted in pots (~2 kg each). After 15 days of incubation *C. ladanifer* was sown (0.5 g seeds, collected in the mine area, per pot) in half of the pots from each treatment (n = 3), while the other three pots remained bare and were kept for 505 days in greenhouse. Comparing the values of several parameters in the control (C) and the treatments combining amendments and plants (AP), the GW properties were improved: pH 4(C)-5.8(AP); $C_{organic}$ (g/kg) 2.3 (C)-22.1(AP); N (g/kg) 0.2(C)-1.2(AP); $P_{extractable}$ (g/kg) <0.02(C)-10.7(AP); $K_{extractable}$ (g/kg) 1.4(C)-2.1(AP). Enzymatic activities were used as biological indicators to monitor the assisted phytostabilisation of GW. Soil microorganisms, evaluated by the dehydrogenase activity, benefited from the presence of plants and amendment application, especially at higher application rates (75 and 150 Mg/ha) (Figure 1). The microbial activity associated to the carbon cycle (β-glucosidase activity) and phosphorus cycle (phosphatase) increased at the end of the experiment being positively influenced (≈1.4-fold greater) by the presence of the plants (Figure 1).

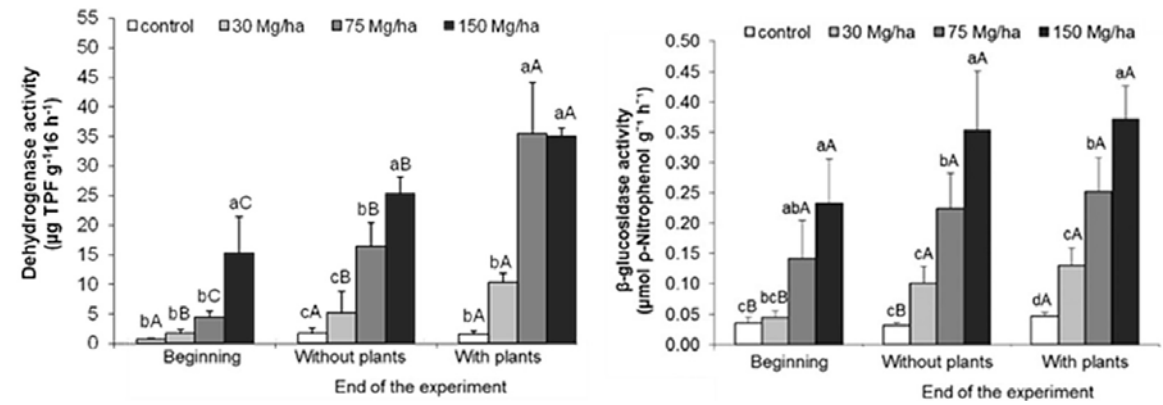

Figure 1. Effects of treatments (control and gossan wastes with amendment at 30, 75 and 150 Mg/ha) and presence of Cistus ladanifer on enzymatic activities at the beginning (after 15 days of incubation and before sowing) and end of the experiment (505 days after sowing) (mean±SD; n=3). Values followed by a different letter are significantly different (p<0.05), small letters indicate comparisons between treatments from the same sampling date and capital letters indicate comparisons between treatments in different dates and with or without plants. (Adapted from Santos et al., 2014).

At the end of the experiment *C. ladanifer* plants were taller, accumulated more fresh biomass in roots and shoots, and had larger young leaves in the amended GW than in the control (Figure 2). Concentrations of As, Ca, K, Pb, Fe, Na and Zn in roots and concentrations of K, Mg and N in shoots were similar, independently of treatment. But plants from control had larger concentrations of Al, N, Cu and Zn in roots and of Al, As, Cu, Fe, Na, Pb and Zn in shoots than those from amended treatments and the values of As and Pb in shoots from control reached values considered phytotoxic for plants. The concentration of As and Pb in the plants growing on the amended GW are considerable lower than those from control, especially for Pb (≈12-fold lower) seeming to be the treatment 75 Mg/ha the most favorable for both elements (Figure 2). The application of a mixture of amendments to *gossan* wastes improved its physical (structure) and chemical properties (fertility, pH, concentrations of nutrients in the available fraction) and increased enzymatic activities related to nutrient cycling and microbial activity. Plants from the amended treatments had lower concentrations of hazardous elements in shoots than those from the control, minimizing potential bioaccumulation in the food chain.

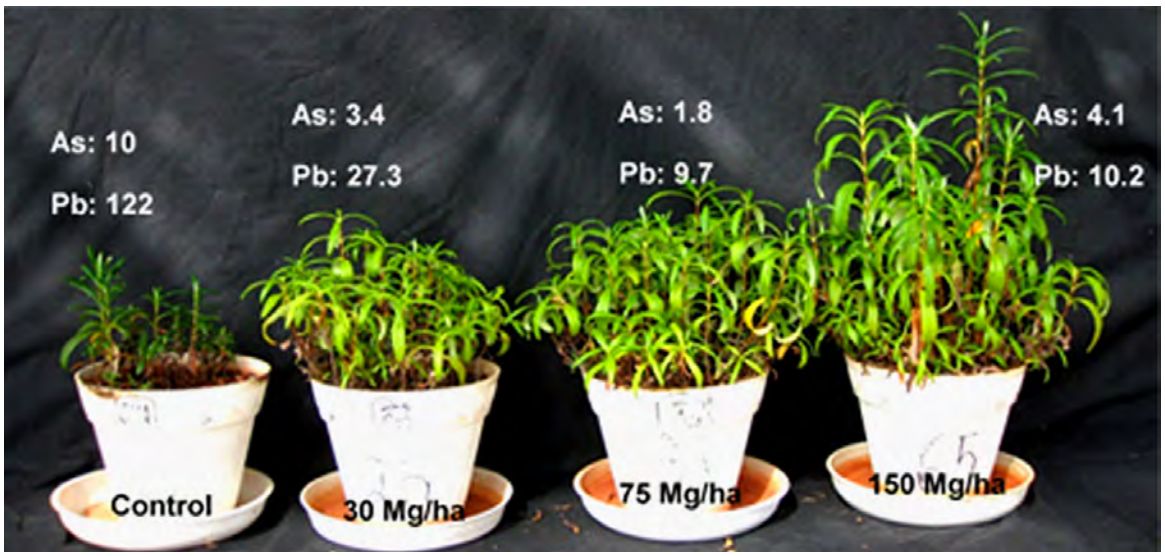

Figure 2. Effects of treatments (control and gossan wastes with amendment applied at 30, 75 and 150 Mg/ha) on the concentration of As and Pb in C. ladanifer shoots at the end of the experiment (505 days after sowing).

Although *C. ladanifer* already colonizes spontaneously contaminated soils/wastes in São Domingos mine area, the *gossan* wastes improvement (physical, chemical and biological) by amendments addition stimulates its growth and decreased the concentration of potentially hazardous elements in shoots, providing a better and faster plant cover during the assisted phytostabilisation process.

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VI Land rehabilitation and rural development

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1. Economic growth versus sustainable development

Since the 50s decade, the world has witnessed changes in its land systems faster than in the rest of human history (MA, 2003). The unsustainable economic growth behind these changes, evidenced in a crescent demand for food, water and other natural resources (Steffen et al., 2015), is leading to a decline of biodiversity and the loss of associated ecological functions. Human-induced climate change is other significant consequence of this global transformation (IPCC, 2013; Steffen et al., 2015), which compromises the capacity of ecosystems to meet the steadily growing demands of people. Far from slowing down, the consumption of biological and physical resources, as well as escalating impacts on ecosystems and the services they provide are predicted to increase: the current estimates for 2050 are of 3 billion more people and a quadrupling of the world economy.

Development has often been defined in terms of economic growth, implying a continuous increase of the value of production and consequently the continuous consumption of resources. Metrics as the Gross Domestic Product (GDP) have an extensive use supporting this perspective. Nevertheless, development does not necessarily mean economic growth. Societies can grow in terms of GDP but not being developed (Costanza et al., 2014b). In fact, an overexploitation of forests or fisheries would signify an increase of the annual growth rate of GDP and thus being interpreted as an economic growth, despite it would imply a loss in resources needed for future development. Economic development must suggest an improving of qualitative potentialities of a society being related to a major access to education, health, jobs and incomes or life expectations, in conclusion, human well-being.

Criticism towards GDP as an indicator of development from an economic, social and environmental point of view is not something new. As Michel Redclift —one of the fathers of sustainable development concept— stated, from an environmental standpoint, the GDP is a particularly inadequate guide to development. "It treats sustainable and unsustainable production alike and compounds the error by including the costs of unsustainable economic activity on the credit side, while largely ignoring processes of recycling and energy conversion which do not lead to the production of goods or marketable services" (Redclift, 1987).

Main international organizations as the United Nations have describe and reported about many negative impacts of human activity in environment. World fisheries are now declining due to overfishing, and a significant amount of agricultural land has been degraded by erosion, salinization, compaction, nutrient depletion, pollution, and urbanization. Other human-induced impacts on ecosystems include alteration of the nitrogen, phosphorous, sulphur, and carbon cycles, causing acid rain, algal blooms, and fish deaths in rivers and coastal waters, along with contributions to climate change. The degradation of ecosystems and their services has many causes, including excessive demand for ecosystem services stemming from economic growth, demographic changes, and individual choices (MA, 2003).

Sustainable Development would aim to deal with these negative effects of economic growth, as a concept for guiding towards a better appraisal of environmental, social and economic sustainability in economic decisions. That means to let to future generations the same development opportunities that we have and to better distribute income among present generations.

There have been many interpretations about how to achieve Sustainable Development since from the concept broadly spread by the Brundtland Commission several contradictions arise. In any case, some principles are well known for achieving sustainability in the use of natural resources.

As Daly and Townsend (1993) state, the renewable resources should be exploited in a manner such that first, harvesting rates do not exceed regeneration rates and second, waste emissions do not exceed the renewable assimilative capacity of the local environment. The regeneration capabilities and assimilation should be considered natural capital. Failure to maintain these capabilities must be considered capital consumption and therefore as unsustainable.

As well, non-renewable resources should be depleted at a rate equal to the rate of creation of renewable substitutes. Actions based on exploitation of non-renewable resources should be paired with other actions that develop renewable substitutes. The net rents from the non-renewable extraction should be separated into an income component and a capital liquidation component. The capital component would be invested each year in building up a renewable substitute (Daly and Townsend, 1993).

2. The concept of natural capital

How much capital we should let to future generations? To answer this question, is necessary to consider that it is the stock and quality of the different forms of capital what determines the capacity of the society for improving its level of well-being. We can consider as the total amount of capital the sum of: natural capital (no built by human beings) and human made capital. Into human made capital we can distinguish material capital (machinery, constructions, technology…), human capital (culture, knowledge, attitudes…) and social capital (institutions and organizations).

Starting from the fact that from a given stock of capital we can obtain a flow of services, the way in which such stock and flow are maintained through time defines sustainability in two diverse ways: weak and strong. Based on the works of Hicks (1939), Page (1977), Hartwick (1977) and Solow (1974), it has been settled the constant capital rule that states that we must maintain the level of utility or well-being constant along the time. Nevertheless, while some academics sustain that this could be achieved by maintaining the total amount of capital constant —regardless of its composition— others state that it is necessary to maintain a stock of each form of capital. The former is an example of weak sustainability, and the latter one of strong sustainability. In both cases, investment is needed along the time for maintaining the stock of capital.

The weak sustainability implies to consider that we can substitute one form of capital by another. Strong sustainability implies to consider a critical level of capital stock for each form, especially for natural capital. Since ecosystems are the support for life, there is some uncertainty about its evolution and resilience capacity, and its destruction could have irreversible consequences. A critical level of natural capital as a safe minimum standard, non-replaceable for any other form of capital, must be conserved and protected. But the degree of uncertainty and ignorance about which level of natural capital must be critical or must constitute a minimum standard is large imposing the application of the precautionary principle. As an example, if we don't know the scope of the role that a forest is having in the water regulation cycle of a given region, the precautionary principle advises not to destroy that forest or not to diminish its dimension, even if the harvested timber would reach a high price in the market.

In 2000, the United Nations Millennium Declaration was signed and committed world leaders to combat poverty, hunger, disease, illiteracy, environmental degradation, and discrimination against women. The Millennium Development Goals are derived from this Declaration, and one of them is to ensure environmental sustainability. But for achieving this goal we need sound scientific information. Kofi Annan stated that: “While major advances in data collection have been made in many areas, large gaps in our knowledge remain. In particular, there has never been a comprehensive global assessment of the world’s major ecosystems” (MA, 2003). In this sense, U. N. launched at the same time, the Millennium Ecosystem Assessment as a major international collaborative effort to map the health of the planet to give response to this need.

3. How to assess natural capital: The ecosystem services approach

The concept of ecosystem services was developed through the 1990s (Costanza & Daly, 1992; Costanza et al., 1997; Daily, 1997), from the base of earlier concerns regarding the relationship of society and environment in the late 1970s. It starts with the utilitarian framing of beneficial ecosystem functions as services in order to increase public interest in biodiversity conservation (Westman, 1977, Gómez-Baggethun at al., 2010). Now, they are defined as the benefits that people obtain from ecosystems (Costanza et al, 1997).

In 2000, the United Nations create the Millennium Ecosystem Assessment (MEA). It was established with the involvement of governments, the private sector, nongovernmental organizations, and scientists to provide an integrated assessment of the consequences of ecosystem change for human well-being. At the same time, it looks for analysing options available to enhance the conservation of ecosystems and their contributions to meeting human needs.

In a world economy market is the resource allocation mechanism intended to be the most efficient to take decisions about the allocation of resources. However, there is a concern from MEA regarding economy market and ecosystem management with ecosystem services in sight. Markets fail specially in the allocation of natural capital and environmental assets as well as considering uncertainties about the future and inter-intra-generational equity issues associated to the management of ecosystems. Markets are intended to be efficient, not equitable. At the same time, inefficiencies are frequent regarding the provision of ecosystem services such as cultural or regulatory services. In this sense "(…) *institutions are now only beginning to be developed to enable those benefiting from carbon sequestration to provide local managers with an economic incentive to leave a forest uncut, while strong economic incentives often exist for managers to harvest the forest. Also, even if a market exists for an ecosystem service, the results obtained through the market may be socially or ecologically undesirable. Properly managed, the creation of ecotourism opportunities in a country can create strong economic incentives for the maintenance of the cultural services provided by ecosystems, but poorly managed ecotourism activities can degrade the very resource on which they depend*." (MA, 2003).

The assessment framework developed for the MEA is anthropocentric, placing human well-being and their linkages with ecosystems at the central focus for assessment (Figure 1). It intends to offer decision-makers a mechanism to:

- Identify options that can better achieve core human development and sustainability goals. The MEA process, at all scales, was designed to bring scientific support to decision-makers concerning the links between ecosystems, human development, and sustainability.
- Better understand the negative trade-offs involved—across sectors and stakeholders—in decisions concerning the environment.
- Align response options with the level of governance where they can be most effective. Effective management of ecosystems will require actions at all scales, from the local to the global. Human actions may affect directly or inadvertently virtually all of the world's ecosystems. MEA developed a multiscale assessment framework for analysing policy options at all scales-from local communities to international conventions.

The concept of ecosystem provides a framework for analysing and acting on the linkages between people and the environment. The United Nations Convention of Biological Diversity (Nairobi, 1992) states that the ecosystem approach is a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way.

This approach recognizes that humans, with their cultural diversity, are an integral component of many ecosystems. At the same time, ecosystems play several functions for human life. An ecosystem function is the technical term used to define the biological, geochemical and physical processes and components that take place or occur within an ecosystem. These functions may supply or contribute to provide benefits for human life called "ecosystems services" (Costanza et al, 1997).

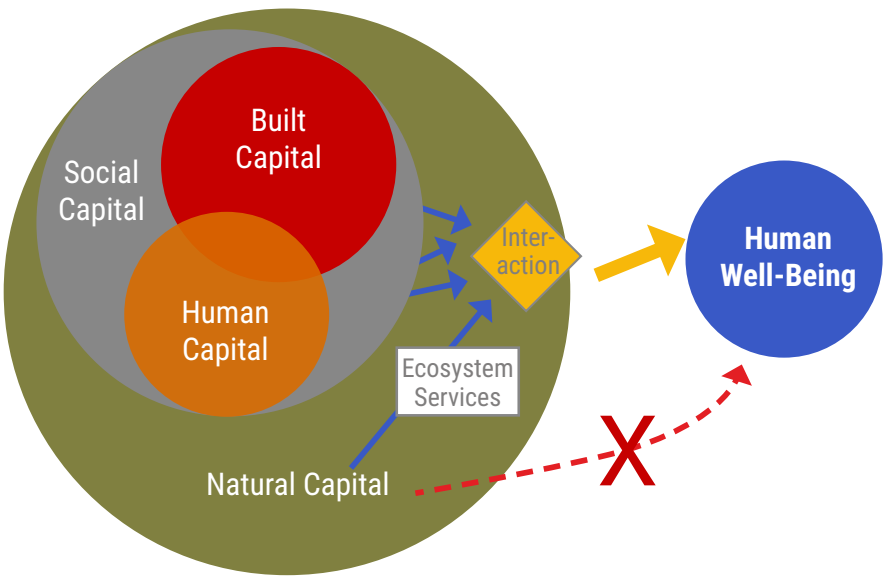

Figure 1. Relationship between natural and other forms of capital, ecosystem services and human well-being. Source: Costanza et al (2014).

Functions and services are different concepts. Different ecosystem functions contribute to the provision of different ecosystem services in different quantities or magnitudes. Each ecosystem function can contribute to more than one ecosystem service, and it takes more than one ecosystem function to provide any ecosystem service.

Ecosystems functions have been defined as: 'the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly' (De Groot et al. 2002) and they have been classified into main four categories -even though several classifications exist nowadays as the Common International Classification of Ecosystem Services (CICES) or the TEEB (2010):

- Supporting or Habitat Functions are defined as the capacity to provide habitat (suitable living space) for wild plant and animal species at local and regional scales. Without this function life would be impossible.

- Regulation functions are related to the maintenance of essential ecological processes and life support systems. They would be climate regulation, gas regulation, disturbance regulation, water regulation, soil retention, nutrient regulation, waste treatment and assimilation, pollination, biological control and barrier effect of vegetation.
- Production functions have been defined as the capacity of provision of natural resources through photosynthesis and nutrient uptake by autotrophs that convert energy, carbon dioxide, water and nutrients into a wide variety of carbohydrate structures which are then used by secondary producers to create an even larger variety of living biomass.
- Information function has been defined as providing life fulfilment opportunities and cognitive development through exposure to life processes and natural systems contributing to human health.

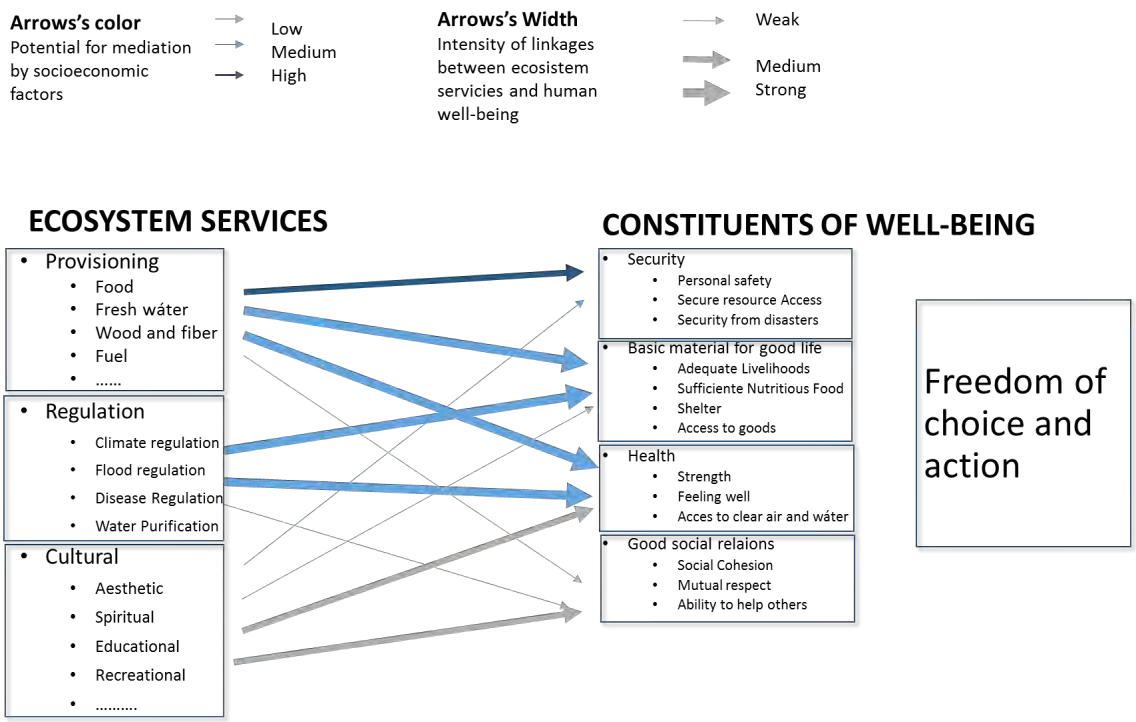

Figure 2. Relationship between ecosystem services and well-being.
Source: Millennium Ecosystem Assessment Report (2003).

Functions can be reconceptualised as ecosystem services when we identified benefits or values that ecosystem functions provide to humans. Ecosystem services concept implies an anthropocentric vision. They were classified as: provisioning, regulating, and cultural services. Provisioning services are the products people obtain from ecosystems, such as food, fuel, fibre, fresh water, and genetic resources. Regulating services are the benefits people obtain from the regulation of ecosystem processes, including air quality maintenance, climate regulation, erosion control, regulation of human diseases, and water purification. Cultural services are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

The figure depicts the strength of linkages between categories of ecosystem services and components of human well-being that are commonly encountered and includes indications of the extent to which it is possible for socioeconomic factors to mediate the linkage. The strength of the linkages and the potential for mediation differ in different ecosystems and regions. In addition to the influence of ecosystem, services on human well-being, and ecosystems are in turn affected by changes in human well-being (MA, 2013).

4. Taking into account ecosystems services for decision making

Valuation is useful to assess trade-offs existing toward achieving a goal. All decisions that involve trade-offs involve valuation, either implicitly or explicitly (Costanza et al., 2014). Current decision-making processes often ignore or underestimate the value of ecosystem services for several reasons. The most important involve the character of public goods that ecosystem services have. They cannot be appropriated for their transformation in market values.

Decision-making concerning ecosystems and their services can be particularly difficult because different disciplines, philosophical views, and schools of thought assess the value of ecosystems differently. From Economy two different paradigms offers diverse ways to value ecosystems: ecological and environmental economics.

The first step to consider the ecosystems and their services is to identify them and to assess changes in their biophysical state. Planning for and managing the impacts of development is an important part of ensuring that natural environment can continue to support community well-being and economic security for current and future generations. Plans should identify ecosystems and their services. The MEA used 10 categories of systems; each of them contains a number of ecosystems not mutually exclusive. They can overlap, and this made not easy to assess their complexity. Also, the assessment of biophysical changes has its difficulty since changes are not linear and take time to develop.

The second step is to give a value to the services identified in order to include them in decision making processes. The economic science has defined the total economic value as the addition of different concepts: utilitarian and non-utilitarian. The first refers to the use value that people give to things. People value ecosystem services because they benefit from them directly or indirectly. It is that we call Use Value. Also, people can value goods and services provided by ecosystems not for using them but for having the possibility to use them in the future. It is that we call Option Value (Weisbrod, 1964). Arrow and Fischer also defined in 1974 the Quasi Option Value as the value that people give to natural assets by the information that they provide for conservation. Finally, we have to consider non-utilitarian values such the Heritage Value or the Existence Value, ascribed for the resource's existence even if people never use that resource directly. These often involve the deeply held historical, national, ethical, religious, and spiritual values people ascribe to ecosystems. Then, the Total Economic Value will be the sum of all these values that we have described. How we can assess them?

In 2002 de Groot et al. proposed the following framework for an integrated assessment and valuation of ecosystem functions, goods and services.

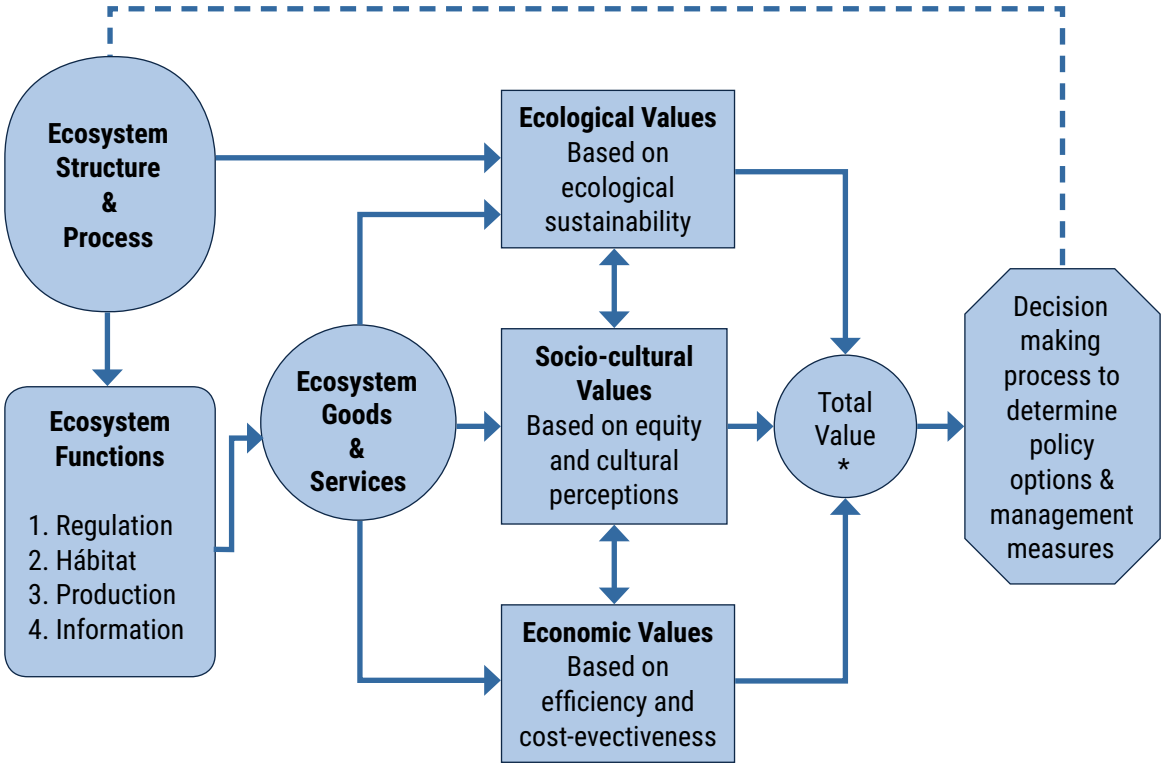

Figure 3
Source: de Groot et al. (2002).

The Economic valuation is one of the different criteria that we must consider for decision-making but not the only one. Even in economic terms, proposals for valuation are different depending of the economic approach adopted. The most spread techniques are based on human preferences and on the Economic Utility Approach.

In this sense, Costanza (2014) states that "*The relative contribution of ecosystem services can be expressed in multiple units — in essence, any of the contributors to the production of benefits can be used as the ''denominator'' and other contributors expressed in terms of it. Since built capital in the economy, expressed in monetary units, is one of the required contributors, and most people understand values expressed in monetary units, this is often a convenient denominator for expressing the relative contributions of the other forms of capital, including natural capital. But other units are certainly possible (i.e. land, energy, time, etc.) — the choice is largely about which units communicate best to different audiences in a given decision-making context*".

Within the approach based on human preferences, economic valuation methods fall into four basic types, each with its own repertoire of associated measurement issues (de Groot et al., 2002): (1) direct market valuation, (2) indirect market valuation, (3) contingent valuation, (4) group valuation.

Direct market valuation (1) is the exchange value that ecosystem services have in trade, mainly applicable to the 'goods' (i.e. production functions) but also some information functions (e.g. recreation) and regulation functions. When there are no explicit markets for services, we must

resort to more indirect means of assessing values (2). A variety of valuation techniques can be used (MA, 2003):

- Avoided Cost (AC): services allow society to avoid costs that would have been incurred in the absence of those services. Examples are flood control (which avoids property damages) and waste treatment (which avoids health costs) by wetlands.
- Replacement Cost (RC): services could be replaced with human-made systems; an example is natural waste treatment by marshes which can be (partly) replaced with costly artificial treatment systems.
- Factor Income (FI): many ecosystem services enhance incomes; an example is natural water quality improvements which increase commercial fisheries catch and thereby incomes of fishermen.
- Travel Cost (TC): use of ecosystem services may require travel. The travel cost is a reflection of the implied value of the service. An example is recreation areas that attract distant visitors whose value placed on that area must be at least what they were willing to pay to ravel to it.
- Hedonic Pricing (HP): service demand may be reflected in the prices people will pay for associated goods; an example is that housing prices at beaches usually exceed prices of identical inland homes near less attractive scenery.

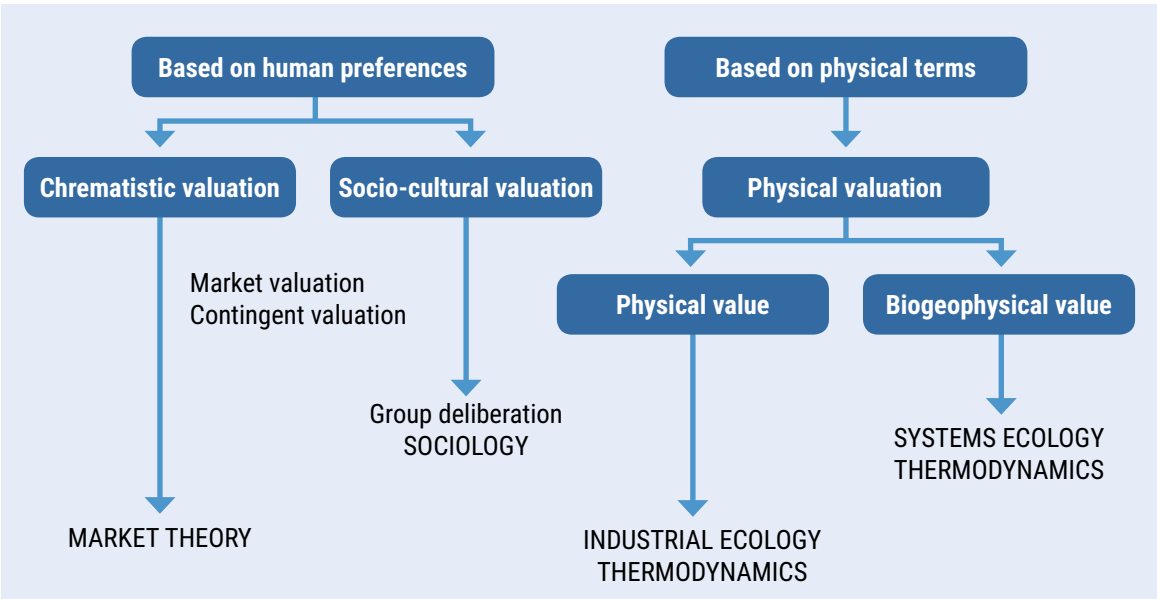

Figure 4. Valuation methods for Natural Capital
Source: Adapted from Gómez-Baggethum & de Groot. (2007).

Contingent Valuation (3) presents hypothetical scenarios that involve the description of alternatives in a social survey questionnaire for eliciting service demand. For example, a survey questionnaire might ask respondents to express their willingness to pay (or their willingness to be compensated), to increase the level of water quality in a stream, lake or river so that they might enjoy activities like swimming, boating, or fishing.

Finally, derived from social and political theory, the Group Deliberation valuation approach (4) is based on principles of deliberative democracy and the assumption that public decision making should result, not from the aggregation of separately measured individual preferences, but from open public debate.

Each of these methods has its strengths and weaknesses. Not all measure the total economic value and not all applies to the complete set of ecosystem services. Following de Groot et al. (2002) Regulation Functions were mainly valued through Indirect Market Valuation techniques (notably Avoided Cost and Replacement Cost), Habitat Functions mainly through Direct Market Pricing (i.e. money donated for conservation purposes), Production Functions through Direct Market Pricing and Factor Income methods, and Information Functions mainly through Contingent Valuation (cultural and spiritual information), Hedonic Pricing (aesthetic information) and Market Pricing (recreation, tourism and science).

The value resulting from these methods is not necessarily a market value. It is useful to insert ecosystem services in the decision-making process as other goods and services with market value. Natural capital and their services are often hidden for public and private agents when they take investment and economic decisions. For instance, in a housing planning, usually the natural capital value destroyed is not reflected in the value of houses. As well, the investment in natural capital as the restauration of lands after a wildfire could be an expense (and then evitable) instead of as an investment if we are not aware about the value of benefits of restoring. In any case, these monetary valuations must be complemented by other physical or social valuations to help decisions. The most important problem to confront is the uncertainty and the unacknowledged about ecosystems and their functions for present and future generations.

5. Why restoration of ecosystems is needed? New demands for rural areas

Investments in natural capital are necessary for conserve it for future generations. Restoration techniques can help us to conserve functions and services from ecosystems. Like the benefits of increased education or improved governance, the protection, restoration, and enhancement of ecosystem services tends to have multiple and synergistic benefits (Benayas et al., 2009).

Rural areas where agriculture and forestry activities have place, often own valuable ecosystems. Agriculture and forestry have evolved with ecosystems creating along the time what we call agri-ecosystems from which human's profit of a wide range of services. It is there where often the trade-offs involved in land use scenarios occur: scenarios that maximize biodiversity conservation and ecosystem services versus scenarios that maximize profit from a single commodity.

Traditionally services that humans demand from human areas were productive ones as food, fibres, minerals, timber and so on. But the more and more, society demand from rural areas regulation and cultural services. At the same time, the increases in productivity, favoured by industrial farming, has created a wide range of negative externalities (harms) over natural capital as well as over human capital in rural areas.

Some examples of negative consequences for natural capital from industrialized farming and forestry systems are soil erosion and degradation, loss of biodiversity, soil and water pollution, droughts … All these phenomena compromise the supply of regulation services and cultural services especially when production is concentrated in some areas.

At the same time, in the European context of the two last decades, increases in productivity and the decline of agricultural prices have driven out people from rural areas with the subsequent loss of human capital. This contributes to marginalization of agricultural land where agriculture has been abandoned because farming ceases to be viable under an existing land use and socio-economic structure. Marginalization and abandonment have become a threat to biodiversity and landscape in many European areas like Spain (López-Iglesias et al., 2013). Abandoned land generally turns into forest over time but with a lesser value for endangered species than original ecosystems. Also, the lack of management increases the risk of fire, being one of the biggest environmental problems in rural areas for South European countries.

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Vulnerability to marginalization depends on the local or regional, social, economic, political and environmental conditions (Brouwer, 2004). Economic marginality is a major driving factor behind the vulnerability of rural areas in Europe but also natural constraints. Marginal areas for agricultural production have often climatic constraints, poor soils and poor accessibility of agricultural lands. Human choices, farmer education, credit, infrastructure, markets and culture are other driving forces behind marginalization.

Natural capital is seen as a new opportunity for these areas since the demand for nature and landscape has become greater in our society. In this sense, a more effective management of ecosystems must contribute to the maintaining of life-support but also to create economic opportunities for rural areas. Actions for ecosystems restoration, for instance after a forest fire, are needed not only for restoring the flow of ecosystem services but for giving to these areas capital (Nature) for economic development. The valuation of the ecosystem services of these areas can enforce the willingness of society for financing the conservation and restoration of these systems. As an example, Barrio & Loureiro (2010) results show that willingness to pay estimates for forest management programs are sensitive to the program's objectives, particularly when linked to the provision of recreational services. In terms of the ecological and physical characteristics of forests, they found that rain forests and a combination of deciduous and perennial forests are more valued than coniferous. Another example of public preferences about Nature is reported by Nuss-Girona & Castañer (2016) who found that, in Spain, the most valued services by public were water quality, biodiversity conservation, gene pool, local ecological knowledge, and erosion control.

6. Instruments for sustainable management of ecosystems

Policymakers have available a variety of responses for sustainable management of ecosystems for ensuring human well-being. These instruments were classified by the MEA approach according to a typology of legal, economic, social and behavioural, technological and cognitive interventions. They intend to modify the behaviour of actors towards the conservation and sustainable management of natural capital.

RESPONSES	
Legal	Treaties; international soft law; international customary law; international agreements; domestic environmental regulations; domestic administrative law and constitutional law;..
Economic	Command-and-control interventions; incentive-based; voluntarism-based; financial-monetary measures; international trade policies.
Social and behavioral	Population policies; public education and awareness; empowering youth, communities and women; civil society protest and disobedience
Technological	Incentives for innovation R&D
Cognitive	Legitimization of traditional knowledge Knowledge acquisition and acceptances

Figure 5. Types of responses for sustainable management of ecosystems for ensuring human well-being. Source: from Millennium Ecosystem Assessment Report (2003).

Legal instruments: The responses are guided by an institutional framework that sets the rules (formal or informal) of the game. Legal responses serve a ''command and control'' function. Formal laws guide many of the other responses. With growing recognition of the dangers of environmental degradation and the need to protect ecosystems for intra- and intergenerational well-being, legal responses gain strength. All legal responses usually remain static without implementation, compliance, and enforcement in respective jurisdictions (MA, 2003).

Economic and financial interventions: These response options are based on the premise that human beings want to maximize their economic welfare. We can find market mechanisms and financial mechanisms. The effectiveness of the economic intervention mechanism, however, is moderated by the fact that socioeconomic conditions vary from society to society (MA, 2003).

Empowering people through the conferral of rights, liberties, and responsibilities, and through education and information dissemination: Women, civil society, local communities, and youth tend to demonstrate a strong aptitude for ecosystem stewardship because they are more directly dependent on ecosystem services for sustenance. Participation and inclusiveness are important for instilling attitudes of stewardship (MA, 2003).

Technological responses allow humans to mitigate their effects on ecosystems by allowing less dependence on them, by lowering anthropogenic impact, or by helping to restore degraded ecosystems. But, the risk of side effects and unintended consequences of technological fixes make it imperative that proper evaluation and risk assessment be carried out before resorting to this response (MA, 2003).

Knowledge underlines all types of responses. Given the role that knowledge plays in forging cognitive processes, creating knowledge, applying it to concrete problems, and disseminating it are important options for policy response. New knowledge creates human capital and, at the same time, inspires and guide institutional change. (MA, 2003).

7. Payments for ecosystem services schemes at rural areas

7.1. Definitions and types

Among the economic and financial interventions, it is now in discussion the concept of Payments for Ecosystem Services Schemes (PES). At the same time, several experiences have been put into function across the world. Following Pascual & Corbera (2011) projects that link direct payments with the maintenance or provision of environmental services are considered in general terms as PES. It has been said that a PES is a volunteer transaction where a well-defined ecosystem service is bought by a buyer from a service provider if and only if the provider secures its provision (conditionality) (Engel et al., 2008). Then, they argue that there are at least three necessary conditions for the design of a 'genuine' PES scheme: a) the relationship between the type of land use being promoted and the provision of the ecosystem service must be clear; b) stakeholders must have the possibility to terminate the contractual relationship (it is a voluntary transaction); and c) a monitoring system must accompany the intervention, in order to ensure that the provision of services is taking place (additionality and conditionality of payments).

At the same time, it is common to refer to PES as pure PES if they cover the five conditions defined by Engel et al (2008), or quasi-pure PES if they don't cover one of the cited conditions, most of times the definition of the service or the additionality condition. In this sense, the literature uses to claim for a good definition of the ecosystem service in order to identify and evaluate a PES scheme (Matzdorf et al., 2013; Engel et al., 2008). There are other authors who use a larger concept of PES (Muradian et al., 2010) including for instance, public instruments already existent to subsidy agriculture as a source of positive externalities. Vatn (2010) points out that a wide variety of PES cases depend strongly on State and community engagement, and therefore cannot be considered as voluntary market transactions, at least from the buyer's point of view. Even if private transactions occur, sometimes the voluntary condition is not met.

Nowadays, following we consider PES as transfers of resources between social actors in order to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources as Muradian et al., (2010) stated. And this can be met by using market mechanisms, public intervention or community or private based instruments.

A classification of PES can be done distinguishing between those based on activities and those based on results or performance. Activity-based schemes (like those rewarding CO2 capture) are the most prevalent in practice due the fact that it is often impossible to link the payment to the demanded service because ES will be delivered along the time and being measured in the medium or long run. In results-based schemes farmers and landowners are rewarded not for performing activity-based standards but for achieving set environmental outcomes (it is the case of payments for maintaining local endangered breeds where the payment is per UGM).

Following the economic approaches where they take inspiration, we can talk about Coasian PES or Pigouvian PES. To date, the mainstream conceptual basis for PES has been Coasian economics. In the case of environmental problems, it proposes that, as long as transaction costs are low enough and property rights are clearly defined, individuals, communities and even supra-national entities would trade their rights away until a Pareto-efficient provision of environmental goods and services has been achieved (Muradian et al., 2010). Creating markets would be the solution for providing ecosystem services that are undersupplied. This inspired the definition of Engel at al. (2008). On the contrary, in the Pigouvian approach States use diverse types of instruments such as taxes (nitrogen emission taxes) or payments as financial aids (agri-environmental schemes) to spur ES provision (Matdorf et al., 2013).

Because of the broad variety of PES approaches and different combination of governance types, Matzdorf et al. (2013) distinguishes four types of instruments depending on their institutional characterization, analysing actors and actors' motivations: a) User- and non-government financed payments; b) Government payments (Pigouvian approach); c) Compliant payments; d) Compensation payment for legal restriction.

A. User- and non-government financed payments

Here, the buyers of ES are mainly motivated by individual utility or profit maximization at least on the demand side. Thus, there is high self-interest on the user side to design such a contract ensuring conditionality because otherwise the user would have a high financial risk. Another situation is done when a private company is the buyer acting for an intrinsic interest embedded in a corporate social responsibility strategy or for marketing purposes. Finally, we can describe in this kind of PES the contracts between providers and environmental NGOs where the later acts as intermediaries behind individual financers or act using own financial means. Some examples are when upstream users of a water stream bargain with downstream users to be paid for maintaining water in good conditions. The case of Vittel in France where the water company pays upstream farmers for not using fertilizers is well known example of that. As well in the Paso de Caballos River Basin in Nicaragua upstream landowners are paid by private downstream households for reforestation and conservation efforts in order to maintain good water conditions (Lipper and Neves, 2011).

For this type of payments, the state is only relevant for enforcement of laws or to provide an institutional framework to promote this kind of agreements including the definition of rights to enjoy the service.

B. Government payments

Based on the Pigouvian approach, the most relevant example are the agri-environmental payments under the Common Agricultural Policy in Europe. Here, the State represents the whole society as the beneficiary of the ecosystem services delivered. Farmers are the providers through a voluntary contract, and they are paid by doing certain activities beyond the compulsory minimum legal levels. These schemes have been criticized because a lack of monitoring, since some of them has been conceived as an income complement. Like this, this type of PES has been classified as pseudo-PES.

Also the first PES labelled programs in Latin America are governmental payments, e.g. Costa Rica's national PES program'Pagos por Servicios Ambientales (PSA)'. An interesting case of a PES scheme between user-financed and government-financed and compliant payment scheme is when the state regulates demand by levying a user fee. For example, the Costa Rican PSA program raises part of the funding for PES through a water tariff inducing a major interest for ensuring conditionality. In any case, this type of schemes applied to large territories reduces transaction costs between buyers and providers.

C. Compliant payments

This is the case of cap-and-trade systems. In the case government payments, it is the payment that is creating the incentive for protecting biodiversity or ensuring sustainable use. In the case of cap-and-trade, it is the cap that plays this role. The market is established to reduce the costs of imposing the cap (Vatn et al., 2011). Examples can be found in developed countries, such as the US wetland mitigation banking where US law allows developers to drain wetlands only if they pay for the reconstruction of an equivalent area of wetland elsewhere. Matzdorf et al. (2013) consider this mixed mechanism as a PES scheme because the payment to providers exist and ensure the continuity of this provision. These authors underline that, regarding this model, the actors on the provider side use to involve a broad variety from NGOs over social entrepreneurs to commercial enterprises, partly as direct ecosystem services sellers, partly as intermediaries. Not all have an intrinsic interest on the provision but in the compliance of the legal requirements. At the same time the bulk of the monitoring work is often borne by the State and the, by the whole society indirectly.

D. Compensation payments for legal restrictions

States can use legal restrictions over the use of certain natural resources in order to avoid negative externalities or to foster the provision of a critical level of ecosystem services, or to ensure the performance of the support function. Some PES are designed to compensate this legal restrictions for equity reasons or for avoiding actors resistances. This is the case of the application of Habitat Directive at the European Union where some payments are designed to compensate farmers or other landowners for the loss of benefits carried out by the compliance of conservation restrictions. The conditionality here is ensured.

7.2. PES Promoting Land Restauration: Some examples

7.2.1. Reducing Emissions from Deforestation and Forest Degradation

Nowadays, PESs schemes are related with water basins management, biodiversity and landscape conservation, and carbon sequestration. In last decades, the so-called REDD schemes ('Reducing Emissions from Deforestation and Forest Degradation') has been developed in developing countries while international climate negotiations take place. In 2007, during the 13th Conference of the Parties (COP 13), the United Nations Framework Convention on Climate Change (UNFCCC) launched negotiations on REDD. The concept was later broadened to include conservation of forests, sustainable management and the enhancement of forest carbon stocks. This is referred to as REDD+ ("REDD plus"). International negotiations to define REDD+ and design an international REDD+ framework are ongoing. It creates a financial value for the carbon stored in forests by offering incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to development. The idea is to pay landowners or land users for adopting management strategies that improve the carbon storage or reduce carbon emissions. Developing countries would receive results-based payments for results-based actions. Deforestation and forest degradation account for approximately 17 per cent of carbon emissions, more than the entire global transportation sector and second only to the energy sector.

In addition, it has been said that REDD+ aims to promote other advantages, often referred to as the non-carbon benefits like biodiversity protection; poverty reduction;

food and water security; and improved livelihoods for communities that depend on forests, including through clarifying land tenure and strengthening governance. As well, several authors state that PES have advantages over other mechanism of conservation because they are direct, volunteer and subject to compliance, and thus make them potentially equitable or socially cost-efficient, minimizing the distortion effects in the general economy. But, the extension of this mechanism in Asian and Latin-American countries have revealed that the implementation of market instruments where social realities don't recognize neither understand this institution, leads to a reallocation of use rights not always demanded or accepted by local population.

Börner et al. (2011) examine five examples of REDD schemes in Latin America. The studied schemes were applied over the base of existent management forest regulations that must be observed in order to get payments. Then, we can classify them as compliant payments as we saw at the previous section. They have important costs of implementation as the payment is not the single incentive and others like technical assistance or other social services, increases the cost. These schemes cannot compensate equally every affected actor and we would find winners and losers. Data from Peru and Brazil show that a great part of the potential of carbon reduction comes from avoiding the expansion of deforestation practices for agriculture and breading purposes. This fact has two consequences depending on the social structure of the area, the lack of employment for people not remunerated by the REDD scheme or a most intensive use of labour force in farming land. The introduction of REDD schemes in areas where the access to land is not equitable and perceived as unfair, may present problems of equity, especially when financial sources are public.

7.2.2. Agri-environmental Schemes at the European Union

Agri-environmental Schemes (AES) have been qualified as quasi-PES. Initially they were conceived as financial incentives to lead farmers, on a voluntary basis, towards less environmental harmful practices. Actually, there have been two types of schemes, those which compensate for the profit loss of modifying farming practices beyond legal requirements, and those which pay for conserving practices considered positive for providing ecosystem benefits. The payment rewards positive externalities provided by agricultural activities. Farmers commit themselves, for a minimum period of at least five years and the schemes can be design at national, regional or local level.

But some criticisms were raised about first agri-environmental payments (EC, 2005) related to the efficiency of the measures defined, the definition of environmental objectives and monitoring. Because of that, it seems that agri-environmental policy is slowly moving towards PES concept. Right now, the definition given by the official site focus on the provision of environmental services with two types of payments following the classification already given: compensation payments and government payments. A key concern is the amount of payments and it is in the agenda not to overcompensate farmers.

Thereby contrary to the developing countries, where PES offers an innovation for the governance of ecosystems considered more efficient than former instruments within a weak institutional framework (Engel et al., 2008; Ezzine de Blas et al., 2011; Hrabanski et al. 2013), in Europe PES have been introduced through the Rural Development Policy adapting the former existent instruments. Three long-term strategic objectives for EU rural development policy in the 2014-2020 period can be identified: (i) fostering the competitiveness of agriculture; (ii) ensuring the sustainable management of natural resources, and climate action; and (ii) achieving a balanced territorial development of rural economies and communities including the creation and maintenance of employment.

The European Agricultural Fund for Rural Development (EAFRD) finances an important number of examples of ES provision not only by the agri-environmental policy but through other financial mechanisms included in Rural Development Programs as the LEADER approach. Member States are required to build their RDPs based upon at least four of the six common EU priorities among them we can find: (i) restoring, preserving and enhancing ecosystems related to agriculture and forestry and (ii) promoting resource efficiency and supporting the shift toward a low-carbon and climate-resilient economy in the agriculture, food and forestry sectors.

For the first one of the priorities quoted the focus areas that have been identified are: restoring and preserving biodiversity (including in NATURA 2000 areas and areas of High Nature Value farming) and the state of European landscapes; improving water management; and improving soil management. Examples of commitments covered by national/regional agri-environmental schemes are, for example, environmentally favourable extensification of farming; management of low-intensity pasture systems; integrated farm management and organic agriculture; preservation of landscape and historical features such as hedgerows, ditches and woods and finally, conservation of high-value habitats and their associated biodiversity. Land restauration is nowadays promoted by EAFRD in several countries. Some examples are quoted in the following table:

Country/ Region	Title	Objective	Topic/Focus
Sweden/ Öster Götland	A HNV restoration project in the Öster Götland region of Sweden, using agri-environment funding to support land management actions and promote a cooperative approach between landowners.	Biodiversity conservation; preservation of landscape; Other (preservation of natural and cultural heritage)	Topic: Restoration of HNV area Focus: Combination of several measures; Small farms; Implementation of collective approaches; Involvement of local communities (Restoration support, environmental support for pastures and mown meadows) *in combination with a pre-scheme pilot project
Spain/ Castilla-León	Maintaining extensive grazing in the monte farming systems in the Castilla Y Léon region of Spain.	Fire prevention	Topic: Fire prevention through extensive grazing Focus: To prevent wildfires on 'monte' (Forest and grassland) through the re-introduction of farming in abandoned areas. Plan 42 is the forest fire prevention strategy of Castilla y León, set up by the regional Ministry of Environment in 2002.
Portugal/ Tejo	Using the forest environment measure to deliver Natura 2000 site management requirements under the Integrated Territorial Intervention approach in Portugal.	Biodiversity conservation	Topic: Restoring High Nature Value agroforestry. Focus: Implementation of a single measure.
Poland	Targeting and tailoring agri-environment schemes to maintain natural grassland areas in the mountain areas of Poland.	Biodiversity conservation; preservation of landscapes; water management; soil functionality	Topic: Agri-environmental programmes to reduce the decreasing of natural grassland areas, based on regionalized approach in mountainous areas. Extensive farming methods in the meadow to reduce the loss of biodiversity and for water and soil protection - farm scale Focus: Implementation of single measure
Finland / Mainland	Using simple agri-environment management measures to improve soil functionality and provide forage and feed resources for wildlife in Finland.	Biodiversity conservation; Soil functionality; Water management	Topic: Nature management fields Focus: Implementation of single measure
Latvia/ Levlīči	Restoring storm damaged forests using Rural Development funding in Latvia.	Objective: Biodiversity conservation; Other (long term sustainability and resilience of the forest ecosystem) Topic: Restoring a storm damaged forest Focus: Implementation of a single measure	

Figure 6. EAFRD promoted examples of land restauration through RDP.
Source: European Network for Rural Development. 2012.

In this sense, the mechanism of payment has presented some challenges to solve in their design. In the Veneto region there is an agri-environmental scheme focusing on conservation agriculture, where the main objective is soil protection. Important questions remain in terms of administration, a critical issue is delayed payments, and in some regions, payments are also considered too low to attract or retain the commitment of farmers. To overcome these challenges and improve the environmental impact of rural development, greater recognition must be given to the role of such services in providing public goods and farmers must be adequately rewarded for their contribution. Another aspect to improve in payment schemes is the monitoring of the environmental results. In Germany, the Contractual Nature Conservation technique uses targeted, site-specific contracts with land-users who receive agri-environmental payments. Each contract is designed to fit with the individual needs of each holding. Contracts are prepared through collaboration between the land-users and environmental experts. As with standard agri-environment operations, land management practices are agreed to support particular environmental services. In addition, quantifiable targets are agreed for outcomes to be achieved as a result of the land management practices.

The Commission states that the combination of RDP environmental services support is expected to create synergies and could involve a variety of integrated measures. These include measures to co-finance environmental works, training, advisory services, cooperation, innovation, and competiveness, as well as other rural development actions deemed relevant by individual Member States.

7.3. PES and rural development

Payments for environmental services have attracted increasing interest as a mechanism to translate external, non-market values of the environment into real financial incentives for local actors to provide environmental services. Latin América has been one of the first regions in the world in adopting PES schemes as an instrument for natural conservation but also to foster economic development in these areas. However, some concerns arise about the capacity of PES to deal with poverty. The PES approach was conceptualized and undertaken as a mechanism to improve the efficiency of natural resource management, and not as a mechanism for poverty reduction (Engel et al., 2008). Nevertheless, many have assumed that PES will contribute to poverty reduction by making payments to poor land users, while others have warned of potential dangers. Three key questions have been raised (Pagiola et al.; 2005): (1) Who are the actual and potential participants in PES programs, and how many of them are poor? (2) Are poorer households able to participate in PES programs? And (3) are poor households affected indirectly by PES programs? There is evidence on locations and situations where the poor are likely to benefit from PES (Bulte et al., 2008). However, the analyses also indicate that tying PES and poverty reduction may result in lower efficiency in meeting either objective — being better to focus programs on one or the other objective separately. For instance, the potential economic benefits of forests allocated to the voluntary carbon market are reported to be much lower than the estimated benefits from oil palm, shedding doubts about the competitiveness of the former (Muradian et al. 2013). If the price of commodities remains high, it is unlikely that PES will be able, by them, to stop the current expansion of the commodity production into natural ecosystems. PES could not compensate commodity production incomes and they will not be efficient for Nature conservation purposes. Moreover, when the recipient of payments is considerably wealthier than the local beneficiaries of ecosystem services, compensation raises important equity concerns.

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9. Study cases

9.1. Study case 1: Agri-environmental policy and local development: a case study of Ribeira Sacra in Galicia (Spain)

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La política agroambiental en el contexto del desarrollo local: la Ribeira Sacra en Galicia, un estudio de caso. Ager, Revista de estudios sobre despoblación y desarrollo rural = *Journal of depopulation and rural development studies*, nº 9, pp: 63-86.

Background and Aims

As pure Payments for Ecosystem Services are difficult to find in the European policies, certain schemes promoting ecosystem services have been developed since 1992 within the agri-environmental policy. In this situation the state itself is as buyer a market actor "acting on behalf of service buyers" (Engel et al., 2008; Matdorf *et al*., 2013). These programs provide payments to landowners who make a voluntary commitment to promote environmental objectives beyond the relevant legal mandatory requirements. Landowners, as farmers, get payments for more environmentally friendly land management practices. This was the case for the Galician scheme "Conservation of landscape and soil protection in vineyards of the Ribeira Sacra", that aimed preserving and recovering terraces landscape avoiding land degradation through agricultural practices and contributing, as well to the maintenance of local population who is a key factor for reducing the risk of fires. This scheme has been in place since 1996 until 2001when the first application period of agri-environmental policies finished.

One of the key characteristics of this measure is their local initiative origin linked to other measures aiming the recovery of landscape and vineyards culture like the constitution of Ribeira Sacra Designation of Origin (D.O.), that take place at the same time in a moment of risk of abandon by the population. In fact, the Designation of Origin Council did the promotion of the scheme among farmers and landowners and gives a technician for the monitoring of plots.

This study case reports results of a study carried out among wine producers in the region. We have studied the factors influencing farmers' willingness to adopt agri-environmental practices for landscape conservation, as well as the interactions with other policies aimed at

local economic promotion. Analysis from in-depth interviews and secondary data shows the importance of transaction costs for both the administration and the farmers. Therefore, we have identified synergies between simultaneous actions to promote agricultural production and landscape conservation.

The study area

The Ribeira Sacra (in Galicia-Northwest of Spain) is a singular area formed by the canyons that the rivers Miño and Sil conform when they confluence. The slopes are very steep (falling from until 500 meters) and the special climate conditions favor the culture of wine in terraces. The characteristic landscape has driven the development of the tourism sector at the same time that the recovering of vineyards and terraces.

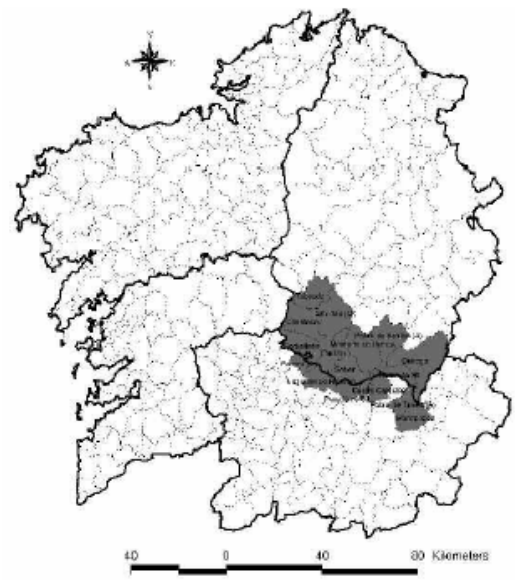

Figure 1. Study Area.

This area suffers from a deep demographic recession. Considering the previous period to the measure implementation, population drops by 26.3% between 1981-2001 for the whole area and by 37.4% excluding the most dynamic towns. As well, the population density was of 31.2 inhabitants per square kilometre in 2001. To this, we have to add a high medium age around 57 years old in 2001 and a high proportion of people over 64 years old (35.2%).

Winegrowers were 2,600 in 2001 and represented 60% of labour occupation in agriculture even if most of them grow grapes as a revenue complement, in a part-time way. The property is very small with an average of 0.43 ha per winegrower and a yield of 2,200 kg of grapes per ha.

Material and methods: To study the assessment and effects of the agri-environmental measure for the conservation of terraces on rural development, we use secondary data given by the administration and primary data coming from in-depth interviews with 10 winegrowers and two technicians from the D.O. Council, conducted in 2003 once the program was finished.

The in-depth interviews comprised five groups of questions about:

a) social-economic characteristics;
b) satisfaction with the program;
c) agri-environmental concerns of winegrowers;
d) winegrower expectations about the future;
e) financial concerns.

In this contribution we focus on results from a) and d) parts of the interview to winegrowers and on results from the technicians interview.

Results

We can say that for the period of application the reception of the measure among winegrowers was good since the contracted area represented in 1999 the 45% of D.O. members and the 28% of the D.O. area. Despite this, the average payment perceived by beneficiary was around 200 € which is clearly consistent with the small dimension of plots hold by producers and could be considered as very limited. But this amount was 8.4% of the local average income generated in the gross operating surplus / mixed income item.

Table 1. *Program Scope*

Year	1997	1998	1999	2000	2001
Nº of applicants	591	667	844	1023	668
Nº winegrowers inside Designation of Origin	1433	1500	1600	2500	2600
Area under contract (ha) per beneficiary	0.51	0.52	0.48	0.49	0.52

Source: Galician Administration and Designation of Origin Council.

The socio-economic characteristics of the beneficiaries are quite similar with the whole group of D.O. members but certain differences must be mentioned: growers under agri-environmental contract are younger (50 years old) and with bigger holdings (2.1 ha on average). Most of them are part-time winegrowers. In our case study 6 beneficiaries had their main labour activity in other sectors than agriculture.

Furthermore, the commitments for growers under contract consisted in carrying out the necessary works for the maintenance of walls and terraces using original materials, perform traditional cultivation practices using mainly organic fertilizers (eliminating mineral nitrogen fertilization) and manual digging, perform a maximum of one herbicide treatment using non-residual products, and cultivate only authorized or recommended vine castes so that the percentage of total area included in the program occupied by castes is greater than 60%.

These practices were not different from traditional ones, what means that financial amounts were perceived as payments for the ecosystem services delivered.

We have to consider these payments as part of a strategy to develop the wine sector in this territory as well as the tourism sector linked to the wine culture. The D.O. Council was the responsible of the spread of the measure among stakeholders since the 90% of interviewed owners declared to know the program by the D.O. As the technicians of the D.O. Council have said, here the production of wine depends on the conservation of soil through terraces, shaping a characteristic landscape.

In 2003 70% of interviewed owners declared payments insufficient. However, technicians agree that these payments helped to the involvement of growers in the improvement of the vineyards and their quality as the same time that the approval of the Designation of Origin label, other public programs for improving the wine characteristics, and the rise of the grapes price. Technicians also said that all these actions created among producers a feeling of territorial belonging where the landscape was a key factor. In this sense, 6 of interviewed persons declare to be optimists about future of vineyards thanks to the evolution of the wine prestige and high prices and despite the aged population and the depopulation of the area. As well, 7 owners declared to have done investments in order to improve the production of their holdings.

Discussion and Conclusions

Agri-environmental schemes are a key mechanism for supporting a wide range of environmental services from farmland (Westerink et al., 2017) and they are likely to be more effective if they are designed at the landscape scale. It has been said that this requires coordination of environmental management across multiple farm holdings and collaboration among governmental and other actors, including, groups of farmers. The present contribution shows a specific example of an environmental service, landscape, ensured by land rehabilitation, which was the key driver of an economic development process with the participation of the administration and groups of farmers collaborating at a territorial level.

We have seen like the rehabilitation of terraces and the vineyards landscape has contributed to a process of economic and social dynamism in a territory confronting serious risk of abandon. The searching of synergies with other policy mechanisms and across activities like wine production and tourism plus the collaboration of winegrower's groups have been of a key importance (van der Ploeg et al., 2000) for the effectiveness of this process. Nowadays, the disposable income per capita has grown by 48.2% between 1996 and 2009 in real terms. In addition, the number of general business has grown by 21.5% between 1999 and 2009 meanwhile the number of business related to tourism and beverage production grew by 17%. The payments offered by the agri-environmental scheme could be reedited under the new CAP for this territory. To improve the environmental impact of rural development, greater recognition must be given to the role of environmental services in providing public goods and farmers must be adequately rewarded for their contribution.

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9.2. Study case 2: Terraced landscape protection and local economic development: the study-case of Cinque Terre National Park, Italy

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Background and Aims

It has been said that terraces and dry-stone walls are among the most prominent human signatures on the landscape. They cover large areas in the world, for instance, areas in Nepal, China, Peru, Africa and several Mediterranean countries where they symbolize an important cultural heritage and still now are known to be a local identity milestone. Basically, they represent an ancient practice of steep hillslope cultivation playing a strategic role on water and soil conservation. Dry-stone terraces consist of a series of nearly levelled platforms built along contour lines at suitable intervals. During past centuries, the need of cultivable and well-exposed areas led man to shape the natural landscape creating an extensive anthropogenic terracing of hillslopes.

Especially today, the value protection of terraced landscape arouses a lot of interest: for this, several projects and scientific associations, such as the "International Terraced Landscapes Alliance", arise everywhere with the main goal to disseminate the relevant roles of terracing.

The study area

In Italy, terracing has been practicing since the Neolithic and it is well documented from Middle Age onward. Liguria is among the Italian Regions with the higher percentage of man-building terraces with on-site stones. Liguria Region is a narrow coastal strip located between the Ligurian Sea and the Alps and Apennines, hence its extraordinary high percentage of mountain (65.1%) and hill (34.9%), with hard slopes almost everywhere. In this context, terraces reduce the slope gradient and they represent for centuries an agricultural source for local people's survival. Cinque Terre area, in the eastern coast of Liguria, comprises five villages Monterosso al Mare, Vernazza, Corniglia, Manarola and Riomaggiore and it represents the most extraordinary example of 'heroic' terracing; it is the place where grapes and olives have been growing for thousands years. Here, the linear development of dry-stone wall is over 6000 km considering an area of ca. 2000 ha.

Since the early 1900s, but especially over the last 50 years, the terracing cultivation in the Cinque Terre area decreased and the original 1400 ha of cultivated terraces nowadays dropped down to ca 100 ha, 88 of which are for the production of Origin Protected Denomination wine.

The abandonment of rural activity is due to marginalization and to socio-economic reasons with a negative impact not only on the landscape, but also on geomorphological stability of the territory. In fact, here the rural abandonment and the miss-maintenance of dry-stone wall are the main factors of landslides and land degradation.

Figure 1. The location of the study area: A = Italy, B = Liguria Region, C = Cinque Terre National Park.

Figure 2. Panoramic view on the Cinque Terre.

A high level of hydrogeological vulnerability characterizes the Cinque Terre territory because of both geomorphological features (high slope) and urbanization reasons, which have changed the natural water systems and drainage. In fact, the rivers with a typical torrential regimen were "closed" under the roads often to create parking areas. Actually, many hydrogeological problems are due to mismanagement of water networks that gave rise to the catastrophic flood occurred on the 25 October 2011. In addition to the land abandonment, wild boar represents a further degradation factor because of its recent population increase, which has a negative impact both on agroforestry and on dry-stone walls.

In 1997 UNESCO inscribed the Cinque Terre area in the World Heritage List considering that *is a cultural site of outstanding value, representing the harmonious interaction between people and nature to produce a landscape of exceptional scenic quality that illustrates a traditional way of life that has existed for thousand years and continues to play an important socio-economic role in the life of the community*. After that, the Italian Government established the "Cinque Terre National Park" with the goal to protect both the natural and the cultural value; concurrently, tourism begins to spread and recently reached 2,5 million tourists a year, becoming an outstanding business sector.

The pie chart (Figure 3) shows that tourism in the Cinque Terre is the local driving economy and about 50% of the companies belong directly to the tourism area or to the accommodation and food, but also other activities are in some way connected to this sector.

Despite tourism explosion, rural abandonment continues and from 2005 to 2015 the local population, especially young people, decreased by 10% ca. Rural abandonment process has many economic effects, regarding not only the agricultural production, but it is also concerns the territory maintenance.

Cinque Terre landscape is a popular attraction because many famous hiking trails run on its terraces. Therefore, terracing restoration is so important and encouraging farmers work is the best way to reach the goal: cultivated terraces are crucial to reduce hydrogeological risk and they are useful in the wildfire stoppage.

Programme description

Cinque Terre rural importance is well recognized and land managers and owners have already started projects of land protection and terracing rehabilitation although it is not easy cause of severe slopes, low accessibility, difficulties on finding both suitable stones on-site and specialized works able to rebuilt the damaged dry-stone walls.

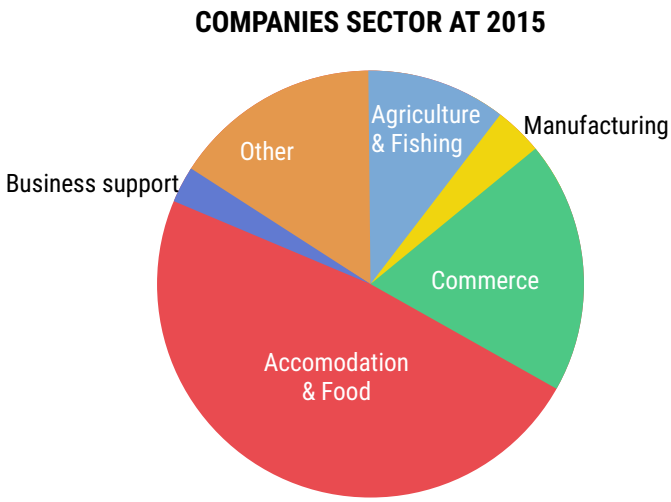

Figure 3. Economic sector in the Cinque Terre National Park (from Camera di Commercio della Spezia).

Generally speaking, the public cost-benefit ratio of terracing restoration is in favor of the benefits because it guarantees several ecosystem services, such as the mountain slope stability, that are not properly monetized, but their real value is highlighted when natural disasters occur (e.g. floods, landslides, hydrogeological instability, groundwater pollution, biodiversity loss…).

Figure 4. Dry-stone wall restoration in Cinque Terre vineyards.

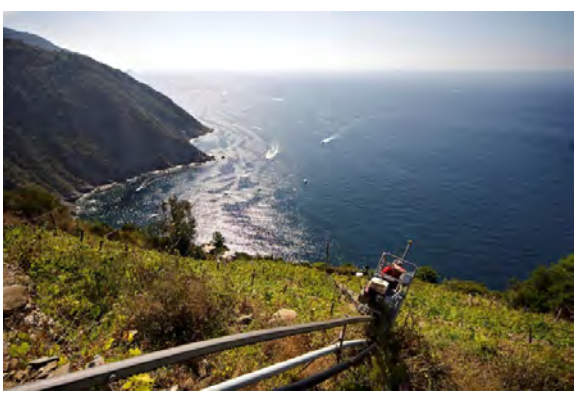

Figure 5. Monorails systems in Cinque Terre vineyards.

Two are the key factors of land rehabilitation processes: on one hand, direct public funding such as the UE Rural Development Programmes, the specific Life Projects (e.g. Life 00 ENV/IT/000191 PROSIT) and the economic support from the National Park such for terracing restoration and for vineyards replanting; on the other hand, the improvement of local economy due to tourism increase that supports local production.

Results

To avoid land abandonment, the National Park has installed a monorail system in order to encourage the wine-growers to reach their steep terraced areas, the Park Administration develops specific training course to have proper workers able to rebuild the ruined dry-stone wall and it gives for free wooden poles and vine plants to those who decide to cultivate new terraces.

Furthermore, terraces have important environmental functions: not only in the mitigation of hydrogeological and land erosion issues, but also terracing systems represent micro-ecosystem that houses many synanthropic flora and fauna species. Definitely, terraces restoration produces tree positive externalities in favor of agriculture, environment and tourism, which overall help in spreading of the Green Economy (Figure 6).

Figure 6. *How terracing contributes to the Green Economy development.*

Acknowledgements

We would like to thank all the Cinque Terre National Park staff for the kind support and for the photos.

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9.3. Study case 3: The Rural Development Programme 2014-2020 for Mainland Portugal: The case of land conservation measures

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Background and Aims

Nowadays the European rural development policy is funded through the European Agricultural Fund for Rural Development (EAFRD). Since 2007 each Member State receives a financial allocation for 7-year period, and they must draw up their Rural Development Programmes (RDP). RDPs are programming documents designed by the Member States according with what is established in the R(EU)1305/2013 and subsequently they are adopted by the European Commission.

Programme Description

These Programmes shall contribute to achieving three objectives: (a) fostering the competitiveness of agriculture; (b) ensuring the sustainable management of natural resources, and climate action; (c) achieving a balanced territorial development of rural economies and communities including the creation and maintenance of employment. RDP should be based on the regional resources and needs, but the EU establishes six common priorities (UE, 2013), two of them related with sustainable development and environmental services:

- Priority 4 - Restoring, preserving and enhancing ecosystems related to agriculture and forestry.
- Priority 5 - Promoting resource efficiency and supporting the shift toward a low-carbon and climate-resilient economy in the agriculture, food and forestry sectors.

It is important to note that at least 30% of the total EAFRD contribution shall be reserved for measures relevant to these priorities. The rural development priorities are broken down into focus areas, the Member States determine the quantified target within each focus area, and finally they set out the measures that will use to achieve the selected targets and how much funding they allocate to each measure. The fourth focus area of Priority 4 is directly linked with preventing soil erosion and improving soil management.

Results

This case study analyses the Rural Development Programme for Mainland Portugal 2014-2020 from the point of view of the integration of land conservation measures. The Programme was adopted by the European Commission on December 12, 2014 and it was divided in four intervention areas, under which are grouped 10 Measures and 62 Actions. This planning tool allows the execution of the European and National funds available for the 7 year period 2014-2020. The budgetary allocation for this programme is € 4.1 billion EUR of public funds (€ 3.5 billion from the EU budget and € 0.59 billion of national co-funding). In relation with the public budget it should be noted that the funds for the programme have been reduced by 2.5 percent in current euros in relation de 2007-2013 period.

The Table 1 shows the Measures included in the RDP for mainland Portugal and the distribution of funds between Measures. Measures 7, 8 e 9 are included in an intervention area called; Environmental, Resource Efficiency and Climate. These Measures are related to environmental objectives; therefore, they contribute to the 4 and 5 priorities[2]. The financial amount perceived by this area was 47.4% of Total Public Spending. But we have to underline that if we exclude the Measure 9 amount (Agricultural Activity of Maintenance in less Favored Areas) – a direct payment for carrying on agricultural practices in less favored areas – the percentage decreases sharply falling to 28.2%.

€ Millions	Public Funds	EAFRD
Measure 1 - Innovation	42,1	37,6
Measure 2 - Knowledge	52,9	45,2
Measure 3 - Agricultural Production Value	1.658,9	1.408,6
Measure 4 - Development of Forest Resources	37,7	31,3
Measure 5 - Production Organization	24,6	22,0
Measure 6 - Management of Risk and Restoring the Potential Productivity	75,0	62,5
Measure 7 - Agriculture and Natural Resources	671,5	577,4
Measure 8 - Protection and Rehabilitation Of Forest Stands	506,2	425,7
Measure 9 - Agricultural Activity of Maintenance in less Favored Areas	801,9	705,2
Measure 10 - Leader	220,6	197,7
Technical Assistance	82,3	70,0
Total RDP	4.173,7	3.583,2

***Table 1**. Public funds by Measure*
Source: PRD 2014-2020 Portugal Continental. Gabinete de Planeamento, Políticas e Administração Geral (2014).

2 The established priorities contribution for P4 and P5 is higher given that other measures included in the RDP indirectly contribute to both priorities.

Measures 7 and 8 are broken down into 14 different actions: 12 actions for Measure 7 and 2 for Measure 8. They include those aimed to the fostering of organic farming, integrated production, Natura 2000 directive payments, efficient use of water, traditional Permanent Crops, Grazing Extensive, Genetic Resources Agroforestry, Forest-Environment, Beekeeping, preserving and enhancing ecosystems related to traditional agriculture, Sustainable Forestry Resources Management Cynegetic And Aquaculture and Land Conservation as well.

Increase the productivity of land, including improvements in soil fertility is one of the objectives defined by the RDP and the correspondent measure is Action 7.4 Soil conservation. This action is described at the national regulation R1305/2013, article 28. It aims to contribute for obtaining direct environmental benefits from soil through the adoption of beneficial practices for its conservation that can decrease erosion, improve soil structure and increase organic material content. It is a voluntary scheme for private or public organizations involved in agricultural production, based in a 5 years contract extended to 7 years under decision of the Managing Authority.

This scheme is not new in the Portuguese RDP. In the last period of programming, 2007-2013, the measure 2.2.4 Land Conservation had similar objectives. The eligibility criteria were stricter, but the budget involved had a sensible lesser amount: €3.6 Million. This is a question that had to be underlined given the general context of budget reduction at the current period: 2.5% of total public expense in current euros.

Finally, before the describing of new schemes, we have to say that in the last period the Land Consevation scheme reached the 100% of execution, even with more beneficiaries than initially projected (103%). In any case, in the Ex-post Evaluation (Domingos et al, 2016) a certain geographical concentration has been detected: 87% of beneficiaries and 92% of the covered area were at the Alentejo Region.

In the period 2014-2020 the total of provided funds for this scheme increases till €12 Million. The implementation criteria are lesser stricter and two types of actions were defined: those addressed to permanent cultures and those for non-permanent cultures:

- 7.4.1 - Soil conservation for non-permanent cultures: €6.3 Million for farmers practicing direct-seeding or tillage in stripes in a minimum of 3 ha of arable land. Measuring the organic content of soil is compulsory. Payments are defined between €145 and 35€/ha depending on the practices adopted and the land area under commitment.
- 7.4.2 - Soil conservation for permanent cultures: €5.7 Million, for farmers practicing alley soil management in a minimum of 0.5 ha. Measuring the organic content of soil is still compulsory as well as the maintenance of natural vegetal coverture of soil. Minimum tillage techniques are required. In this case payments are defined between €100 and 13€/ha.

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9.4. Study case 4: Rural development and sustainable agriculture in Thessaly Plain (Greece). Study of nitrate levels on agricultural soils included in the project "European Economic Community - Action Program Against Nitrate Pollution of Agricultural Origin"

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Background and Aims

Greece covers an area of 13.196.887 ha, of which 97.1% are designated as rural areas (73.9% mainly rural and 23.2% intermediate rural) according to the OECD (Organization for Economic Co-operation and Development - OECD) criteria, in which approximately the 2/3 of the country's total population inhabit (37.2% mainly rural and 27.2% intermediate rural).

The agricultural sector has always been a structural feature of Greek society as well as a part of the Greek economy and development, both at national and regional level. In addition, the accession of Greece to the European Union (EU) in 1981 had a catalytic effect on the subsequent development of Greek agriculture. Nowadays, agriculture is fully governed by the rules of the Common Agricultural Policy (CAP), which form and control the entire framework and operation of the agricultural sector.

In Greece the state is required to develop the agricultural sector through the training and guidance of farmers, the design and the implementation of development programs. This promotes the increase and improvement of production, the modernization of farm facilities and the creation of viable agricultural holdings with more modern aims, such as improving food quality and safety, protecting the environment and the landscape and improving the quality of life in rural areas.

The study area

In this case study will present the methods that used at national level and specific at Thessalian plain (*Figure* 1), related to rural development through land restoration, to deal with the environment pollution caused by the use of large amounts of nitrate and generally nitrogen-containing fertilizers.

As is well known, the use of large amounts of nitrate and generally nitrogen-containing fertilizers has accumulated large amounts of nitrate in water, soils and plants over the past decades, varying the nitrate pollution problem from local to international; resulting in human health, damage to living organisms and aquatic ecosystems or damage to leisure facilities or other legitimate uses of water (Liacata, 2006). In particular, the problem of groundwater nitrate pollution is a global problem.

Programme description

The control of nitrate pollution on the rural fields of Thessaly plain, according to Council Regulation (European Economic Community - EEC) No 2078/92, started in 1996 and is scheduled to continue until 2020.

Figure 1. The area of Thessaly in Greece.

Figure 2. Areas of action in Greece that trying to reduce water pollution from agricultural activity. Green area: Nitrate vulnerable zones / Blue area: Other important wetland areas.

The Region of Thessaly occupies an area of approximately 14000 km^2, ie 10.6% of the Greek territory. From this, the cultivated land accounts for 36.1% ~ 2 million ha, of which the 1.01 million ha are irrigated (Karyotis et al, 2002). Thus, 30% are irrigated by surface waters and 70% are irrigated by underground waters of Thessaly (Legos, 2009). However, the intensity of the agricultural exploitation —coupled with the lack of a rational water resource management system— led to the over-pumping of groundwater, as a result the fall of the groundwater level. At the same time, nitrogen fertilizers have caused degradation of groundwater. The above's led to the adoption of an action plan which predicted the reduction in the dispersal of nitrogen fertilizers.

This reduction can be achieved by a number of measures, including:

- Improving the efficiency of nitrogen use.
- Introduction of new irrigation techniques.
- Cultivation of appropriate varieties.

It is well known that Council Directive 91/676/EEC of 12 December 1991, requires Member -States to identify waters that are been inflected at nitrogen fertilizers and to designate vulnerable zones. The Council aims at reducing the water pollution caused directly or indirectly by nitrates of agricultural origin and to prevent further pollution of this kind.

In order to achieve the above objectives, Greece has established (eg. at Thessaly) the Codes of Good Agricultural Practice (COGAP), which, in connection with the subsidized program "Action Program Against Nitrate Pollution of Agricultural Origin" are aimed to sustainable management of agricultural land and natural resources. Main concerns are to protect and preserve the rural landscape and its characteristics and to protect the health of farmers and consumers (*Figure* 2).

In order to achieve the above objectives, the following agricultural activities shall be implemented:

- Soil treatment. The soil treatment is the preparation of the field for the next crop.
- Crop rotation. Crop rotation is the practice of growing a series of dissimilar or different types of crops in the same area in sequenced seasons.
- Fertilizing of the rural areas. It is necessary the fertilizer being suitable for the soil and the crop, to check the quantities added to the soil each time, as well as the way and the time of their application.
- Water resources management.
- Plant protection. The use of plant protection products must be justified by the existence of the disease, the extent of the infestation or the presence of weeds.

The initial program was adopted for the period 1995-1999. The implementation of this period was based on the obligation of beneficiaries to reduce the quantities of nitrogen (N) applied to COGAP, while at the same time achieving specific objectives (protection of slopes, reduction of use of irrigation water) by using other methods, such as sequential culturing at autumn legumes on sloping land, rotational set-aside of 30% or 50% of the eligible area, etc. Equally same programs were implemented, every 6 years from 2000 to 2020.

For the period 2014-2020 more than 4.190 beneficiaries with a total area of 857,500 hectares will be included in the Denitrification Measure with a total budget of 260 million euro. The intervention areas of the action have now been expanded (from seven in the period 2007-2013) to thirty-seven, covering an area of 30% of the country's total area, increasing the amount of financial assistance to 260 million euro in the current period. The selection of the areas was based on the poor chemical condition of the groundwaters, compared to the Natura areas, since the primary objective of the action is to protect the water quality.

Corn	Sugar beet	Vegetables	Cotton	Heliotrope
30.8	35.8	60	51.5	31.6

Table 1. *Amount of financial support per product at Thessaly, €/acre*

Results: The results of the primal project on nitrate pollution in Thessaly, which is part of the National Action Plan, suggest that farmers have begun to change their attitude towards crop fertilization, focusing on a more rational and more scientific approach. So far they have found that the reduction in nitrogen quantities does not necessarily entail a corresponding loss of production, especially for cotton.

Over the period 1996-2000, more than 3,200 farmers and 5000 ha land was participated in the project. It was estimated that a reduction in the nitrogen fertilizers used was recorded by about 10 ton for the region of Thessaly (eg-30% for cotton, from 140 to 100 kg N / ha or - 25% for tomatoes from 270 to 200 kg N / ha) (Karyotis et al, 2002). At that period the project was based on the obligation of the beneficiaries, to:

- Significantly reduce (20%) the applied nitrogen amounts per crop and soil class in relation to COGAP,
- Reduce the consumption of irrigation water (35%) in the groundwater depletion areas (drip irrigation, dry crops)
- Take measures to control erosion in sloping areas (drip irrigation).
- Apply a fixed uncultivated margin equal to 3% of the area included in the program.

Those results encourage farmers to adopt those new practices as it reduces the dispersal of nitrogen fertilizer and restoring the provision of the underground water ecosystem.

Discussion and Conclusions: The program will be applied to a total area of 242.811 ha, covering the most important crops of the plain. The program requires a 50% reduction in the applied N, without this being accompanied by a reduction in production and incomes, in order to be socially acceptable. The neutral impact on production is achieved through the recycling and use of irrigation water, nitrate-loaded liquid fertilizer and the improvement of fertilizer application methods aimed at limiting nitrogen losses and consequently increasing the fertilizer utilization rate. The action program does not eliminate the nitrogen flush, but the degree of nitrogen recovery from the polluted groundwater is significantly greater than the degree of nitrogen flushing. This situation requires the continuation of the program for the Nitrates Reduction, in a much more targeted form as regards nitrogen management as well as the extension of the Program to other actions related to the conservation of the natural environment and the reduction of irrigation water consumption.

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Deben incluírse ata 5 palabras clave situadas despois de cada resumo distintas das incluídas no título.

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A estructura do artigo debe axustarse na medida do posíbel á seguinte distribución de apartados: Introdución, Material e métodos, Resultados e discusión, Agradecementos e Bibliografía.

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Grave o seu arquivo de texto nun formato compatíbel con Microsoft Word.

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Cada táboa e figura gardarase nun arquivo distinto co número da táboa e/ou figura. Os formatos preferidos para os gráficos son: Para os vectores, formato EPS, exportados desde o programa de debuxo empregado (en todo caso, incluirán unha cabeceira da figura en formato TIFF) e para as ilustracións en tons de grises ou fotografías, formato TIFF, sen comprimir cunha resolución mínima de 300 ppp. En caso de enviar os gráficos nos seus arquivos orixinais (Excel, Corel Draw, Adobe Ilustrator, etc.) estes acompañaranse das fontes utilizadas. O nome do arquivo da figura (un arquivo diferente por cada figura) incluirá o número da ilustración. En ningún caso se incluirá no arquivo da táboa ou figura a lenda, que debe figurar correctamente identificada ao final do texto. O material gráfico escaneado deberá aterse aos seguintes parámetros: Debuxos de liñas: o escaneado realizarase en liña ou mapa de bits (nunca escala de grises) cunha resolución mínima de 800 ppp e recomendada de entre 1200 e 1600 ppp. Figuras de medios tons e fotografías: escanearanse en escala de grises cunha resolución mínima de 300 ppp e recomendada entre 600 e 1200 ppp.

Recepción do manuscrito

Os autores enviarán un orixinal e dúas copias do artigo completo ao comité editorial, xunto cunha copia dixital, acompañados dunha carta de presentación na que ademais dos datos do autor, figuren a súa dirección de correo electrónico e o seu número de fax, á seguinte dirección:

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Enviar o texto e cada unha das ilustracións en arquivos diferentes, nalgún dos seguintes soportes: CD-ROM ou DVD para Windows, que irán convenientemente rotulados indicando o seu contido. Os nomes dos arquivos non superarán os 8 caracteres e non incluirán acentos ou caracteres especiais. O arquivo de texto denominarase polo nome do autor.

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Cos arquivos inclúa sempre información sobre o sistema operativo, o procesador de texto, así como sobre os programas de debuxo empregados nas figuras.

Copyright

Unha vez aceptado o artigo para a publicación na revista, o autor(es) debe asinar o copyright correspondente.

Decembro 2015

Recursos Rurais

Revista do Instituto de Biodiversidade Agraria e Desenvolvemento Rural (IBADER)

Proceso de selección y evaluación de originales
Recursos Rurais publica artículos, revisiones, notas de investigación y reseñas bibliográficas. Los artículos, revisiones y notas deben ser originales, siendo evaluados previamente por el Comité Editorial y el Comité Científico Asesor. Los trabajos presentados a Recursos Rurais serán sometidos a la evaluación confidencial de dos expertos ajenos al equipo editorial, siguiendo criterios internacionales. En el caso de que los evaluadores propongan modificaciones en la redacción del original, será responsabilidad del equipo editorial —una vez informado el autor— el seguimiento del proceso de reelaboración del trabajo. Caso de no ser aceptado para su edición, el original será devuelto a su autor, junto con los dictamenes emitidos por los evaluadores. En cualquier caso, los originales que no se ajusten a las siguientes normas técnicas serán devueltos a sus autores para su corrección, antes de su envío a los evaluadores.

NORMAS PARA LA PRESENTACIÓN DE ORIGINALES

Procedimiento editorial
La Revista Recursos Rurais aceptará para a su revisión artículos, revisiones y notas vinculados a la investigación y desenvolvimiento tecnológico en el ámbito de la conservación y gestión de la biodiversidad y del medio ambiente, de los sistemas de produción agrícola, ganadera, forestal y referidos a la planificación del territorio, tendientes a propiciar el desarrollo sostenible de los recursos naturales del espacio rural y de las áreas protegidas.Los artículos que no se ajusten a las normas de la revista, serán devueltos a sus autores.

Preparación del manuscrito

Comentarios generales
Los manuscritos no deben exceder de 20 páginas impresas en tamaño A4, incluyendo figuras, tablas, ilustraciones i la lista de referencias. Todas las paginas deberán ir numeradas, aunque en el texto no se incluirán referencias al número de pagina. Los artículos pueden presentarse en los siguientes idiomas: galego, castellano, portugués, francés o inglés. Los originales deben prepararse en un procesador compatible con Microsoft Word®, a doble espacio en una cara y con 2,5 cm de margen. Se empleará la fuente tipográfica "arial" a tamaño 11 y no se incluirán tabulaciones ni sangrías, tanto en el texto como en la lista de referencias bibliográficas. Los parágrafos no deben ir separados por espacios. No se admitirán notas al pie. Los nomes de géneros ye especies deben escribirse en cursiva y no abreviados la primera vez que se mencionen. Posteriormente el epíteto genérico podrá abreviarse a una sola letra. Debe utilizarse el Sistema Internacional (SI) de unidades. Para el uso correcto de los símbolos y observaciones más comunes puede consultarse la última edición do CBE (Council of Biology Editors) Style manual.

Página de título
La pagina de título incluirá un título conciso e informativo (en la lengua original y en inglés), el nombre(s) de los autor(es), la afiliación(s) y la dirección(s) de los autor(es), así como la dirección de correo electrónico, número de teléfono y de fax del autor con que se mantendrá la comunicación.

Resumen
Cada articulo debe estar precedido por un resumen que presente los principales resultados y las concusiones más importantes, con una extensión máxima de 200 palabras. Ademas del idioma original en el que se escriba el articulo, se presentará también un resumen en inglés.

Palabras clave
Deben incluirse hasta 5 palabras clave situadas después de cada resumen, distintas de las incluidas en el título.

Organización del texto
La estructura del articulo debe ajustarse en la medida de lo posible a la siguiente distribución de apartados: Introducción, Material y métodos, Resultados y discusión, Agradecimientos y Bibliografía. Los apartados irán resaltados en negrita y tamaño de letra 12. Si se necesita la inclusión de subapartados estos no estarán numerados y se tipografiaran en tamaño de letra 11.

Introducción
La introdución debe indicar el propósito de la investigación y proveer una revisión corta de la literatura pertinente.

Material y métodos
Este apartado debe ser breve, pero proporcionar suficiente información como para poder reproducir el trabajo experimental o entender la metodología empleada en el trabajo.

Resultados y discusión
En este apartado se expondrán los resultados obtenidos. Los datos deben presentarse tan claros y concisos como sea posible, si es apropiado en forma de tablas o de figuras, aunque las tablas muy grandes deben evitarse. Los datos no deben repetirse en tablas y figuras. La discusión debe consistir en la interpretación de los resultados y de su significación en relación al trabajo de otros autores. Puede incluirse una conclusión corta, en el caso de que los resultados y la discusión lo propicien.

Agradecimientos
Deben ser tan breves como sea posible. Cualquier concesión que requiera el agradecimiento debe ser mencionada. Los nomes de organizaciones financiadoras deben escribirse de forma completa.

Bibliografía
La lista de referencias debe incluir unicamente los trabajos que se citan en el texto y que estén publicados o que hayan sido aceptados para su publicación. Las comunicaciones personales deben mencionarse solamente en el texto. En el texto, las referencias deben citarse por el autor y el año y enumerar en orden alfabética en la lista de referencias bibliográficas.

Exemplos de citación en el texto:
Descripciones similares se dan en otros trabajos (Fernández 2005a, b; Rodrigo et al. 1992).
Andrade (1949) indica como...
Según Mario & Tineti (1989) los factores principales están...
Moore et al. (1991) sugieren eso...

Exemplos de lista de referencias bibliográficas:

Artículo de revista:
Mahaney, W.M.M., Wardrop, D.H. & Brooks, P. (2005). Impacts of sedimentation and nitrogen enrichment on wetland plant community development. Plant Ecology. 175, 2: 227-243.

Capítulo en un libro:
Campbell, J.G. (1981). The use of Landsat MSS data for ecological mapping. En: Campbell J.G. (Ed.) Matching Remote Sensing Technologies and Their Applications. Remote Sensing Society. London.
Lowel, E.M. & Nelson, J. (2003). Structure and morphology of Grasses. En: R.F. Barnes et al. (Eds.). Forrages. An introduction to grassland agriculture. Iowa State University Press. Vol. 1. 25-50

Libro completo:
Jensen, W (1996). Remote Sensing of the Environment: An Earth Resource Perspective. Prentice-Hall, Inc. Saddle River, New Jersey.

Una serie estándar:
Tutin, T.G. et al. (1964-80). Flora Europaea, Vol. 1 (1964); Vol. 2 (1968); Vol. 3 (1972); Vol. 4 (1976); Vol. 5 (1980). Cambridge University Press, Cambridge.

Obra institucional:
MAPYA (2000). Anuario de estadística agraria. Servicio de Publicaciones del MAPYA (Ministerio de Agricultura, Pesca y Alimentación), Madrid, España.

Documentos legales:
BOE (2004). Real Decreto 1310/2004, de 15 de enero, que modifica la Ley de aprovechamiento de residuos ganaderos. BOE (Boletín Oficial del Estado), nº 8, 15/1/04. Madrid, España.

Publicaciones electrónicas:
Collins, D.C. (2005). Scientific style and format. Disponible en: http://www.councilscience.org/publications.cfm [5 enero, 2005]

Los artículos que fuesen aceptados para su publicación se incluirán en la lista de referencias bibliográficas con el nombre de la revista y el epíteto "en prensa" en lugar del año de publicación.

Ilustraciones y tablas
Todas las figuras (fotografías, gráficos o diagramas) y las tablas deben citarse en el texto, y cada una deberá ir numerada consecutivamente. Las figuras y tablas deben incluirse al final del artículo, cada una en una hoja separada en la que se indicará el número de tabla o figura, para su identificación. Para el envío de figuras en forma electrónica vea más adelante. Dibujos lineales. Por favor envíe impresiones de buena calidad. Las inscripciones deben ser claramente legibles. El mínimo grosor de línea será de 0,2 mm en relación con el tamaño final. En el caso de Ilustraciones en tonos medios (escala de grises): Envíe por favor las impresones bien contrastadas. La ampliación se debe indicar mediante barras de escala. Se aceptan figuras en color.

Tamaño de las figuras
Las figuras deben ajustarse a la anchura de la columna (8.5 centímetros) o tener 17.5 centímetros de ancho. La longitud máxima es de 23 centímetros. Diseñe sus ilustraciones pensando en el tamaño final, procurando no dejar grandes espacios en blanco. Todas las tablas y figuras deberán ir acompañadas de una leyenda. Las leyendas deben consistir en explicaciones breves, suficientes para la comprensión de las ilustraciones por si mismas. En las mismas se incluirá una explicación de cada una de las abreviaturas incluidas en la figura o tabla. Las leyendas se deben incluir al final del texto, tras las referencias bibliográficas y deben estar identificadas (ej: Tabla 1 Características...). Los mapas incluirán siempre el Norte, la latitud y la longitud.

Preparación del manuscrito para su envío

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Grave su archivo de texto en un formato compatible con Microsoft Word.

Tablas y figuras
Cada tabla y figura se guardará en un archivo distinto con número da tabla y/o figura. Los formatos preferidos para los gráficos son: Para los vectores, formato EPS, exportados desde el programa de dibujo empleado (en todo caso, incluirán una cabecera de la figura en formato TIFF) y para las ilustraciones en tonos de grises o fotografías, formato TIFF, sin comprimir con una resolución mínima de 300 ppp. En caso de enviar los gráficos en sus archivos originales (Excel, Corel Draw, Adobe Ilustrator, etc.) estos se acompañaran de las fuentes utilizadas. El nombre de archivo de la figura (un archivo diferente por cada figura) incluirá el número de la ilustración. En ningún caso se incluirá en el archivo de la tabla o figura la leyenda, que debe figurar correctamente identificada al final del texto. El material gráfico escaneado deberá atenerse a los siguientes parámetros: Dibujos de lineas: el escaneado se realizará en linea o mapa de bits (nunca escala de grises) con una resolución mínima de 800 ppp y recomendada de entre 1200 y 1600 ppp. Figuras de medios tonos y fotografías: se escanearan en escala de grises con una resolución mínima de 300 ppp y recomendada entre 600 y 1200 ppp.

Recepción del manuscrito
Los autores enviarán un original y dos copias del artículo completo al comité editorial junto con una copia digital, acompañados de una carta de presentación en la que ademas de los datos del autor, figuren su dirección de correo electrónico y su número de fax, a la siguiente dirección:

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Con los archivos incluya siempre información sobre el sistema operativo, el procesador de texto, así como sobre los programas de dibujo empleados en las figuras.

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Una vez aceptado el artículo para su publicación en la revista, el autor(es) debe firmar el copyright correspondiente.

Diciembre 2015

Recursos Rurais

Revista do Instituto de Biodiversidade Agraria e Desenvolvemento Rural (IBADER)

Selection process and manuscript evaluation

The articles, reviews and notes must be original, and will be previously evaluated by the Editorial Board and the Scientific Advisory Committee. Manuscripts submitted to Recursos Rurais will be subject to confidential review by two experts appointed by the Editorial Committee, which may also consider choosing reviewers suggested by the author. In cases of dispute the intervention of a third evaluator will be required. Finally it is for the Editorial Committee's decision on acceptance of work. In cases in which the reviewers suggest modifications to the submitted text, it will be the responsibility of the Editorial Team to inform the authors of the suggested modifications and to oversee the revision process. In cases in which the submitted manuscript is not accepted for publication, it will be returned to the authors together with the reviewers'comments. Please note that any manuscript that does not adhere strictly to the instructions detailed in what follows will be returned to the authors for correction before being sent out for review.

INSTRUCTIONS TO AUTHORS

Editorial procedure

Recursos Rurais will consider for publication original research articles, notes and reviews relating to research and technological developments in the area of sustainable development of natural resources in the rural and conservation areas contexts, in the fields of conservation, biodiversity and environmental management, management of agricultural, livestock and forestry production systems, and land-use planning.

Manuscript preparation

General remarks

Articles may be submitted in Galician, Spanish, Portuguese, French or English.

Manuscripts should be typed on A4 paper, and should not exceed 15 pages including tables, figures and the references list. All pages should be numbered (though references to page numbers should not be included in the text). The manuscript should be written with Microsoft Word or a Word-compatible program, on one side of each sheet, with double line-spacing, 2.5 cm margins on the left and right sides, Arial font or similar, and font size 11. Neither tabs nor indents should be used, in either the text or the references list. Paragraphs should not be separated by blank lines. Species and genus names should be written in italics. Genus names may be abbreviated (e.g. Q. robur for Quercus robur), but must be written in full at first mention. SI (Système International) units should be used. Technical nomenclatures and style should follow the most recent edition of the CBE (Council of Biology Editors) Style Manual.

Title page

The title page should include a concise and informative title (in the language of the text and in English), the name(s) of the author(s), the institutional affiliation and address of each author, and the e-mail address, telephone number, fax number, and postal address of the author for correspondence.

Abstract

Each article should be preceded by an abstract of no more than 200 words, summarizing the most important results and conclusions. In the case of articles not written in English, the authors should supply two abstracts, one in the language of the text, the other in English.

Key words

Five key words, not included in the title, should be listed after the Abstract.

Article structure

This should where possible be as follows: Introduction, Material and Methods, Results and Discussion, Acknowledgements, References. Section headings should be written in bold with font size 12. If subsection headings are required, these should be written in italics with font size 11, and should not be numbered.

Introduction

This section should briefly review the relevant literature and clearly state the aims of the study.

Material and methods

This section should be brief, but should provide sufficient information to allow replication of the study's procedures.

Results and discussion

This section should present the results obtained as clearly and concisely as possible, where appropriate in the form of tables and/or figures. Very large tables should be avoided. Data in tables should not repeat data in figures, and vice versa. The discussion should consist of interpretation of the results and of their significance in relation to previous studies. A short conclusion subsection may be included if the authors consider this helpful.

Acknowledgements

These should be as brief as possible. Grants and other funding should be recognized. The names of funding organizations should be written in full.

References

The references list should include only articles that are cited in the text, and which have been published or accepted for publication. Personal communications should be mentioned only in the text. The citation in the text should include both author and year. In the references list, articles should be ordered alphabetically by first author's name, then by date.

Examples of citation in the text:
Similar results have been obtained previously (Fernández 2005a, b; Rodrigo et al. 1992
Andrade (1949) reported that...
According to Mario & Tineti (1989), the principal factors are...
Moore et al. (1991) suggest that...

Examples of listings in References:

Journal article:
Mahaney, W.M.M., Wardrop, D.H. & Brooks, P. (2005). Impacts of sedimentation and nitrogen enrichment on wetland plant community development. Plant Ecology. 175, 2: 227-243.

Book chapter:
Campbell, J.G. (1981). The use of Landsat MSS data for ecological mapping. En: Campbell J.G. (Ed.) Matching Remote Sensing Technologies and Their Applications. Remote Sensing Society. London.
Lowel, E.M. & Nelson, J. (2003). Structure and morphology of Grasses. En: R.F. Barnes et al. (Eds.). Forrages. An introduction to grassland agriculture. Iowa State University Press. Vol. 1. 25-50

Complete book:
Jensen, W (1996). Remote Sensing of the Environment: An Earth Resource Perspective. Prentice-Hall, Inc. Saddle River, New Jersey.

Standard series:
Tutin, T.G. et al. (1964-80). Flora Europaea, Vol. 1 (1964); Vol. 2 (1968); Vol. 3 (1972); Vol. 4 (1976); Vol. 5 (1980). Cambridge University Press, Cambridge.

Institutional publications:
MAPYA (2000). Anuario de estadística agraria. Servicio de Publicaciones del MAPYA (Ministerio de Agricultura, Pesca y Alimentación), Madrid, España.

Legislative documents:
BOE (2004). Real Decreto 1310/2004, de 15 de enero, que modifica la Ley de aprovechamiento de residuos ganaderos. BOE (Boletín Oficial del Estado), nº 8, 15/1/04. Madrid, España.

Electronic publications:
Collins, D.C. (2005). Scientific style and format. Available at: http://www.counckjnce.org/publications.cfm [5 January 2005]

Articles not published but accepted for publication:
Such articles should be listed in References with the name of the journal and other details, but with "in press" in place of the year of publication.

Figures and tables

Numbering:
All figures (data plots and graphs, photographs, diagrams, etc.) and all tables should be cited in the text, and should be numbered consecutively.

Figure quality.
Please send high-quality copies. Line thickness in the publication-size figure should be no less than 0.2 mm. In the case of greyscale figures, please ensure that the different tones are clearly distinguishable. Labels and other text should be clearly legible. Scale should be indicated by scale bars. Maps should always include indication of North, and of latitude and longitude. Colour figures can be published.

Figure size:
Figures should be no more than 17.5 cm in width, or no more than 8.5 cm in width if intended to fit in a single column. Length should be no more than 23 cm. When designing figures, please take into account the eventual publication size, and avoid excessively white space.

Figure and table legends:
All figures and tables require a legend. The legend should be a brief statement of the content of the figure or table, sufficient for comprehension without consultation of the text. All abbreviations used in the figure or table should be defined in the legend. In the submitted manuscript, the legends should be placed at the end of the text, after the references list.

Preparing the manuscript for submission

Text

The text should be submitted as a text file in Microsoft Word or a Word-compatible format.

Tables and figures

Each table and each figure should be submitted as a separate file, with the file name including the name of the table or figure (e.g. Table-1.DOC). The preferred format for data plots and graphs is EPS for vector graphics (though all EPS files must include a TIFF preview), and TIFF for greyscale figures and photographs (minimum resolution 300 dpi). If graphics files are submitted in the format of the original program (Excel, CorelDRAW, Adobe Illustrator, etc.), please ensure that you also include all fonts used. The figure or table legend should not be included in the file containing the figure or table itself; rather, the legends should be included (and clearly numbered) in the text file, as noted above. Scanned line drawings should meet the following requirements: line or bit-map scan (not greyscale scan), minimum resolution 800 dpi, recommended resolution 1200-1600 dpi. Scanned halftone drawings and photographs should meet the following requirements: greyscale scan, minimum resolution 300 dpi, recommended resolution 600-1200 dpi.

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As noted above, the text and each figure and table should be submitted as separate files, with names indicating content, and in the case of the text file corresponding to the first author's name (e.g. Alvarez.DOC, Table-1.DOC, Fig-1.EPS). File names should not exceed 8 characters, and must not include accents or special characters. In all cases the program used to create the file must be clearly identifiable.

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December 2015

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