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The european framework for soil sustainability: mapping soil quality in model areas in Catalonia

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The European Framework for Soil Sustainability: Mapping Soil Quality in Model Areas in Catalonia

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SUMMARY

Soil is a vital non-renewable resource delivering multiple functions simultaneously. Healthy soil gives us clean air and water, bountiful crops and forests, productive rangelands, diverse wildlife, and beautiful landscapes. Soil health, or quality, can be broadly defined as the capacity of a living soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health. Soil quality and health change over time due to natural events or human impacts.

Land degradation is the consequence of multiple processes that both directly and indirectly reduce the utility of land. Hence, soil degradation is defined as a decrease in soil quality, which is caused by non sustainable soil management. When the soil loses its functionality, this is directly related with decreasing soil quality. In this context, the concept of soil quality could be used to assess multifunctional land health qualities and pedogenesis with the use of soil quality indicators.

Soil quality indicators should describe ecosystem processes and soil properties. These indicators should be accessible to a large number of users and applicable to a diverse range of soil and climatic conditions and soil management practices.

This PhD proposes a scheme of intrinsic soil indicators for determining soil quality. These schemes include three different sets of soil quality indicators derived from basic soil use criteria. Such criteria are based upon indicator availability, suitability and usefulness. These indicators are grouped according to three different soil threats; soil salinity, declining organic matter and desertification. The indicators should be interpretable in the context of soil quality, whilst also providing an auditable pathway through which soil management decisions can be made.

The study tested several approaches to characterize the 3D spatial variability of soil quality as defined in the UE framework based upon detailed field surveys, using appropriate indicators. Apart from this, the thesis also assessed the temporal variability of soil salinity and sodicity indicators –as soil degradation processes. The results obtained were validated according to the possibilities of deriving appropriate management practices or recommendations of land use planning.

These selected indicators were tested in two different areas in Catalonia. One was located on the left side (N) of Ebro Delta (Delta de l'Ebre region), and the other was located between Canalda and Odén (Solsona region). These areas were chosen based upon their differences in soil quality, land use and environmental conditions.

Soil salinity study showed that the selection of the right method is less important than the use of the right data at the proper scale. Therefore, the knowledge and the understanding of the hydrological and soil processes and the use of detailed data representing these processes, is more relevant than the proper method applied, and according to this, there is a spatial and temporal variability after 12 year in the study area using whatever method tested.

Notably, land use is a strong factor affecting soil organic carbon (SOC) distribution in space and in depth, because land use type significantly altered its vertical distribution. A good management of the cropland such as soil-friendly practices should maintain SOC, however, a conversion of crop land to pasture – which happened in the past in the area- could cause a substantial C accumulation below 1-m depth. Moreover, afforestation of pasture by forest (pine woods) could increase SOC and would provide protective cover in vulnerable, steep and mountainous areas.

The use of a detailed soil map (1:25,000) for mapping SOC showed satisfactory results, when it is compared with other digital mapping methods. Moreover, it illustrates SOC differences between soil mapping units.

Desertification and erosion were studied applying MEDALUS (Mediterranean Desertification and Land Use) and RUSLE (Revised Universal Soil Loss) approaches. MEDALUS approach defined environmentally sensitive areas classes and RUSLE quantified the erosion that takes place in the study area. The MEDALUS model can assess the extent, intensity and severity of desertification process in the target area. However, there is scope for improvements.

The MEDALUS model can assess the extent, intensity and severity of desertification process in the target area. However, there is scope for improvements. It uses socio-economic factors that are quantified through scoring, which are not always as objective as the terrain and soil factors. It should also be more flexible, giving the opportunity to add new parameters into the model, such as the slope length and steepness factors of the RUSLE that have more importance when working at detailed scales.

RESUM

El sòl és un recurs no renovable de vital importància amb la capacitat de tenir múltiples funcions alhora. La qualitat i la salut del sòl són conceptes equivalents, tot i que no sempre han estat considerats sinònims. La qualitat s'ha d'interpretar com la utilitat del sòl per un propòsit específic a gran escala de temps. L'estat de les propietats dinàmiques del sòl com el contingut de la matèria orgànica, diversitat d'organismes, o productes microbians en un temps particular constitueixen la salut del sòl. Així doncs, cal considerar que el sòl com a ésser viu, que ha de tenir una gestió sostenible per una òptima productivitat vegetal i animal, i poder mantenir o millorar la qualitat de l'aire i l'aigua. La qualitat del sòl s'acostuma a veure alterada a causa d'esdeveniments naturals o per l'acció de l'home.

La degradació dels sòls es defineix com la disminució de la seva qualitat causada per un mal ús per part de l'espècie humana, o bé per causes generals. Així doncs, la pèrdua de funcionalitat del sòl està lligada a la disminució de qualitat d'aquest.

Els indicadors de qualitat del sòl han de complir amb una sèrie de requisits, com ser descriptors de processos dels ecosistemes, integrar propietats i processos físics, químics i biològics del sòl; ser accessibles a diferents usuaris i aplicables a diferents condicions de camp; ser sensibles a les variacions de maneig i de clima; i provenir d'una base de dades existent.

En aquesta tesi s'ha estudiat el comportament d'indicadors de qualitat del sòl escollits sota un marc polític de la Unió Europea (COM(2002)). En concret, s'han estudiat indicadors relacionats amb tres amenaces del sòl, contingut de matèria orgànica, grau de desertificació de les terres i estat de salinitat dels sòls, amb l'objectiu de validar la seva funcionalitat per qualificar el sòl.

En particular estudia l'aplicació diversos mètodes per caracteritzar la variabilitat espacial en 3D de la qualitat del sòl, tal com es defineix en el marc de la UE, utilitzant indicadors adequats i fent servir dades de camp detallades. A més a més, també s'ha avaluat la variabilitat temporal dels indicadors de la salinitat del sòl i sodicitat. Els resultats obtinguts s'han validat per tal de poder fer recomanacions pràctiques d'ordenació del territori i una millora de la gestió d'aquest.

Els estudis s'han portat a terme en dues árees ben diferenciades de Catalunya. La salinitat i sodicitat s'ha estudiat al marge esquerra (N) del Delta de l'Ebre, mentre que carboni orgànic del sòl i desertificació ha estat estudiat a una zona compresa entre Canalda i Odèn, (El Solsonès) a la Catalunya central. Aquestes àrees han estat seleccionades en base a les seves diferències en la qualitat del sòl, ús del sòl i les condicions ambientals.

A partir dels resultats es discuteix la validesa dels mètodes pels objectius d'avaluació de la qualitat del sòl, la factibilitat de les anàlisis per a la disponibilitat actual de cartografia edàfica, i es proposen mesures per a la seva millora.

L'estudi de la salinitat del sòl ha mostrat que en un periode de temps de 12 anys, es constata, tot i l'aparent canvi climàtic, un descens de la salinitat al delta, probablement provocat per canvis en maneig del reg i per la regulació de l'Ebre a través dels seus embassaments; així doncs el maneig del reg i del riu tenen encara marge de maniobra per poder compensar, per ara, la possible concentració de sals causada per un increment de l'evapotranspiració al delta, deguda al canvi climàtic. Quedaria per avaluar l'efecte de l'augment del nivell del mar, i les previsions de l'evolució del delta.

L'ús del sòl condiciona i afecta la distribució del carboni orgànic del sòl (COS) en l'espai i en profunditat. Tot i que l'aplicació de bones pràctiques agràries a les terres cultivades fa que es mantinguin el nivel de COS fins i tot millorar-los, una conversió d'aquestes terres a pastures, podria afavorir l'acumulació fins 1 m de profunditat de COS. D'altra banda, el repoblament de pastures a pins també podria fer que s'augmentés COS, a més a més li proporcionaria una coberta protectora. L'ús d'un mapa detallat per cartografiar COS ens mostra uns resultats satisfactoris quan el comparen amb resultats obtinguts d'altres mètodes de cartografia digital. A més a més, ens mostra les diferències de COS entre unitats cartogràfiques de sòls.

La desertificació i l'erosió es van estudiar aplicant els models MEDALUS i RUSLE. El model MEDALUS defineix classes d'àrees ambientalment sensibles mentre que el model RUSLE quantifica els valors d'erosió que es produeix en l'àrea d'estudi.

El model MEDALUS avalua la magnitud, intensitat i severitat dels procesos de desertificació. No obstant això, hi hauria d'haver un marge per la millora del model, ja que utilitza factors socioeconòmics que es quantifiquen a través de puntuacions. Aquestes puntacions no són sempre tan objectives com els factors del terreny i del sòl. Hauria de ser un mètode més flexible, donant l'oportunitat d'afegir nous paràmetres al model, com podrien ser la longitud del pendent i els factors d'inclinació de la RUSLE que tenen més importància quan es treballa a escales detallades.

RESUMEN

El suelo es un recurso no renovable de vital importancia con la capacidad de tener múltiples funciones a la vez. La calidad y la salud del suelo son conceptos equivalentes, aunque no siempre han sido considerados sinónimos. La calidad debe interpretarse como la utilidad del suelo para un propósito específico a gran escala de tiempo. El estado de las propiedades dinámicas del suelo como el contenido de la materia orgánica, diversidad de organismos, o productos microbianos en un tiempo particular constituyen la salud del suelo. Así que, hay que considerar el suelo como un ser vivo, que debe tener una gestión sostenible para una óptima productividad vegetal y animal, y poder mantener o mejorar la calidad del aire y el agua. Normalmente la calidad del suelo se ve modificada a causa de acontecimientos naturales o por la acción del hombre.

La degradación de los suelos se define como la disminución de su calidad causada por un mal uso por parte de la especie humana, o bien por causas generales. Así pues, la pérdida de funcionalidad del suelo está ligada a la disminución de calidad de éste.

Los indicadores de calidad del suelo han de cumplir con una serie de requisitos, como ser descriptores de los procesos de los ecosistemas, integrar propiedades y procesos físicos, químicos y biológicos del suelo; ser accesibles a diferentes usuarios y aplicables a diferentes condiciones de campo; ser sensibles a las variaciones de manejo y de clima; y provenir de una base de datos existente.

En la presente tesis se ha estudiado el comportamiento de indicadores de calidad del suelo escogidos bajo un marco político de la Unión Europea (COM(2002)). En concreto, se han estudiado indicadores relacionados con tres amenazas del suelo, contenido de materia orgánica, grado de desertificación de las tierras y estado de salinidad de los suelos, con el objetivo de validar su funcionalidad para cualificar el suelo.

Se ha estudiado la aplicación de varios métodos para caracterizar la variabilidad espacial en 3D de la calidad del suelo, tal como se define en el marco de la UE, utilizando indicadores adecuados y utilizando información de campo detallada. Además, también se ha evaluado la variabilidad temporal de los indicadores de la salinidad del suelo y sodicidad. Los resultados obtenidos se han validado para poder hacer recomendaciones prácticas de ordenación del territorio y una mejora de su gestión.

Los estudios se han llevado a cabo en dos áreas bien diferenciadas de Cataluña. La salinidad se ha estudiado en el margen izquierdo del Delta del Ebro, mientras que el carbono orgánico del suelo y la desertificación se ha estudiado en una zona comprendida entre Canalda y Odén, (El Solsonès) en la Cataluña central. Estas áreas han sido seleccionadas en base a sus diferencias en la calidad del suelo, uso del suelo y condiciones ambientales.

A partir de los resultados se discute la validez de los métodos para una correcta evaluación de la calidad del suelo, la factibilidad de los análisis para la disponibilidad actual de cartografía edáfica, y se proponen medidas para su mejora.

El estudio de la salinidad del suelo ha mostrado que en un periodo de tiempo de 12 años, se constata, a pesar del aparente cambio climático, un descenso de la salinidad en el delta, probablemente provocado por cambios en manejo del riego y por la regulación del Ebro a través de sus embalses; así pues el manejo del riego y del río tienen todavía margen de maniobra para

poder compensar, por ejemplo, la posible concentración de sales causada por un incremento de la evapotranspiración en el delta. Quedaría para evaluar el efecto del aumento del nivel del mar, y las previsiones de la evolución del delta.

El uso del suelo condiciona y afecta a la distribución del carbono orgánico del suelo (COS) en el espacio y en profundidad. Aunque la aplicación de buenas prácticas agrarias en campos de cultivo hace que se mantengan el nivel de COS, incluso mejorarlos. Un cambio de estas tierras a pastos, podría favorecer la acumulación hasta 1 m de profundidad de COS. Por otra parte, la forestación de pastos a pinos también podría hacer que aumentara COS, además le proporcionaría una cubierta protectora al suelo. El uso de mapas detallados de suelos para cartografiar COS nos muestra unos resultados satisfactorios cuando se comparan con otros resultados obtenidos a partir del uso de técnicas de cartografía digital.

La desertificación y la erosión se estudiaron aplicando los modelos MEDALUS y RUSLE. El modelo MEDALUS define clases de áreas ambientalmente sensibles mientras que el modelo RUSLE cuantifica los valores de erosión que se produce en el área de estudio.

El modelo MEDALUS evalúa la magnitud, intensidad y severidad de los procesos de desertificación. Sin embargo, debería tener un margen para la mejora del modelo, ya que utiliza factores socioeconómicos que se cuantifican a través de puntuaciones. Estas puntaciones no son siempre tan objetivas como los factores del terreno y del suelo. Debería ser un método más flexible, dando la oportunidad de añadir nuevos parámetros al modelo, como podrían ser la longitud de la pendiente y los factores de inclinación de la RUSLE que tienen más importancia cuando se trabaja a escalas detalladas.

Chapter 1

GENERAL INTRODUCTION

AND

GENERAL OBJECTIVES

1 GENERAL INTRODUCTION

1.1 Soil quality and health

Soil is a vital non-renewable resource delivering multiple functions simultaneously. Healthy soil gives us clean air and water, bountiful crops and forests, productive rangelands, diverse wildlife, and beautiful landscapes. Soil does all this by performing seven essential functions (European Commission (EC) (EC, 2006a, 2006b, 2012). These multiple functions include food and fibre production, nutrient retention and cycling, carbon storage, filtration of water, habitat for soil biodiversity, physical and cultural environment for humans and human activities, source of raw materials and an archive of geological and archaeological heritage (EC, 2006a, 2012). The ability of the soil to deliver multiple functions simultaneously is called functional soil capacity (Schulte *et al.*, 2014).

Soil health refers to self-regulation, stability, resilience, and a lack of stress symptoms in a soil ecosystem, i.e. soil health describes the biological integrity of the soil community-the balance among organisms within a soil and between soil organisms and their environment (Curell *et al.*, 2012). Soil health is a description of the condition or status of a soil and may comprise multiple factors including soil quality characteristics that come together to create a hospitable environment for soil life (Curell *et al.*, 2012). Hence, soil quality is the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation (Karlen *et al.*, 1997; Doran, 2002). Soil quality and soil health are considered equivalent concepts, but are not always considered synonymous with one another (Doran and Parkin, 1994). Soil health and soil quality change overtime due to natural events or human impacts (Doran, 2002). The quality of the soil could be interpreted as a function of the land use when used for a specific purpose for a long time scale (Carter *et al.,* 1997).

McBratney *et al.* (2012) introduced the concept by of Soil Security. Soil security covers all the major needs for soil including maintenance and improvement of the world's soil resources so that they can continue to provide food, fibre and fresh water, make major contributions to energy and climate sustainability, and maintenance of biodiversity and the overall protection of ecosystem goods and services. As soil security is a concept of securing soil for the sustainable development of humanity we need to consider more than the biophysical stocks, functioning and ecosystem services, we also need to embrace the economic, social and policy dimensions (McBratney *et al.*, 2014). Soil protection, or perhaps more appropriately termed Soil Security, is conceptually a far wider assessment framework than soil quality in its attempt to encompass all human and natural activities associated with soil (McBratney *et al.*, 2012; Koch *et al.*, 2013).

Soil has both inherent and dynamic qualities. Inherent soil quality refers to the soil's natural ability to function; therefore these characteristics do not change easily. Conversely, dynamic soil quality relies on soil management, since soils respond differently to management depending upon the inherent properties of the soil and the surrounding landscape. In short, the quality of a soil is an assessment of how it performs all of its functions now and how those functions are being preserved for future use. Soil degradation is defined as the loss of the soil's capacity to develop its functions; e.g., support for plant growth, hydrological regulation of watersheds, environmental filtering, and support for buildings, among others (Poch and Martínez-Casasnovas, 2016). The delivery of soil functions is enhanced by management and land-use decisions that consider the multiple functions of soil whilst the functional capacity of the soil may be impaired by decisions which focus only on single functions, such as crop productivity (Doran, 2002).

Costanza and Daly (1992) reported that soil natural capital could be defined as a stock of natural assets yielding a flow of either natural resources or ecosystem services. Robinson *et al.* (2009) and Dominati *et al.* (2010) incorporated the idea of soils as natural capital into a conceptual framework. Doing so, they create the opportunity to value the natural capital of soils and also to track the changes in these values for a given human use. The natural capital of soils can be characterised by soil properties, that, on the other hand, is the way in which soil scientists and agronomists describe and characterise soils. As measurable quantities, soil properties enable soil scientists to compare soils based on different criteria (Dominati *et al.*, 2010).

Following and extending the definitions of Robinson *et al.* (2009) and Dominati *et al.* (2010), 'soil natural capital', according to McBratney *et al.* (2012) comprises natural stocks (the compositional state of the soil system), ecosystem services (functions performed by the soil for the whole ecosystem), and ecosystem goods (products of the ecosystem supplied by soil). Ecosystem services include: clean air, water and soil, conservation of biodiversity, nutrient cycling and wildlife habitat protection. These services are difficult to quantify in economic terms but are fundamental to society. Research is still required to combine basic and applied hydrological, biological, geological and soil sciences with socio-economic science to reveal new ways in which managed ecosystems can provide ecosystem services.

It has been widely reported that soil functions are currently declining, which is affecting soils' ability to provide ecosystem services and goods (Lal, 2010). Land degradation is the consequence of multiple processes that both directly and indirectly reduce the utility of land. Land degradation represents a remarkable issue which is widespread over large areas of the world where soils have suffered from a loss of biological production and resilience caused by both, natural and anthropogenic factors (Mainguet 1994; Blum, 1998). The phenomenon involves a reduction of the renewable resource potential by one or a combination of processes acting upon the land. Soil degradation refers to changes in the soil health that result in its diminished capacity to deliver ecosystem services (FAO, 2015). Erosion and pollution are major sources of soil degradation and globally an estimated 17 % of the land surface has already been strongly degraded (van Lynden, 1997). A majority of soils are used for agricultural purposes where management practices can affect soil quality. Agriculture is often associated with negative impacts on the soil such as those resulting from the tillage system, or related to the use of pesticides or inorganic nitrogen or phosphate fertilisers (UK National Ecosystem Assessment (NEA), 2011). Hence, agricultural management has an important role in the sustainability of this resource. Soil quality is conceptualized as the major linkage between the strategies of conservation management practices and achievement of the major goals of sustainable agriculture (Parr *et al.*, 1992; Acton and Gregorich, 1995). Depending on how soils are managed, they can be important sources or sinks for carbon dioxide (CO_2) and other gases that contribute to the greenhouse gases (GHG) resulting in climate change. Soils store, degrade or immobilise nitrates (NO_3^-), phosphorus (P), pesticides and other substances that can become pollutants in air or water (Rosewell, 1999). Assessment of soil quality or health is invaluable in determining the sustainability of land management systems (Karlen *et al.*, 1997).

1.2 *EU regulations in environmental and soil degradation*

Despite its importance for our society, and unlike air and water, there is no EU legislation specifically targeting the protection of soil. Different EU policies for water, waste, chemicals, industrial pollution, nature protection, pesticides and agriculture contribute indirectly to soil protection. However, as these policies have other aims, they are not sufficient to ensure an adequate level of protection for all soil in Europe. Furthermore, the prevention of soil degradation is also limited by the lack of data. In this context, in 2006, the European Commission adopted a Soil Thematic Strategy (COM(2006) 231) (EC, 2006a) and a proposal for a Soil Framework Directive (COM(2006) 232 (EC, 2006b), which highlighted the pressing need for research which combines the analysis of processes related to threats to soil (decline of soil organic carbon; soil erosion and desertification, compaction; salinisation and sodification; landslides, contamination; and declining soil biodiversity) and the development and harmonisation of methods for soil monitoring (EC, 2006a). Figure 1 shows the causes of soil degradation as a consequence to the impact of human activities.

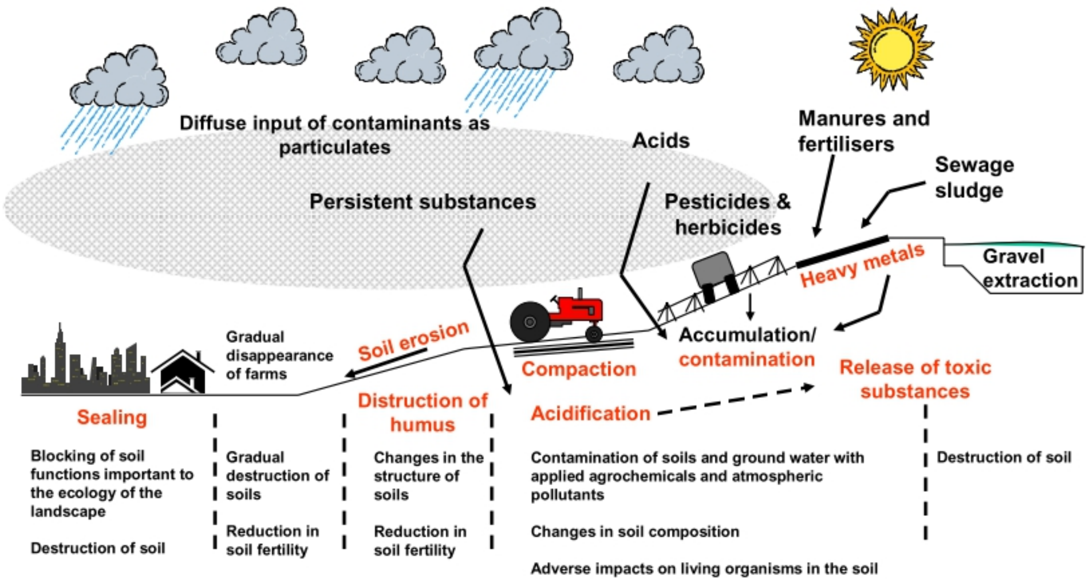

Figure 1. The impact of human activities on soil, causing risk of soil degradation. Source: Tóth, *et al.* (2008).

External costs of degradation are often larger than direct private costs. Such external costs are important from a policy perspective because they represent a potential cause of market failure. The total costs of degradation that could be assessed for erosion, organic matter decline, salinisation, landslides and contamination, would be up to €38 billion annually for the EU-25 (Bowyer *et al.*, 2008).

To date, soil protection is not a specific objective of any EU legislation but features in some legislation as a secondary objective. Currently, the most important EU environmental directives with respect to soil quality are the Nitrates Directive (91/676/EEC) (EU, 1991) and the Water Framework Directive (2000/60/EC) (EU, 2000). Others, such as the Habitats Directive (92/43/EEC) (EU, 1992), Birds Directive (2009/147/EC) (EU, 2009), and EU EIA Directive (2011/92/EU) (EU, 2012) through Natura 2000 seek to halt the loss of biodiversity are expected to have beneficial effects on soil quality, but to a lesser extent owing to a more focused set of objectives (EC, 2009).

In 1998 the framework of the Cardiff Process required different Council formations to integrate environmental considerations into their respective activities. This framework reported environmental objectives to be integrated into EU sectoral policies, including the Common Agricultural Policy (CAP). The requirement to keep agricultural land (whether in productive use or not) in good agricultural and environmental condition (GAEC) aims to prevent land abandonment and ensure minimum maintenance of agricultural land. The elements of GAEC specifically target protection of soil against soil erosion, maintenance or improvement of soil organic matter, and maintenance of a good soil structure (EC, 2009).

In recent years the importance of the prevention of soil or the recovery thereof has intensified. In mid-2002, a statement was made, ‘Towards a thematic strategy for soil protection’ of the commission to the council, the European Parliament, the Economic and Social Committee and the Committee of the Regions, exposing a need to admire and preserve soil threats hanging over this regard. This statement should include a global and long-term protection of the soil, preserving the vital functions of the soil and include qualitative and quantitative targets. Then collect information activities, deploy adequate measures to protect the soil and its sustainable use. Overall, a community framework should be based upon scientific knowledge and best available techniques.

The Commission published the Soil Thematic Strategy in 2006 (COM(2006) 231) (EC, 2006a). Its overall objective is the protection and sustainable use of soil, based on the prevention of further soil degradation, preserving soil functions and restoring degraded soils to a level of functionality consistent with current and intended use (EC, 2006a). The Commission identified eight threats to soil: erosion, decline in organic matter, local and diffuse contamination, sealing, compaction, loss of biodiversity, salinisation and landslides. The proposed Soil Framework Directive (COM(2006) 232) (EC, 2006b) requires Member States to identify areas at risk of soil degradation, as well as to set up an inventory of contaminated sites. Some five years after the adoption of the Soil Thematic Strategy, the European Commission published a policy report on the implementation of the Strategy and ongoing activities (COM(2012)46) (EC, 2012). In May 2014, the European Commission decided to withdraw the proposal for a Soil Framework Directive. In response, a petition has been made by more than 1500 soil scientists, to protect European soils, and open a new pathway to a European Soil Directive.

1.3 *Evaluation and monitoring of soil quality and soil degradation*

Even in the absence of direct legislation for soil protection, it is still critical that soil quality is monitored to sustain and guarantee the delivery of ecosystem service into the future. Soil monitoring is the systematic determination of soil properties such that spatial and temporal changes can be detected (FAO/ECE, 1994; GSP & FAO, 2012). According to Morvan *et al.* (2008) a soil monitoring network (SMN) can be defined as a set of sites or areas where a periodic assessment is carried out to allow changes in soil characteristics and changes in an extended set of soil properties to be identified and documented overtime. The spatial distribution of sites must be considered so that adequate representativeness is achieved. Some researchers (Bouma, 2002; Arshada and Martin, 2002) have proposed procedures for evaluating soil quality and functioning by combining and integrating specific elements into soil quality indices. These procedures allow for weighting of various functions, depending upon the user goals and socio-economic concerns. The criteria for selecting quality indicators will differ depending upon the land uses and their dynamic over time (Warkentin, 1995; Noble *et al.*, 2000; Astier, 2002). The soil quality indicators selected must be related to the economic, social and ecological development of the study area and therefore the indicators will vary in number and type according to agro-ecological area, agroclimatic factors and management systems. Soil quality indicators are needed to measure changes in soil functioning that result from management changes. These indicators should be a measure of (quantitatively or qualitatively) physical, chemical and biological properties, processes and characteristics.

Holloway and Stork (1991) suggested some essential requirement for an indicator: (a) be adequately sensitive to change, (b) accurately reflect the functioning of the system, (c) be universal, yet illustrate temporal or spatial patterns and (d) be cost effective and relatively easy and practical to measure. Not all indicators fulfil all of these criteria nor will one indicator alone be sufficient to indicate all changes within a system (Holloway and Stork, 1991). Doran *et al.* (2002) define similar requirements for soil quality indicators: a) to be descriptive of the soil processes; b) the ability to integrate physical, chemical and biological soil properties; c) be accessible to users and applicable to different field conditions; d) to be sensitive to changes in climate and management; and e) to come from an existing database.

Some authors (Larson and Pierce, 1991; Doran and Parkin, 1994) suggested the use of Minimum datasets (MDS), soil parameters which could be used to evaluating soil quality, defined as soil properties or indicators. MDS includes physical, chemical and biological indicators such as nutrient availability, labile fraction and/or total organic carbon, texture, soil water retention, soil structure, maximum roots depth, pH, electrical conductivity, amongst others. Many basic soil properties are useful in estimating other soil properties or attributes which are difficult or too expensive to measure directly and they could be inferred from pedotransfer functions (PTF). PTFs can be defined as predictive functions of certain soil properties from other easily, routinely, or cheaply measured properties (McBratney *et al.*, 2002; Vrščaj, *et al.*, 2008). These indicators could be qualitative or quantitative parameters, or composed indexes obtained from the relationship between different parameters (Etchevers, 1999). The status of the soil quality is represented by increases and decreases of the parameters. A better soil evaluation could be assessed if potential indicators such as ecological-biological parameters could be used for this finality (Astier- Calderon *et al.,* 2002). The MDS should be measured in a short time, represent economic viability and should allow us to identify the multi-functionality of the soil.

Indicators can be used at international and national levels in environment reporting, measurement of environmental performance and reporting on progress towards sustainable development (OECD, 1999, 2002). They can further be used at national level in planning, clarifying policy objectives and setting priorities. The selection of indicators is an evolutionary process that depends on the pressures of society and political decisions.

OECD (1999, 2002) established a pragmatic approach, recognising that there is no universal set of indicators; rather, several sets exist, serving several purposes and audiences. A common framework was created for proposing and defining agro-environmental indicators. This common framework is based on a pressure-state-response (PSR) model, thus pressure indicators, status indicators and indicators of response are established (OECD, 1999, 2002). In some cases, it was found that the limit between causes and response was so close that they could be considered both simultaneously. Soil degradation problems can be successfully addressed through a driving forces–pressures–states–impacts–responses analysis (Görlach, 2004; Poch and Martínez-Casasnovas, 2016). Some applications can be found in Porta and Poch (2011) and Emadodin *et al.* (2012). Conversely, the lack of data may also represent a difficulty in the indicator selection process especially when it comes to issues related to incomplete sets of data, low quality data or where data collection is not systematic and therefore inconsistent.

Overall, the development of a soil monitoring scheme is a complex task owing to the hetreogenous nature of soil coupled with the complex interaction with land management. The ENVASSO (ENVironmental ASsessement of Soil for mOnitoring) project was set up to obtain detailed information on soil monitoring networks to define and document a soil monitoring system in support of a Soil Framework Directive towards soil protection (EC, 2015). The ENVASSO Consortium reviewed existing information related to soil systems across 25 EU member states to serve as a base for a soil monitoring network along with appropriate procedures, protocols and indicators for evaluation. Part of the research carried out within this project was aligned with the ENVASSO project. In particular, the research related to salinity was developed to satify ENVASSO project requirements.

1.4 *Existing information in Spain and Catalonia*

Ibáñez *et al.* (2003) have summarized the existing networks of soil measurements in Spain. A conclusion from their review is that there are few systematic approaches or integral network methods to measure soil quality in Spain. However, a more careful look into the existing systems and information allow us to conclude that with some effort such integration could potentially be achieved. Also, modelling changes of soil properties is a complementary tool in any soil monitoring scheme, but a basic requirement for such modelling is a soil map. A soil map should capture the detailed information of soil properties to support the use of modelling. Catalonia Government is concerned about soil protection and is conscious of the need to have systematic soil cartography of the entire region. Departament d'Agricultura, Ramaderia i Pesca (DARP), produced a soil map at a resolution of 1:25.000 of Catalonia, including 500.000 ha of the most important agricultural lands, (Alcañiz *et al.*, 2005). In 2008, the the Cartographic and Geologic Institut of Catalonia (ICGC) assumed responsibility for the ongoing soil mapping.

As well as salinity, erosion and soil organic matter decline are considered key threats to soil quality in Spain. The Erosion catchments network "RESEL" started in 1995 and provides information about erosion. Several land uses and measurements from MAPA/MIMAN (INIA, 2004) include information about organic matter, heavy metals, etc. Non regular systematic repeated samplings currently done by DARP provide information about soil salinity, nutrient rate, organic matter and heavy metals in the most intensive agricultural areas. For the forested areas measurements from the EU network in addition to other sampling (DARP, CREAF) contain a large amount of information.

1.5 *Land evaluation and soil quality*

The quality of the soil not only relies on the inherent properties of the soil but also on the interaction of soil with land management. Any measure of soil quality should consider land management as this will govern the magnitude of the delivery of soil functions and soil quality. Soil quality assessment is considered an effective method for evaluating the environmental sustainability of land use and management activities (De la Rosa, 2005). Land evaluation was defined as '*the process of assessment of land performance when used for specified purposes, involving the execution and interpretation of surveys and studies of land use, vegetation, landforms, soils, climate and other aspects of land in order to identify and make a comparison of promising kinds of land use in terms applicable to the objectives of the evaluation*' (FAO, 1976). There is a long history of multivariate soil evaluation or 'assessment frameworks' derived for Land Evaluation to Soil Quality assessments, there is a need to demonstrate the value of soil and soil science (McBratney *et al.*, 2012). The basic metric of 'soil quality' is the indicator— which 'measures' the characteristic of some state of the soil system. Soil quality is a concept considered to be somewhat vague in its actual definition. The concept has failed to converge on a common accepted set of indicators in relation to human-centred values, or because the indicators are by necessity temporally changing (McBratney *et al.*, 2012). The land evaluation analysis focuses on different purposes, which can be grouped into two main classes: land suitability, and land degradation approaches.

Land suitability is defined in land evaluation as the fitness of a given land unit for a specified type of land use (FAO, 1976). The term suitability refers to the productive use or uses after taking into account the physical limitations of the land. These physical qualities are frequently far from ideal. Differences between ideal and actual may be regarded as limitations imposed by the physical quality of the soil and the environment (De la Rosa, 2005).

Land capability classification is a qualitative system that was developed by the US Department of Agriculture (USDA), in the 1930s, as part of an erosion control program (Klingebiel and Montgomery, 1961). Land capability refers to the potential of land to sustain a number of predefined land uses in a built-in descending sequence of desirability. Land capability is assessed by comparing the characteristics of a land mapping unit with critical limits set for each capability class. To obtain limits for the capability classes, expert knowledge is related to land characteristics. Land capability has been used in Spain for many years, and many authors to assess land suitability.

However, another way to evaluate soil is to know the vulnerability of the soil or soil degradation. Soil degradation means loss of soil or soil quality for specific functions (Blum, 2008). Risk of soil degradation can result from extreme natural events, such as long-lasting torrential rainfall, potentially resulting in erosion, inundations, landslides and further adverse effects. Notwithstanding this, human activities are still regarded as the main causes of soil degradation risk (Blum, 2002). Soil that is lost due to degradation processes (e.g. erosion, pollution) would need hundreds or thousands of years to recover naturally. Compared to the lifespan of human beings, soil loss is not recoverable, which means that soil must be regarded as a non renewable resource. It is estimated that about 15% of the total land area of the world has been degraded by soil erosion and physical and chemical degradation, including soil salinisation (Wild, 2003). In Catalonia, some of the key threats to soil quality identified include salinity, soil organic matter decline and erosion/desertification. This introduction continues with a review of these threats with each section concluding with a review of the specific threat in relation to the study area of the current research.

1.6 *Processes of soil degradation: salinity, application in Spain and Catalonia*

Saline soils often occur in closed depressions and other poorly drained areas in arid environments, which can lead to the development of saline wetlands. Historically, farmers disdained those areas because they had little or no agricultural value; however, today, biodiversity conservation and other environmental issues have increased the concern for the protection of wetlands (Ramsar Convention Secretariat, 2010), even though the degradation and destruction of saline wetlands commonly occurs.

Saline soils are soils in which the content of salts more soluble than gypsum, (basically sulphates of magnesium and sodium, magnesium chloride and sodium carbonate), is high enough to impact on agricultural production, environmental health, and economic welfare (Rengasamy, 2006).

Soluble salts most commonly present are the chlorides and sulphates of sodium, calcium and magnesium. Nitrates may be present in considerable quantities but this only occurs rarely. Sodium and chloride are by far the most dominant ions, particularly in highly saline soils, although calcium and magnesium are usually present in sufficient quantities to meet the nutritional needs of crops. Many saline soils contain appreciable quantities of gypsum ($CaSO_4$, $2H_2O$) in the profile, although its sole presence is not diagnostic for salinity, since it only contributes in about 2 $dS \cdot m^{-1}$ at 25ºC in the overall electrical conductivity of the soil. Soluble carbonates are always absent. The pH value of the saturated soil paste is always less than 8.2 and more often near neutral (Gupta and Abrol, 1990).

A majority of saline soils have a whitish colour or have a white salt crust on the surface, highlighted by the presence of almost widespread saline efflorescence, at least during the dry season. The salinity of the soil is measured from the electrical conductivity (EC) in saturation extracts, and it is accepted worldwide that values higher than 4 $dS \cdot m^{-1}$ are indicative of salinity (Richards, 1954). This phenomenon is related to low values of organic matter, mainly due to the repercussions of the salt content in the chemical balance, hence, a resultant change in the biological activity and the vegetation cover (van de Graaff and Patterson, 2001). Clay soils dominated by clay minerals that have a high cation-exchange capacity (CEC), such as smectite, have higher EC than clay soils dominated by clay minerals that have a low CEC, such as kaolinite (Qadir and Schubert, 2002). Soils with restrictive layers, such as clay-pans, typically have higher EC because salts cannot be leached from the root zone and accumulate on the surface (Cardon *et al.*, 2006).

The sodic or alkaline soils have high amounts of sodium (Na^+) cations in the exchange complex. This Na^+ has a high potential of dispersion of the organo-mineral clay and humus complexes in the soil that can adversely affect soil structure and crop growth. These soils are classified as sodic soils, having an exchangeable sodium percentage (ESP) higher than 15% (Richards, 1954), and are qualified as alkaline soils when Na comes from sodium carbonate, which produces pH values above 8.5. Nevertheless, soils that have more than 6% ESP are considered to have structural stability problems related to potential dispersion (Van de Graaff and Patterson, 2001). Sodicity is not the only factor involved in clay dispersibility: clays with a given sodicity are more dispersible with a high pH than with a low one (Van de Graaff and Patterson, 2001).

The main natural factors influencing soil salinisation and sodification are climate, the salt contents of the parent material and groundwater, land cover and topography. The most influential human induced factors are land use, farming systems, and land management, such as the use of salt-rich irrigation water and/or insufficient drainage. A distinction can be made between primary and secondary salinisation processes. Primary salinisation involves salt accumulation through natural processes due to a high salt content of the parent material or in groundwater. Secondary salinisation is caused by human interventions such as inappropriate irrigation practices.

Table 1. Soil salinity effects.

Decreased fertility	Altered micronutrients concentration (Fe, Mg, Mo) and basic nutrients (P, K) for plant assimilation Increased osmotic pressure of the soil solution, which difficult the nutrients exchange from the soil and plants. Alteration of ion exchange capacity, increasing soil degradation: erosion, compaction and decline of organic matter.
Biodiversity loss	Increased soil biotoxicity with excessive sodium and chloride accumulations.

Source: Feliu and Gueorguíeva, 2003.

Salinisation and sodification are often associated with irrigated areas where low rainfall, high evapotranspiration rates or soil textural characteristics impede the washing of salts out of the soil, which subsequently build up in the surface layers. Irrigation with water that has a high salt content dramatically worsens the problem.

In coastal areas, salinisation may be associated with the over-exploitation of groundwater caused by the demands of growing urbanisation, industry and agriculture. Over-extraction of groundwater can lower the normal water table and lead to the intrusion of seawater (Alcañiz *et al.*, 2005).

Worldwide, approximately 950 million ha of land are estimated to be salt affected, with salinity affecting 23% of arable land and saline-sodic soils affecting a further 10% (Szabolcs, 1994). They occur mainly in the arid–semiarid regions of Asia, Australia and South America. Soil salinity is estimated to affect one to three million hectares of land within the EU, mostly in Mediterranean countries. The countries most affected by salinisation or sodification are Spain, Hungary and Romania. Other countries show localised occurrence of these conditions, which could have a devastating effect locally (EC, 2009). Salinisation is regarded as a major cause of desertification and is therefore a serious form of soil degradation. With the increases in temperature and decreases in precipitation characteristic of climate in recent years, the problem of salinisation in Europe is getting worse. The antropic salinisation and sodification are processes, especially the sodification, which could become irreversible as the cost recovery for washing is not acceptable in agricultural soils (Pizarro, 1985). In Spain there are about 840,000 ha affected by salinization processes (Szabolcs, 1989). In addition, 3% of the 3.5 million hectares of irrigated land is severely affected, reducing markedly its agricultural potential while another 15 % is under serious risk (Jones, *et al.*, 2003).

In Catalonia, most of the saline soils are located in the central area of the Ebro valley and on the Mediterranean coast (Llobregat and Ebro deltas, Low Empordà). In the central area of the Ebro Valley has approximately 20% of the irrigated land, which is considered to have a high risk of salinisation. In the mid 19th century 'Canal d'Urgell' was built and it transformed 64 345 ha from dryland into an irrigated (Porta and Poch, 2011). The irrigation techniques were neither well known nor the effects that could occur. The irrigation caused salt to wash from the parent materials and to move downstream. It also resulted in a rise of the water table in depressed areas. Combined, all of these factors resulted in the soils becoming saline. In response to this situation, generalized drainage systems were built with the aim of loading irrigation waters with salts into the Segre River, an Ebro tributary. The overexploitation of aquifers is another way of salinisation taking place in the province of Tarragona and in the Llobregat delta near Barcelona, while salinisation by marine aerosols is present in Girona and in Ebro delta.

The Ebro delta has a socio-economic and environmental interest (agronomic, ecological, landscaping and because of the wetland preservation) in soil salinity, as the delta formations are linked to the salt accumulation because of the deltaic cycle. Deltas are areas with important soil fertility because of the benefits from the fertile depositions of sediments that naturally arise from flooding. These areas are characterised as having plenty of water for irrigated agriculture. Salinisation cycle in these areas is a result of the complex interactions between the sea water from flooding during storms, the fresh water from the river, the salt content of the water from the water table.

In the Soil Thematic Strategy of the European Commission (COM (2006) 231; COM (2012) 46 final) the environmental threat of salinisation/sodification is approached with the delineation of actual salt affected areas as 'hot spots' and identification of potential salt affected areas due to the influence of changing environmental conditions or various human activities (EC, 2006a; EC, 2012). In this regard, soil salinity is studied in the south of Catalonia, in the Ebro delta, where the deltaic cycles are present. Soil salinity is present because of the irrigation water, land management and natural factors that occur in the area.

1.7 *SOM stock assessment, application in Spain and Catalonia*

Soils are the largest terrestrial pool for organic carbon in the biosphere as it is shown in Figure 2 (Lal, 2001). Large-scale changes in land use like deforestation and agricultural activities, including biomass burning, plowing, drainage, and low-input farming have resulted in significant changes in soil organic carbon (SOC) pools (Lal, 2003). By mineralization, leaching, erosion, or changes in land use, 50 to 70% SOC is lost as CO_2. Thus, world soils historically have been a major source of atmospheric enrichment of CO_2 (Lorenz and Lal, 2005). About 20% of the global emissions presently come from land use change (Intergovernmental Panel on Climate Change, 2001). SOM, in particular organic C (SOC), has been in the spotlight of soil research for decades. However, the high variability and diversity of data make comparisons difficult (Lugato, *et al.*, 2014). Globally, SOC amounts to about 1,500 Pg C in the upper meter of the soil, ranging from 3.0 kg C m^{-2} in arid climates to 80.0 kg C m^{-2} in organic soils of colder wetter regions (Lal, 2004). Calvo *et al.* (2015) reported that soils in northern Spain, in Atlantic climate, have an average stock 260 $t \cdot ha^{-1}$ (0-30 cm), whilst Doblas-Miranda *et al.* (2013) observed an average stock 59 $t \cdot ha^{-1}$ (0-30 cm) in areas with a Mediterranean climate (Rodríguez-Murillo 2001; Rodeghiero *et al.* 2011). A general view on the average content in SOC of European soils is that most of Southern Europe is covered by soils with less than 2% SOC (Jones, *et al.*, 2005). This is related to both climate change and historical land use.

Organic matter is an important component of soils because of its influence on soil structure and stability, water retention, cation exchange capacity, soil ecology and biodiversity, and as a source of plant nutrients. Soil organic matter plays a major role in maintaining soil functions. Soil organic matter (SOM) consists of partially decayed plant residues and microorganisms and the by-products of microbial growth and decomposition. SOM is generally agreed to contain 58% soil organic carbon (SOC), and in most cases, it is effectively measured as organic carbon. SOM is a key component of soil as it influences its structure, aggregate stability, nutrient availability, water retention and resilience (Hallett *et al.*, 2012; Robinson *et al.*, 2012). Thus, soil organic matter content is essential to determine the behaviour with plants and organisms.

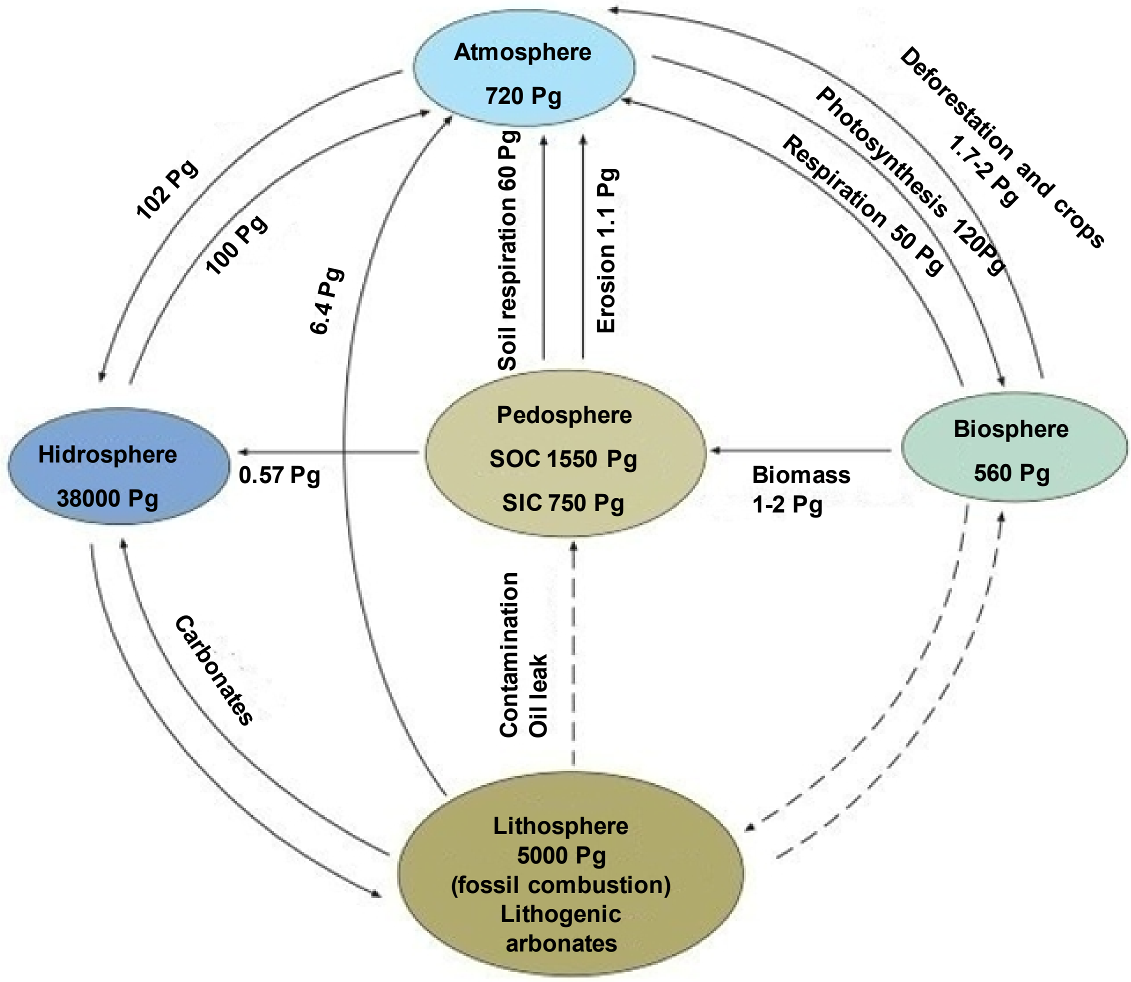

Figure 2. Major C reservoirs on the Earth. (SOC: Soil Organic Carbon; SIC: Soil Inorganic Carbon). Source: Lal (2001)

SOM is composed of organic materials (remains of roots, leaves and dung), living organisms (bacteria, fungi, earthworms and others) and humus with varied organic substances, dark brown - black resulting from the decomposition of organic materials of plant origin (Julca-Otiniano *et al.*, 2006). The decomposition of SOM has two different phases: humification and mineralisation (Porta *et al.*, 2003). The humus mineralisation can vary from 2 to 5% of the carbon annually, the amount of carbon lost should have been returned of organic matter from the remains of vegetation, but there are several factors that break this balance, such as environmental conditions that affect the microbial activity and agricultural use that influence to the destruction of organic matter. Accordingly, the distribution of SOC follows a similar gradient as does the accumulation of biomass, thus in colder climates with low temperature and regular rainfall, there is more SOC than in warmer weather with less rain, where organic matter tends to be mineralized quicker. Soil pH also influences microbial activity. In soils with high pH due to the presence of limestone, or as a contribution of calcium cations; the mineralization of SOM is quicker than in soil dominated by acidic conditions.

SOM has a strong affect on soil properties such as water infiltration rate, erodibility, water holding capacity, nutrient cycling, and pesticide adsorption (Campbell *et al.*, 1996; Wander and Yang, 2000 and Ding *et al.*, 2002). Soil texture has a direct influence on the SOC. Low SOC stocks are more associated with sandy soils than with fine texture soils. This is because the sandy soils are more aerated and less humid; both factors that directly impact the mineralisation of SOM. Nevertheless, soils with poor drainage have higher SOC than well drained soil associated with the higher soil moisture content and decreased metabolic activity of aerobic organisms due to reduced aeration (Dessureault-Rompré *et al.*, 2011).

The distribution of SOM within aggregates is an important factor for its turnover (Yamashita *et al.*, 2006). Temperature is one of the most important factors influencing SOM decomposition, and is considered to be positively correlated with the decomposition rate of SOM (Tan *et al.*, 2013). This fact is most pertinent where conditions shift from anaerobic to aerobic resulting in a speeding up the mineralisation process, hence drying of peatland due to climate change could convert peatland from carbon sink to source. Under global warming scenarios, the response of SOM decomposition in different soil aggregates to temperature change is unquestionably important for predicting the global pattern and magnitude of SOC storage in the future (Fang *et al.*, 2005)

Cropping systems have a significant effect on SOM content and quality. Consequently, small changes in the SOM content are significant to the agricultural production of the region. Intensive agricultural practices change SOM characteristics greatly, and are generally associated with a substantial loss of SOC (Ding *et al.*, 2002) overtime. Tillage aerates the soil and raises its average temperature, thereby contributing to an increased rate of organic matter decay due to accelerated oxidation and mineralisation (Wild, 2003). Loss of organic matter also occurs because erosion washes away topsoil and humus. Overall, arable cropping returns less organic matter to the soil than native vegetation does. It is well-known that some agricultural practices cause SOM decline, but this is also occurring in natural and semi-natural areas where agricultural influences are weaker (Baritz *et al.*, 2004).

Unlike Northern Europe, Mediterranean agriculture is mainly focused on the production of fresh fruit, citrus, olives, grapes, vegetables and cereals (wheat, barley and corn). Most of these soils are characterized as having low organic matter content. As mentioned above, the major sources of organic matter in agricultural areas traditionally comes from tillage residues, stubble, waste plant pruning, or animal manure. It has been proven that the replacement of the organic matter in arable crops is not enough due to specialization and monoculture. This specialization has led to the separation of livestock and arable crops, and consequently the livestock/cropping rotation farming practices have disappeared resulting as decline of soil organic matter.

The accumulation of organic matter in the soil is a slow process and enhanced through appropriate techniques such as conservation agriculture, organic farming, permanent pastures, and cover crops, applications of manure and compost, crop and cultivation in terraces using contours. These techniques have proven their efficiency when preventing erosion, increasing fertility and enhancing soil biodiversity. This historical land use includes the conversion of grassland, forests and natural vegetation to arable land, deep ploughing of arable soils, intensive tillage operations, over-fertilization (Jones *et al.*, 2012), drainage, liming, fertilizer use and tillage of peat soils, crop rotations with reduced proportion of grasses, soil erosion, and wildfires. The latter two are of particular importance in Mediterranean countries (Shakesby, 2011).

Most Catalonian soils have a long tradition of agriculture use, for this reason the organic matter is scarce but stable, therefore its loss will be at a low rate (Alcañiz *et al.*, 2005). Virgili (1994) and Costa (2004) observed that most of the soils in Catalonia range between 1.4 – 4.5 % organic matter, depending on the land use and land management. Bowyer *et al.*, (2008) argues that soils with less than 1.7% of organic matter should be considered as in pre-desertification process;

hence several areas in Catalonia would be classified as at risk. A soil with low organic matter implies a decrease in fertility, structure, and is therefore, more exposed soil degradation.

The extent of SOM decline is unacceptable and has implied consequences for soil functions and the services which these support, such as food production, groundwater protection, and other biodiversity conservation. EC (2006a) in the Soil Thematic Strategy for soil protection propose a base content of SOM below which further SOM declines may lead to irreversible damage to soil functionality, (Eckelmann *et al.*, 2006). These thresholds or critical limits should reflect soil types, soil characteristics and specific use. The definition of SOC thresholds is very problematic since some soils have naturally low SOC, with a very small likelihood of further SOC losses, while some soils with intermediate SOC contents may be at high risk of continuing losses. Notably, technology is available that supports sustainable management of soil with low SOC levels (Van Camp *et al.*, 2004).

Table 2. Proposed maximum and minimum thresholds.

Soil > 8% COS	Drained, current or formerly wet soils under arable crops or intensive livestock management.
Soil < 2% COS	Arable soils, in particular those that are managed in continuous arable production, especially where tillage is intensive.

Source: Eckelmamm *et al.*, 2006.

SOC stocks are regulated by inputs (decrease vegetation, biomass decreased as decreasing the accumulation of organic matter) and outputs (erosion, leaching, heterotrophic respiration, dispersion and solubility the soil organic matter due to salinity), therefore a modification of these will result in a variation in the SOC stock. The highest concentration of organic carbon is in the first 30 cm of soil, in the topsoil horizon. This horizon is more susceptible to wind and water erosion, causing the loss of organic matter. Decline of organic carbon is one of the threats described in described in Soil Thematic Strategy (COM(2006) 231) (EC, 2006a). In the ENVASSO project it was reported that the most appropriate soil quality indicators for SOM status were 1) Topsoil organic carbon content; 2) Soil organic carbon stocks and 3) Peat stocks. Alternative options such as ‘total carbon stocks down to 1 m depth’, ‘soil organic matter profiles down to 1 m depth’ and ‘soil organic matter stratification ratio’. However, the geographical coverage of SOC measurements to this depth in existing soil monitoring networks is very poor however their introduction could be achieved but would require novel approaches utilising new mapping technologies to derive this indicator. Historically reporting for carbon emissions has been limited to the top 30 cm due to policy requirements.

1.8 *Processes of soil degradation: desertification, application in Spain and Catalonia*

According to the United Nations Convention to Combat Desertification (UNCCD) definition, desertification comprises land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors including climatic variations and human activities. UNCCD argues that desertification is a consequence of physical, biological, political, social, cultural and economic interactions (Kannan, 2012). Subsequently it is an issue of sustainable development, because it is a sign of a breakdown of the balance between the natural system and socio-economic system. Climate change affects directly to desertification and erosion however moreover as aridity, drought, among others, do not explain alone the phenomenon of desertification. Overall, climate change accelerates soil degradation processes such as desertification, soil erosion and salinisation.

Global assessments of land degradation estimate 15% of the world's total land area shows evidence of damage, mainly a consequence of erosion, nutrient loss, salinisation and physical compaction (Wild, 2003; Bowyer *et al.*, 2008). Desertification currently affects about one-sixth of the world's population and one-quarter of the world's land (Bullock and Houérou, 1995), 6 to 7 million hectares (Mha) are lost annually due to soil erosion, and up to 20M ha of irrigated land are affected by salinisation (World Resources Institute, 1998). Most of these areas experience water deficits are home to approximately 37% of the world population lives. Desertification is threatening the survival of population in the developing world, with special emphasis to the continents of Africa and Asia, where are most of the arid areas of the planet (Millenium Ecosystem Assessment, 2005).

The impacts of the different degradation processes vary across Europe with southern European countries generally considered to experience the most severe water erosion linked to extreme and intermittent rainfall (Bowyer *et al.*, 2008). The Mediterranean area is identified as sensitive to desertification due to a combination of climate conditions, soil and terrain characteristics, agriculture and exploitation of water resources (Castillo *et al.*, 2004). Socio-economic factors have played a key role in the transformation of the landscape; the abandonment of marginal land and sometimes of whole communities, the practice of intensive agriculture, overexploitation of water resources and the loss of land for housing developments are facts that considerably affect desertification (Rosell *et al.*, 2004). More than half of the Mediterranean region registers a degree of aridity more or less pronounced. 70% of landscapes records a moderate desertification risk, while the remaining 30% is affected by other degradation processes including *inter alia* erosion, reclamation of marginal lands, fires, abusive groundwater exploitation, seawater intrusion, salinisation, contamination by pesticides, soil acidification, land use changes (López Bermúdez, 2001).

Desertification in Spain is mainly generated by soil erosion caused by water. Here, a good part of desertification is due to the ancient destruction of vegetation in a hostile and fragile natural environment. The only definitive remedy is the reconstruction of the protective vegetal cover (Martínez-Fernández *et al.*, 2005). Desertification was considered initially a key issue within the threat Soil Erosion and this has been recognized by many scientists and some governments as a major problem (Bullock, 1995). Centro de Investigaciones sobre la Desertificación (CIDE) in Spain reported that 37% of the country suffers a level of desertification and in some areas; soil erosion could over raise in 40% the thresholds accepted (Suárez, 2013). Spain, having ratified the UNCCD, drafted the Programme of Action against Desertification (PAND), which aims to contribute to sustainable development in arid, semi-arid and dry sub-humid areas of the country, and the prevention or reduction of land degradation, rehabilitation and recovery of the affected lands (2000). Simulatenously, the Biodiversity General Directorate in Spanish Government took action with a national inventory of soil erosion.

Since 1981 the LUCDEME project is underway to combat desertification in the Mediterranean. There were different aims in the framework of the LUCDEME such as a Network of Monitoring Stations and an evaluation of erosion (RESEL). RESEL had the aim of obtaining real data in relation to erosion and the impact on certain measures. Boix-Fayos *et al.* (2005) reviewed a large number of erosion studies in south-east Spain, which is one of the most severely affected areas in Europe.

The Soil Thematic Strategy Technical Working Group on Soil Erosion undertook a detailed analysis of the monitoring of soil erosion and concluded that it should be an indicator-based approach (Vandekerckhove *et al.*, 2004). In 2008, ENVASSO project reported a selection of the most appropriate soil quality indicators according to the soil threats described in the Soil Thematic Strategy (COM(2006) 231) (EC, 2006a). One of the indicators selected for erosion was the ‘estimated soil loss by rill, inter-rill, and sheet erosion’ (EC, 2006a). An accurate estimation of soil loss can be obtained from erosion models that already exist, for example: PESERA (Kirkby *et al.* 2004); USLE (Wischmeier and Smith, 1978); RUSLE (Morgan *et al.*,1984; Morgan, 2001 and Renard *et al.*, 1997).

In the context of the EC Project MEDALUS (Mediterranean Desertification and Land Use), the focus was primarily on European Mediterranean environments where physical loss of soil by water erosion, and the associated loss of soil nutrients, was identified as the dominant problem. In more arid areas, there is greater concern with wind erosion and salinisation problems, but these are considered to be less significant than water erosion for the northern Mediterranean area (Kirkby and Kosmas, 1999). Some indicators proposed in previous projects (MEDALUS project) demonstrated that desertification may have proceeded to a state where soils are infertile, or highly sodic, or the land has become a rock desert. Kirkby and Kosmas (1999) concluded that the most useful indicators are those that predict the potential risk of desertification while there is still time and scope for remedial action. ENVASSO project reported (Huber *et al.*, 2008) a selection of the most appropriate indicators for desertification: 1) Vulnerability to desertification; 2) Wildfires (burnt land area); 3) Soil loss from burnt areas; 4) Soil organic carbon content; 5) Salt content and 6) Soil biodiversity decline. The vulnerability to desertification could be assessing using the MEDALUS approach.

Land use in Catalonia is largely dedicated to agricultural and forestry, which ultimately could lose production capacity. Soils with woody crops such as vineyard, olives trees, and nuts trees along with irrigated crops are the most susceptible to degradation and salinisation. Poorly managed forestry is prone to fires and also to soil degradation, in particular soil erosion.

2 GENERAL OBJECTIVES AND STRUCTURE OF THE THESIS

This PhD builds upon approaches recommended under the European framework, and refines the use of potential indicators to assess the functional aspects of environmental quality and degradation, in particular of soils.

These selected indicators were tested in two different areas in Catalonia. One was located on the left side (N) of Ebro Delta (Delta de l'Ebre region), and the other was located between Canalda and Odèn (Solsonès region). These areas were chosen based upon their differences in soil quality, land use and environmental conditions.

The general objective is:

- To test several approaches to characterize the spatial variability of soil quality and land degradation as defined in the EU framework based upon detailed field surveys in model areas.

The hypotheses include:

- That temporal and spatial distribution of the soil salinity in the Ebro Delta can be assessed through field work using electomagnetic sensors (EM-38) together with mapping and geostatistical techniques.
- The SOC stock in mineral soils, in the main soil types and land uses, of a study area in the Iberian Pre-Pyrenees (Canalda river basin) can be obtained and quantified in depth; using an on-purpose soil survey for SOC mapping included in the ordinary survey for a detailed soil map (1:25,000).
- Different land suitability or degradation degree in a 10 km^2 – model mountainous area can be identified with the use of different models dealing with: i) Environmental Sensitive Areas; ii) Land suitability or land capacity; iii) Quantification of erosion; and checked with the actual land performance.

STRUCTURE OF THE THESIS

This document consists of five parts, three of them are independent chapters presented in the format of a journal article, the other two are a general introduction and general conclusions. For this reason, some parts, such as the material and methods section of the three inner parts, may contain a certain degree of repetition. Each chapter has its own reference section with the citations referred to. A brief description of the contents of each chapter is described below:

Chapter 1. General Introduction: Mapping soil quality and processes soil degradation.

In this chapter the concepts of soil quality, soil degradation, land evaluation and soil indicators are reported. In the frame of the European environmental legislation soil threats and soil indicators are discussed. Issues as scale and different approaches on modelisation are presented for the purpose of studying soil salinity, soil organic carbon, desertification and erosion.

Chapter 2. Soil salinity monitoring in the Ebro delta: temporal and spatial variability.

This chapter focuses on the study of the temporal evolution and spatial distribution of the soil salinity and sodicity in the Ebro Delta in 12 years time, with the use of an electromagnetic sensor (EM- 38) and chemistry data. This chapter was partly presented in a poster format in the 5th International Conference on Land Degradation:

> **Simó, I.**, Poch, R.M., Grañana, S., Boixadera, J. 2008. Soil Salinity monitoring in the Ebro Delta: Temporal and Spatial distribution. 5th International Conference on Land Degradation. Italy.18-22 September 2008.

Chapter 3. Quantifying, modelling and mapping soil organic carbon stock in depth. A case Study in a Mediterranean mountainous area (Catalan Pre-pyrenees).

In this chapter is quantified and mapped soil organic carbon stock at different depths. The mapping was done using different techniques and compared each other. Part of this chapter was already presented as poster presentation in 19th World Congress of Soil Science:

> **Simó, I.**, Poch, R.M., Herrero, C., Boixadera, J. 2010. Accuracy of soil organic carbon inventories in Mediterranean mountain areas. 19th World Congress of Soil Science, Soil Solutions for a Changing World. Brisbane, Australia. 1 – 6 August. http://www.iuss.org/19th%20WCSS/Symposium/pdf/1.5.1.pdf

Another part was presented as oral presentation and published in the proceedings book of the 1st GlobalSoilMap Conference:

> **Simó, I.**, Herrero, C., Boixadera, J. and Poch, R.M. 2014. Modelling soil organic carbon stocks using a detailed soil mapping a Mediterranean mountainous area. In: Arrouays, D., McKenzie, N., Hempel, J., Richer, A., McBratney, A. (eds).GlobalSoilMap: Basis of the global spatial soil information system- Proceedings of the 1st GlobalSoilMap Conference. Taylor & Francis Group. UK. pp. 421-427

Chapter 4. Land evaluation of a study area applying GIS-based approaches using MEDALUS model, Agrological classes and RUSLE model: A case study in the Catalan Pre-Pyrenees.

In this chapter we describe the different sensitivity to land suitability or degradation of the study area using different quality data and different approaches. This chapter was partly presented as poster in a national conference, Congrés de Sòls de Muntanya i Canvi Global:

> **Simó, I.**, Poch, R.M., Oller, M., Boixadera, J. 2010. Delimitació de zones de risc a la desertificació d'una àrea pilot situada a la comarca del solsonés, mitjançant tècniques SIG. Congrés de Sòls de Muntanya i Canvi Global. Catalonia-Spain 14-17 July.

Chapter 5. General discussion

This chapter outlines the findings back in chapter 3, chapter 4 and chapter 5 related to the general introduction and describes the cross-fertilization between chapters.

Chapter 6. General conclusions

The closing chapter outlines the main conclusions out of the previous research reported in chapter 3, chapter 4 and chapter 5 and the recommendations for future work.

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Chapter 2

MONITORING SALT AFFECTED SOILS IN THE EBRO DELTA:

TEMPORAL AND SPATIAL VARIABILITY

List of acronyms and abbreviations

EM	Electromagnetic induction
EMv	Uncorrected reading in EM-38 at the vertical orientation of the coils
EMh	Uuncorrected reading in EM-38 at the horizontal orientation of the coils
EC	Electrical conductivity (dS m^{-1})
ECe	Electrical conductivity of the saturation extract (dS m^{-1})
$EC_{1:5}$	Electrical conductivity of a 1 to 5 soil to water extract (dS m^{-1})
ECa	Apparent soil electrical conductivity (dS m^{-1})
$ECa_{1:5}$	Apparent soil electrical conductivity of a 1 to 5 soil to water extract (dS m^{-1})
ECh	Reading in EM-38 divided by 100 at the horizontal positions of the coils
ECe estimated	Electrical conductivity of the saturation extract (dS m^{-1}) estimated using $EC_{1:5}$
GPS	Global positioning system
SAR	Sodium adsorption ratio
ESP	Exchangeable sodium percentage
pHe	pH of the saturation extract
$pH_{1:2.5}$	pH of a 1 to 2.5 soil to water extract
Ca^{++}	Calcium in the the saturation extract
Mg^{++}	Magnesium in the the saturation extract
K^+	Potassium in the the saturation extract
Na^+	Sodium in the the saturation extract
Cl^-	Clorur in the the saturation extract
SO_4^{-2}	Sulfate in the the saturation extract
CO_3^{-2}	Carbonate in the the saturation extract
HCO_3^-	Bicarbonate in the the saturation extract
NO_3^-	Nitrate in the the saturation extract
PTF	Pedotransfer function
PCA	Principal Components Analysis
MPE	Mean Predicted Error
SDPE	Standard Deviation of the Prediction Error
RMSPE	Root Mean Square Prediction Error
R^2_p	Prediction coefficient of determination
OK	Ordinary Kriging

MONITORING SALT AFFECTED SOILS IN THE EBRO DELTA: TEMPORAL AND SPATIAL VARIABILITY

1 INTRODUCTION

Soil salinity or salt affected soils occurs when salts more soluble than gypsum are present excessively in the soil solution, to the point that it affects plant growth. It can occur either by natural processes, or by human activities that disturb natural ecosystems, for instance, by irrigation. Salinisation refers mainly to chlorides or sulphates of sodium, magnesium, potassium and/or calcium (but not gypsum), in soils. Soil sodicity is the excessive amount of adsorbed sodium with respect to adsorbed calcium and magnesium in the soil's exchange complex, poses critical agricultural and environmental hazards because of its negative effects on soil structural stability and hydraulic conductivity (Amezketa, 2007).

Soil salinity and sodicity can occur naturally in low, poorly drained areas, in hot and dry climates, where surface water is collected and evaporated. In addition, these facts can be exacerbated by human activities, in particular due to inadequate irrigation of agricultural land (European Commission, 2006b, 2012). Due to the high speed of the reactions involved (dissolution/reprecipitation), salinity has a high temporal and spatial variability. The salinity can be assessed according to crop production decline or by the presence of halophytic vegetation, but the use of analytical indicators, (Huber *et al.*, 2008; Jones, 2008) such as electrical conductivity (ECe) in soil extract at 25ºC and the sodium adsorption ratio (SAR), are the ones used for assessing salinity and sodicity, respectively, that allow the application of temporal monitoring (Herrero, 2004).

Soil salinisation is one of the soil degradation processes that significantly decreases the soil quality. The Commission of the European Communities established a communication called Towards a Thematic Strategy for Soil Protection (European Commission, 2002), being salinisation one of the eight main threats identified, besides erosion, organic matter decline, contamination, compaction, soil biodiversity loss, sealing, landslides and flooding (European Commission, 2002; 2006a; 2006b, 2012).

The processes by which soluble salts cause salinity include: (a) the application of waters containing salts; (b) mineral weathering in soils; (c) saline water table or marine aerosols. The processes for sodic soils are different than saline soils but they are related. Sodic soils contain a large amount of sodium cations attached to clay particles. The accumulation of dispersive cations such as Na in the soil solution and the exchange phase (K, Mg, Ca) affects the physical properties of the soil, such as the structural stability, hydraulic conductivity, infiltration rate and erosivity (Navarro-Pedreño *et al.*, 2007). The physical behaviour of salt-affected soils has previously been described in terms of the combined effects on soil salinity including Richards (1954); Szabolcs (1989) and Navarro-Pedreño *et al.* (2007).

Some effects of salinity are manifested in loss of stand, reduced rates of plant growth, reduced yields, and in severe cases, total crop failure (Rhoades and Loveday, 1990). Salinity limits water uptake by plants by reducing the osmotic potential and thus the total soil water potential. Salinity may also cause specific ion toxicity or upset the nutritional balance (Corwin and Lesch, 2003).

Soil salinity is measured by the electric conductivity in soil extract at 25ºC (ECe) (USSL, 1954). When salts more soluble than calcium carbonate and gypsum are present in the soil and affect crop growth and/or crop yield, these soils are considered salt affected. Most of these soils have an ECe of more than 4 dS/m at 25ºC. In some cases, salinity is linked with soil sodicity.

The sodification process involves the presence of soluble sodium salts in the soil solution and sodium adsorption in the exchange complex (Van-Camp *et al.*, 2004), that can be assessed by measuring the soluble Na^+ concentration relative to soluble divalent cation concentration in soil solution, i.e. sodium adsorption ratio (SAR), which is related to the Exchangeable Sodium Percentage (ESP) (Qadir *et al.*, 2002). Too much sodium leads to excessive swelling of the soil, which may result in a structural collapse referred to as dispersion. Therefore, both ECe and SAR or ESP are needed to predict soil structural stability degradation (Amezketa, 2007).

Salt and sodium affected soils can be classified in terms of the dominant management problem as:

- Saline soils: High salt content, ECe ≥ 4 dSm^{-1} at 25ºC.
- Sodic soils: Soils with high sodium content in the exchange complex. ESP > 15 or SAR > 13, pH > 8.5 (typically 9-9.5, referred as alkaline soils).
- Saline-sodic soils. Soil high in both salt content (ECe ≥ 4 dSm^{-1} at 25ºC) and adsorbed Na (ESP > 15 or SAR > 13).

Soils with SAR values well below these thresholds have been found to be prone to clay dispersion and unfavourable physical conditions when the soil solution concentrations are below their flocculation values (Amezketa *et al.*, 2003). Thus, both ECe and SAR are needed to predict soil structural stability.

There are several methods that have traditionally been used for determining soil salinity at catchment scale, mainly: a) visual crop observations; b) ECe, the reference method to measure the soil salinity which is used for plant tolerance to salinity, production, and water management (i.e., leaching requirement, crop pattern, etc) and; c) non-invasive measurement of electrical conductivity with electromagnetic induction (EM). The technique of EM measures the apparent soil electrical conductivity (ECa). For soil salinity, ECa measurement should be calibrated against the standard ECe which is used in salt-tolerant plant studies. The laboratory method of ECe is expensive, time consuming, and tedious (e.g., sampling, soil preparation, and measurement) (Bouksila, 2011). The application of Electromagnetic Induction (EM) measurements of ECa in soil science first appeared in late 1970‘s and early 1980‘s in efforts to measure soil salinity (Corwin and Rhoades, 1981; Corwin and Rhoades, 1982). The EM-38 (Geonics Ltd. Mississauga, Ontariom Canada) is considered one of the best methods for soil salinity measurement in a geospatial context (Corwin and Lesch, 2003; 2006; Terron *et al.*, 2011). EM-38 is easily hand-held because it weighs only about 3.6 kg and is only 1.05 m long. The EM-38 is designed to measure salinity in the root zone.

Most salinity problems in paddies occur in river deltas or other coastal areas, and are caused by seawater intrusion or low-quality irrigation water; however, few studies have used electromagnetic induction (EM), to measure the soil salinity of paddies (Enrique *et al.,* 2005; Li *et al.,* 2013, Herrero and Hudnall*,* 2014). Rather, most EM surveys in paddy plots have been conducted on saline-sodic soils in humid areas for purposes other than soil salinity mapping (Herrero and Hudnall*,* 2014).

Most soil properties of scientific interest vary continuously in space and time, and cannot be measured or recorded everywhere. To represent their variations, the values of individual variables or class types at unsampled locations must be estimated from information recorded at sample sites. Geostatistics is the tool to deal with this continuous spatial variation (Navarro-Pedreño *et al.,* 2007), since it represents a great aid to explore/model patterns of space/time dependences between and within soil data (Qadir *et al.*, 2000).

For almost 50 years many attempts have been made to predict some complex soil properties from some easily available soil properties using empirical models. In soil science, such empirical models are named pedotransfer functions (MacDonald, 1998; Krogh *et al.*, 2000; Rashidi and Seilsepour, 2008). A pedotransfer function is a mathematical equation between two or more soil parameters which shows a reasonably high level of correlation coefficient. This relationship is used to facilitate the estimation of a non-measured soil parameter from one or more measured ones. It may also be useful for the prediction of the evolution of a soil parameter under different future conditions. Many mathematical models have been developed to predict soil salinity (Raes *et al.*, 2002; Srinivasulu, 2004; Askri *et al.*, 2010). These models usually need a significant number of input parameters, for example, pH and Base Saturation, Cation Exchange Capacity (CEC) and clay and organic matter contents, or salinity, pH and Exchangeable Sodium Percentage (ESP). For instance, Sodium Adsorption Ratio (SAR) are often determined using laborious and time consuming laboratory tests, but it may be more suitable and economical to develop a pedotransfer function. Pedotransfer functions have been developed to increase cost effectiveness, because they reduce the effort and cost involved in soil sampling and laboratory analysis. They are based on general data sets and verification with ‘true’ field data is often lacking. Their general character can produce a high level of uncertainty (Finke *et al.*, 1996).

Previously, researchers also reported a relationship between soil SAR and soil EC (Richards, 1954; Emerson and Bakker, 1973). However, these pedotransfer functions have been shown not to be constant, but to vary substantially with both solution ionic strength and the dominant clay mineral present in the soil (Shainberg *et al.*, 1980; Marsi and Evangelou, 1991; Evangelou and Marsi, 2003; Rashidi and Seilsepour, 2008). Therefore, the pedotransfer functions are not constant and should be determined directly for the soil of interest.

Some authors such as McBratney *et al.* (2000) argue that there is a need to improve estimates of the horizontal variation in soil properties through improved geostatitsical techniques but they emphasized also, the importance of identifying the vertical variation in soil attributes within a profile, and Li *et al.* (2013) discussed the importance of understanding the vertical variation in soil salinity and the reasons why there have been few three-dimensional studies of soil salinity.

During recent years, land use in salt affected soils has undergone a change from reclamation for agricultural production to preservation for nature conservation and biodiversity. The emphasis now is to keep these salt affected lands as protected areas, maintaining the original biota, flora, fauna and rural life (Huber *et al.*, 2008). Currently, more than 900 million hectares (ha) worldwide are affected by soil salinity, of which 23% of these lands are under agricultural use, and a further 10% are considered saline-sodic agricultural lands (Szabolcs, 1989).

The soils under Mediterranean climate are the most likely to be affected by increased salinisation if temperature increases and precipitation decreases. Salt affected soils occupy approximately 2.8 M ha of the Mediterranean, and are found mainly in the dry areas of the Iberian Peninsula (Ebro Valley, Castilla, and Mediterranean coast) (Van-Camp *et al.,* 2004), with areas in south-western

France, Italy, Sicily (Dazzi and Fierotti, 1996), Sardinia, Corsica, Romania, Hungary and the Dalmatian coast of the Balkans.

In Spain, in particular, 3% of the 3.5 million hectares of irrigated land is severely affected by salinity, and another 15% is under serious hazard. There are no estimates on the total economic cost of this phenomenon nor how global change will affect it. In Catalonia, most of the saline soils are located on the Mediterranean coast, in the Llobregat and Ebro deltas. The Ebro Delta has a socio-economic and environmental interest in soil salinity, as the delta formations are linked to the salt accumulation because of the deltaic cycle. In Catalonia, the delta formations are areas of great importance because of the existing fauna and the agricultural soil use. These soils have benefited from the fertile depositions of sediments that naturally result from flooding. Most of these areas have plenty of water and are favourable for irrigated agriculture. In the Ebro Delta the main land use is irrigated paddy rice and vegetables, and the water for irrigation comes from the Ebro River. At the present time, this delta is in recession because of the number of reservoirs existing along the Ebro River (Térmens, 2014).

The soil salinity of protected wetlands, which often is unknown, must be maintained if the conservation of the ecosystems is to be achieved. The knowledge of soil salinity is also important if land is to be used for agriculture. Wetland rice cultivation is spread over the major part of the Ebro Delta, accounting for more than 77% of land use. Under present conditions, one rice crop per year is harvested, yielding more than six tonnes per hectare of paddy rice. Rice production relies on water from the Ebro river. The “Ebro Delta Natural Park” was established in 1986 to preserve and improve the natural inheritance of the site, including the abundant lagoons (16% of the area) occupied by *Phragmites australis*. The Delta plays a key role as a nature reserve with over 500 recorded flora species, an extensive range of entomological fauna and its internationally important 300 bird species (60% of all the European species) (Casanova, 1998).

Characterizing spatial and temporal variability of soil properties at field and landscape scales is tremendously important for a variety of agronomic and environmental concerns. A soil quality assessment study was conducted on salt affected soils or soil salinity in a pilot area in the Ebro Delta. The main hypothesis of this study was that the salt content was changed over a 12 year time period. Hence, the main goal of this study was to assess and monitor the spatio–temporal distribution of salt affected soils in the Ebro Delta, through field work using electomagnetic sensors together with mapping and geostatistical techniques; and the use of it to assess salinity and sodicity evolution under future scenarios by developing pedotransfer functions.

2 MATERIALS AND METHODS

2.1 *The study area*

This study was carried out in agricultural soils in the south of Catalonia, located in the Ebro Delta. The Ebro Delta is one of the most important wetland in the Western Mediterranean. It occupies an area of 32000 ha, of which approximately 21500 ha is cultivated, while the remaining area is occupied by natural spaces (including beaches, marshes, lagoons, etc.). The area is divided into two sides; the northern side (left bank, where our study site is located, covering 102 km^2) and a southern side (right bank). The current delta dates from the end of the last ice age when static changes lead to its growth. Sediment was transported by the Ebro river and acted upon by marine, wind and river processes while successive changes in sea level allowed progressive and rapid growth over the sediment platform. Thus the delta is a relatively recent feature on the Mediterranean coastline.

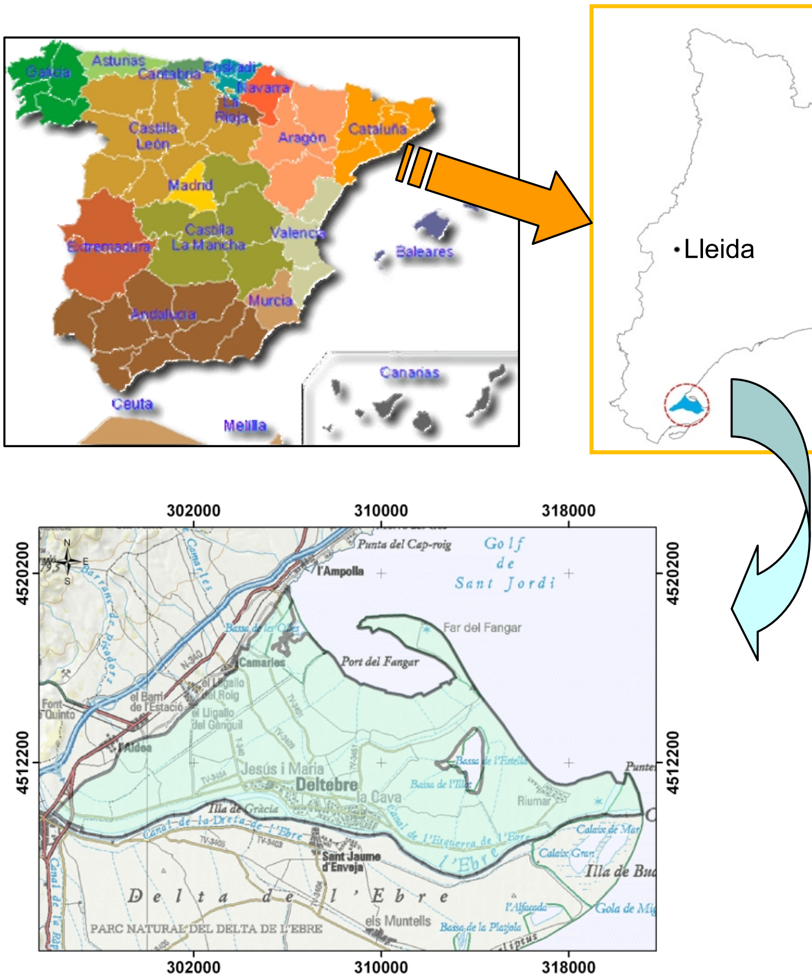


Figure 2. Location of the study area.

The Ebro Delta has a Mediterranean climate, with temperatures seldom higher than 35ºC or lower than 0ºC. The annual average temperature is 16.2ºC, with a monthly maximum average of 24.2ºC in August and a minimum of 9ºC in January. The yearly average rainfall is 530 mm, very irregularly distributed, being the wet season in spring and autumn and the dry season in winter and summer.

Figure 3 shows the average weather conditions and the Ebro river flow discharges prevailing in the study area over a time period of 20 years. There are evidences of very dry years with annual precipitation < 400 mm, and mean temperature > 18°C (1995 and 2009). The annual Ebro river discharges (m^3/s) reported in the graph show remarkable fluctuations, with apparent lower values at the end of the period (2005 – 2010).

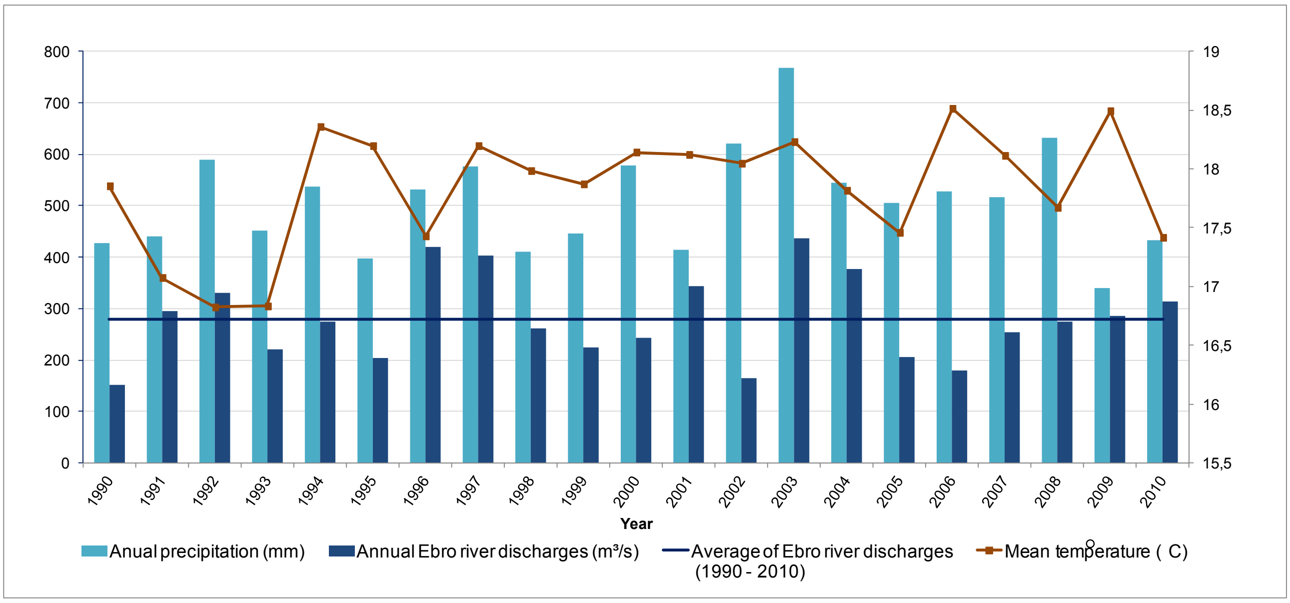


Figure 3. Average weather conditions and Ebro river flow prevailing in the study area over 20 years. Data source: Ebro River Authority (SAIH, 2015).

The Ebro Delta is the most vulnerable part of the Ebro River, altered by the drastic reduction of water and sediment flows, due to the construction of several dams and the corresponding increase of water demand and river regulation (Sierra *et al.,* 2004). The groundwater has a marine origin in its outer part, although it mixes with water from continental origin, therefore its electrical conductivity is highly variable (from 2 to 60 dS·m^{-1}). This phenomenon is known as saline wedge. This saline wedge could go 32 km^2 inland from the estuary of the Ebro river (Ibañez, 1993), consequently, the irrigation water could be affected by salinisation. The EC (electrical conductivity) of the irrigation water during the rice growing season in 1994-1996 was, on average, 1.055 ± 0.088 dS·m^{-1} at 25ºC, and in 2007-2008 was, on average, 1.052 ± 0.27 dS·m^{-1} at 25ºC. The EC of the irrigation water was always below 1.2 dS·m^{-1} in 1998 year (Casanova, 1998).

Notably, in 1998, the Catalan rice growers welcomed massively (96% of the total) for agroenvironmental aid from the European Union. This aid promoted the conservation measures or Best Management Practices (BMPs) for rice production implemented primarily for the purpose of protecting the environment (enhance yields, wildlife habitats, improve overall production, sustainability of soil and water sources). Mostly it was applied to wetlands included in the Ramsar Convention, as in the case of the Ebro Delta. One of the requirements of these environmentally friendly methods was to keep flooded acreage for an additional four months during the autumn – winter. The major change of the agroenvironmental aid was to leave the paddy rice flooded for a longer period of time than it was before, thus, from September to mid January. This aid was applied since 1998 to nowadays, however it will change in 2016 because of the problems caused by the apple snail (*Pomacea maculate)* the paddy rice of the Delta Ebro.

A scheme of the rice farming activities before than 1998 year in the Ebro Delta is presented in the Figure 4.

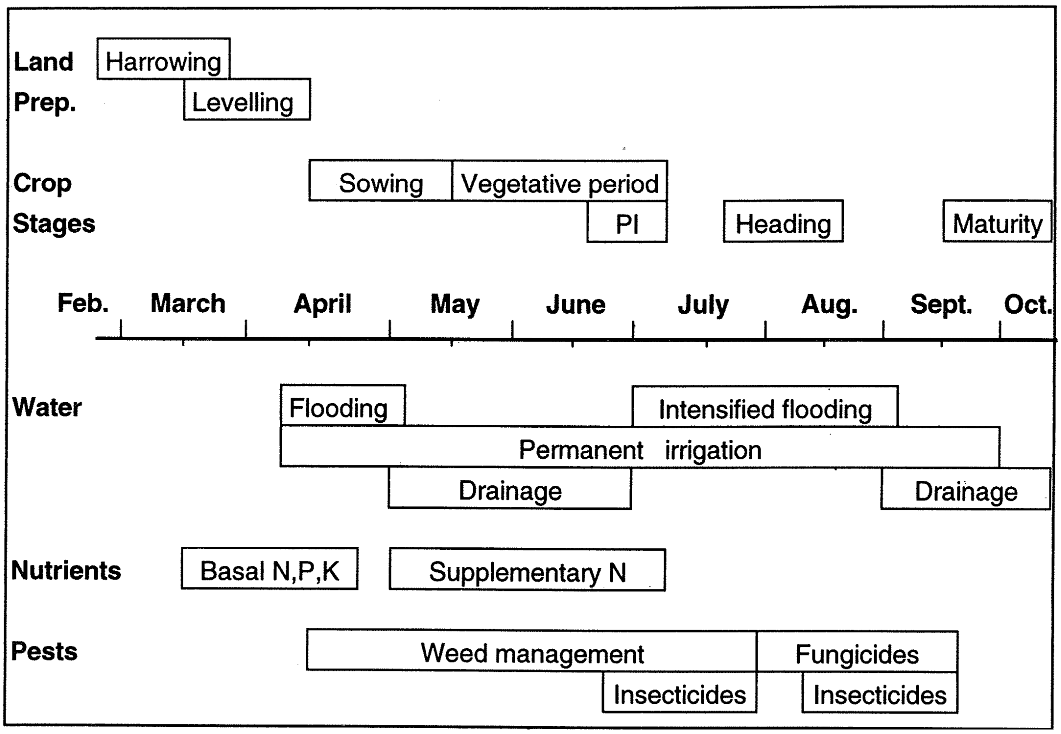


Figure 4. Rice farming calendar in the Ebro Delta before than 1998 year. Data source: Casanova (1998).

Regarding the life cycle of the rice plant, the following stages are distinguished: sowing, panicle initiation (PI), heading and maturity. According to this life cycle, the water flooding and the irrigation water stages were important from April to ending September in 1994-1996, however, these stages changed with the agroenvironmental aid from the European Union, removing the drainage period from September to October, and promoting to keep flooded acreage for an additional four months from October to January. In 2008 the irrigation water canals started to be improved and expanded, improving then the water irrigation plan.

The main characteristic of paddy soils is that paddy soils are highly modified by anthropogenic activities. The formation of these Anthrosols is induced by tilling the wet soil (pudding), and the flooding and drainage regime associated with the development of a plough pan and specific redoximorphic features. This anthropogenic cycle might be accompanied by the progressive loss of clay during paddy cultivation and migration into the bottom layers (Kögel-Knabner *et al.*, 2010). The migration of clay into the plough layer, in addition to mechanical pressure exerted by tractors, contributes to forming a plough pan (Li, 1992; Kögel-Knabner *et al.*, 2010). Repeated pudding over many years increases the plant available water capacity in the puddle layer, but reduces the small coarse meso pores in the plough pan (Janssen and Lennartz, *et al.*, 2006; Kögel-Knabner *et al.*, 2010).

In the Ebro Delta, the dominant soils are Fluvisols, Arenosols and Calcisols (IUSS Working Group WRB, 2014), however, the paddy soil are affected by soil salinity in some places, through the deltaic cycles. They are complex because of the interaction between sea water, water from the Ebro River and the water table. The hydrological functioning of the Delta ponds is very different from most Mediterranean coastal lagoons, due to the artificial water regime of the cultivation of rice. In most of the coastal lagoons, where the hydrological regime is not manipulated by humans, salinity of water is lower in the months of autumn and winter. However, the salinity of Delta ponds is lowest in the spring and summer (driest) months. This is because fresh waters are used on paddy fields for rice cultivation and later (autumn and winter) is mixed with sea water of ponds entering through the canals or by infiltration, as a way of management (Figure 5). This fact affects salt affected soils to different degrees.

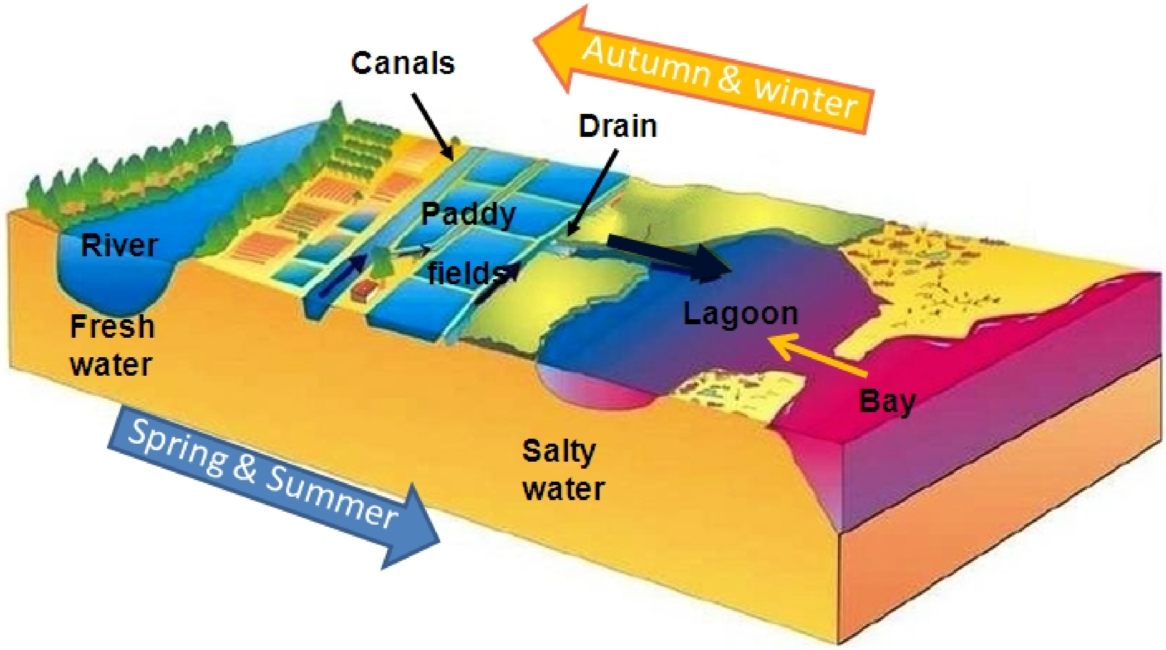


Figure 5. Hydrologycal cycle and most important components in this cycle. Source: Gonzalez, A., 2008.

2.2 *Electromagnetic sensor management*

The salt affected soil and its evolution over time must be investigated, either by direct soil sampling or by other measuring techniques. The most feasible, for most situations, is to survey by electromagnetic induction (EM) measurements. Hendrickx *et al.* (1992) argued that the EM has become the first choice for measuring soil salinity or salt affected soils in a geospatial context because (i) measurements can be taken as quickly as one can move from one location to the next, (ii) the large column of soil measured reduces local scale variability, and (iii) measurements in relatively dry or stony soils are possible because no contact is necessary between the soil and the EM sensor.

The EM-38 (Geonics Ltd., Ontario, Canada), is a hand-held sensor that has two parallel coils, which reads the electromagnetic response of its surrounding space. For a given soil, this response varies with moisture, salt content, and temperature. Obtaining EM-38 values are easy because the instrument does not need to be in contact with the soil (Herrero *et al.*, 2011). The EM-38 has an intercoil spacing of 1 m, this allows a penetration depth of about 1 m and 1.5 m in the horizontal (EMh) and vertical (EMv) dipole orientations respectively (Corwin and Lesch, 2003). However, several factors influence Eca besides soil salinity, as water content, porosity, structure, temperature, clay content, mineralogy, cation exchange capacity, and bulk density (McNeill, 1980; Friedman, 2005; Rodriguez-Pérez *et al.*, 2011). Therefore, for accurate ECa and ECe calibration, the EM-38 measurements are preferably made at field capacity and in a specific soil type (McKenzie *et al.*, 1989; Herrero and Aragüés, 2003).

The EM-38 must be calibrated by determining soil salinity from soil samples taken immediately after the EM reading at some of the surveyed sites in order to obtain a valid soil salinity survey. The EM readings must be conducted at adequate soil moisture content to maximize the harmonic structure of the emitted electromagnetic wave at each wave emission location. Soil salinity can vary within a few meters in the area of interest, which compromises the validity of representative auger samples. EM readings can reduce this problem, because of the larger soil volume explored.

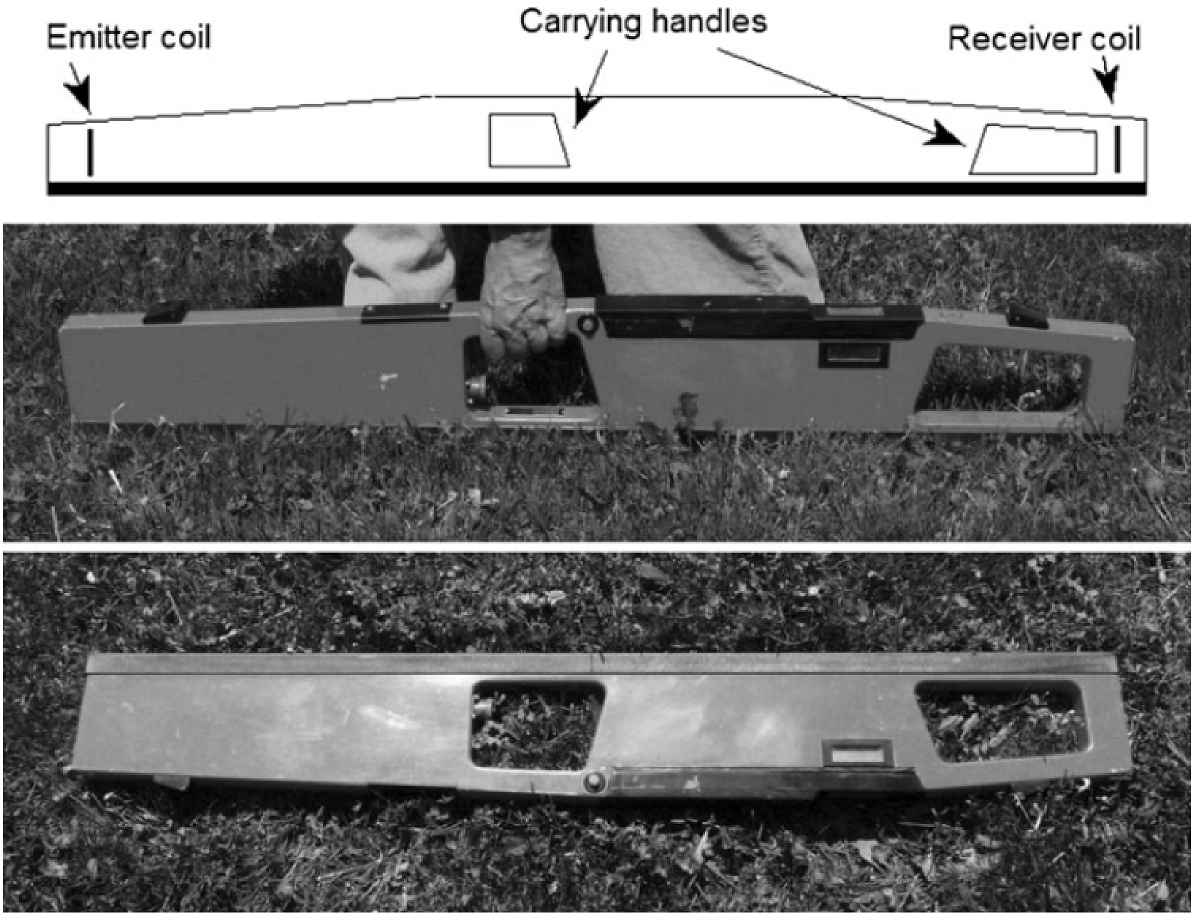


Figure 6. Diagram of an EM38 sensor (a), the EM38 in vertical (b) and horizontal (c) position. Source: Herrero, *et al*. (2014).

2.3 *Experimental setup*

This study was developed over two field campaigns with a temporal variability of ten years: 1994-96, 2007-2008.

Table 1. Schema of the two sampling campaigns.

Sampling campaing	**Year**	**Task applied**
1st	1994	Eighty profiles were described during the winter
	1995-1996	EM-38 measurements in a grid survey (500x500) and 410 auger –holes taken. Soil samples were taken one in every five EM-38 measurements at each 30 cm depth to 120 cm. Additional closely-spaced EM-38 measurements were incorporated into the sampling scheme (10% of the total data set, approximately 50 observations) to allow for geostatistical treatment of the data. This soil sampling was carried out during the winter in 1995 and in 1996.

Sampling campaing	Year	Task applied
2nd	2007	Resampling with the EM-38 of the 420 nodes of the grid survey sampled in 1995-1996, from the end of February till the end of March of 2007. We took soil samples at 56 grid nodes at depths of 0-30 cm, 30-60 cm, 60-90 cm and 90-120 cm.
	2008	Resampling of the 55 soil profile sites that were described in 1994 (legacy data) at depths of 0-30 cm, 30-60 cm, 60-90 cm and 90-120 cm.

2.3.1 *First sampling. Procedure applied during 1994-1996:*

The aim of this campaign was to perform a soil survey for soil mapping at semi-detailed scale (1:50.000) and to assess soil salinity of the northern part of the Ebro Delta The soil survey was conducted in two steps:

i) Eighty profiles were described during the winter of 1994 by Casanova (1998). These profiles where distributed according to the geomorphologic and geologic units. Each profile was described using SINEDARES (CBDSA, 1983) methodology. Some soil physical and chemical properties, such as sand, silt and clay content (% by weight) and $pH_{1/2,5}$, pH_e, $EC_{1:5}$ (electric conductivity of a 1:5 soil:water suspension.), EC_e, anions, cations and SAR of the soil samples were measured according to Soil Survey Staff (1996).

ii) A second field campaign was carried out by Casanova (1998) during the winters (mid Febrary till the end of March) of 1995 and 1996. Aerial photographs were used for recording the coordinates. A simple design using a regular grid survey (500x500 m) was performed for the use of an electromagnetic sensor. Augering at grid intersection (a total of 410 auger-holes) were completed each to a depth of at least 150 cm. At each auger hole, an electromagnetic reading was taken using a Geonics EM-38 (EM) sensor. Samples were taken for further laboratory analysis of $EC_{1:5}$ (Richards, 1954), at depths of 0-30 cm, 30-60 cm, 60-90 cm and 90-120 cm, for one in every five measurements. Sampling sites were geolocated with the use aerial photographs at 1:25,000 scale, and coordinates were stored in a GIS database. Additional closely-spaced EM-38 measurements were incorporated into the sampling scheme (10% of the total data set, approximately 50 observations) to allow for geostatistical treatment of the data. Figure 7 shows the soil sampling locations in the study area.

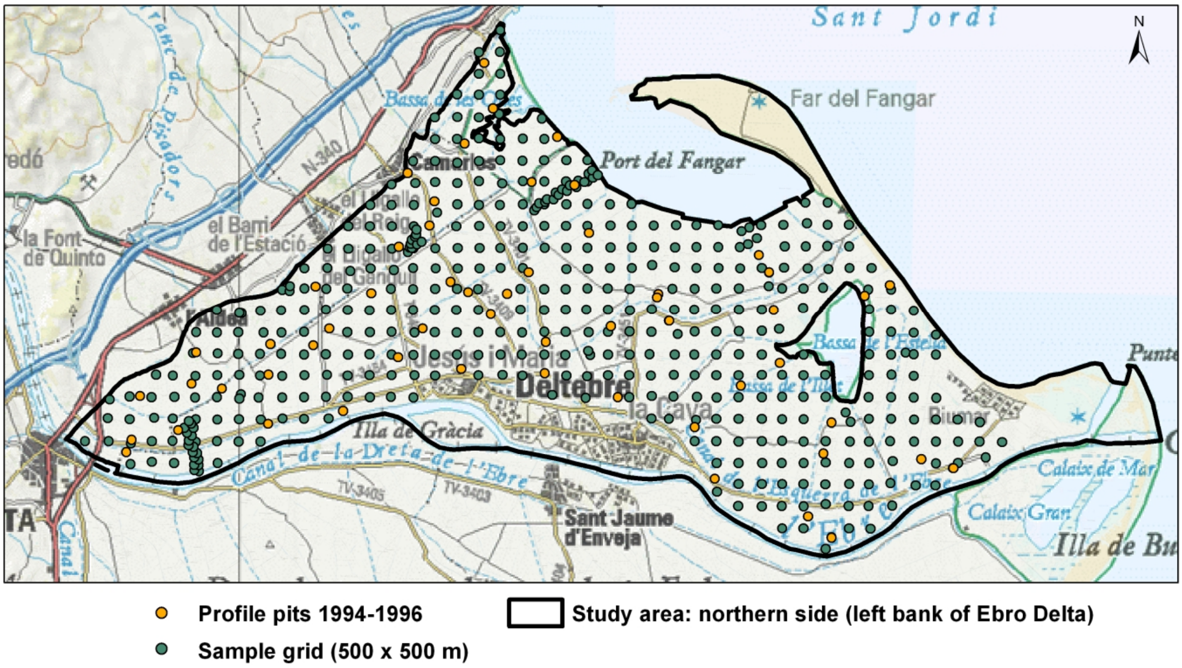


Figure 7. Soil sampling locations is the study area for 1994-1996 sampling campaign (Casanova, 1998).

2.3.2 *Second sampling. Procedure applied during 2007-2008:*

For comparison purposes, it was essential to adopt a very similar sampling strategy for both field campaigns. Salinity was again measured with the EM-38 on the same surveying grid as in the previous sampling campaign in 1995-1996, having 420 nodes with a 500 m interval.

The correct site location for resampling is critical, but difficult because many previous sample locations were completed without maps at adequate scale and were not georeferenced. Today, the location of new sample sites is simple with global positioning systems (GPS). Hence, in 2007-2008 GPS was used for the EM-38 survey, bringing us to the supposed points sampled in the first sampling in 1995-1996.

An electromagnetic induction Geonics EM-38 (EM) sensor was used to measure an apparent soil electrical conductivity (ECa) in horizontal and vertical dipole position at each grid node. The electromagnetic induction Geonics EM-38 (EM) sensor has been used in different soil studies (De Jong *et al.*, 1979) and in soil mapping (Baden *et al.*, 1982) and in the earlier field campaign. The soil survey was conducted in two steps:

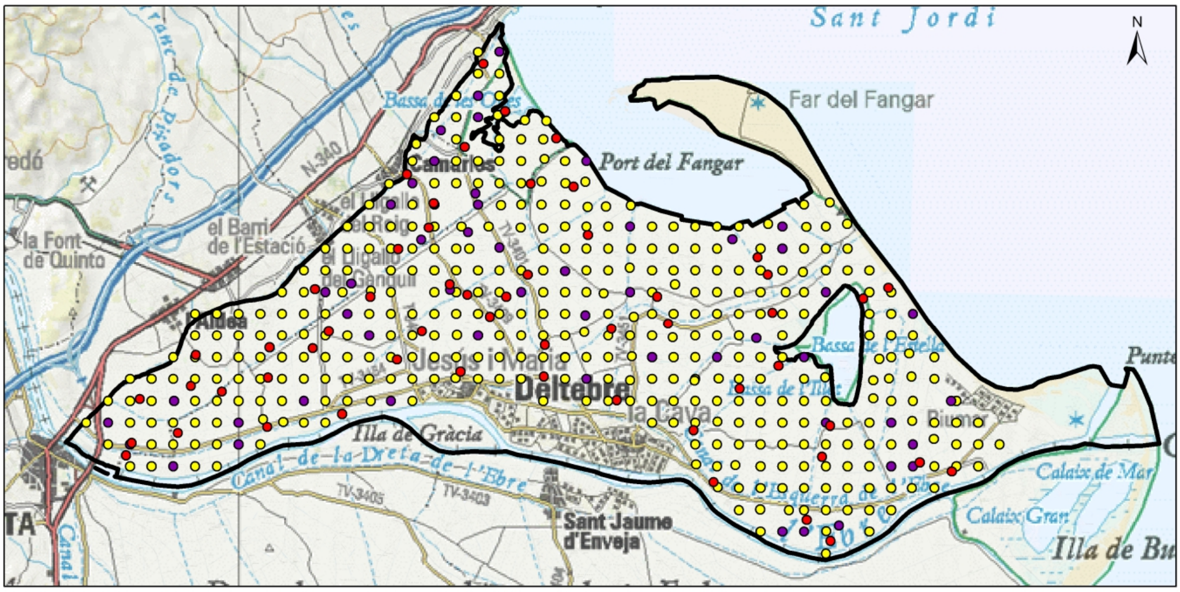
i) In 2007, we obtained one EM reading at each node grid (420 EM readings in total). The field work was carried out from the end of February till the end of March, when the soil of Ebro Delta is drained enough to have adequate moisture conditions.

Together with the EM readings, we took soil samples at 56 grid nodes at depths of 0-30 cm, 30-60 cm, 60-90 cm and 90-120 cm. These samples were analyzed in the laboratory for soil moisture, sand, silt and clay content (% by weight), organic matter, $EC_{1.5}$, $pH_{1/2,5}$, Hs, EC_e, pH_e, and anions and cations in the saturated paste. Groundwater samples were also collected and analyzed.

The 56 grid nodes were distributed randomly across the study area but using the soil surface texture of the soil map units as criterion. The soil surface texture was obtained with the existing soil map (Casanova, 1998) of the study area, . The number of samples to be taken at each soil surface texture was selected according to the soil surface texture extension. Once the number of samples to be taken was established, the nodes to sample were selected randomly.

ii) In order to be able to compare salt profile variation in 12 years, a second sampling was carried out in 2008. The aim was to sample the 55 soil profile sites that were described in 1994 (legacy data). Soil samples were taken at depths of 0-30 cm, 30-60 cm, 60-90 cm and 90-120 cm, and EM readings were measured, at each profile site. The field work was carried out from the end of February till the end of March in 2008, again drained enough to be able to find the soil under adequate moisture conditions (as close as possible at field capacity).

Sampling sites were geolocated with a GPS (Global Positioning Systems) Trimble (GeoExplorer 3), Buren and Stuhlinger (2006) reported accuracy of 0.179 m to the Trimble (GeoExplorer 3). The sampling sites coordinates were stored in a GIS database. Figure 8 shows the soil sampling locations in the study area.



Boreholes observations in 2008 **Sample grid (500 x 500 m)**

Boreholes observations in 2007 **Study area: northern side (left bank of Ebro Delta)**

Figure 8. Soil sampling locations is the study area for 2007- 2008 sampling campaign.

2.4 *Calculations and statistical analysis*

2.4.1 *Lateral variability of the soil electromagnetic sensor readings and their calibration*

Most soil properties have strong lateral variability and soil salinity is one of these. The use of the EM-38 sensor although supporting the study of soil salinity, sensor readings could vary significantly with distance (Pérez-Coveta, 2000; Herrero and Pérez-Coveta, 2004). In order to cut possible distortions of the EM variability, related to the change of measurement scale for the EM38 dial, we took five readings at each node, one in the centre of the node and four more at a 2 m distance from the centre defined by the cardinal points (N, E, S, W). Then we calculated the average of these four readings to have one per node. At each survey site, EM readings were taken with the coil configurations oriented in the horizontal (EMh) and vertical (EMv) positions, providing for effective measurement depths of roughly 1 and 1.5 m, respectively. The direct EM readings were corrected by dividing 100 to simplify formulations; these EM-38 readings were designed as ECh for horizontal coils and ECv for vertical coils.

Corwin *et al.* (2003), Corwin *et al.* (2006), Herrero *at al* (2011) noted that the temperature influences in the ECa (apparent soil electrical conductivity). Electrolytic conductivity increases at a rate of approximately 1.9% per degree centigrade increase in temperature (Corwin, 2003). Customarily, electrical conductivity (EC) is expressed at a reference temperature of 25 ºC, for purposes of comparison (United States Salinity Laboratory Staff, 1954). However, we compared the ECh data and the ECh corrected according to the soil temperature with one way ANOVA however no significant differences (ANOVA) were found in any of the 2 sampling campaigns (1995/1996; and 2007), therefore any correction factor was applied to the ECh.

The calibration of the EM-38 sensor is possible using simple regression, polynomial regression or using equations from the sensor design (McNeill, 1980). From them, lineal and polynomial regressions were applied to check the relationship of ECe with ECh. In addition, another regression was applied in order to check relationship of laboratory data $EC_{1.5}$-ECe.

With the sites samples, the ECe was regressed on ECh to estimate ECa. With ECh of those sites without soil samples and the regression equation calculated, we estimated ECa. Independent calibrations of the EM-38 with ECe were tested for the sites with high contribution to EM response from the deep layers (Herrero and Hudnall, 2014).

2.4.2 *Obtaining soil salinity and SAR for the whole profile (salt profile and SAR profile)*

Jones *et al.* (2008) describes salt profile content as an appropriate indicator for soil salinity. The indicator 'salt profile' describes the horizontal and varying vertical extent of salinisation. Salt can be measured either as the total concentration of salts, or electrical conductivity (EC) of a saturated paste or saturation extract. Salt profile is a good indicator for soil quality because it gives a complete picture on the salinity state of the soil, or more exactly the salt-affected area (Huber *et al.*, 2008). SAR profiles can equally be obtained.

Using soil profile data from 1994-1996 and the fifty-five sites sampled at different depths in 2007 and 2008 we calculated the total soil salinity of the profile and the SAR.

The following equations were applied to calculate soil salinity for the profile and SAR profile:

$$SCi = \frac{ECe * e}{\mathrm{D}} \tag{1}$$

$$SCP = \sum_{i=1}^{i=n} CSi \tag{2}$$

$$SAR = \frac{Na^{+}}{\sqrt{0.5\,(Ca^{2+} + Mg^{2+})}} \tag{3}$$

$$SARhi = \frac{SAR * e}{D} \tag{4}$$

$$SARP = \sum_{i=1}^{i=n} SARhi \qquad (5)$$

where SCi, salinity in the i depth; ECe, electric conductivity from soil-paste; e, thickness of the soil sample; D, the thickness of the profile (90 or 120 cm); SCP, salt profile; SAR, the equation for SAR. SARhi, SAR in the i depth; SARP, SAR profile.

2.4.3 *Calculating spline function for soil salinity.*

Soil attributes in general vary continuously with depth in a soil profile (Russell and Moore, 1968). In contrast to this, the traditional method of sampling soil involves dividing a soil profile into horizons. The number of horizons and the position of each are generally based on attributes easily observed in the field, such as morphological soil properties (Bishop *et al.*, 1999). In order to estimate the continuous variation of salinity along the profile from discrete samples, spline functions can be used (Bishop *et al.*, 1999). This spline function fits a smooth curve through any set of data points (Jauregui and Quirino, 1985).

The aim of this work was to obtain the best spline function using polynomials and exponential decay functions. These functions describe the ECe and ECa in depth. These calculations were applied for each year of sampling, 1994-96, 2007 and 2008. We decided to group the sampled sites into four classes according to the surface texture (Tóth *et al.,* 2008a) having the hypothesis that the salinity content could depend on the soil texture. Specific surface area is an important property in terms of adsorption of dissolved materials in the water surrounding the soil particles (Van de Graaff and Patterson, 2001). The classes considered where 1) Coarse texture: sandy and loamy sandy; 2) Moderate coarse texture: sandy loam and fine sandy loam; 3) Medium texture: very fine sandy loam, silty loam and silt; 4) Fine texture: clay loam, sandy clay loam, silty clay loam, sandy clay, silty clay and clay.

2.4.4 *Determining the pedotransfer functions for soil sodicity*

Wösten *et al.* (2001) reported that one remedial approach for obtaining PTF was the use of principal components analysis to find a small number of new parameters that are linear combination of the original inputs and can explain a large percentage of variability within samples. Hence, in order to select the analytical parameters to be able to obtain the best pedotransfer function (PTF) for SAR, two statistical methods were explored. The first was principal components analysis (PCA), which helped to select the best physical and chemical soil properties for SAR prediction. The second was a linear model: multiple linear regression (MLR), that predicted a set of properties (SAR) from another set of properties (selected with PCA). Using MLR with the soil properties selected in the PCA, we were able to obtain the PTF for SAR.

The data for sampling years 1994, 2007 and 2008 was treated using these models. For each year, 80% of the data were used for applying MLR model to obtain the PTF of the SAR, and the remaining 20% of the data were used for the validation of these equations. A comparison between the results obtained with statistical data and real random data set division was used to evaluate the performance of the PTF obtained using MLR model. This data analysis was carried out using the software XLSTAT (XLSTAT, 2014).

2.4.5 *Validation method for the pedotransfer functions and for the spatial distribution of soil salinity*

Additional indices were applied for validation such as De Vos indices (De Vos *et al.*, 2005). They were applied to establish the prediction quality of the PTFs. These were Equation 6, the mean predicted error (MPE) and Equation 7, the standard deviation of the prediction error (SDPE), Equation 8, the root mean square prediction error (RMSPE); and Equation 9, the prediction coefficient of determination (R_p^2) as shown below (De Vos *et al.*, 2005).

$$\text{MPE}=\frac{1}{n}\sum_{i=1}^{n}(\widehat{Pb,\imath}\text{-Pb,i}) \tag{6}$$

$$\text{SDPE}=\sqrt{\frac{1}{n-1}\sum_{i=1}^{n}\left((\widehat{Pb,\imath}-Pb,i)-MPE\right)^2} \tag{7}$$

$$\text{RMSPE}=\sqrt{\frac{1}{n}}\sum_{i=1}^{n}(\widehat{Pb,\imath}\text{-Pb,i})^2 \tag{8}$$

$$R_p^2=\frac{[\text{cov Pb,i}, \widehat{\text{Pb,i}}]^2}{\text{var(Pb,i)}-\text{var}(\widehat{Pb,\imath})} \tag{9}$$

where Pb,i and $\widehat{\text{Pb},\imath}$ are the observed and predicted salinity values, respectively; n the number of observations; and var and cov the variance and the covariance function, respectively.

These indices were applied to validate the PTF using the 20% of the data, which was selected applying a random process.

The same validation method was used for validating the map of spatial distribution of soil salinity in 1995-1996, obtained applying geostatistical analysis. The soil salinity map for the 1995-1996 was validated using the additional closely-spaced EM-38 measurements, which is the 10% of the total data set, approximately 50 observations. However, the soil salinity map for the 2007 was validated with the 10% of the 420 ECh. This 10% was selected randomly of the total data set for the 2007 year.

2.4.6 *Assessment of spatial distribution of soil salinity*

In order to compare soil salinity variation after a period of 12 years, we used the EMh readings converted to ECh. These ECh values were converted to ECa using regressions. The spatial distribution of soil salinity was plotted using geostatistics.

We used Co-kriging for the calculation of the spatial distribution of soil salinity for the 2 campaign in 1994-1996 and 2007-2008. The soil profiles with EC laboratory measurements were used as the input “hard” data of such a co-kriging and ECa estimated from EM38 being the “soft” input data respectively.

The ArcInfo ArcGIS software was used for geostatistical analysis (Johnston *et al.,* 2001) and for soil salinity mapping, by co-kriging. This method is a linear combination of primary and secondary data values that minimizes the variance of the estimation error by exploiting the cross-correlation between several variables. The cross-correlated information contained in the secondary variable should help reduce the variance of the estimation errors. The development of the co-kriging system is identical to the development of ordinary kriging system, but using two set of data values.

Co-kriging is simply an extension of autokriging in that it takes into account additional correlated information in the subsidiary variables (Webster and Oliver, 2007). Co-kriging provides an estimate at an unobserved location of variable z, based on the weighted average of adjacent observed sites within a given area. The theory is derived from that of regionalized variables (Matheron, 1965, 1971) and can be briefly described by considering an intrinsic random function denoted by $\check{Z}_u(S_0)$, where S represents all sample locations, i = 1,...,n; and V variables, l = 1, 2, ..., V. An estimate of the weighted average given by the ordinary kriging predictor at an unsampled site, $\check{Z}_u(S_0)$, is defined by:

$$\check{Z}_u(S_0) = \sum_{l=1}^{v} \sum_{i=1}^{n} \lambda_{il} Z_l(S_i) \tag{10}$$

where the subscript i refers to the sites, of which there are n_l where the variable l has been measured. The λ_{il} are the weights, satisfying:

$$\sum_{i=1}^{nl} \lambda_{il} \begin{cases} 1 & l = u \\ 0 & l \neq u \end{cases} \tag{11}$$

The weights are calculated from the matrix equation:

$$c = \boldsymbol{A^{-1}b} \tag{12}$$

where **A** is a matrix of semivariances between the data points; **b** is a vector of estimated semivariances between the data points and the points at which the variable z is to be predicted; and c is the resulting weights and the Lagrange Multipliers ψ (Triantafilis, 2001).

The cokriging (prediction) variance is given by:

$$\delta_u^2(x_o) = b^T \lambda \tag{13}$$

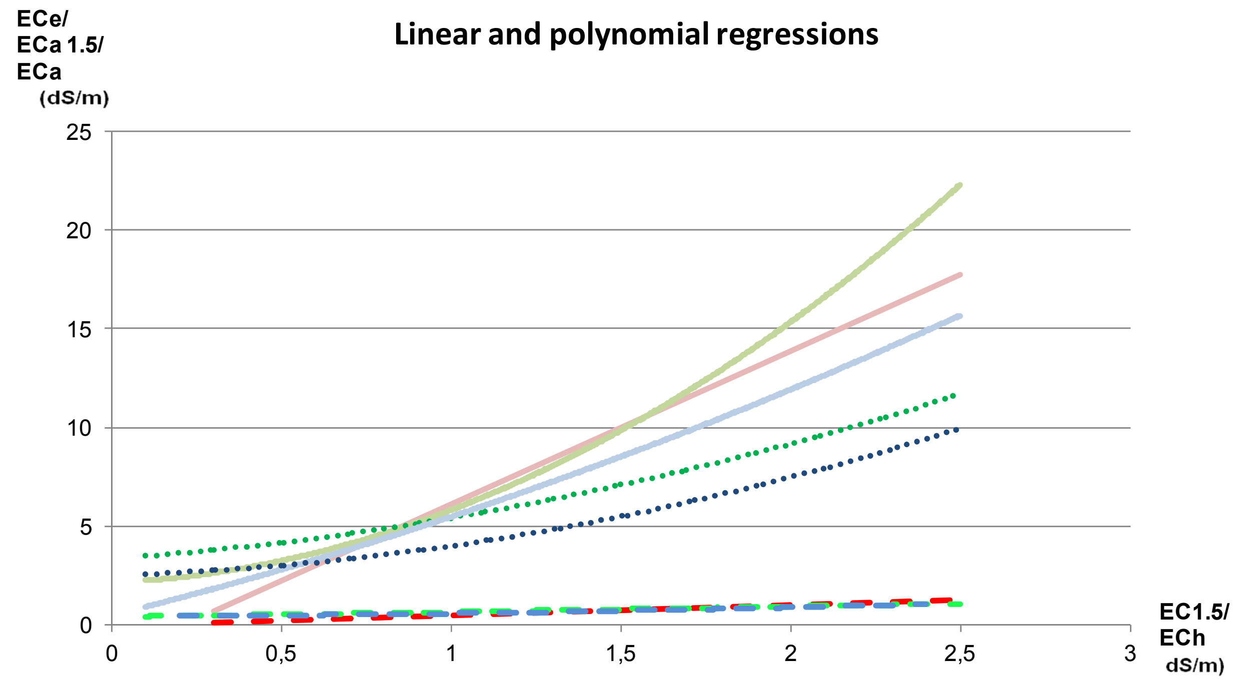
3 RESULTS AND DISCUSSION

3.1 <u>*General regressions: EC EM sensor and EC soil analysis*</u>

Relationships ECe – $EC_{1:5}$, $ECa_{1:5}$ – ECh and ECa – ECh. were found using data of the sampling periods (1994-96, 2007, 2008). Figure 9 shows these relationships (regressions), where $ECa_{1:5}$ is the $EC_{1:5}$ estimated with ECh, ECa is the ECe estimated with ECh.

In general, the adjustments for ECa-ECh attain a higher r^2 than for $ECa_{1:5}$-ECh. In 1994-1996, we could not have adjustment for ECa-ECh because of the lack of laboratory ECe data, however, we had data for the regression between $ECa_{1:5}$ and ECh with r^2 = 0.83, being the lowest r^2 taking into account all the adjustments.

Conversely, the regression between ECe and $EC_{1:5,}$ the best r^2 is for the 1994-1996 sampling campaign. These regressions have similar r^2 than other authors obtained such as Herrero and Aragüés (2003) and, Herrero and Hudnall (2014).



Period of sampling	Regression between ECe and $EC_{1:5}$ (solid line)			Regression between $ECa_{1.5}$ and ECh (dashed line)			Regression between ECa and ECh (dotted line)		
		r^2	n		r^2	n		r^2	n
1994-1996	$ECe = 7.75 \cdot EC_{1:5} - 1.65$	0.92	55	$ECa_{1.5} = 0.53 \cdot ECh + 0.06$	0.83	55	-		
2007	$ECe = 2.92 \cdot (EC_{1:5})^2 + 0.76 \cdot (EC_{1:5}) + 2.14$	0.86	194	$ECa_{1.5} = 0.01 \cdot (ECh)^2 + 0.24 \cdot ECh + 0.41$	0.84	48	$ECa = 0.84 \cdot (ECh)^2 + 1.23 \cdot (ECh) + 3.35$	0.95	48
2008	$ECe = 0{,}72 \cdot (EC_{1:5})^2 + 4.28 \cdot (EC_{1:5}) + 0.48$	0.87	216	$ECa_{1.5} = 0.08 \cdot (ECh)^2 + 0.06 \cdot ECh + 0.44$	0.85	55	$ECa = (ECh)^2 - 0.48 \cdot ECh + 2.52$	0.94	55

Figure 9. Linear and polynomial regressions between several salinity indicators. Refer to the text for ECe, ECa1:5 and ECh.

Figures 10, 11 and 12 show results of soil salinity distribution in the area using three different ways of calculating it; ECe from the laboratory (a), regressions between ECe and $EC_{1:5}$ (b); and regression between ECa or $ECa_{1:5}$ and ECh (c), as it is shown in Figure 9, for 1994-1996, 2007 and 2008. In general, these figures show that 0-1.75 and 1.75-4 classes are underestimated with respect to saline classes comparing soil salinity obtained with regressions (Figures 10b, 10c, 11b, 11c, 12b and 12c) and laboratory data (Figures 10a, 11a and 12a).

The distribution of salinity based on ECe shown in Figure 10a, Figure 11a and Figure 12a is calculated using laboratory data (ECe). Soil salinity for the whole profile was calculated with equation 1 and equation 2. Distribution of salinity based on ECe estimated (Figures 10b, 11b and 12b) with ECa1:5 or ECa (Figures 10c, 11c and 12c), calculated using regressions in Figure 9. In Figure 10, the distribution of non saline soils (EC < 4, sum of the two lowest classes) in 1994-1996 is quite similar using whichever of the three methods (a, b and c). Around 60% of the profiles were non saline in 1994-1996.

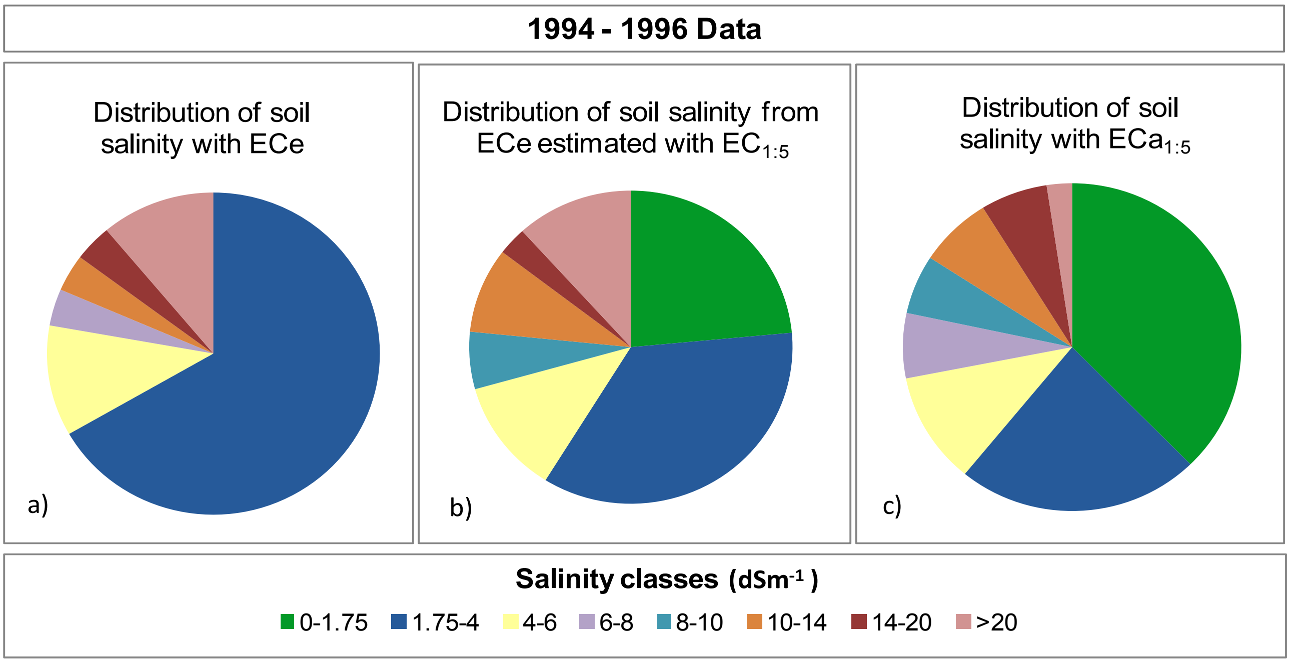


Figure 10. Distribution of soil salinity in 1994-1996, a) using laboratory data (N=37); b) ECe estimated with $EC_{1:5}$ (N=34) and c) Eca estimated with EMh (N=327).

ECe estimated using $EC_{1:5}$ (Figure 10b) highlights that the highest class of soil salinity (> 20 dSm^{-1}) is quite similar when compared with the real laboratory data (Figure 9a), however for $ECa_{1:5}$, is much less (Figure 10c). The middle salinity classes (4 – 20 dSm^{-1}) are quite diverse when comparing the three parameters (Figures 10a, 10b, 10c). ECe estimated using $EC_{1:5}$ and $ECa_{1:5}$ tends to underestimate saline soils in 1994-1996. Part of the ECe estimated with $EC_{1:5}$ and $ECa_{1:5}$ are classified as 1.75-4 dSm^{-1}, however it appears as non saline at all (values between 0-1.75) when salt profile is calculated with ECe. Figure 11 shows the comparison using the three parameters for the sampling in 2007. Non saline soils are underestimated using ECe estimated with $EC_{1:5}$ (Figure 11b) and overestimated in ECa (Figure 11c). ECa is the closest parameter to real ECe in 2007.

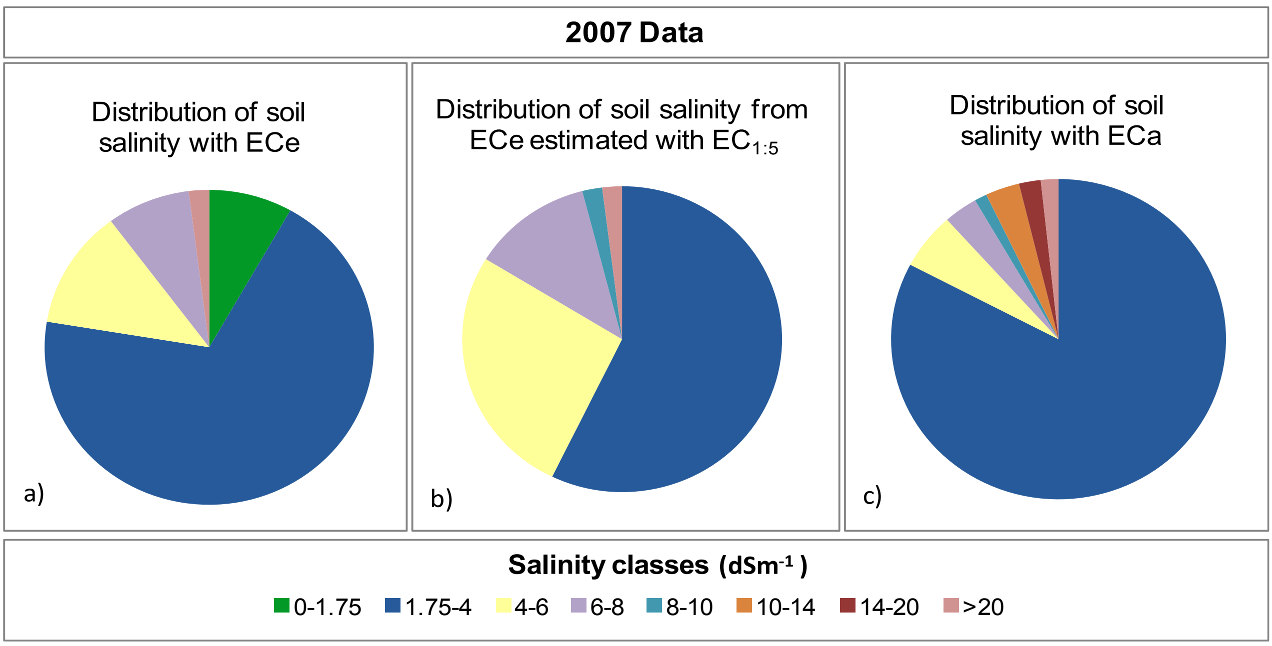


Figure 11. Distribution of soil salinity in 2007, a) using laboratory data (N=49); b) ECe estimated with $EC_{1:5}$ (N=49) and c) Eca estimated with EMh (N=421).

Overall, Figure 12a supports the ECe values for soil salinity from 2007. Figure 12b shows a variation with ECe estimated data, therefore ECe estimated with $EC_{1:5}$ works well in terms of salinity prediction in 2008. Regarding ECa (Figure 12c), soil salinity is very uneven with respect to ECe or ECe estimated with $EC_{1:5}$. This is a consequence of the low number of EM-38 readings taken in 2008 compared with 1994-1996 or 2007.

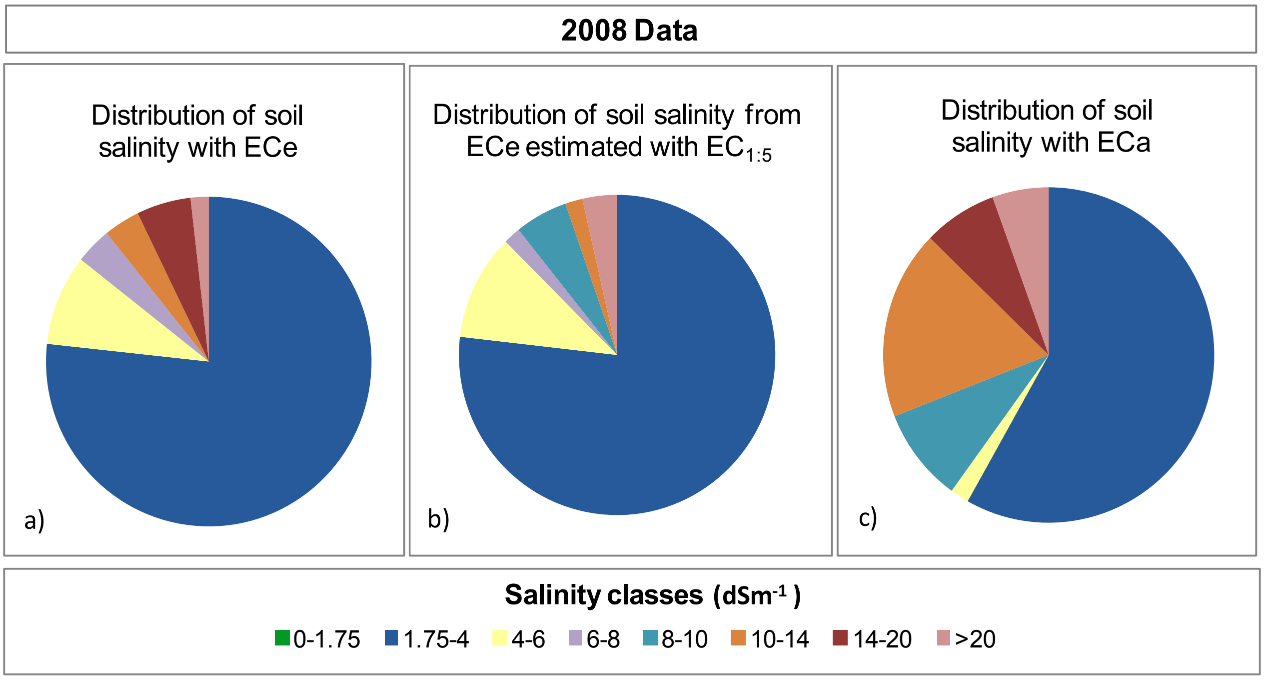


Figure 12. Distribution of soil salinity in 2008, a) using laboratory data (N=56); b) ECe estimated with $EC_{1:5}$ (N=55) and c) Eca estimated with EMh (N=56).

3.2 <u>*Salinity profiles according to soil surface textures*</u>

We calculated splines for soil salinity distribution at depth for each soil surface texture groups with the data in 1994-1996, 2007 and 2008. The total number of salt profiles is 56 in 1994-1996, 53 salt profiles in 2007 and 55 salt profiles in 2007. These splines for soil salinity distribution at depth were calculated using laboratory ECe data and ECe estimated with $EC_{1:5}$, correlations in Figure 9 (regression between ECe and $EC_{1:5}$).

Figure 13 shows soil salinity distribution at depth, for the four categories suggested by Tóth *et al.* (2008a) according to the surface texture, using laboratory ECe data or using ECe estimated with $EC_{1:5}$. We hypothesised that salinity content could depend on the soil texture; Figure 13 shows that this suggestion is positive. Soil salinity content varies between soils with coarse, moderate and medium/fine texture. We did not recognise differences between medium and fine texture, thus this classes could be grouped in future studies.

Splines calculated with 1994-1996 data have an r^2 ranging 0.68-0.95, with the exception of the moderate coarse texture soils, for the reason that we have very few soil sites with this soil surface texture. Splines show good r^2 in medium texture for the laboratory ECe and ECe estimated with $EC_{1:5}$, however the salt profile is higher in depth with ECe than with ECe estimated.

High values of soil salinity (laboratory ECe) are found in the topsoils for all surface soil textures, having the the highest values in 1994-1996 campaign. This fact happens for ECe estimated with $EC_{1:5}$ for all soil textures, with the exception of the coarse textured soils. Similar results in paddy fields were found by Herrero and Castañeda (2015).

Soils with coarse textures are mainly located close to the coast and they have seawater entrance at depth, this is the reason why we have a soil salinity increase at depth. This influence sea water is present in 2007 and 2008, when we calculate salt profile with ECe obtaining acceptable $r^2 = 0.43$ and $r^2 = 0.53$, respectively. The graphics show that using ECe estimated with $EC_{1:5}$, this curve is inverted for 1996 and 2008 data. This inverse curve is due to low number of samples for the coarse texture group in 1996 and 2008.

Splines calculated with 2007 data have an acceptable r^2 ranging 0.60-0.95, with exception of the coarse texture (r^2=0.43 and 0.07, laboratory ECe and ECe estimated, respectively). Moderate and medium surface textures have very good r^2 (0.61-0.95), but the graphs show a slightly higher salt profile using ECe estimated with $EC_{1:5}$. Splines calculated with 2008 data have good r^2 (0.5-0.92), except for the coarse textured soils. Splines for medium textures show a similar trend regardless of the parameters (ECe or ECe estimated). Splines for fine textures soils reflect good r^2, but the salt profile is slightly higher at depth using ECe estimated.

Comparing the trend of all the splines at depth, the graph shows that the soil salinity at depth in 1994-1996 was higher than in 2007 and 2008. The leaching associated with the annual inundation of paddies with fresh water plus the modest salinity of the groundwater can explain the strong desalination for the profiles of the graph in the moderate, medium and fine surface textures.

These graph show that the regression for the calculation of ECe estimated from EC1:5 could perform with acceptable results once we want to calculate saline profiles using splines. Splines work well when we want to study vertical variation of soil salinity (ECe).

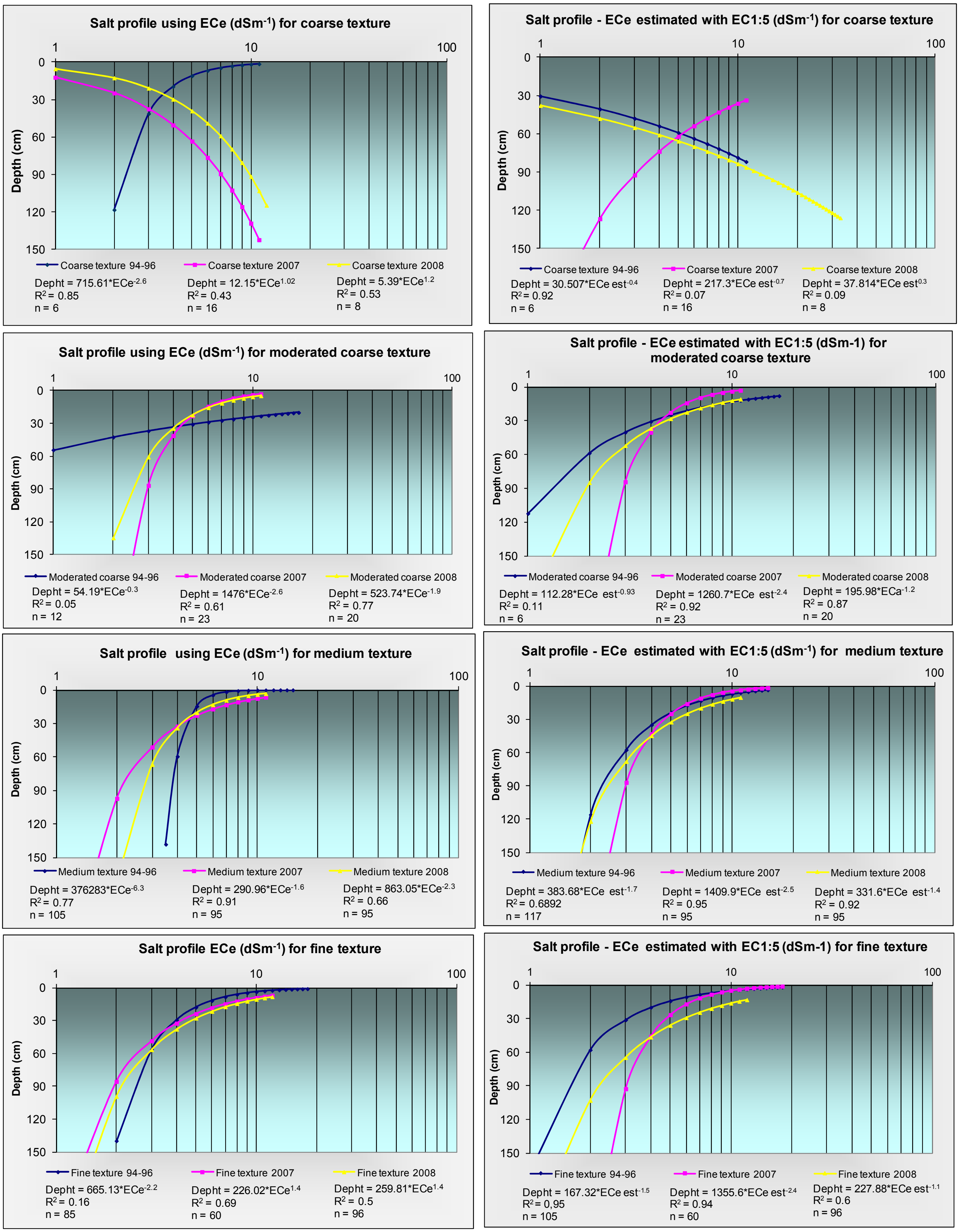


Figure 13. Salinity profiles according to different superficial soil textures for the all sampling years.

3.3 *Pedotransfer functions for soil sodicity*

Table 2 displays the analytical results for the analyzed samples ranked by their salinity; pHe, ECe, Ca^{++}, Mg^{++}, Na^{+}, K^{+}, Cl^{-}, SO_4^{-2}, CO_3^{-2}, HCO_3^{-}, NO_3^{-} are measured in the extract from the saturated paste.

Table 2. Statistical summaries of the chemical properties for each year of sampling.

	Variable	N	Minimum	Maximum	Average	Standard deviation
1994	$pH_{1:2.5}$	154	7.7	9.2	8.2	0.3
	pHe	154	7.0	8.4	7.9	0.3
	$EC_{1:5}$	154	0.1	8.7	1.1	1.2
	Calcium Carbonate equivalent (CAR) (%)	154	17.0	56.0	31.8	5.2
	ECe	154	0.5	59.7	5.9	8.5
	SAR	154	0.3	44.5	3.7	5.2
	Ca^{++} (meq/l)	154	2.7	75.5	22.5	13.9
	Mg^{++} (meq/l)	154	0.3	152.0	11.0	17.3
	Na^{+} (meq/l)	154	1.4	460.6	29.6	55.2
	K^{+} (meq/l)	154	0.1	7.9	0.7	1.1
	Cl^{-} (meq/l)	154	0.6	536.1	32.1	81.2
	SO_4^{-2} (meq/l)	154	1.3	85.8	23.3	15.0
	CO_3^{-2} (meq/l)	154	0.0	0.4	0.0	0.1
	HCO_3^{-} (meq/l)	154	2.4	19.2	6.6	3.8
	NO_3^{-} (meq/l)	154	0.0	16.0	1.3	2.7
2007	$pH_{1:2.5}$	201	7.8	9.2	8.3	0.3
	pHe	201	7.5	8.7	8.0	0.8
	$EC_{1:5}$	201	0.2	4.3	0.7	0.5
	Calcium Carbonate equivalent (CAR) (%)	201	24.0	59.0	36.9	6.0
	ECe	201	1.04	51.0	3.9	5.4
	SAR	201	0.75	92.0	6.0	12.5
	Ca^{++} (meq/l)	201	0.0	41.1	15.7	10.6
	Mg^{++} (meq/l)	201	0.3	80.4	6.3	8.3
	Na^{+} (meq/l)	201	3	628.2	21.5	62.4
	K^{+} (meq/l)	201	0.1	8.3	0.7	1.0
	Cl^{-} (meq/l)	201	2.2	591.6	18.4	59.1
	SO_4^{-2} (meq/l)	201	3.1	58.3	19.9	12.5
	CO_3^{-2} (meq/l)	201	0.0	1.4	0.1	0.1
	HCO_3^{-} (meq/l)	201	1.4	12.2	3.6	1.8
	NO_3^{-} (meq/l)	201	0.0	1.0	0.1	0.2
	Organic matter (%)	201	0.04	6.6	1.9	1.5

	Variable	N	Minimum	Maximum	Average	Standard deviation
2008	$pH_{1:2.5}$	219	7.7	9.4	8.3	0.3
	pHe	219	7.3	8.6	8.1	0.2
	$EC_{1:5}$	219	0.2	5.8	0.7	0.7
	Calcium Carbonate equivalent (CAR) (%)	219	13.0	63.0	37.8	5.9
	ECe	219	0.6	36.1	4.4	5.1
	SAR	219	1.3	66.6	7.1	10.1
	Ca^{++} (meq/l)	219	0.7	31.0	15.1	9.1
	Mg^{++} (meq/l)	219	0.9	48.6	8.0	7.1
	Na^{+} (meq/l)	219	2.7	330.0	24.6	42.2
	K^{+} (meq/l)	219	0.1	21.2	0.8	1.6
	Cl^{-} (meq/l)	219	2.0	351.3	22.8	46.4
	SO_4^{-2} (meq/l)	219	1.9	54.8	21.7	13.8
	CO_3^{-2} (meq/l)	219	0.0	2.1	0.03	0.2
	HCO_3^{-} (meq/l)	219	1.5	9.8	3.3	1.4
	NO_3^{-} (meq/l)	219	0.0	5.6	0.1	0.5
	Organic matter (%)	219	0.01	16.1	1.7	2.2

In 1996, the ECe ranges from 0.5 to 59.7 dSm^{-1}, and the pHe from 7.0 to 8.4. Clays remain flocculated at these pH values and at high salt content levels. Using soil salinity classes listed by the Soil Survey Division Staff (1993), the average ECe of these samples (5.9 dS m-1) qualifies as slightly saline. In 2007, the salinity range of these samples was lower than in 1996. ECe ranged from 1.04 to 51.0 dSm^{-1} with an average of 3.9 dSm^{-1} with an average pHe of 8. In 2008, the ECe ranged from 0.6 to 36.1 dSm^{-1} and the pHe averaged 8.1. Highly saline soils (high Ece) also cause the clay to be flocculated, even when Na is present.

The ions in the soil saturation extracts can be ordered from highest to lowest concentration as follows:

Anions	Cations
$Cl^{-} > SO_4^{-2} > HCO_3^{-} > NO_3^{-} > CO_3^{-2}$	$Na^{+} > Ca^{++} > Mg^{++} > K^{+}$

This means that the groundwater composition has almost not changed, and that seawater is the main origin of the soil salinity (Murray, 2004).

PCA analysis was carried out for each year of sampling. Two principal components were selected for the 1994 data, 2007 data and 2008 data that accounted for 63.8 %, 65% and 59.5% of the variance in the correlation matrix respectively, and that were retained after eigenvector extraction.

The PCA studies (Figure 14) show a good relationship between $EC_{1:5}$ and ECe with chlorides reflected as the origin of the soil salinity in first PC (F1) whilst the second PC (F2) is related to SAR and pH with sodicity and alkalinity.

The F1 accounts for 43.80%, 43.02% and 34.49% of the variance in the 1994 data, 2007 data and 2008 data, respectively (Figures 14a, 14b and 14c). In 1994 data, this PC is highly correlated with electrical conductivity in water (r=0.95) and with electrical conductivity in soil saturation extracts (r=0.96), magnesium (r=0.94), K^+ (r=0.94), and chloride (r=0.913) ions. Besides being highly correlated, chloride, sodium, sulphate and calcium in this order account for the major portion of ions in the soil saturation extracts (Table 2). In 2007, F1 show a high correlation with electrical conductivity in soil saturation extracts (r=0.98), sodium (r=0.95), chloride (r=0.947) and magnesium (r=0.93) ions. Similar results were found and described by Herrero and Castañeda (2015) in previous studies. These results show that F1 has strong correlation with ions related to soil salinity. Various types of Na^+, Mg^{++} and Ca^{++} salts are present, mainly chloride and sulphate, this determines salinisation.

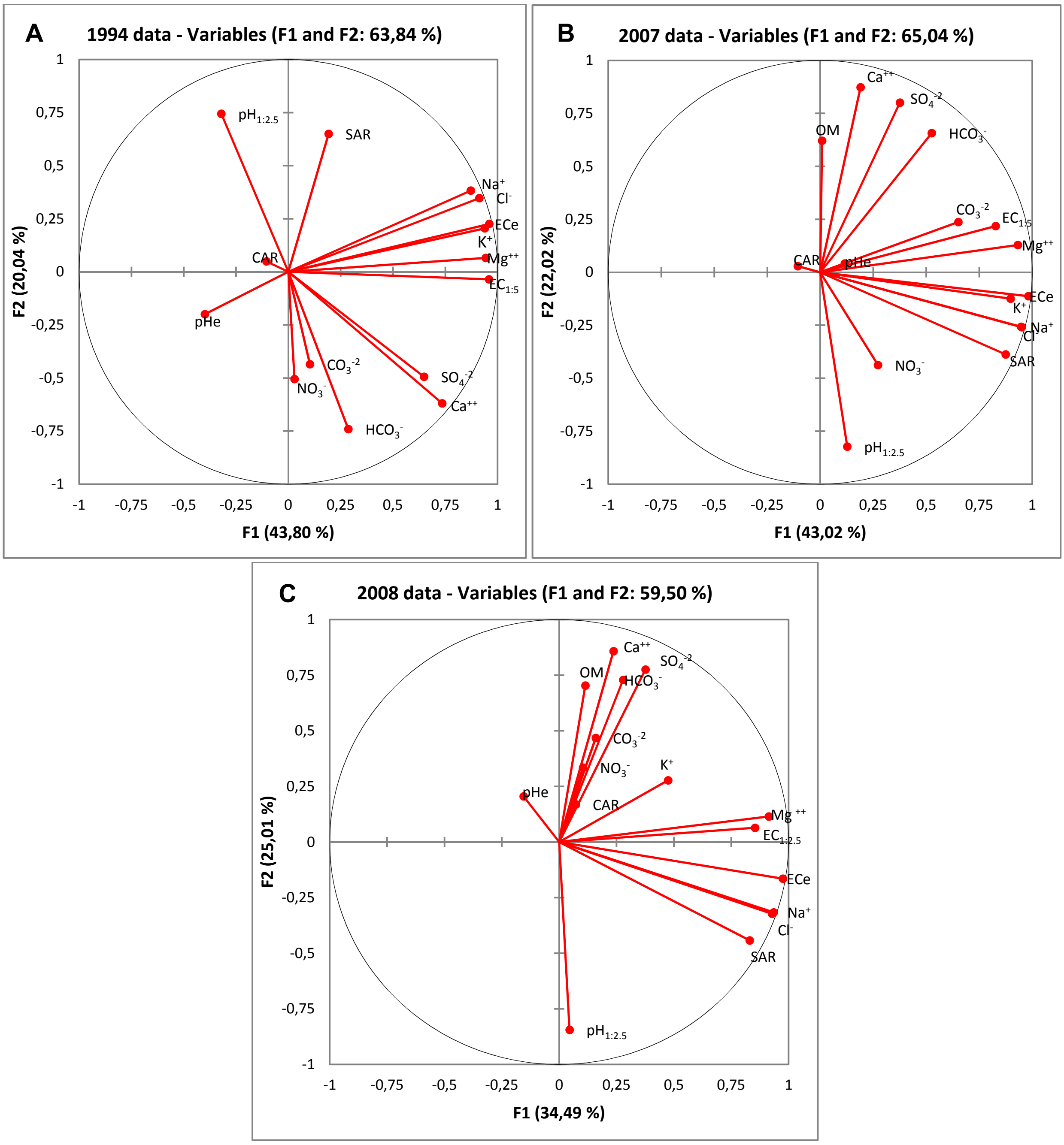


Figure 14. Loading plots of the study data. A) 1994 data; B) 2007 data and C) 2008 data. See Table 2 for symbol explanation.

The F2 (Figures 14a, 14b and 14c) is dominated by sulphates and calcium negatively correlated with sodicity.

The F2 accounts for 20.03%, 22.05% and 25.01% of the variance in the 1994 data, 2007 data and 2008 data respectively (Figures 14a, 14b and 14c). In 1994 data and 2007 data, this F2 is highly correlated with alkalinity (r=0.74), SAR (r=0.65) and Ca^{++} (r=-0.62) for 1994 data, and pHe (r= -0.82) and Ca^{++} (r=0.87) for 2007 data. At a first glance this F2 can be interpreted as representing saturation extract alkalinity. A high correlation coefficient between organic matter and alkalinity in calcareous soil was also found by González *et al.* (2007).

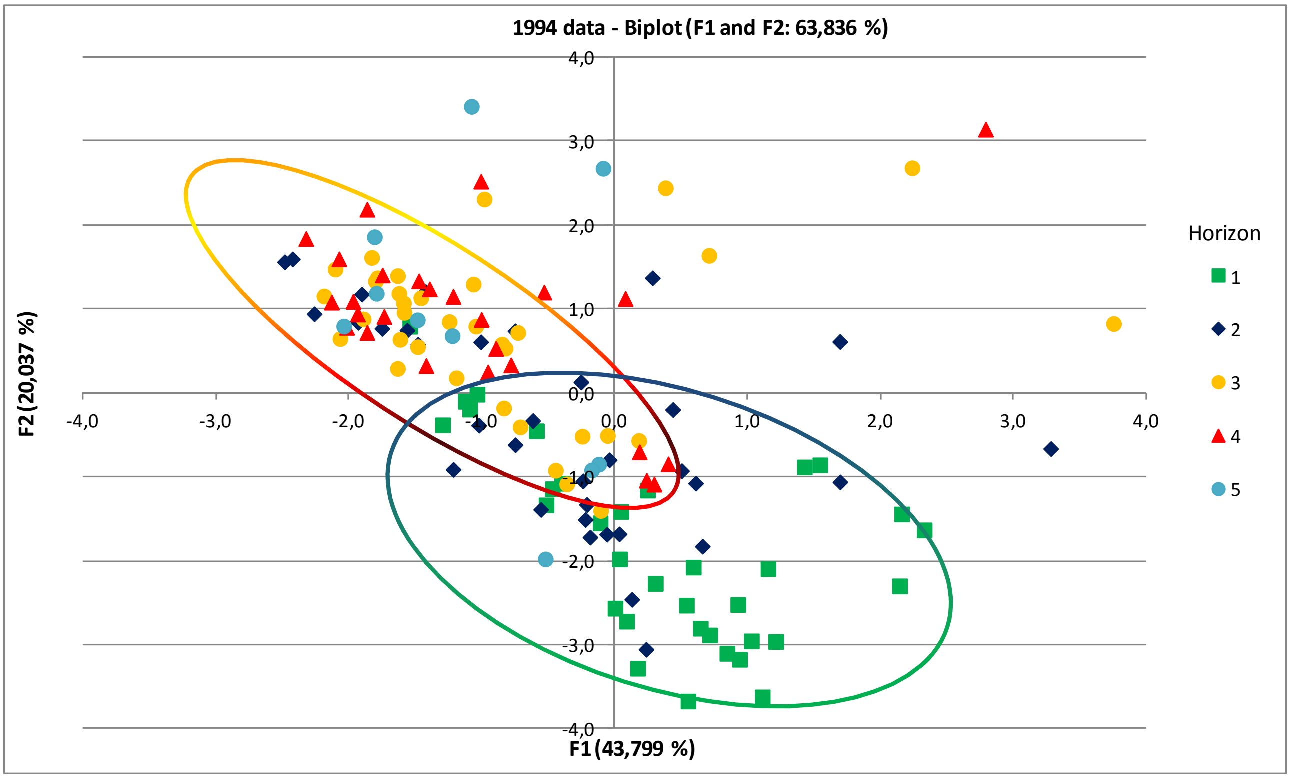


Figure 15. Plot of the (scaled) first two principal components for 1994 data.

The results of PCA are typically displayed as score plots (showing sample groupings) and loadings plots (showing relationships between the geochemical variables used in the analysis) of the principal components extracted from the analysis.

Figure 15 and Figure 16 show the data for the first and second PC for 1994 and 2007 respectively. This graph shows clear clusters when we plotted the data using horizons level. Horizon levels are diagnostic horizons as we use data from soil profiles for this analysis in 1994-1996. The topsoil horizons (1 and 2) are better correlated with variables than deeper horizons (3 and 4).

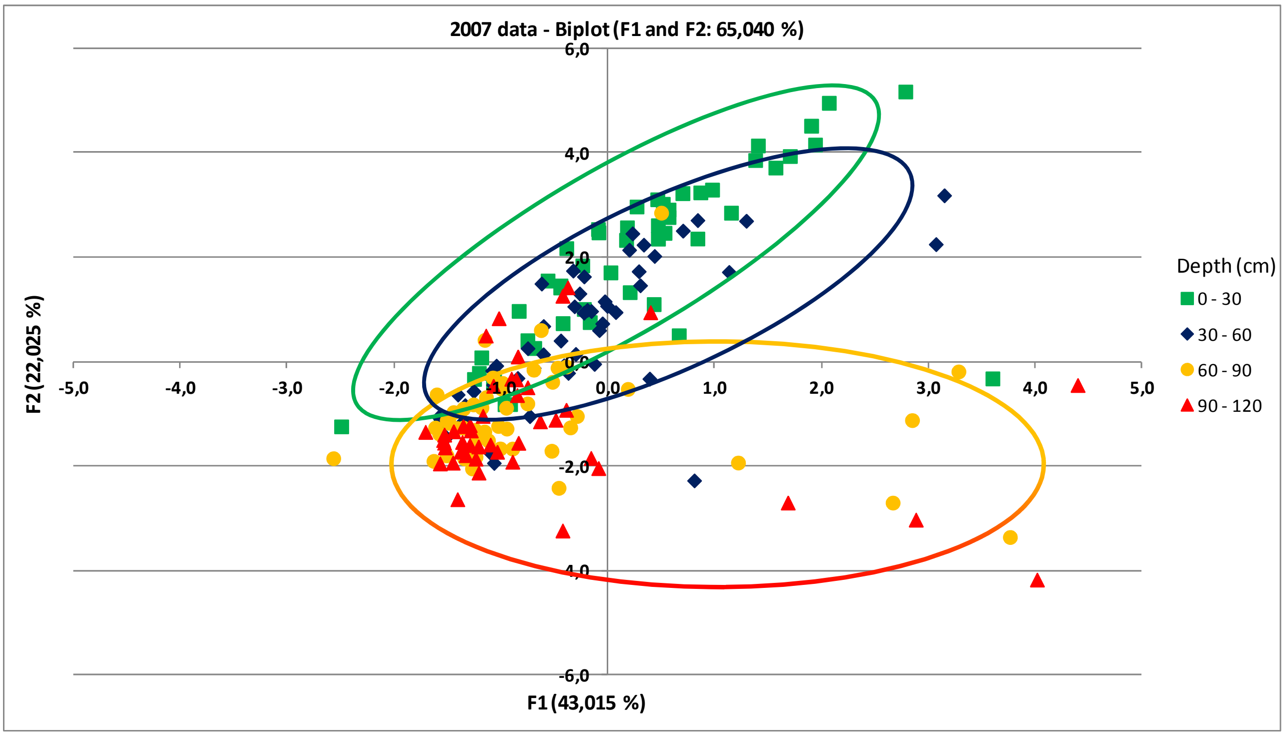


Figure 16. Observation plot of the (scaled) first two principal components for 2007 data.

Figures 15 and 16, show a projection of a multidimensional set of axes onto a two-dimensional plane (F1 and F2). 2007 data and 2008 data have been studied separately. These data were showing a better correlation between F1 and F2 once they were studied separately than together. Figure 16 shows that the depths in that figure observe similar behaviour compared to those in Figure 15. However the correlations of the variables are different in the 1994 data than in 2007, or in the 2008 data according to the F1 and F2.

Using multiple linear regressions, one PTF per sampling campaign was calculated. Equations were calculated based on the best variables that explained SAR correlation in PCA.

ECe, $pH_{1:2.5}$ and $EC_{1:5}$ were chosen for calculating the PTF for 1994 data, which has the lowest multiple correlation coefficient with 0.68, while PTF in 2007 and 2008 have very good correlation with 0.96. For the PTF, all variables selected were significant ($P < 0.05$).

Table 3. PTF calculated using multiple regression and evaluation indices (Vos indices (De Vos *et al.*, 2005)) for the PTF. Refer to text for symbols.

	Pedotransfer function		PTF validation			
Any	**Equation**	**Multiple Correlation Coefficient R^2**	**MPE**	**SDPE**	**RMSPE**	**Prediction Coefficient of determination R_p^2**
1994	SAR = - 63.22 + 0.17·ECe + 7.93·pHa + 0.67·ECa	0.68	-1.51	7.0	7.0	0.32
2007	SAR = -129.234 + 15.34·pHa - 0.28·pHe + 6.63·ECa + 1.35·ECe	0.96	-1.02	5.0	5.0	0.95
2008	SAR = -185.90 + 13.98·pHa + 8.59·pHe + 1.003·ECa + 1.65·ECe	0.96	0.61	3.4	3.4	0.87

These PTF should help in further studies to calculate SAR without the needed to have laboratory data of anions involved in the SAR equation. Thus, these PTF would economise the budge for future studies on that topic.

The PTF were validated using Vos indexes (De Vos *et al.*, 2005). The MPE, RMSPE and R_p^2 (equations 6, 7, 8 and 9) were calculated for a subset of the data (20%).

De Vos *et al.* (2005) described that the MPE, SDPE, and MSPE should be as small as possible and the prediction coefficient of determination (R_p^2) is a measure of the strength of the linear relationship between measurements and predictions, and indicates the fraction of the variation that is shared between them. Table 3 shows results for the validation highlighting a negative MPE for the functions of 1994 and 2007 and indicating a systematic underestimation of the salinity. However, for 2008 data we obtained an overestimation for the total dataset. MPE ranged from -1.51 to 0.61. The prediction coefficient of determination was moderate and showed little variation for 2007- 2008 data. This is not the case for the 1994 data, that the prediction coefficient of determination is quite low 0.32. Similar validation results were obtained in previously studies for PTF and soil salinity (Bouksila, 2011). The best PTF was obtained for the 2008 data, which shows the lowest MPE, SDPE and RMPE values.

3.4 *Temporal assessment of spatial distribution of soil salinity*

Salt profiles were calculated for 1994-1996 and 2008 using equation 1 and equation 2. We used the same profiles to be able to see the tendency of the soil salinity in the profile. After 12 years, there is a noticeable change in soil salinity. Comparing the same profiles with analytical salinity measures a decline of ECe is evident.

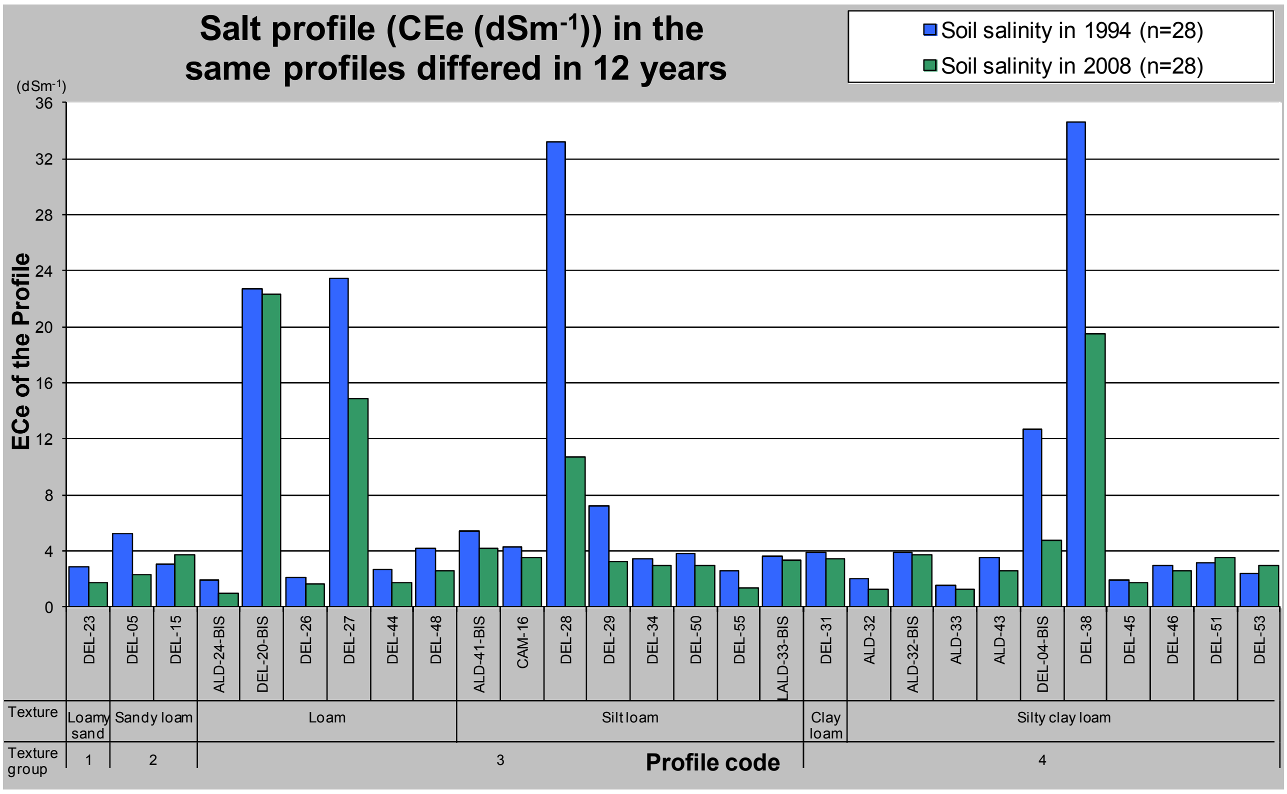


Figure 17. Assessment of the salt profile variation in 10-12 years time for 25 soil profiles

Figure 17 provides a comparison of the ECe in the same profiles in 1994-1996 and 2008. In Figure 17 there is a clear decrease of salt content in the profiles samples in a period of 12 years, however, in this figure there are four profiles, which this decrease is significantly evident (DEL-27; DEL-28; DEL-04-BIS; DEL 38). These profiles could be outliers, but we decided to keep them because they are located in the areas with highest salt content values in the salt distribution maps

(Figure 18a and Figure 18b). These four profiles were studied in more detail in depth, and the highest salt content is stored between 30-90 cm depth, whilst from 0 to 30 cm depth, the salt content is medium (< 4 dSm^{-1}).

Figure 18 shows continuous data or the extent of salt-affected variation in the 12 year period. In the 1994-1996 sampling campaign we used 2 regressions (Figure 9) to be able to obtain ECe. We used Regression between $ECa_{1.5}$ and ECh; and after it, the regression between ECe and $EC_{1:5}$.The ECe data estimated from ECh (EM38) (Figure 9) and the EC laboratory data were used to apply co-kriging to obtain the Figure 18a.

In 2007 sampling campaign, ECh gives good information on apparent soil electrical conductivity (ECa) to 100 cm depth. We used the correlation equation ECh-ECe to obtain ECa (Figure 9) in the 2007 sampling campaign. Eca data and the EC laboratory data were used in the co-kriging model to obtain soil salinity distribution in the study area as Figure 18b is shown.

The goal of this assessment is to show how the salinity varies for the whole study area. In general, there is a clear decline of the soil salinity as it is shown in figure 17. The noticeable decrease of soil salinity could be attributed the application of the environmental aid started on 1998, keeping longer continuous flooding of the paddies with fresh, running irrigation water. This agro-environmental measure or Best Management Practices, had an important requirement which could has influenced in soil salinity, it was to keep flooded the paddy rice fields for an additional four months during the autumn – winter. This fact could have affected positively on the salt content decrease of the Ebro Delta soils in 12 years time. Additionally, in 1995 was a very dry year with extreme weather conditions (Figure 3), this fact affected the Ebro river water flow and may the groundwater of the Ebro Delta. The Ebro river water flow was 1000 m^3/s less in 1995 than in 2007 and 2008. The saline wedge was probably very active in 1995, influencing electric conductivity of the Estella lagoon and the groundwater, making them more saline. Figure 18a shows the spatial salinity distribution in 1994-1996. Most of the highest values of ECe were around of Estella lagoon (Figure 18a). A hypothesis could be that during this dry period water from Estella lagoon could be used for irrigation, rising the salt content of the affected soils as well.

Figure 18b shows that, after 12 years, salts were washed out of the soils, since the salinity levels are lower. During this period of time, with no extremely dry periods, higher Ebro river water discharges and having the agro-environmental aid applied with four more months of fresh irrigation water to the paddy rice fields, all these facts affected positively to the washed up of the salts in the soils.

At the highest locations, soils are well developed, well drained and are non saline. There are some saline soils located in the flatter areas near the coast and have moderate coarse or coarse textures. The saline-sodic soils represent the maximum ECe/ECa values in the map. These soils are located in ancient fluvial soils or river lakes. The remaining soil types, except for soils near the water recharge areas of adjacent uplands, are moderately to strongly saline.

Most salinity problems in paddies occur in river deltas or other coastal areas, and are caused by seawater intrusion or low-quality irrigation water; however, few studies have used EM to measure the soil salinity of paddies (Enrique *et al.*, 2005; Li *et al.,* 2013, Herrero and Hudnall, 2014).

The study shows that there is both spatial and temporal variability of the salts as some authors found such as Herrero and Aragüés (2003) or Herrero and Hudnall (2014), but in different environments. These variations in soil salinity are due to management practices, specifically the use of water for irrigation coming from the Ebro River. For much of the year (April to February) the paddy fields are flooded with water. This water comes directly from the Ebro River, and it usually has low EC values (1994-1996 was, on average, 1.055 ± 0.088 $dS \cdot m^{-1}$ at 25ºC, and in 2007-2008 was, on average, 1.052 ± 0.27 $dS \cdot m^{-1}$ at 25ºC). In irrigated areas such as the Ebro Delta, the salt concentration of the drainage water is normally higher than that of the irrigation water.

The spatial distribution of soil salinity maps obtained with co-kriging for the 2 campaigns (1994-1996 and 2007-2008) were validated using De Vos indices (De Vos *et al.*, 2005). Table 4 show the validation results for the two maps.

The MPE, SDPE, RMSPE, and R_p^2 were calculated for a subset of data for the 2 campaigns (1994-1996 and 2007-2008). The MPE, SDPE, and MSPE should be as small as possible (De Vos *et al.*, 2005). Results showed a negative MPE for the both maps, indicating a systematic underestimation of the predicted soil salinity. Prediction errors were highest in the soil salinity map for the 2007-2008 than for the 1994-1996. The prediction coefficient of determination is highest and very good for the soil salinity map in 1994-1996, however the accuracy for the soil salinity map in 2007-2008 is good enough (R_p^2 = 67.8), as in previous papers describing the accuracy assessment of predictive maps, recommend that acceptable confidence levels should be 50-80% (Moran and Bui, 2002; Minasny and McBratney, 2007; Simo *et al.*, 2015).

Table 4. Results of the validation maps using De Vos indices (De Vos *et al.*, 2005).

	Validation maps			
Map	**MPE**	**SDPE**	**RMSPE**	**Prediction Coefficient of determination R_p^2**
Soil salinity in 1994-1996	-0.03	0.4	0.4	91.6
Soil salinity in 2007-2008	-0.02	1.7	1.7	67.8

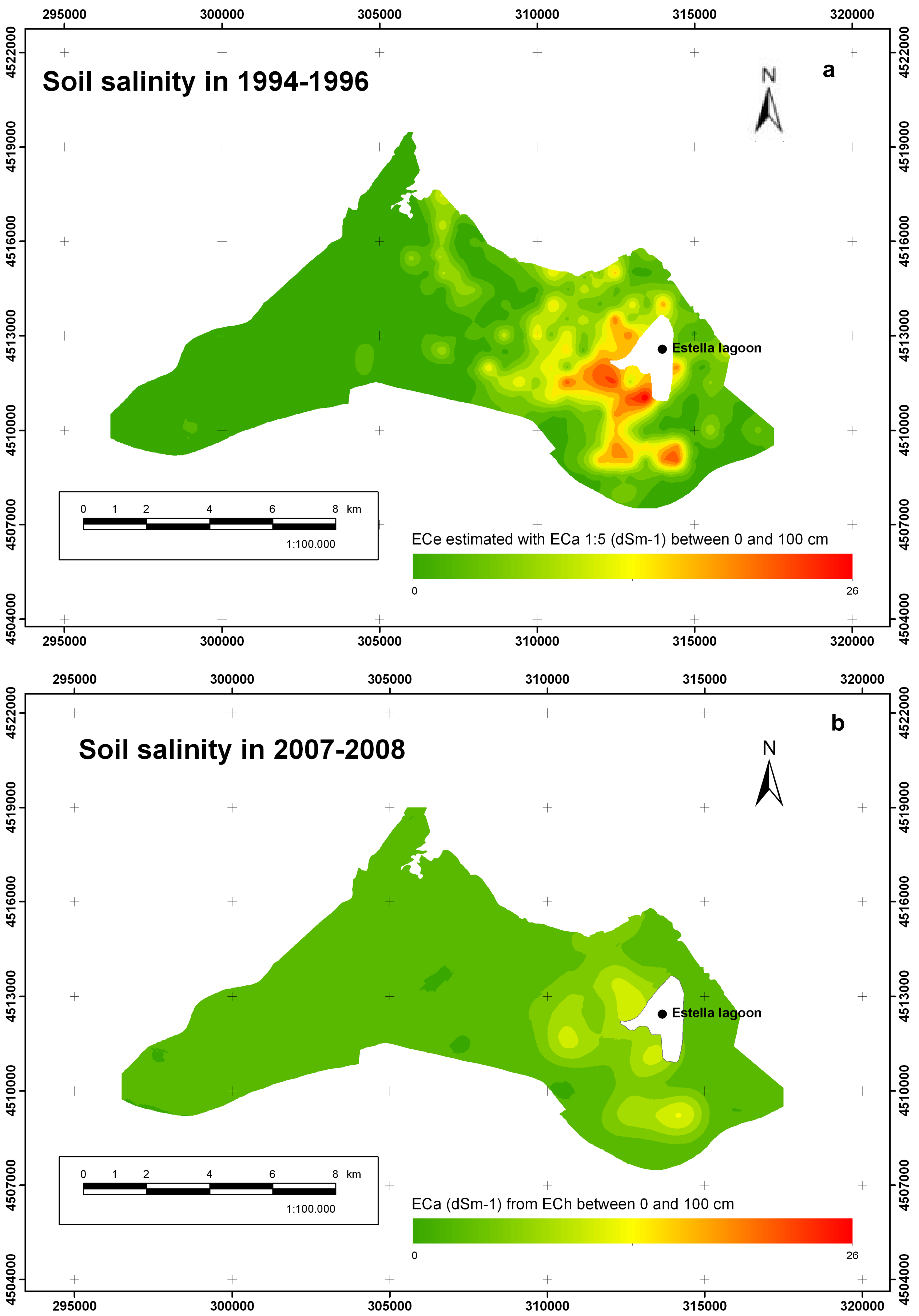


Figure 18. Salinity prediction map (ECa in dSm^{-1}) starting from the surface down to 100 cm. using co-kriging. 18a) Salinity map obtained with 1994-1996 data. 18b) Salinity map obtained with 2007-2008 data.

3.5 *Sodicity (SAR) for the whole profile*

The SAR profile is calculated with equations (3), (4) and (5). Sodicity is increasing despite a decline in salinity, similar results were found by Herrero and Castañeda (2015) in a similar environment in paddy fields after 20 years. This data is plotted in Figure 19 showing the distribution of 28 profiles studied in both sampling campaigns according to the salinity classification. Most of the profiles are non saline soils, although we have few sodic-saline and sodic soils in the study area (Figure 20). In Figure 19 there is a clear increase of SAR in the soil profiles in a period of 12 years, however, there are four profiles, which have had a significant increase (DEL-20-BIS, DEL-27; DEL-28; DEL 38, DEL-51). These profiles could have been considered as outliers, although we did not because they are the same profiles than have high salt profile (Figure 19).

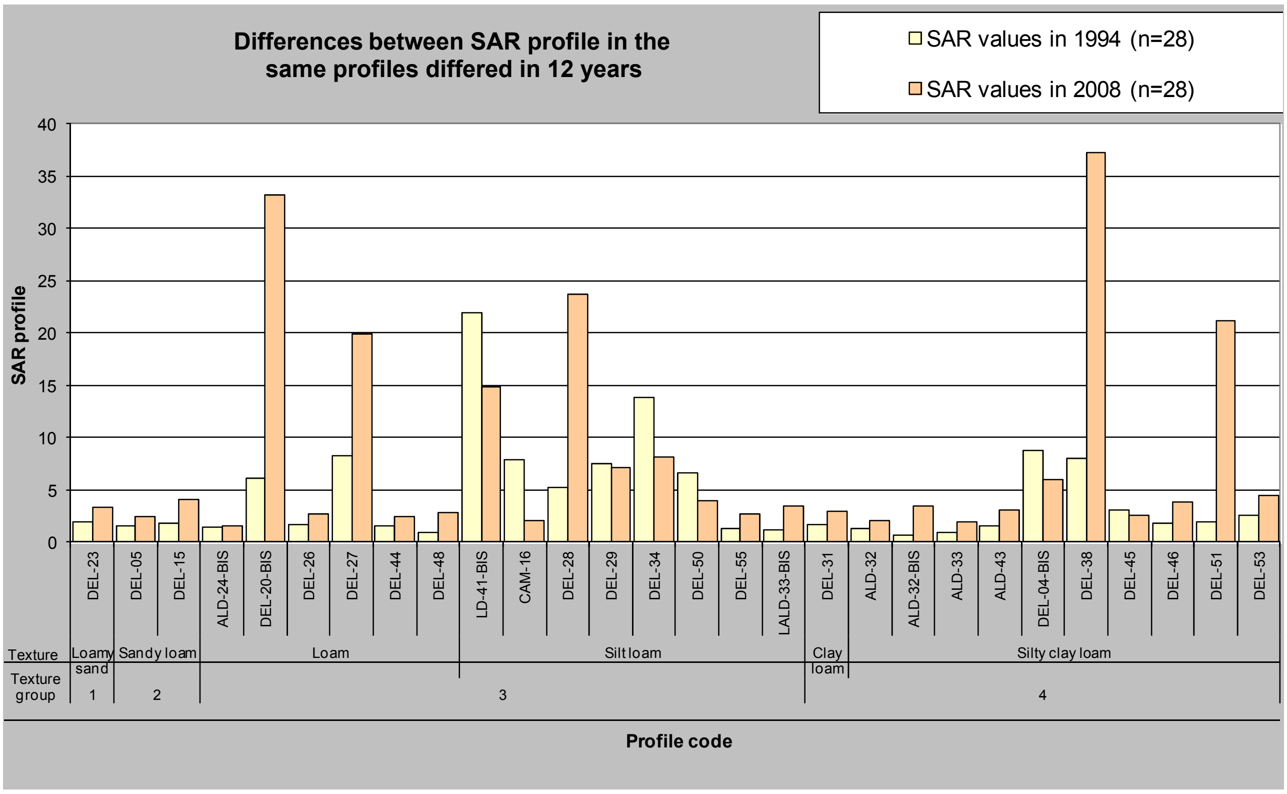


Figure 19. Assessment of the SAR variation in 10-12 years time for 28 soil profiles

After the 12 year period, a SAR increase is noticeable in the same profiles sampled. The severe decrease of Ca and Mg (Table 2) after 12 years, it did help to the SAR increase. The average SAR = 3.7 (Cl = 0.6 - 536.1; Na = 1.4 – 460.6; Ca = 2.7 -75.5; Mg =0.3 – 152) in 1994-1996, and 7.1 (Cl = 2 – 351.3; Na = 2.7 – 330; Ca = 0.7 - 31; Mg =0.9 – 48.6) in 2008, which indicated that SAR did change significantly after 12 years. This increase is mainly because Ca and Mg were washed out from the soils relative to Na, due to the fresh irrigation water during 12 years period and not using groundwater affected by the wedge that happened before.

Soils exposed to high SAR remain permeable because the clays remain flocculated because of the high salinity water. However, high SAR and low salinity cause the soil to become much less permeable; this depends on the clayeyness of the soil and the type of clay mineralogy (Van de Graaff and Patterson, 2001). Further studies related to clay mineralogy and soil structure in these sodic soils would be suggested.

Sodic soils has soil organic carbon (SOC) loss by increasing dispersion of aggregates, which increases SOC mineralisation, and increasing bulk density which restricts access to substrate for mineralisation (Wong *et al.,* 2010).

These results show that these soils should have special attention because of the high sodicity. These soils are water logged for most of the year because of the paddy fields. Notably, sodicity soils when excessively wet cause clay to swell, weakling the aggregates in the soil, causing structural collapse and closing-off of soil pores and water infiltration problems (van de Graaff and Patterson, 2001; Herrero and Castañeda, 2015).

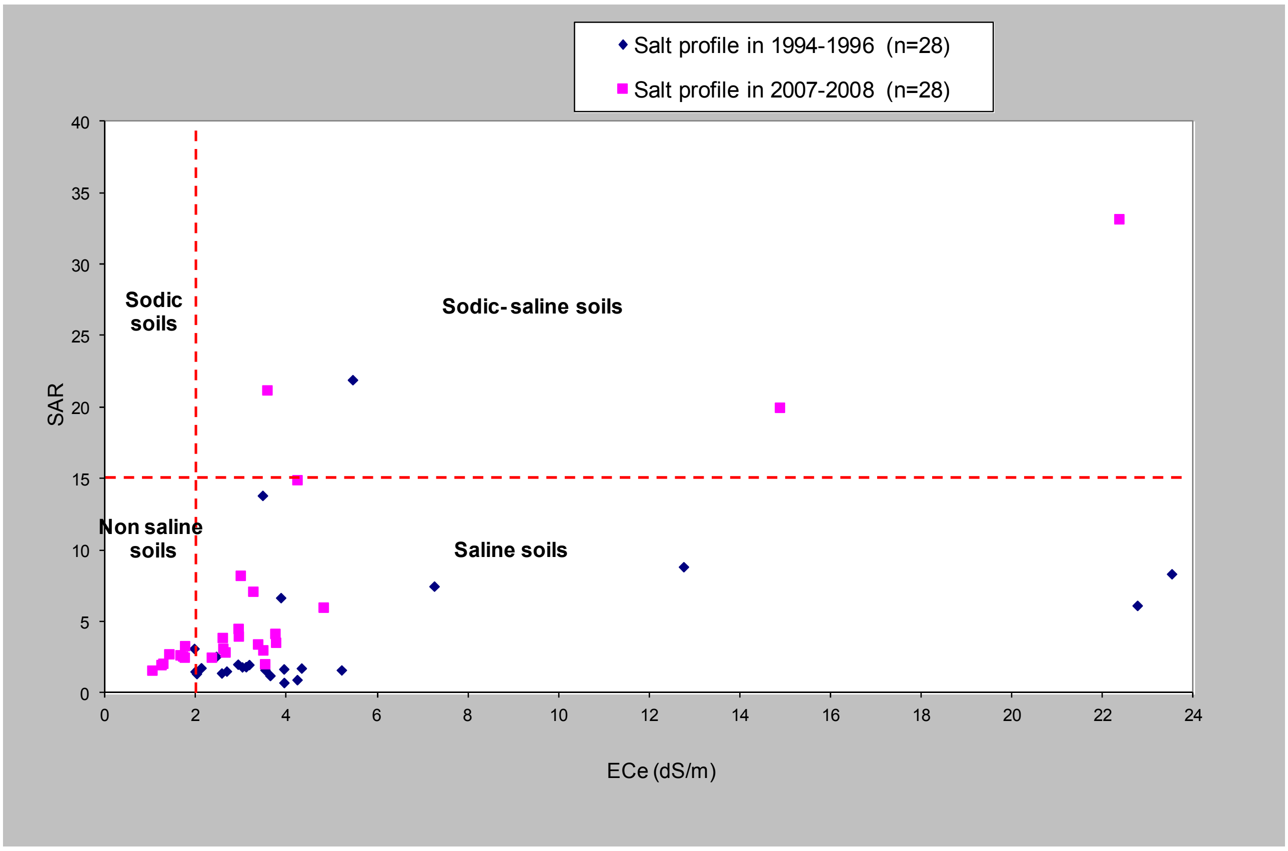


Figure 20. Relation between SAR and ECe for 28 soil profiles.

Irrigation water with low salinity, such as the water used in the Ebro Delta, is suitable for non-saline and non-sodic soils, but represents a long-term risk to saline soils. It leaches salts from the soil solution, allowing Na+ to come off the exchange sites and cause sodicity problems by dominating the soil solution (van de Graaff and Patterson, 2001).

4 CONCLUSIONS

- **Sampling campaign strategy and methods**

Although a laborious and time-consuming procedure when used over a large area, the application of a regular grid is a simple design. The appropriate sampling strategy must be determined based on achieving adequate precision of spatial information of soil properties. Otherwise the sampling might be excessively intensive or too sparse to provide an accurate spatial interpolation. Further research should be conducted to obtain the minimum optimal distance for a sampling design.

The use of legacy data difficult the re-sampling data, as we should consider that the profiles coordinates of legacy data are fine, however the coordinates could have a variation on precision and this variation would affect the re-sampling.

Sensor-based geospatial EM measurements provide relevant information on within-field variability of soil salinity. Therefore, techniques for rapid determination of soil salinity based on electrical conductivity were assessed and proved to be satisfactory.

- **Statistical approach**

This research provides a starting point towards improving an existing methodology for study soil salinity and this can be used to prevent the risk of soil salinity for land management in Mediterranean conditions. We suggest that the method used prevents an overestimation in predictions of soil cover degradation due to salinisation. The method used can also reveal line areas linked with discharges of saline waters or saline aquifers.

The developed PTFs for the estimation of SAR only requires the determination of pH and EC in the saturation extract, besides the EM measurements, therefore it reduces time and costs of sodicity estimation, without affecting substantially its accuracy.

As a result, splines work well when we want to convert soil profile data into standard depths, because with the splines we should be able to know the ECe at certain depths.

This study has elucidated the spatial correlations and variations in soil measures related to salinity.

The results of the various models applied can be used in making decisions regarding environmental monitoring remediation, land management and planning. The spatial predictions have special and particular importance before agricultural transformation of the land (from an economical point of view) or environmental restoration (selection of the most appropriate species adapted to soil salinity). This study shows methodologies that can be used for environmental quality assessment and planning at large and medium scales.

- **Spatio–temporal distribution of salt affected**

The measurements of parameters such as $EC_{1:5}$, ECe, ECa could be used as indicators. These allow both consultants and determining authorities to better predict appropriate design for sustainable practices and management of the hydrologic cycle.

The study shows that there is a spatial and temporal variability after 12 year. Applying this methodology with electromagnetic sensor (ECe-ECh-EM), the extent of saline soils has declined by 22% in 2007 with respect to 1994-96.

While soil salinity has declined, soil sodicity is increasing in the same profiles sampled. Soils exposed to high SAR and high salinity water remain permeable because the clays remain flocculated.

The salinity maps developed through this approach should be used to delineate potential saline areas and to locate additional soil sampling sites for a deeper characterization of saline and/or sodic-saline conditions and the subsequent potential deterioration of soil physical properties.

At the highest locations, soils are well developed, well drained and are non saline. The saline soils are located in the flatter areas near the coast and have moderate coarse or coarse textures. The saline -sodic soils represent the maximum ECa values in the map. These soils are located in ancient fluvial soils or river lakes. The remaining soil types, except for soils near the water recharge areas of adjacent uplands, are moderately to strongly saline.

High values of soil salinity (ECe) are found in the topsoils for all surface soil textures, with the highest values found in 1994-1996 campaign. This fact happens for ECe estimated with $EC_{1:5}$ for all soil textures, with the exception of the coarse texture. Soils with coarse textures are located mainly close to the coast and they have seawater entrance in depth, this in the reason why we have a soil salinity increase in depth. In the Ebro Delta, the salinity content depends on the soil texture.

- **Predictions of soil salinity in the study area by the global change**

Regardless of the climate change, in this time period of 12 years, there is a decrease of soil salinity in the Ebro Delta, probably caused by changes in irrigation management and regulation of the Ebro river through dams. This means that the management of irrigation and the river still have leeway to compensate the possible concentration of salts caused by increased evapotranspiration of the Ebro Delta. Further work is recommended in focussing on the effect of rising sea levels, and how may this affects to the evolution of the delta and the soil salinity in this area.

Overall, the selection of the right method is less significant than the use of the right data at the proper scale. Therefore, the knowledge and the understanding of the hydrological and soil processes and the use of detailed data representing these processes, is more relevant than the proper method applied.

Acknowledgment

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Chapter 3

QUANTIFYING AND MAPPING SOIL ORGANIC CARBON STOCK: A CASE STUDY IN A MEDITERRANEAN MOUNTAINOUS AREA (CATALAN PRE-PYRENEES)

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1 INTRODUCTION

Soil properties play an important role in land management activities such as agriculture, erosion control, environmental protection and nature conservation. Soil organic carbon (SOC) is an important indicator of land and soil health as it integrates several inherent soil properties, for example, clay content influences the capacity of soils to protect organic matter against mineralization. Also, responds strongly to aboveground landscape dynamics, including land-use change and land degradation (Vagen *et al.*, 2013). The assessment of SOC pools is essential for the evaluation of the ability of soil to sequester atmospheric carbon. This is particularly pertinent in relation to responding to climate change. The world´s mineral soils represent a large reservoir of carbon (C), with estimates ranging from 1115 to 2200 Pg (1Pg = 10^{15}g) C in the first metre (m) (Post *et al.*, 1982; Eswaran *et al.*, 1993; Berboux *et al.*, 2002). Almost half (approximately 45%) of soils in Europe have a low or very low organic matter content (meaning 0-2% organic carbon) and 45% have a medium content (meaning 2-6% organic carbon) (European Commission (EC), 2006, 2012). Soils with low and medium SOC content can provide a good platform for crop production and good pasture for grazing animals, but the expansion and intensification of agriculture during the 20th century, has resulted in a decline in the organic carbon contents of many soils (Sleutel *et al.*, 2003). This decline in SOC has important implications for agricultural production systems, in addition to implications for soil properties such as soil structure, water retention, cation exchange capacity, nutrient retention and release, bulk density, biological activity, and so on.

Soils can mitigate or contribute to carbon budgets depending on management and are seen as having a crucial influence on climate change. The Kyoto Protocol allows carbon sinks to be selected in the categories of grazing, cropland and forest management (Smith, 2004). SOC stocks are a key factor when defining national greenhouse gas inventories. An accurate measure of SOC stocks and consequent changes over time are required for monitoring when reporting to the United Nations Framework Convention on Climate Change (UNFCCC). The decline in SOC is recognized as one of the eight soil threats identified in the EU Thematic Strategy for Soil Protection (EC, 2006, 2012). One of the key goals of the Soil Thematic Strategy is to maintain and enhance SOC levels across the EU, however the EC policy document, the *Roadmap to a Resource Efficient Europe* (EC, 2011), sets the objective of increasing current levels of organic matter in the EU by 2020 (Lugato *et al.*, 2013).

The global carbon cycle project identified the major factors influencing carbon sequestration (IPCC, 2000, Robert and Saugier, 2003). SOC storage varies mainly in the long-term as a result of climate (altitude and latitude), and soil-forming factors; whereas vegetation and changes in land management and land use patterns affect storage in the short-term (Batjes, 1996; Carré *et al.,* 2007). The highest SOC concentrations were observed in the upper soil layers but large amounts are also stored between 1 and 2 m depth (Batjes, 1996). The review indicated that soil organic matter and biological activity of the litter decomposition are the main factors that could modify the sequestration of C in soil. The biomass brings fresh organic matter to soil as litter and roots which are decomposed by soil organisms. The rate of decomposition depends on soil moisture content, temperature and oxygen content. Furthermore, decomposition can be accelerated by land-management practices, such as intense tillage practices. Climate change modeling is complicated by these factors and interactions (Carré *et al.,* 2007). For example, SOC storage in mountain areas is highly heterogeneous, mainly as a result of local-scale variability in the soil environment (topography, stoniness, parent material) and microclimate.

Globally, Jobbágy and Jackson (2000) found that soil C stocks are positively correlated with mean annual precipitation and negatively with mean annual temperatures. SOC is more deeply distributed in arid scrublands than in arid grasslands, and sub-humid forests have shallower SOC distribution than sub-humid grasslands. The overall quantity of organic carbon in a given soil is determined largely by climate and organic inputs but can also be significantly affected by land use (McKenzie *et al.,* 2006). Our capacity to predict and ameliorate the consequences of global change depends in part on a better understanding of the distributions and controls of soil organic carbon (SOC) and how vegetation change may affect SOC distributions with depth (Jobbágy and Jackson, 2000).

SOC pool reporting depends upon suitable data in terms of organic carbon content and soil bulk density and on the methods used to upscale point data to comprehensive spatial estimates. In order to quantify SOC, soil surveys should be conducted to determine, characterize and quantify parameters for SOC stock, such as organic matter content, bulk density, rock fragment and soil depth. Moreover, the sampling should be georeferenced and carried out at an adequate scale to be representative.

The prediction of soil SOC stocks across the landscape has been increasingly studied in many areas of the world (Garcia-Pausas *et al.*, 2007; Miller, *et al.*, 2015). However, most SOC studies and inventories are confined to 30 cm soil depth (Zdruli *et al.,* 2004; Stolbovoy *et al.,* 2007) but the amount of SOC stored below 30 cm is of relevance in many ecosystems (Adhikari *et al.*, 2014). Generally, the highest levels of SOC occur in the topsoil and decrease with depth. Some soils store about 37 to 39% of their total SOC between 1 and 2-m depth (Lorenz and Lal, 2005).

There are several tools to be used for modelling and quantifying SOC stocks in depth, such as; a) spline functions (Odgers *et al.*, 2012), b) exponential decay functions (Minasny *et al.,* 2006) and c) soil-type specific or profile depth (Batjes, 1996; Batjes, 2008). The present study shows how the use of profile depth could show a variation on the quantification of SOC stocks and how this variation could be mapped using a detailed soil map or digital soil mapping techniques.

This study aims to produce a detailed cartographic soil inventory and identify the SOC stock in mineral soils, in the main soil types and land uses, of a study area in the Iberian Pre-Pyrenees (Canalda river basin). This study assessed carbon stocks in the soils and evaluated the effect of land use change on SOC storage in agro-silvo-pastoral ecosystems, along altitudinal gradients, through the determination of the soil potential for carbon storage and the quantification of this C reservoir. We compared alternative mathematical characterizations of the vertical distribution of SOC below the first meter and compared with other soil properties such as texture and calcium carbonates. Additionally, SOC maps are produced with the objective of identifying and securing existing information for SOC and to show the spatial distribution and geographical variation of SOC stock at different depths.

2 METHODS

2.1 *The study area*

The study area was located in the Canalda river basin, a tributary to the Ebro Valley (Catalan Pre-Pyrenees, NE Spain) with an area of 10 km^2 (Figure 1).

The study area is mountainous with altitudes of between 1100 and 2100 m and, slopes between 10-50%. The parent materials are calcareous conglomerates, calcilutite and limestone. The most common soils in the area are Inceptisols and Entisols, but Mollisols are also present. Soil depths vary from <40 cm to 120 cm but in general range from 70 cm to 100cm. All soils are stony.

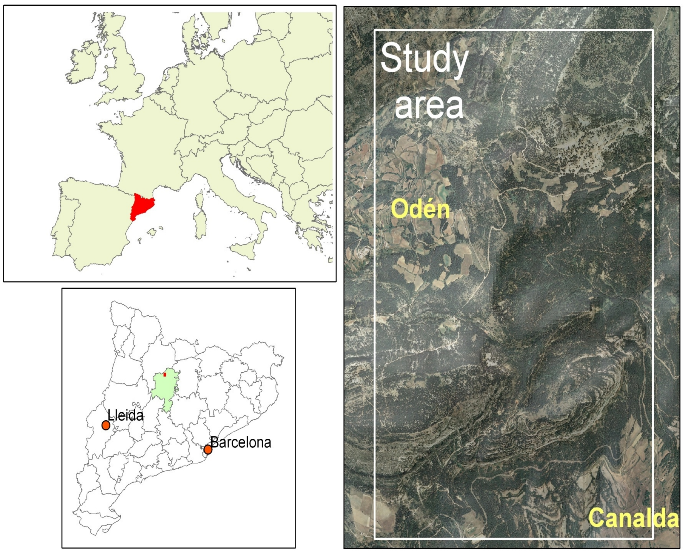

Figure 1. Map showing location of the study area.

The climate is Mediterranean in the lowest parts to Subalpine in the highest parts according to Ubalde (1997). Most of the rain falls in autumn and spring, occurring typically as isolated, often violent storms. The annual rainfall ranges from 500 to 850 mm, and it's distributed along an altitudinal gradient. The study area is described as sub-humid and humid climate in the highest altitudes. The hottest and the coldest months are August (19ºC) and January (0ºC), respectively, with a mean annual temperature of 12.1 ºC (Figure 2). The driest season is in winter and wetter periods are in spring and summer. Soil temperature and moisture regimes are mesic-frigid and ustic-udic, respectively (Estruch *et al.*, 1999, Orozco *et al.*, 2006, Loaiza, 2007).

The area has been subjected to strong land use changes over the last 100 years, mainly the abandonment of agricultural land and its conversion to pasture or forestry (Ubalde *et al.*, 1999). Forestry and agriculture are the main land-uses of the area with 71% and 24%, respectively. The forest use varies from diverse forest environments (*Pinus nigra, P. sylvestris* and *P. uncinata*) to subalpine and Sub-Mediterranean vegetation (*Quercus ilex* sp *ballota*). Rocky areas occur in 2% of the study area. These areas are mainly dominated by slopes exceeding 35%. The principal agricultural uses include cereals, potatoes and pastures. The potato crop is a main agricultural product in these high mountain areas, as the high altitude (1100 to 1600 m) reduces the prevalence of pests, reducing the need for pesticide applications. The crop fields have a minimal management; this means that soil-friendly tillage practices or conservation agriculture are applied being mainly contour farming and manure application. Agricultural practices generally reflect a reliance on indigenous knowledge.

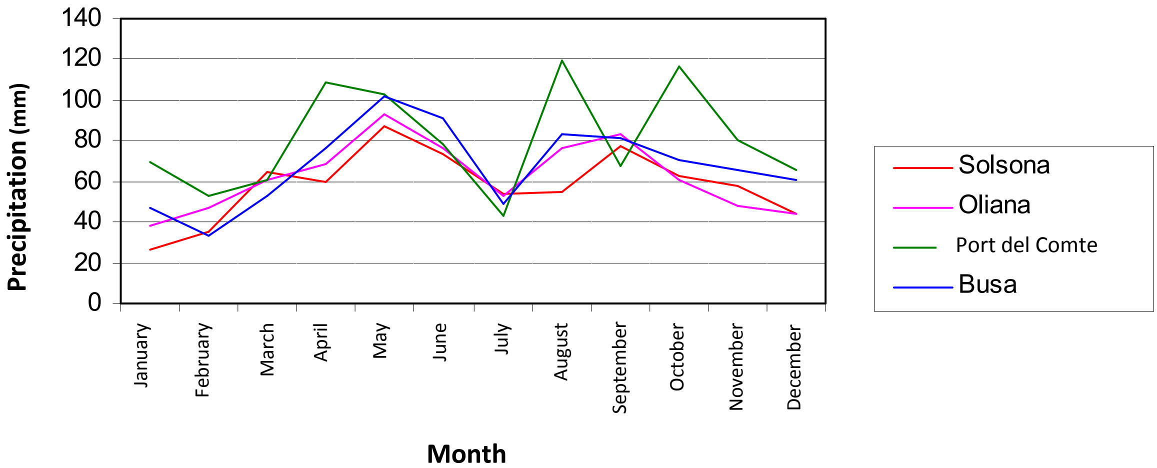

Figure 2. Monthly rainfall (mm) of the closest weather stations within a 12 km radius (Solsona (690 m), Oliana (469 m), Port del Comte (1800 m) and Busa (1200 m)) to the study area, Canalda river basin. Source: Estruch (1999).

2.2 Cartographic soil inventory

The aim of the field work was to collect the necessary data to be able to produce a soil map at a detailed scale (1:25,000), with an intensity of 0.04-0.2 observations/ha (Porta, *et al.*, 2005; McKenzie *et al*., 2008), obtaining a total of 53 pits described in the area.

There are five major control factors for soil development: parent material, climate, vegetation, topography, and time (Jenny, 1941). Although Jenny characterised these factors as being key to soil formation, they are all conceptually linked with SOC dynamics as SOC is intimately linked with soil development (Hobley *et al.*, 2015). Today this information can be obtained from existing, large-scale soil maps, climatic data, land use/cover maps, digital terrain models and their derivatives, parent material/geology, and landscape position. Soils units were delimited by photo interpretation based on field observation (soil pits, soil cuttings and augering) and with the consideration of existing control factors mentioned above.

The soil survey was focused on being able to obtain enough field data for the production of the soil organic carbon map. Each field observation was described according to CatSIS methodology (Boixadera *et al.*, 1989), and samples were taken for each horizon. All samples were analyzed in the laboratory to determine physico-chemical characteristics, such as clay content, pH, $CaCO_3$, Organic Matter (OM) by wet oxidation, soil texture (%), and cation exchange capacity (Porta, 1986). The soil classification system used was Soil Taxonomy (Soil Survey Staff, 2014a), producing a soil map with 26 different soil types. The profiles were correlated with World Reference Base (WRB) classification system (IUSS Working Group WRB, 2014).

All the information related to the soil map and climate, relief, parent material, and landscape factors were recorded in a Soil Geodatabase (SGDB) using ArcInfo ArcGIS software, which was the software used to produce the final soil map.

2.3 *Calculating soil organic carbon stock*

Thirty-seven profiles were described and characterised. In order to quantify SOC, all profile horizons were sampled to characterise bulk density and rock fragments in more detail.

The standard soil survey procedure used in the soil map of Catalonia used the following determinations:

- Assessment of rock fragment volume. 2 kilograms of soil were taken for each horizon, down to a depth of 200 mm or to a lithic contact. For each horizon, a 2 mm-mesh sieving was conducted in the field, the weight percentage of coarse fragments was measured, and it was converted to volume percentage.

- The bulk density was measured by three methods: core sampling (Nacci *et al.,* 1999), the aggregate method (Grossman *et al.*, 2002) for each horizon and the hole method (Nacci *et al.*, 1999) for surface horizons, but for SOC calculations only the aggregate method (Grossman *et al.*, 2002) was used.

- Organic carbon was determined by the standard wet oxidation method (Walkley-Black method) (Porta, 1986).

Distribution of SOC in deeper soil layers in different systems may result in different SOC proportion depending on the soil depth sampled (Angers & Eriksen-Hamel, 2008). Thus, this study focused on quantifying SOC at different depths, with the objective of comparing how SOC stock varied according to each map unit.

Calculations of SOC stocks for each profile were carried out according to the classical way of calculating C stock for a given depth: by summing C stock of successive horizons in the profile. This calculation requires the continuity of the profile by depth. The following equation was used for the estimation of the SOC in each horizon (OCh):

$$OCh(Mg/ha) = OC \times BD \times T \times (1 - RFV) \times 0.001 \quad (1)$$

Where OCh = Organic Carbon of the horizon (Mg/ha), OC = Organic Carbon (%), BD = Soil Bulk Density (kg/m^3), T is thickness of the horizon (cm) and RFV is Rock Fragment Volume.

However, in many cases horizons depths did not align with the depth intervals selected for calculation of SOC. Where the 0 – 15 and 0 – 30 cm depth interval fell without a horizon, the following equations were used (50 and 100 cm calculations follow the same procedure) for these:

$$SOC\ (15\ cm) = OCh_1 + \left(\frac{OCh_2}{Dh_2} \cdot (15 - UBh_2) \right) \quad (2)$$

$$SOC\ (30\ cm) = OCh_1 + OCh_2 + \left(\frac{OCh_3}{Dh_3} \cdot (30 - UBh_3) \right) \quad (3)$$

Where OCh_1, OCh_2, OCh_3 = Organic Carbon (Mg/ha) calculated with the previous equation (1) for different horizons, Dh = depth (cm) and UB = upper boundary in this horizon (cm).

SOC was calculated for each modal profile for each map unit. The following depth intervals were considered: 0 - 15 cm, 0 - 30 cm, 0 - 50 cm and 0 - 100 cm. SOC was calculated for each depth, disregarding organic horizons due to the low stability of organic matter. The 0 – 15 cm and 0 - 30 cm depths are chosen because they represent the stock of carbon susceptible to anthropic action.

2.4 *Mapping SOC using several approaches*

Several options were considered for SOC mapping including the use of the detailed soil map and geostatistical methods (ordinary kriging and universal kriging).

2.4.1 *Detailed soil map*

The procedure for mapping SOC using a detailed soil map is the following: the computed and measured ranges of SOC were used to enhance the existing soil map. SOC data was prepared by joining to the traditional vector digital soil map. These SOC stocks were pre-summarized to the map unit level using best-practice generalization methods (Soil Survey Staff, 2014b). The generalisation methods included map unit components from modal profiles.

2.4.2 *Geostatistics methods*

Kriging is a family of estimators used to interpolate spatial data. This family includes ordinary kriging, universal kriging, indicator kriging, co-kriging and others. The choice of which kriging to use depends on the characteristics of the data and the type of spatial model desired. Several options were considered including ordinary kriging and universal kriging (Lefohn *et al.*, 2005). Kriging fits a function to a specified number of points or all points and determines an output value for each location. The basic idea of kriging is to predict the value of a spatial variable at an unobserved location from observations at locations nearby. Kriging assigns values to unknown locations using a simple linear weighted average of neigbouring known points (de Smith *et al.,* 2015). Kriging is typically used when a spatially correlated distance or directional bias in the data is known and is often used for applications in soil science (Childs, 2004). Weighting in kriging uses a sophisticated averaging methodology and based on this along with its suitability for soil applications, kriging was considered the most appropriate method for development of the soil property maps here. The main idea of kriging is to take a reference point x, and compare this value at other locations at increasing distances from the reference point. This is done with any pair of points in the area. As distance increase, this measure will likely increase also, and in general a monotonic increase in squared difference with distance is observed for most of the studies. (Longley *et al.,* 2001). This builds upon the concept of the semivariogram.

Ordinary kriging could readily be developed based upon the available values from the data points. It provides an estimate at an unobserved location of variable z, based on the weighted average of adjacent observed sites within a given area. The theory is derived from that of regionalized variables (Matheron, 1965, 1971) and can be briefly described by considering an intrinsic random function denoted by $Z(S_i)$, where Si represents all sample locations, i = 1,...,n. An estimate of the weighted average given by the ordinary kriging predictor at an unsampled site, $Z(S_0)$, is defined by:

$$Z(S_0) = \sum_{i=1}^{n} \lambda_i Z(S_i) \quad (4)$$

where λ_i are the weights assigned to each of the observed sample sites. These weights sum to unity so that the predictor provides an unbiased estimation:

$$\sum_{j=1}^{n} \lambda_i = 1 \quad (5)$$

The weights are calculated from the matrix equation

$$c = \boldsymbol{A^{-1}b} \quad (6)$$

where **A** is a matrix of semivariances between the data points; **b** is a vector of estimated semivariances between the data points and the points at which the variable z is to be predicted; and c is the resulting weights and the Lagrange Multipliers ψ (Triantafilis, 2001).

Ordinary kriging method assumes no trend in the data as opposed to universal kriging which assumes a general polynomial trend model. Universal Kriging (UK) uses a regression as part of the process, with the unknown values presumed to have a local linear or quadratic trend (de Smith et al 2015).

Universal kriging is used to estimate spatial means when the data have a strong trend and the trend can be modeled by simple functions (Lefohn *et al.,* 2005). Trend is scale dependent. Universal kriging allows the incorporation of both deterministic and stochastic components in kriging:

$$S(x) = \sum_{j=o}^{p} b_j\, q_j(x) + e(x) \quad (7)$$

The first term represents the nonstationary trend, which is modelled as a set of linear functions of the environmental variables Q with parameter vector b, and the second term is the stochastic component modelled by variogram (McBratney *et al.*, 2003). Alternatively, the trend function can be modelled separately, where kriging is combined with regression (Ahmed and DeMarsily, 1987; Knotters *et al.*, 1995). This method involves regression of the soil attributes as a function of predictor variables.

In universal kriging, we assumed both a non-constant mean and the presence of local (spatial) variation. This gives the following decomposition with any of the following sets of terms used interchangeably:

Yi =	large scale variation	+	small scale variation
	trend or drift		residual error
	nonstationary mean		random variation
	deterministic signal		noise

Co-variants used within the universal kriging approach included a land use map (Land use Map of Catalonia in 2002), auxiliary variables predictors derived from a Digital Elevation Model (DEM) (15 m resolution) such as slope, aspect, hillshade and soils-landscape units map. Land-use data was applied as this reflected the soil management types, in terms of fertilisation, conservation agriculture, and other managements, which are important drivers of SOC. The DEM provided information on altitude, slope degree and aspect, and these data types were selected as they represent natural changes in SOC as a result of the major topographical features and provide an indicator of the climatic influence on soils at high altitudes. However, the best co-variants to fit in the universal kriging were slope, land use and soils-landscape units map. All co-variants and SOC stocks were studies using linear models (ls) in R software (R Core Team, 2013), to select the best ones.

R software scripts with ordinary kriging, Inverse Distance Weighting (IDW) and universal kriging models were supplied by ISRIC-World Soil Information Institute in Wageningen, The Netherlands. These models were used to develop the SOC maps. The scripts were adapted to the study area data and were run successfully for the SOC at different depths.

2.5 *Validation approach*

For the spatial prediction of SOC stocks at the different methods were applied, and an additional sampling of soil organic matter was conducted around the modal profiles of each soil mapping unit (Figure 3). The study area was celled by 100 x100 m grid. The number of cells to samples in each soil mapping unit was decided according to the largest of the map unit. Consequently, the largest map units had a maximum of five cells to be sampled and the smallest map units had two cells to be sampled. A total number of 110 cells were sampled. At each cell, eight samples were taken at different depth 0-15 cm, 15-30 cm and 30-45 cm. Soil organic matter was analysed in the laboratory using the Walkley-Black method described by Porta (1986). In order to be able to calculate SOC stock at 0-15 cm, 0-30 cm and 0-45 cm, in this approach, we assumed the rest of the variables to be those of the modal profile, and we applied Equation 1, 2 and 3. At each cell the mean of the eight sampled points of SOC at 0-15 cm, 0-30 cm and 0-45 cm were calculated.

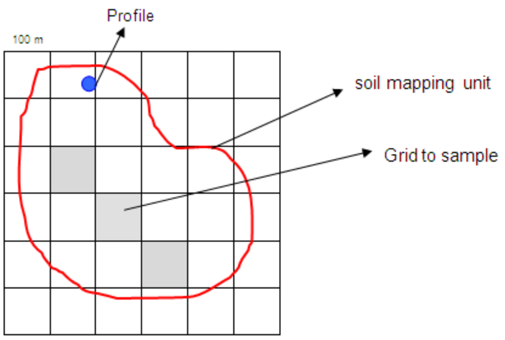

Figure 3. Sampling strategy for the validation approach

These 110 values of SOC stock were used as independent validation data set. For the validation of the maps additional indices such as De Vos indices (De Vos *et al.*, 2005) were calculated. They were applied to establish the accuracy of the maps developed. These were Equation 8, the mean predicted error (MPE) and Equation 9, the standard deviation of the prediction error (SDPE), Equation 10, the root mean square prediction error (RMSPE); and Equation 11, the prediction coefficient of determination (R_p^2) as shown below (De Vos *et al.*, 2005).

$$\text{MPE} = \frac{1}{n}\sum_{i=1}^{n}(\widehat{Pb,i} - Pb,i) \quad (8)$$

$$\text{SDPE} = \sqrt{\frac{1}{n-1}\sum_{i=1}^{n}((\widehat{Pb,i} - Pb,i) - MPE)^2} \quad (9)$$

$$\text{RMSPE} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(\widehat{Pb,i} - Pb,i)^2} \quad (10)$$

$$R_p^2 = \frac{[\mathrm{cov}\ Pb,i,\ \widehat{Pb,i}\,]^2}{\mathrm{var}(Pb,i) - \mathrm{var}\ (\widehat{Pb,i})} \tag{11}$$

where Pb,i and $\widehat{Pb,\imath}$ are the observed and predicted salinity values, respectively; n the number of observations; and var and cov the variance and the covariance function, respectively.

2.6 *Statistics*

SOC was normalized using a lognormal transformation. Significant differences (*P < 0.05*) were tested by one way ANOVA for the land use, depth, and altitude. Significant differences for means were produced by the Fisher test for the same levels of significance as above. Organic carbon (%) and calcium carbonate content were analyzed using Pearson's correlation analysis. The statistics were applied with JMP pro 11(JMP, 2007) and Minitab 15 (Minitab, 2009). The SOC map was developed using ArcInfo ArcGIS software (ESRI, 2012).

3 **RESULTS AND DISCUSSION**

3.1 *SOC stock, sampling depth and driving factors*

SOC stocks were estimated at 0 - 15 cm, 0 - 30 cm, 0 - 50 cm and 0 - 100 cm depths, for a subset of 34 profiles. SOC stocks ranged from 33 to 200 Mg/ha (Table 1).

The SOC stock in the upper 1 m in the study area (33 – 150 Mg/ha) was of the same order of magnitude as those found by Jobbágy and Jackson (2000) in temperate grasslands (117 Mg/ha) or by Doblas-Miranda *et al.* (2013) in forestry, scrublands and grasslands of peninsular Spain (20 – 160 Mg/ha).

Modal profile	Depth	Horizons	Coarse elements (%)	Bulk density (kg/m^3)	OCh (Mg/ha)	SOC (Mg/ha)
Typic Haplustepts	0-60	A1;Bw1;Bw2;R	64.04 - 73.13	1487.2 - 1655.2	A1;Bw1;Bw2 14;33.1;8.5	55.7
Typic Ustifluvents	0-110	A1;A2;AB;Ab;R	43.3 - 68.5	1352.3 - 1554.6	A1;A2;AB;Ab 27.2;26.7;33;39.6	126.7
Lithic Ustorthents	0-40	Oi;Oa;A1;A2;R	25 - 63.5	1000 - 1533.6	Oi;Oa;A1;A2 24.6;53.8;33.7;49.7	83.5
Udic Calciustepts	0-180	Ap1;Ap2;Bw;Bk;C1;Ck	64.5 - 92.88	1390 - 1737.4	Ap1;Ap2;Bw;Bk;C1;Ck 40.1;9.8;3.3;9;14.1	76.6
Typic Hapludolls	0-80	O;A1;Bw1;Bw2	20.19 - 70.64	1000 - 1782.7	A1;Bw1;Bw2 57.7;12.8;33.7	105.3
Typic Udorthents	0-81	O;A1;A2;Bk/R	46.92 - 84.23	1000 - 1686.9	A1;A2;Bk/R 54.8;41.9;34.4	131.1
Typic Calciustepts	0-120	Oi;Oa;A1;Bw;Bk;C	78.89 - 91.68	1000 - 1711.8	A1;Bw;Bk;C 28.6;31.5;15.5;16	91.9
Typic Eutrudepts	0-80	O;A1;Bw1;Bw2;Bk/Ck;Ck1	50.75 - 84.82	1063.6 - 1763.2	O;A1;Bw1;Bw2;Bk/Ck 39.3;9.4;2;2.5;19.8	33.9
Entic Hapludolls	0-140	A1;Bw1;Bw2;Bw3;Bwk	51.10 - 81.10	1346.7 - 2353.5	A1;Bw1;Bw2;Bw3;Bwk 64.9;23.4;6.1;6.9;6.4	107.9
Typic Calciustolls	0-100	O;A1;A2;Bw;Bk	78.88 - 81.11	1000 - 1700	O;A1;A2;Bw;Bk 21;27.4;18.7;10.7;11.6	68.6

Table 1. General characteristics to calculate SOC from a sub set of modal profiles.

Topsoils had more SOC than deeper horizons. The median SOC stock from 0 - 15 cm was 1.8 Mg/ha*cm compared to < 0.5 Mg/ha*cm in 50 – 100 cm (Figure 4). SOC decreased with depth in all land use types.

SOC was calculated at each horizon per cm with the goal to be able to show how varies the OCh (Mg/ha·cm) in depth (Figure 4). These data was correlated with texture at each horizon with the objective to show correlations between these soil properties (Figure 6).

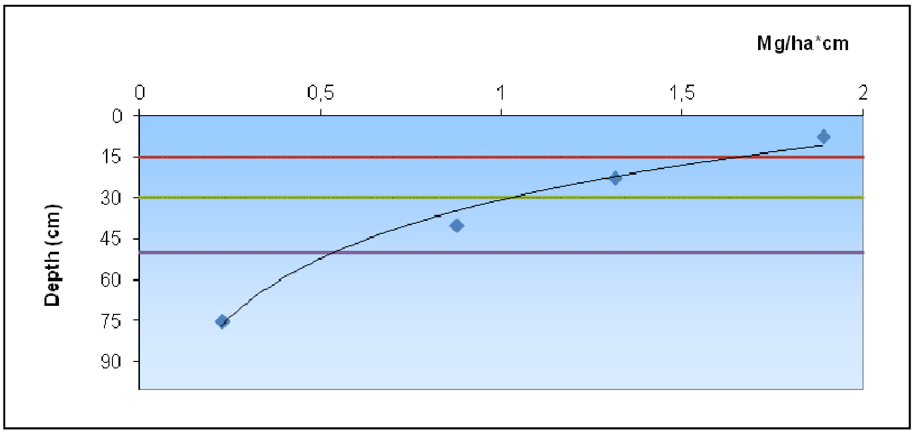

Figure 4. Averaged SOC stocked (Mg/ha) per cm for each calculated depth.

Significant differences were found between different horizon types (Figure 5). Surface horizons tend to have more SOC than deeper horizons, regardless of other genetic qualifiers. Pearson's correlation (r = - 0.38) between $CaCO_3$ (%) and OC (%) for the horizons, shows a weak, inverse relationship: when one increases, the other decreases proportionally constant. Despite this relationship, carbonates contribute to SOC protection and aggregate formation and stabilization (Bronick and Lal, 2005).

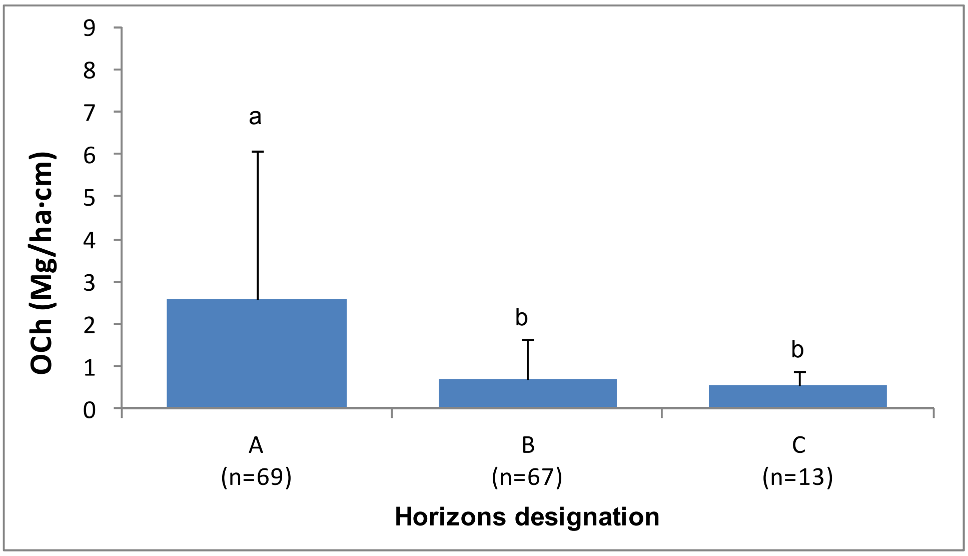

Figure 5. Mean of organic carbon (Mg/ha*cm) in horizon types. Different letters indicate significant differences at *P<0.05* according to Fisher test.

The clay content is known to improve binding between organic matter and soil particles through humic-clay complexes and through the protection of the soil aggregates (Haynes and Swift, 1990). The relationship between SOC stock and clay content was not always clear, however in our study area there was a clear increase in SOC stocks with increase in clay content as found in other studies (Leifeld *et al.,* 2005). Figure 6 shows that soils with coarse texture (< 15% clay and > 70% sand) usually have less organic carbon than soils of finer textures (loam or clay), and the reason for this is because coarse textured soils have lower moisture content and greater aeration that results in more rapid oxidation of organic matter (Jones *et al.*, 2005). This interaction was highlighted by the Pearson's correlation between organic carbon (OCh) and the clay content (r=0.198) at horizon level. The same order of magnitude of this relationship has also been described in other regional studies (Verheijen *et al.*, 2005; Goidts and van Wesemael, 2007).

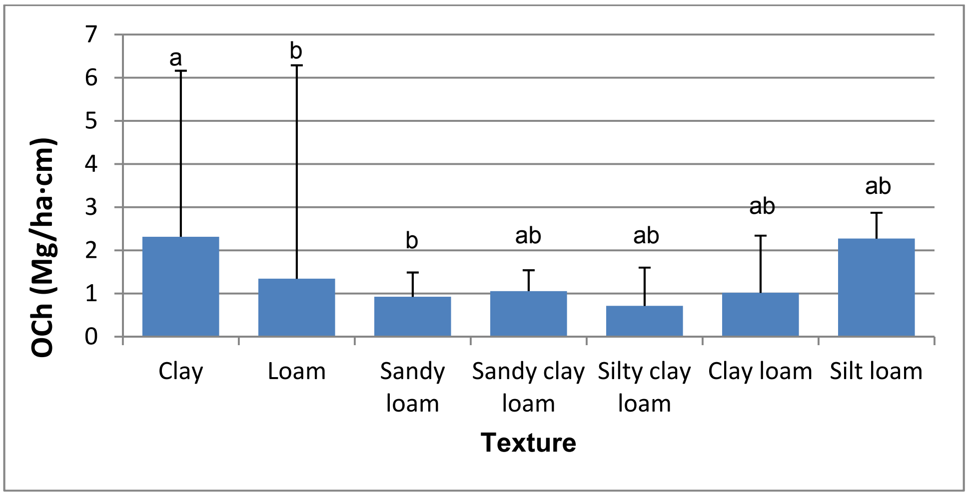

Figure 6. Mean of organic carbon at horizon (Mg/ha*cm) for the main soil textures. Different letters indicate significant differences at *P<0.05* according to Fisher test.

As shown by previous authors, the amount of SOC in the first meter differed significantly with altitude (P<0.05, Table 2), this was in fact related to annual precipitation and annual temperature. Land use types were reclassified into 3 categories of; grazing, cropland and forest management. SOC stock was found to differ significantly between depth, land-use type, and altitude (Table 2). Table 2 shows that the largest SOC contents are associated with forest land use, especially at altitudes 1400-1550 m and increase with increasing clay content. SOC concentrations decreased with soil depth in all land-use types and were consistently higher under forest.

Table 2. Soil organic carbon (SOC) (Mg/ha) for different classes of land-use, soil depth, and altitude.

SOC (Mg/ha)	Land uses			Depths			Altitudes		
	Grazing	Forest	Cropland	0 - 15	0 - 30	0 - 50	1250-1400	1400-1550	> 1550
Median	57.7	52.8	33.6	25.7	45.3	47.7	39.1	51.6	27.3
Mean	62.9 a	77.3 a	36.4 b	35.3 c	58.2 b	78.5 a	48.9 b	72.1 a	36.5 c
S.D	35.4	62.5	17.9	34.4	39.3	47.7	30.7	54.2	24.4
n	27	24	30	27	27	27	33	36	12

Means followed by the same letter are not significantly different (*P* < 0.05) within each source of variation (Fisher test)

The potential for SOC sequestration varied according to the local climate, soil and management conditions where the experiment was set up. At lower altitudes and warmer latitudes, the production of SOC can be limited by water stress, but microbial processes are faster, and as a consequence SOC tends to decrease, as was the case here at 1250 – 1400 m in our study area.

A number of studies indicate an increase of SOC stock with altitude (Ganuza and Almendros, 2003; Miller *et al.*, 2004; Leifeld *et al.*, 2005). At altitudes from 1400 to 1550 m, with lower temperatures at higher altitudes and limited C turnover, the result reflected an increased C accumulation even under conditions of smaller productivity and C inputs (Leifeld *et al.*, 2005). However, Bardgett (2005) and Garcia-Pausas (2007) indicate that this trend may reverse at certain altitudes, when land use change to grassland and that SOC may reach almost zero at the unvegetated substrates of the upper alpine areas. Our results indicate that at high-altitudes where land is used for grazing, the SOC stock decreased.

Grazing and forest land uses had similar means and differed from croplands, but with higher variation between sites (Table 2, Table 3). Table 3 shows increments of the means and standard deviation (SD) in the different land uses studied. In forest sites, the SOC at 0-15 cm depth, has a lower value (30.22 Mg/ha) compared to the grazing sites (31.46 Mg/ha) at the same depth. The inverse occurs when SOC is observed at depth. This variation in SOC with depth can be associated with the quality of the carbon inputs, which often characterized by lignin content, it is an important control of decomposition rates (Austin *et al.,* 1998; Jobbágy and Jackson*,* 2000; Lorenz *et al.,* 2005; Doblas-Miranda *et al.*, 2013).

Table 3. Measured SOC (Mg/ha) in relation to soil depth for three land uses.

	Soil depth							
Land use	n	0-15 cm	n	0-30 cm	n	0-50 cm	n	0-100 cm
Cropland	9	21.04 ± 6.66	9	37.55 ± 12.01	9	49.47 ± 18.59	9	62.57 ± 24.68
Forest	6	30.22 ± 9.30	6	54.79 ± 17.49	6	77.97 ± 21.16	6	116.33 ± 22.75
Grazing	7	31.46 ± 8.31	7	51.14 ± 13.31	7	66.14 ± 15.51	7	88.53 ± 21.95

Data are means ± SD

Our data showed great variability in the SOC stock, which can be partially accounted for by soil depth, climate and topographic variables. In our study, soil depth appears as the most important variable to explain the SOC stocks. We hypothesized that vegetation, through patterns of allocation, would be the major determinant of the relative vertical distribution of SOC. We found significant changes in SOC profiles among vegetation types (Figure 7).

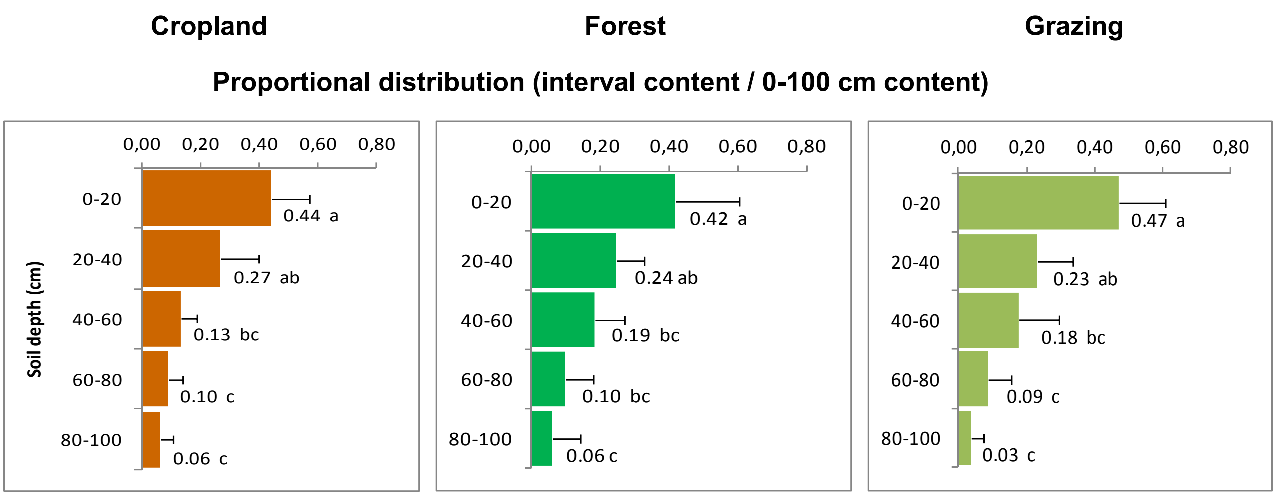

Figure 7. Profiles of soil organic carbon distribution associated with dominant plant functional types (mean+SD). Croplands are sites dominated by potatoes and cereals; forest includes subalpine and Sub-Mediterranean vegetation; Grazing are scrub and pasture. Bars indicate the first meter in 20 cm intervals of SOC proportions. These values were obtained by averaging the actual proportional values of individual soils. Bars in the first meter sum to one. The number and their right show their contribution relative to the first meter. (Fisher test, P < 0.05).

Land use type significantly altered the vertical distribution of SOC. Several researches have proved that deeper layers in the soil profile are able to store a substantial amount of organic C (Batjes, 1996; Jobbágy and Jackson, 2000; Muñoz-Rojas *et al.*, 2012). The relative distribution of SOC in the uppermost 20 cm of soil was deepest in forest, intermediate in cropland and shallowest in grazing (i.e 42%, 44%, and 47% of all SOC in the top 1 m was contained in the uppermost 20 cm; Figure 7). On average more than 50% of SOC is stored in the topsoil. The relative SOC content in the middle 60 cm (20-80 cm), ranges from 49% to 53%, depending on the different land use. Therefore, depth plays an important role on the SOC stock.

The standard deviation of SOC content is large, as observed in other global SOC budgets. Within our study, the average coefficient of variation (CV) of SOC content in the first meter is 60%, in close agreement with the average CV in previous budgets that grouped profiles based on soil taxonomic orders (79%, calculated from Batjes, 1996) or bioclimatic zones (65%, calculated from Post *et al.*, 1982). This variability implies that other factors, perhaps local ones, such as depth or stone content, are important, and that the grouping of SOC data into large, aggregated units may mask meaningful variation.

In most soils, SOC is higher in the surface horizons and it decreases with depth. The quality of carbon inputs, often characterized by lignin content, is another important control of decomposition rates (Menteemeyer, 1978; Melillo *et al.*, 1982; Austin and Vitousek, 1998), and may contribute to observed differences between land use categories. Woody above ground inputs and relatively low decomposability in forests soils could increase SOC storage in surface horizons compared to grazing or cropland systems. With increasing soil depth, land use factors became more important to SOC storage. Six *et al.* (2002) reported that site factors such as soil type, lithology and geology become more important with depth, it can be explained by the fact that the ability of the soil inorganic matrix to retain SOC is linked with mineralogy and texture. Hobley *et al.* (2015) reaffirmed the fact that SOC decreases with depth, and reported that near the surface, SOC content is highest and so fine minerals are most likely to be saturated with SOC, limiting their retention capacity, however, below the surface SOC content generally decreases.

Land use changes, however, may affect the SOC storage in deeper soil horizons. Guo and Gifford (2002) observed that conversions of forest land to pasture or crop land had no effect on SOC stocks below 1 and 0.6-m depth, respectively. In contrast, the change of crop land to pasture caused substantial C accumulation below 1-m depth. Conversely, Halliday *et al.* (2003) reported a range of effects on soil C and increase of it as a result of afforestation of pasture by pinus. Afforestation is specifically used to provide protective cover in vulnerable, steep and mountainous areas. The establishment of a forest cover under good management is an effective means of increasing organic matter production. However, the land must have the productive capacity to support an appropriate forest type, which differs according to climate, soil, slope and the specific purpose of the forest. Regeneration of natural grasslands and forest areas increases biomass production and improves the plant species diversity, resulting in more diverse soil biota and other associated beneficial organisms. Natural regeneration may be more reliable where land is not very productive.

Another important property that directly affects the SOC stock is the different gradient in stone/rock fragment content and profile depth across land use types and altitude (Leifeld *et al.*, 2005). This study was carried out in a mountainous area where most of the soils have more than 40% stone content with some of the shallowest soils could have at least 65%. Thus, C storage in this area is limited by soil depth and stone content, which is further effected by the impacts of management and climate.

3.2 *Soil Organic Carbon mapping approaches*

Figure 8 shows the map of SOC at 0-15 cm, at 0 - 30 cm, at 0- 50 cm and finally SOC stock for each of the soil units represented in the conventional soil map developed. To be able to know the depth of each map unit, we provide a thematic soil depth map (Figure 9).These maps provide an estimation of the SOC stock that is spatially varied in relation to altitude, topographical position but also by depth (Figure 8). However, in most soils, high SOC values are associated with low bulk density values and vice versa; these effects are confounded by differences in land use and management, parent material and soil textural class (Bernoux *et al.*, 1998; Sombroek *et al.*, 2000; Batjes, 2008).

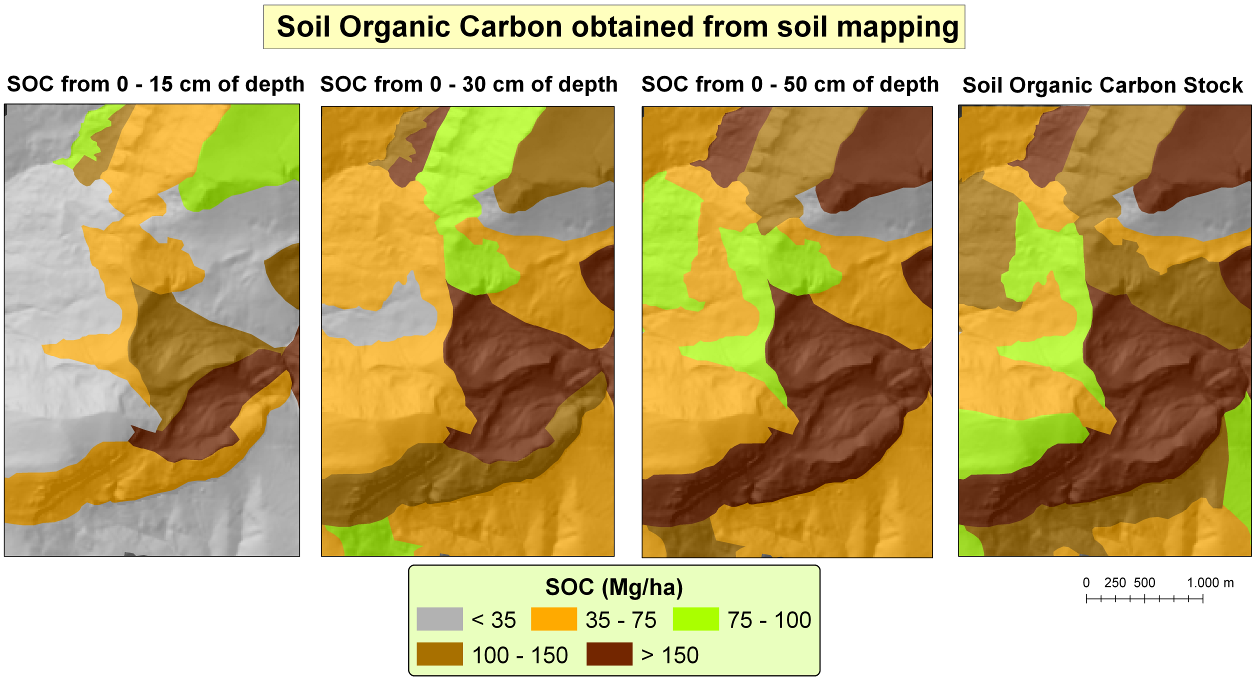

Figure 8. Spatial distribution of SOC at different depths 0-15cm, 0-30 cm, 0-50 cm and SOC total stock.

The high levels of SOC found in shallow soils (Figure 8 and Figure 9), were not expected, but the reason for this is because the majority of shallow soils were found under forest land use which is associated with a higher SOC stock.

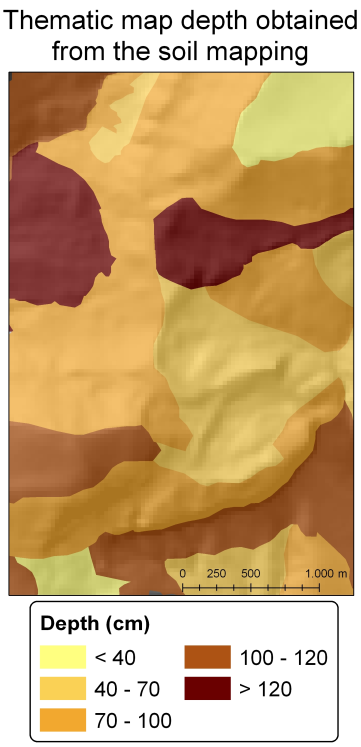

Figure 9. Thematic map showing depth of soils sampled within the mapping unit and the modal profile.

SOC maps and the depth map were developed from generalisation methods which included map unit components from modal profiles using the detailed soil map. Some issues exist in relation to mapping soil properties using this procedure, because conventional soil maps neither delineate all of a field's inherent variability nor represent specific soil attribute variation (Moore *et al.*, 1993). This is to be expected due to the heterogeneity of the soil itself; however, conventional soil maps are general-purpose maps: they do provide information on the three-dimensional spatial distribution of a wide range of soil properties that are inferred from representative soil profile descriptions associated with the map units (Kempen *et al.,* 2012) and are therefore considered fit for purpose from a generalised perspective.

Figure 10 shows the results of SOC distribution in the study area using the ordinary kriging method. The semivariograms of the universal kriging method for each SOC stock map are shown in Figure 11. These results show how the methods predict SOC stocks at different depths using profiles and generating circular patterning radiating from the data point in relation to distance from the point. The semivariogram for each SOC stock map (Figure 11) shows that all adjustments are significant using a spherical model in ordinary kriging.

These maps differ from the SOC maps generated using the detailed soil map, however both have similarities, which are that the highest values of the SOC stocks are located at the same area, since the source modal profiles are the same and in the same location. Modal profiles were used as a main driver in the SOC stock maps using a detailed soil map. However, once the total SOC stock was calculated for the whole 10 km^2 area at different depth, the results show (Table 4) that total SOC stocks (0-15 cm) with ordinary kriging underestimates total SOC stocks for an area of 10 km^2 compared with the other methods, contrary, universal kriging does it for SOC stocks (0-30 cm) or SOC stocks (0-50 cm).

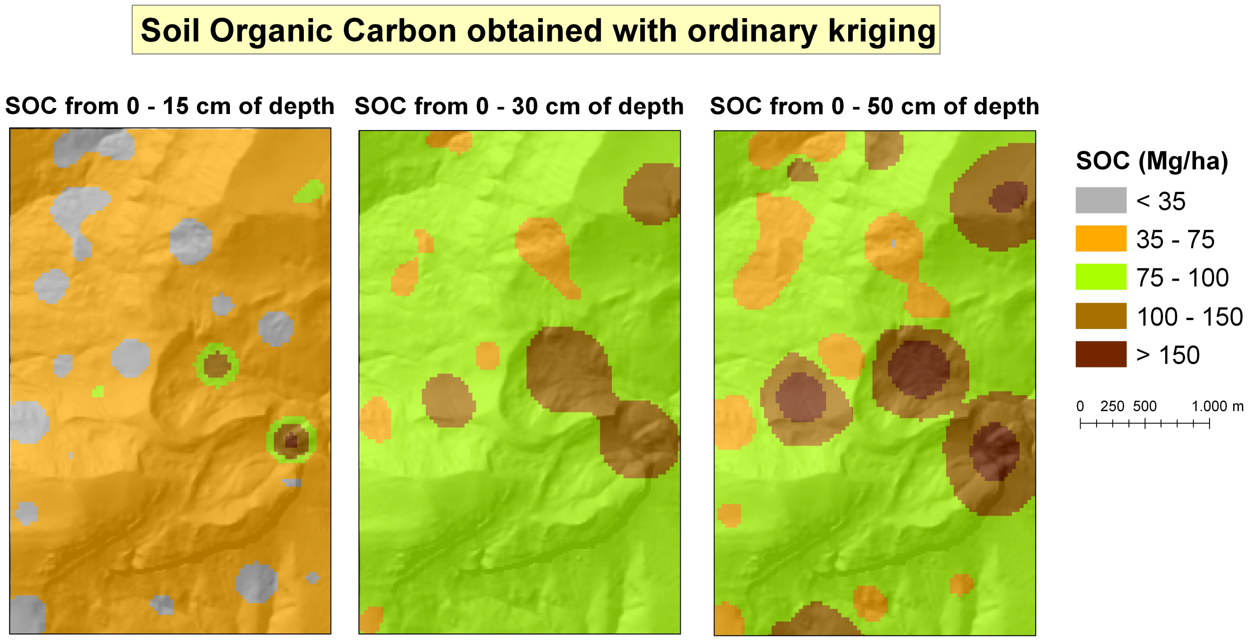

Figure 10. Soil organic carbon stock maps at different depth obtained using ordinary kriging.

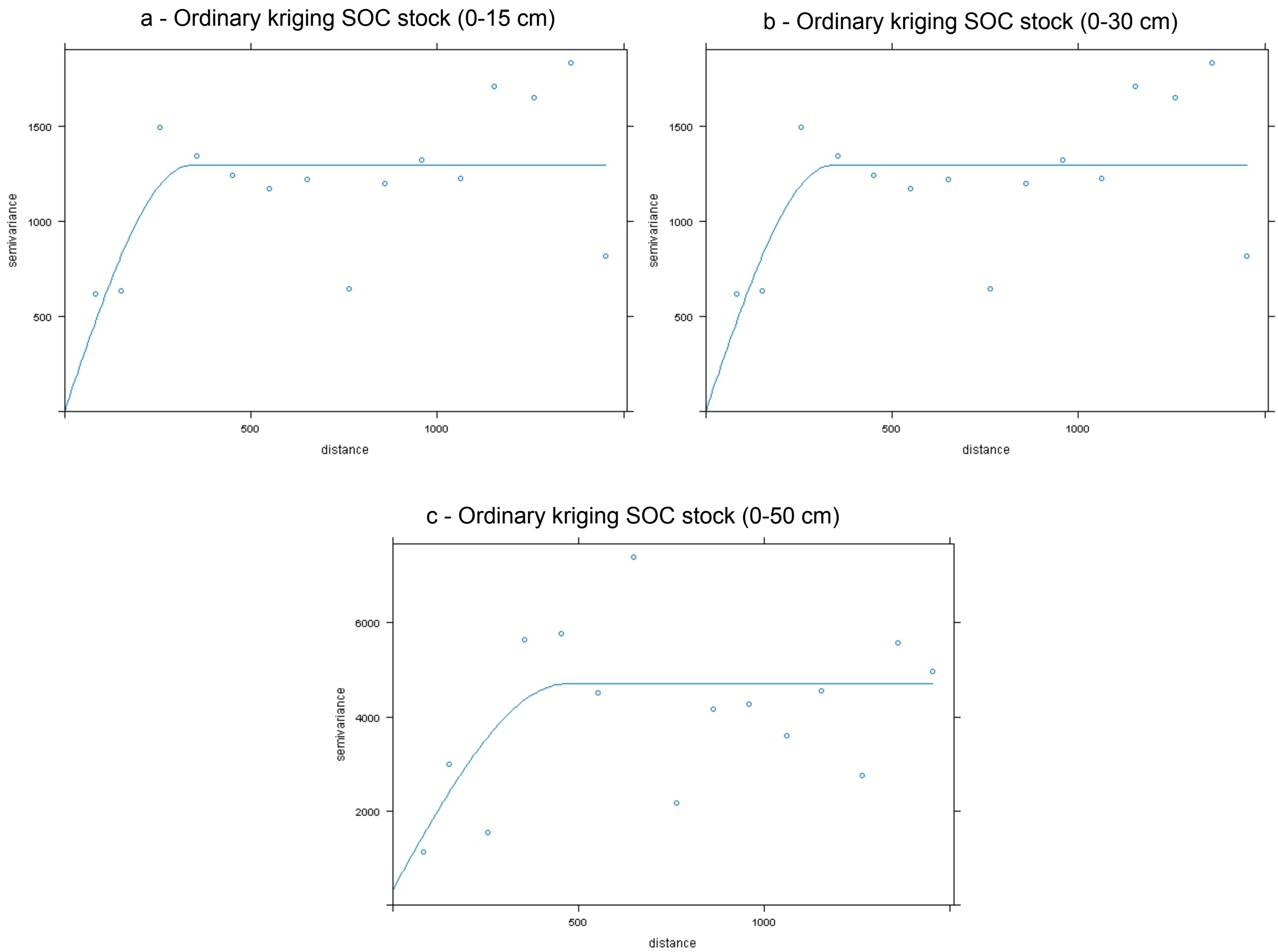

Figure 11. Semivariograms for each soil organic carbon stock map obtained with OK. a) Semivariogram for the ordinary kriging SOC stock (0 - 15 cm); b) Semivariogram for the ordinary kriging SOC stock (0 - 30 cm); c) Semivariogram for the ordinary kriging SOC stock (0 - 50 cm).

Taking into account additional terrain variables, universal kriging maps (Figure 12) have more similarities or follow similar patterns on SOC stocks distribution with the SOC map obtained from the detailed soil map (Figure 8) than the ordinary kriging maps (Figure 10). One of the co-variants used is the soils-landscape map, which has a strong influence in the universal kriging model, depicts a map product similar to the detailed soil map. The semivariogram for each SOC stock map (Figure 13) shows that all adjustments are significant using a spherical model in universal kriging. However, semivariograms (Figure 11 and Figure 13) show a lot of noise, which impedes a well-fitting theoretical semivariogram and noise is an indication of insufficient data points. As it can be noticed from the plots (Figure 13), the nugget variation at the support point is significant with Figure 11c, Figure 13a and Figure 13b all showing a clear nugget effect. It is caused by the absence of data values across the short distances where the samples were taken. In Figure 13b, this effect is even more evident, wherein the co-variants used in the model are more important in the model than the SOC data itself.

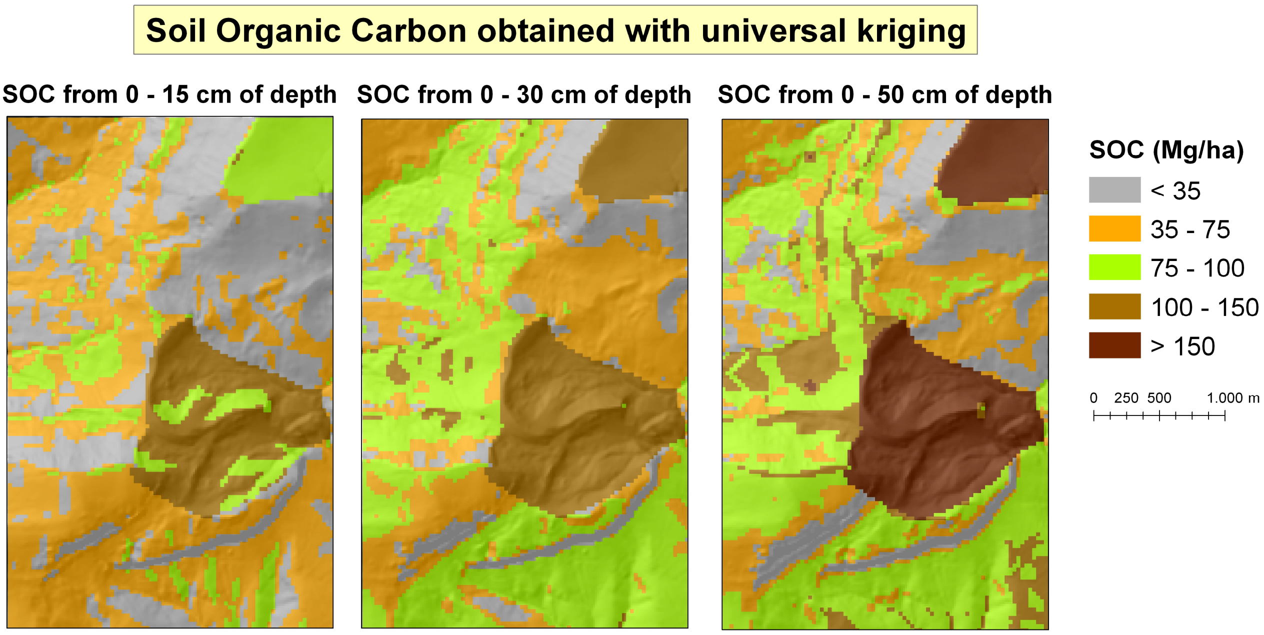

Figure 12. Soil organic carbon stock maps at different depth obtained using universal kriging.

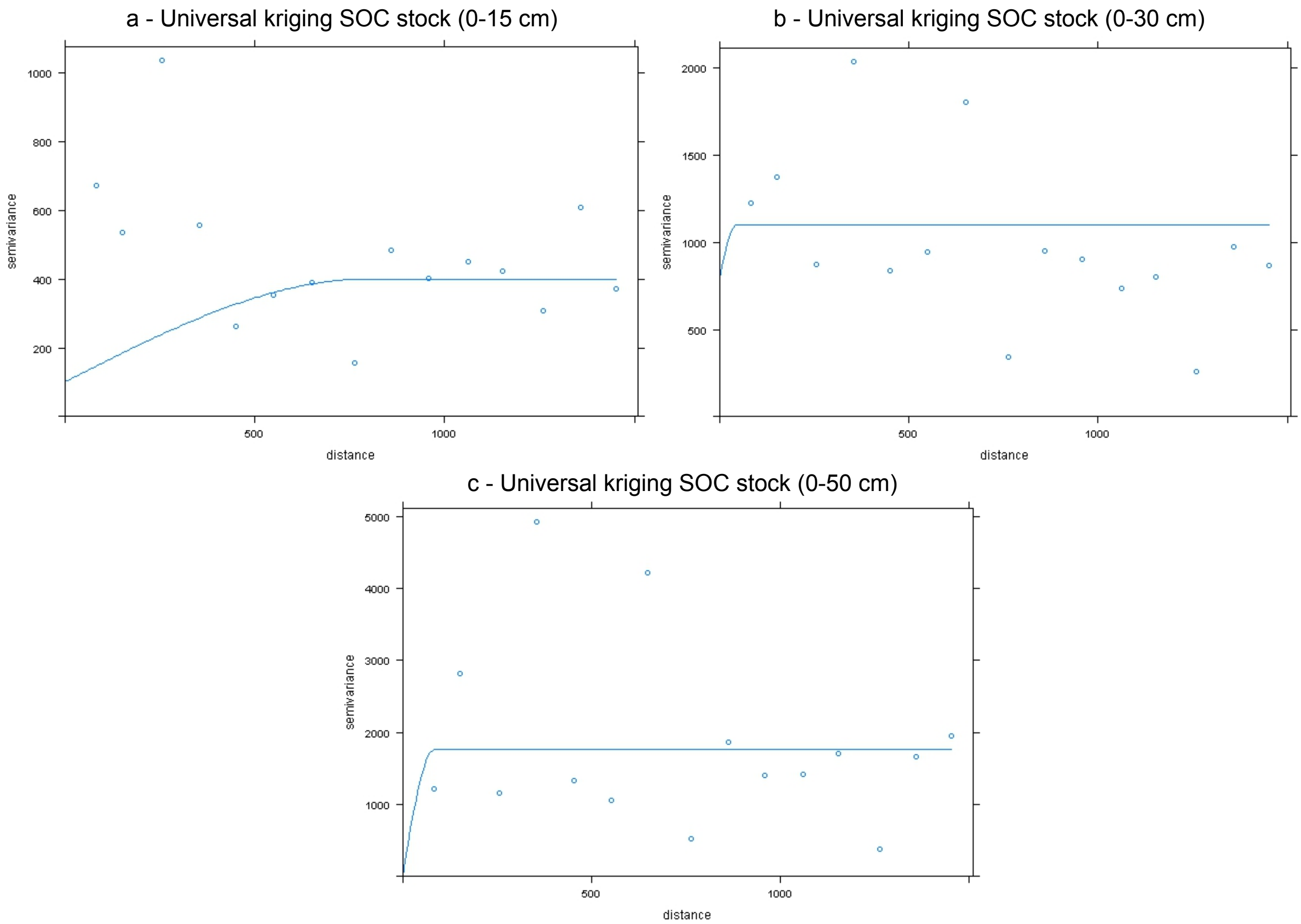

Figure 13. Semivariograms for each soil organic carbon stock map obtained with UK. a) Semivariogram for the Universal Kriging SOC stock (0 - 15 cm); b) Semivariogram for the Universal Kriging SOC stock (0 - 30 cm); c) Semivariogram for the Universal Kriging SOC stock (0 - 50 cm).

Table 4 shows the estimated total SOC stock at 0-15 cm, 0-30 cm and 0-50 cm for the 10 km^2 of the study area with the three methods. The results using UK or a detailed soil map are comparable. However, taking into account detailed soil map method as the reference method, the UK method underestimates by 8% and 17% the total SOC stocks at 0-30 cm and 0-50 cm, respectively, for the 10 km^2 area. Despite the fact that the use of a detailed soil map for SOC mapping could overestimate the real SOC, this method gives similar results to the kriging ones for mapping SOC in the area. Validation SOC maps (De Vos *et al.*,2005; Kempen, *et al.*, 2012; Orton, *et al.*, 2016) are required to validate this hypothesis and to validate OK and UK for accuracy.

Table 4. Total SOC (Gg) for the study area (10 km^2) at different depths.

	Total SOC (Gg) at different depths		
Mapping method	**0-15 cm**	**0-30 cm**	**0-50 cm**
OK	44,50	73,67	96,16
UK	45,37	72,31	83,61
Detailed soil map	45,74	78,65	100,94

3.3 *Validation of SOC maps*

The accuracy of the generated SOC maps using different approaches was estimated using an independent validation data set for one of the variables (OC content). A MPE of 0.01-17.8 were calculated that resulted in an explained variance of 4.5-45.1% depending upon the SOC approach and the depth. Table 5 summarises the validation indexes calculated. Large differences of the prediction accuracy between the depths considered were observed. For the 0-15 cm depth, 0-30 cm depth and 0-50 cm depth, the percentage of explained variance ranged between 4.5 and 49.2 %. The wide variation depends upon the approach applied. The approach that shows better accuracy is the use of the detailed soil map, whilst the universal kriging approach shows high variance at all depths. The comparison of measured versus modelled SOC stocks of the validation data set revealed a wide accordance for any approach (Figure 14). The importance of the predictor variables for SOC storage is estimated by the decrease of prediction accuracy indicated by an increase of MPE.

Table 5. Summary of the validation indexes calculated for the different SOC maps.

Soil Carbon Map	n	MPE	SDPE	RMSPE	Prediction Coefficient of determination R_p^2
OK at 0 - 15 cm	110	3.9	27.8	27.9	14.0
UK at 0 - 15 cm	110	0.01	36.8	36.6	4.5
Soil Map at 0 - 15 cm	110	9.6	38.7	39.7	**25.0**
OK at 0 - 30 cm	108	4.1	36.8	36.9	21.9
UK at 0 - 30 cm	108	0.6	52.0	51.7	7.3
Soil Map at 0 - 30 cm	108	14.4	54.4	56.0	**35.6**
OK at 0 - 50 cm	27	16.9	38.0	37.3	49.2
UK at 0 - 50 cm	27	17.0	41.6	40.8	41.8
Soil Map at 0 - 50 cm	27	17.8	39.8	42.9	**45.1**

The method accuracy varies significantly depending on the approach used. The best coefficients of determination (R_p^2) have been obtained with the use of the detailed soil map in function of 50 cm depth. Overall, the accuracy of the prediction of SOC for the best approach in our study area is the use of the detailed soil map, ranging 25 to 45.1%, depending upon the depth considered. Notably, a high proportion of the variance of SOC stocks cannot be explained by the model. However, studies in the Netherlands which also revealed relatively low explained variances of 21 to 43% for predictions of SOC contents and stocks in different study areas (Schulp *et al.*, 2013). Those authors conclude that only low levels of explanation can be expected for SOC predictions even for models that include a more factors which presumably control SOC storage due to an inherent high spatial variability of SOC.

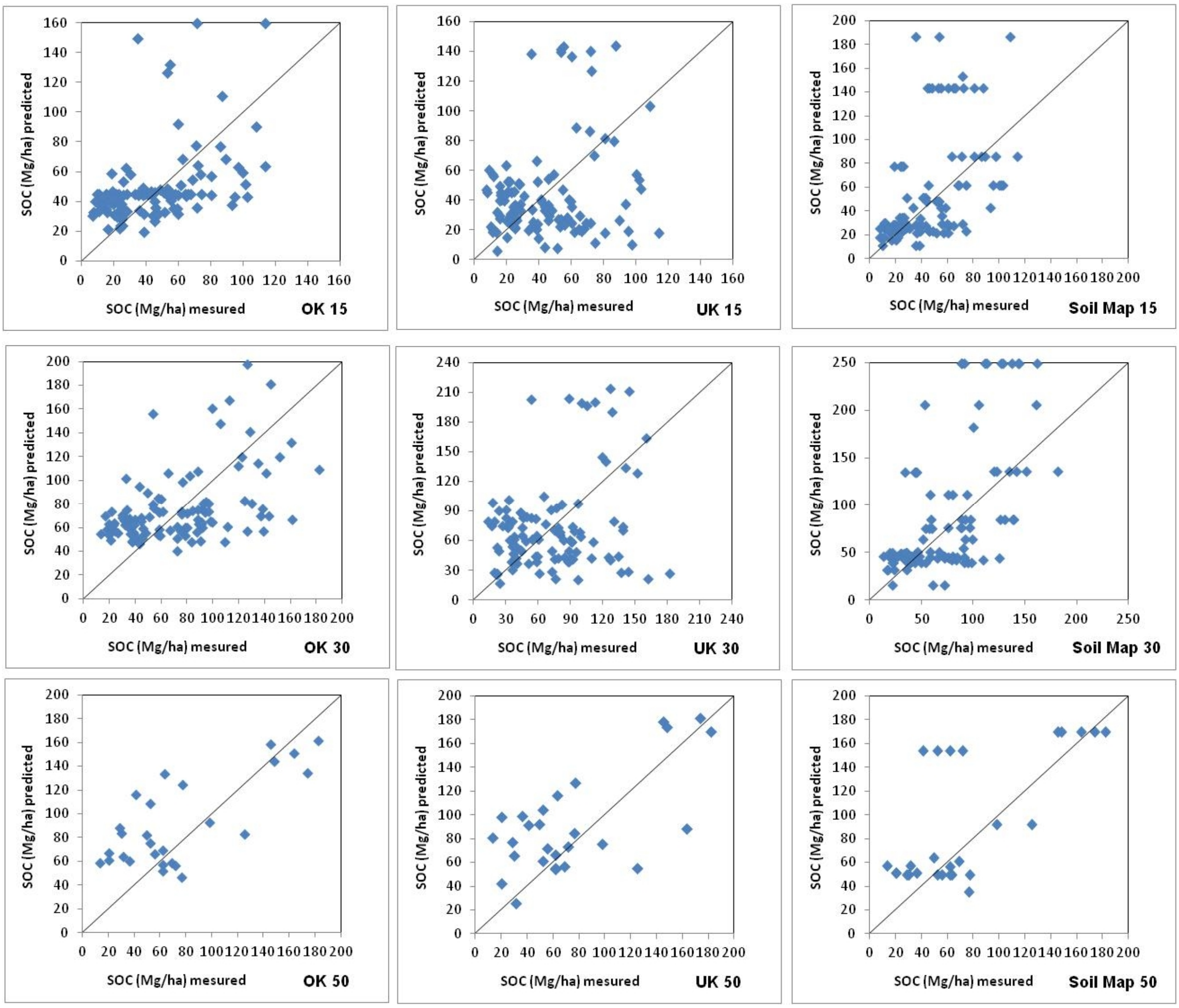

Figure 14. Prediction accuracy of the SOC regionalization by comparison of modeled versus measured SOC stocks of the validation data set (OK= Ordinary Kriging, UK= Universal Kriging, 15= 0-15 cm depth, 30= 0-30 cm depth, 50= 0-50 cm depth)

Most of the studies that aimed to predict SOC contents and stocks obtained comparably moderate prediction accuracies. Meersmans *et al.* (2008) modelled SOC stocks of Belgium using a multiple regression approach that incorporated land use, soil type, texture and soil wetness with an explained variance of 36%, however Goidts *et al.* (2009) reported predicted topsoil SOC stocks of agricultural soils in Belgium with a range of 12–29% of SOC variability. Relatively low proportions of explained variances were even found in studies that modelled SOC stocks separately for the top- and the sub- soil. For Australia, Henderson *et al.* (2005) explained 41% of topsoil SOC variability and 24% for the subsoil using a decision tree-based approach. In the tropics, SOC predictions seem to be even more challenging as an explained variance of only 6% for topsoil and 8–25% for subsoil SOC concentrations was determined using a Random Forests (RF) approach for an island in Panama (Grimm *et al.*, 2008).

There are several possibilities that could potentially improve the prediction of SOC stocks. A further improvement could be achieved by increasing the number of sampling points, the application of a random forest model and the use of higher resolution of the predictor parameters, all of which is likely to result in a more accurate modelling of SOC.

4 CONCLUSIONS

The results of the study showed that land use, altitude and depth account for some of the variations in SOC at different depths and the SOC stock. Soils under cropland use (62.57 Mg/ha) had less SOC than grazing (88.53 Mg/ha) or forest (116.33 Mg/ha) soils, thus land use is a strong factor affecting SOC distribution.

Land use type significantly altered the vertical distribution of SOC. On average more than 50% of SOC is stored in the topsoil. The relative SOC content in the middle 60 cm (20-80 cm), ranges from 49% to 53%, depending on the different land use. Therefore, depth plays an important role on the SOC stock. Using a sampling strategy only at the soil surface would seriously affect SOC estimation of nearly all soils in the region, as most soil profiles are deeper than 15-20 cm. Sampling deeper in the soil profile in subsurface is time consuming, but it will provide much better estimates of ecosystem C budgets and fluxes.

The ability to compare spatially-varying SOC stocks is currently important. The use of the soil map for mapping SOC can satisfactorily do it, when it is compared with other digital mapping methods. Moreover, it illustrates SOC differences between soil mapping units in the study area at the detailed scale used. However, a lot of countries do have general soil maps that have used them in an undisciplined way, leading to wrong decisions being made. The format and the resolution of conventional soil maps is not compatible to the provision of the data needed to calculate SOC. For this reason, only detailed soil maps should be used for this purpose. We therefore considered other techniques such as universal kriging relevant for obtaining SOC stock maps, when environmental variables are available in the study area, others than a detailed soil map.

The method accuracy varies significantly depending on the approach used. Overall, the accuracy of the prediction of SOC for the best approach in our study area is the use of the detailed soil map, ranging 25 to 45.1%, depending upon the depth considered. Notably, a high proportion of the variance of SOC stocks cannot be explained by the model.

Finally, a good management of the cropland such as soil-friendly practices should maintain SOC, however, a conversion of crop land to pasture –which happened in the past in the area- could cause a substantial C accumulation below 1-m depth. Moreover, afforestation of pasture by pines could increase SOC and would provide protective cover in vulnerable, steep and mountainous areas.

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Chapter 4

LAND EVALUATION OF A STUDY AREA APPLYING GIS-BASED APPROACHES USING MEDALUS MODEL, LAND CAPABILITY AND RUSLE MODEL: A CASE STUDY IN THE CATALAN PRE-PYRENEES

LAND EVALUATION OF A STUDY AREA APPLYING GIS-BASED APPROACHES USING MEDALUS MODEL, LAND CAPABILITY AND RUSLE MODEL: A CASE STUDY IN THE CATALAN PRE-PYRENEES

1 INTRODUCTION AND OBJECTIVES

Europe's natural landscape is changing rapidly as economies expand and cities grow. The soil is part of this changing environment and in many places soil quality is declining significantly affecting its physical, chemical and biological properties. Soil quality decline is also associated with both inorganic and organic chemical contamination. In addition, agricultural practices and management systems have generally been adopted without considering soil conservation or recognizing their consequences on environmental quality. However, more recently many people have expressed concern about the way land use and pollution are reducing the resilience of the soil in Europe and its ability to withstand all of the threats that it is facing. There is no doubt that soil degradation (compaction, erosion, loss of biodiversity and organic matter) has resulted in soils becoming both less fertile and less able to regulate water and cycle nutrients (European Commission (EC), 2002, 2006a, 2006b, 2012).

Karlen *et al.* (1997) defined soil quality as the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation, hence, soil quality assessment is considered an effective method for evaluating the environmental sustainability of land use and management activities.

Land evaluation was defined as 'the process of assessment of land performance when used for specified purposes, involving the execution and interpretation of surveys and studies of land use, vegetation, landforms, soils, climate and other aspects of land in order to identify and make a comparison of promising kinds of land use in terms applicable to the objectives of the evaluation' (FAO, 1976). De la Rosa (2005) highlights land suitability or production and land vulnerability or degradation approaches as central to land evaluation analysis. A key point is that for any kind of land use planning, the decision maker needs reliable information about the characteristics of different land areas and their behavior under various land uses; this is the function of land evaluation.

Soil degradation is one facet of environmental degradation. Much research of the natural and social sciences is devoted to the latter, mainly towards the maintenance of biodiversity and to the conservation of the environment.

Net degradation occurs when the degradation processes significantly exceed nature's capacity of restoration to the original state; therefore land degradation is usually referred to as human induced (Blum, 1998). It occurs when one or more land resources (soil, water, vegetation, geological substrate, air, climate, relief) have changed for the worse (Stocking and Murnaghan, 2001). Desertification is the result of degradation processes taking place in arid and semi-arid areas, where water is the main limiting element for ecosystem productivity and other ecosystem services.

Desertification of an area arises when certain environmental thresholds are exceeded and a series of irreversible changes start to occur (Van-Camp *et al*., 2004; EC, 2006a).

Soil erosion has been classified as one of the types of soil degradation, within a wider context of land degradation. Soil erosion is a natural process, occurring over geological time, and indeed it is a process that is essential for soil formation in the first place since it is the source of colluvial and alluvial sediments where soil can form. With respect to soil degradation, most concerns about erosion are related to accelerated erosion, where the natural rate has been significantly increased by human activity (Grimm *et al.,* 2002).

Soil erosion is regarded as one of the major and most widespread forms of land degradation (EEA, 2003). Indeed, about 16% of the total land area in Europe (excluding Russia) is affected by soil erosion to some degree (Oldeman *et al*., 1991, EEA, 2003). In Spain, it is the main cause of desertification, affecting water supply and vegetation as well as soils. This process has been active for centuries, mainly driven by deforestation, which has irreversibly affected soils (Martínez-Fernández and Esteve, 2005).

The Universal Soil Loss Equation (USLE or RUSLE) is an empirical-statistical model used for the estimation of erosion rates for agricultural land evaluation purposes. USLE model predicts the long-term average annual rate of erosion on a field slope based on rainfall pattern, soil type, topography, crop system and management practices. It only predicts the amount of soil loss that results from sheet or rill erosion on a single slope and does not account for additional soil losses that might occur from gully, wind or tillage erosion. This erosion model was created for use in selected cropping and management systems, but is also applicable to non-agricultural conditions such as construction sites. The USLE can be used to compare soil losses from a particular field with a specific crop and management system to "tolerable soil loss" rates (Wischmeier and Smith, 1978). RUSLE (Renard *et al*., 1997) is a reviewed version of USLE.

Another type of land evaluation is based upon land suitability which includes land capability. Land capability classification is a term introduced by Klingebiel & Montgomery (1961) for land evaluation. This approach is used for a ranked system based on the severity of land limitations for general agricultural use and refers in particular to the quality of the land to produce common cultivated crops and pasture without deterioration over a long time.

Conversely, good agricultural practices, oriented to the conservation of the major soil functions, are well documented. Traditional farming practices guaranteed them in many parts of Europe, especially in the Mediterranean region. Different erosion control measures including the integration of organic matter or crop rotation were all traditional farming practices well known and utilised by these southern-European farmers. The European Union, through its Common Agricultural Policy, has recognized the importance of such beneficial agricultural practices. An agri-environmental incentive plan, based on regulation 2078/92 (EC, 1997), plays an important role in preserving these agricultural practices.

The concept of Environmental Sensitivity (ES) arose in industrialised countries about 30 years ago and the increased incidence and severity of soil degradation has stimulated a surge of interest (Rubio, 1995; Basso, *et al.,* 2000). The increased degradation can be associated with uncontrolled forest destruction, water pollution, wind and water erosion, salinisation, and inadequate soil management under both cultivated and uncultivated regimes. An ES can be defined as the response of the environment to a change in one or more external factors and degradation occurs when this response is deleterious the environment.

Environmentally sensitive areas (ESAs) are places that have special environmental attributes worthy of retention or special care. These areas are critical to the maintenance of productive and diverse plant and wildlife populations. The Mediterranean desertification and land use (MEDALUS) approach (Kosmas *et al*., 1999) focuses on recognizing ESAs through multi-factor approaches (Bakra *et al.,* 2012). The ESAs to desertification are classified in different types, using certain key indicators to assess the ability of the soil to resist degradation. These key indicators are stress indicators or indices that can be used at regional or national level. They are classified in four categories defining (1) the soil quality, (2) the quality of the climate, (3) the quality of the vegetation and (4) the quality of management. This approach includes the study of multiple parameters that are associated with information from soils, vegetation and climate. These parameters are texture, coarse elements, drainage, parent material, soil depth for soil quality; rainfall, aridity, and aspects for climate; plant cover, fire risk, erosion protection and resistance to aridity for vegetation quality; and intensity of land use, pastures and forest areas, managerial policies for management practices (Kosmas *et al*., 1999).

Desertification indexes or indicators are diagnostic variables that show when desertification can become irreversible and result in permanently infertile soils, which represents the last stage of the soil desertification. The most useful indicators are those that show the potential risk of desertification while there is still time to react in order to protect the soil and restore it with feasible rehabilitation measures. The MEDALUS model (Kosmas *et al*., 1999) is one of the methodologies applied for the recognition of some potential risk areas. It provides a basis for drawing up guidelines for basin management to enable development strategies to be selected which minimize adverse environmental impacts while still delivering the necessary economic return. This approach is used to determine the Environmental Sensitive Areas (ESAs) and the tendency of desertification in the study area (Kosmas *et al*., 1999). In an integrated view of desertification, MEDALUS requires field surveys of soil erosion and vegetation growth, climate variability studies, computer modeling of river basin responses to change, and socioeconomic issues. Most studies confirm that the MEDALUS model evaluates the desertification rate accurately with acceptable results at scales of 1:250,000 to 1,000,000 (Kosmas *et al*., 1999; Basso *et al*., 2000; Kosmas *et al*., 2003; Sepehr *et al*., 2007, Lavado *et al*., 2009).

The Mediterranean sub-humid and sub-arid areas are especially prone to desertification due to a combination of several environmental factors such as low and irregular rainfall, poor vegetation coverage, hilly relief, high soil erodibility and also, socioeconomic problems such as abandonment of agricultural land; therefore a tool such as the MEDALUS model can be potentially very useful to classify and determine the ESAs in order to prioritize land management and use.

During the last 60 years an abandonment of the traditional crops and rural settlements has taken place in the Catalan Pre-Pyrenees. During the fifties, farmers were attracted by better salaries in the industrial and services sectors, shifting them into urban habitants. This situation worsened in the seventies, resulting in a severe agricultural recession. In the Solsonès region, this has been the prevailing tendency, with agricultural population reducing by 50 % in the last 30 years (DARP, 1996, 1999; Ubalde *et al.,* 1999; Loaiza-Usuga and Poch, 2009).

The Canalda watershed is located in the north of the Solsonès region. It can be found in the Segre river basin, which is located in the north eastern part of the Ebro basin, limited by the Pre-Pyrenees to the North and by the Llobregat basin to the East. The common erosion types existing in the study area are mainly sheet and splash erosion, however neither rill nor gully erosion has been observed (Verdú *et al.,* 2000). The study area is of special interest in terms of hydrology and soil erosion, because the water from the watershed drains to the Rialb reservoir in the Segre river. The water quality is especially important because it is distributed and used along the Ebro basin for irrigation and for general use in the Lleida metropolitan area.

In 2006, the European Union promoted the project "ENVironmental ASsessment of SOil for monitoring (ENVASSO)" based on indicators and criteria to characterise the state of the UE soils. The purpose of this project was to allow users to design common strategies for the use of validation methods, and to obtain data for the European soil protection policy (EC, 2002, 2006a, 2006b, 2012). The Agriculture, Food and Rural Action Department in the Government of the Catalonia (DAR) was a participant of the project, therefore this study was frame in the ENVASSO project as part of the DAR.

The main goal of this work is to identify places with different sensitivity to land suitability or degradation in a 10 km^2 – model mountainous area (Canalda-Odèn in the Catalan Pre-Pyrenees). Three different approaches were carried out in the study area to be able to identify areas of land suitability and land degradation.

In the first instance, MEDALUS approach was applied to identify the Environmental Sensitive Areas; such areas on the basis of an indicator (ESAI, Environmental Sensitive Area Index) in which environmental quality (climate, vegetation, soil) as well as anthropogenic factors (management) are included. This model was developed and proposed by the European Commission in MEDALUS project (Mediterranean Desertification Land Use) (Kosmas *et al*., 1999).

In the second case, to assess land capability (Klingebiel & Montgomery, 1961) were applied.

Finally, in order to add information to the environmental sensitive areas defined, the RUSLE erosion approach (Renard *et al.*, 1997) was applied.

These three approaches give additional information on to the other, with the objective of having better knowledge about the study area, to be able to assess about the management of the land that is taking place, and to establish the optimal method(s) for land use planning in this pre-pyrenean region.

2 MATERIALS AND METHODS

2.1 *Study site*

The study area is located in the Canalda river basin, a tributary to the Ebro Valley (Catalan Pre-Pyrenees, NE Iberian Peninsula) and has an area of 10km^2 (Figure 1).

The study area is mountainous, with altitudes of between 1100 and 2100 m and slopes between 10-70% (Figure 1). The parent materials are calcareous conglomerates, calcilutites and limestones. The most common soils in the area are Inceptisols and Entisols but Mollisols (Soil Survey Staff, 2014) are also present (Orozco *et al*., 2006; Estruch *et al*., 2003). Soil depths vary from <40 to 120 cm but mostly have a range from 70 to 100 cm. All soils are stony.

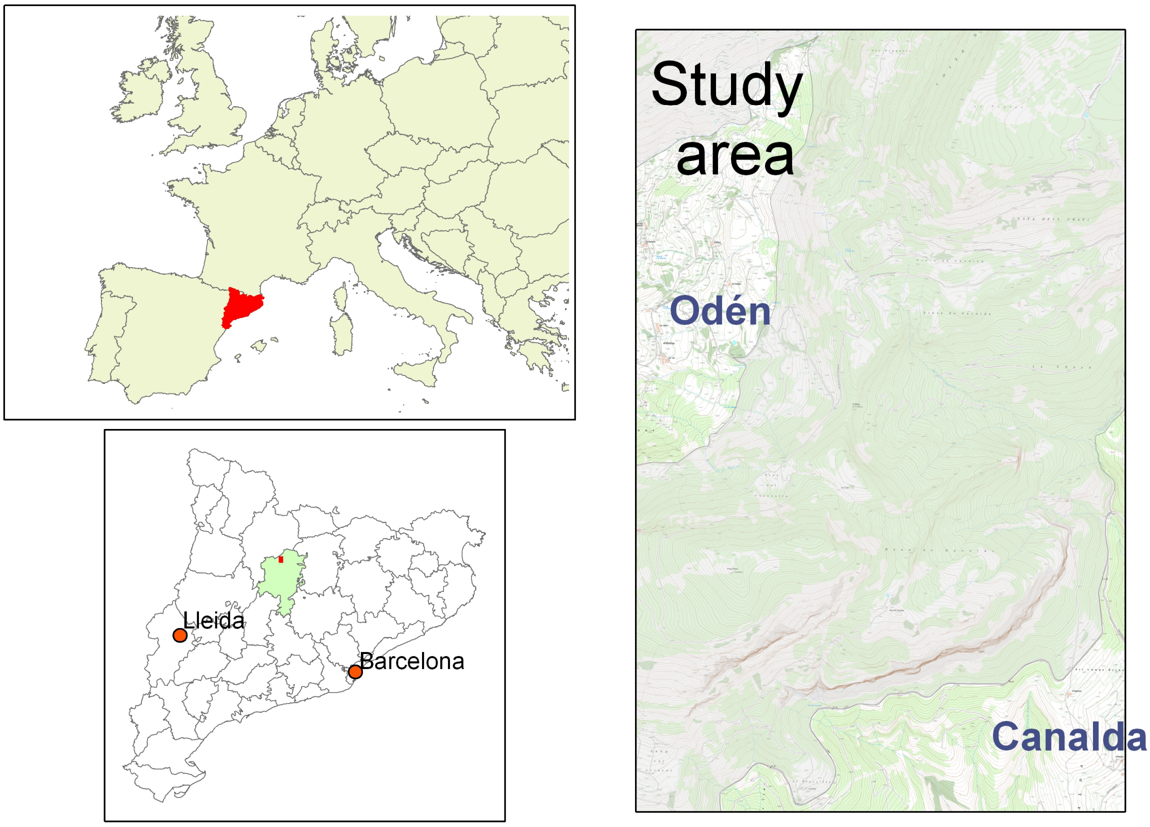

Figure 1. Location of the study area. On the study area, the natural heritage (PEIN) is in grey colour.

The climate is Mediterranean in the lowest parts to Subalpine in the highest parts according to Ubalde (1999). Most of the rain falls in autumn and spring as isolated, often violent, storms. The annual rainfall is 500 to 850 mm, distributed along an altitudinal gradient. The study area is qualified as sub-humid and humid climate in the highest altitudes. The hottest and the coldest months are august (19ºC) and January (0ºC), respectively, with a mean annual temperature of 12ºC (Figure 2). The driest season is in winter and rain periods are in spring and summer. Soil temperature and moisture regimes are mesic-frigid and ustic-udic respectively (Estruch *et al.*, 2003, Orozco *et al.*, 2006, Loaiza-Usuga and Poch, 2009).

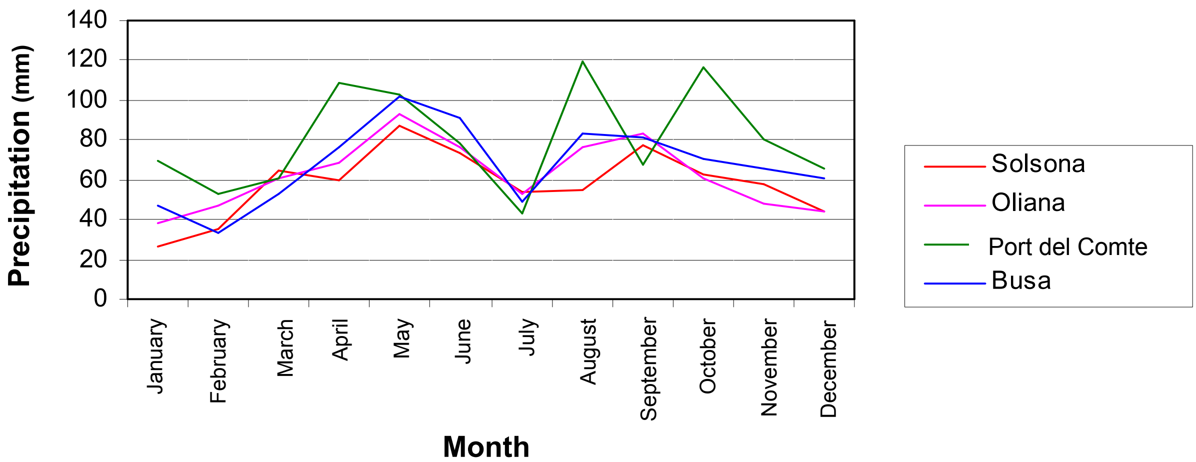

Figure 2. Monthly rainfall (mm) of the closest weather stations in 12 km radius (Solsona (690 m), Oliana (469 m), Port del Comte (1800 m) and Busa (1200 m)) to the study area, Canalda river basin. Source: Estruch, 1999.

The area has been subjected to strong land use changes in the last 100 years, mainly the abandonment of agricultural land and its conversion to pasture or forest (Ubalde *et al.*, 1999). Forestry and agriculture are the main land-uses of the area with 71% and 24% respectively. The study area has 60% of which is considered as natural heritage (PEIN) and it is mainly used for forestry. PEIN areas are delimited and stablished according to required guidelines for the basic protection of the natural spaces which conservation must be guaranteed according to their scientific, ecological, scenic, cultural, social, educational and recreational values.The forestry use varies from diverse forest environments (Pinus nigra, P. sylvestris and P. uncinata) to subalpine and Sub-Mediterranean vegetation (Quercus ilex sp ballota). Rocky areas account for 2% of the study area. These areas are predominantly with a slope of > 35%. The principal agricultural uses include cereals, potatoes and pastures. The potato crop is especially important in the area, because the high altitudes (1100 – 1600 m) and the climate in the area which makes the development of pests difficult, thus the production of excellent quality can be achieved without the application of pesticides. Field management practices are minimal with tillage practices tending to be soil-friendly, mainly contour farming combined with the application of manure.

2.2 *Field work and soil mapping*

The aim of the field work was to collect the necessary data to be able to produce a soil map at detailed scale (1:25,000), with an intensity of 0.04-0.2 observations/ha (Porta *et al.*, 2005; McKenzie *et al*., 2008), obtaining a total of 53 pits described in the area.

Soils are formed through the combined effect of physical, chemical, biological and anthropogenic processes on soil parent material. These factors will affect soil formation in different ways across the landscape. Defined originally by Jenny (1941), these factors are soil, climate, organisms, relief, parent material, age and landscape position (SCORPAN). Today this information can be obtained from existing, large-scale soil maps, climatic data, land use/land cover maps, digital terrain models and their derivatives, parent material/geology, and landscape position.

The soil survey was focus on the goal to be able to obtain enough field data for the production of the soil organic carbon map. Each field observation was described according to CatSIS methodology (Boixadera *et al.*, 1989), and samples were taken for each horizon. All samples were analyzed in the laboratory to determine physico-chemical characteristics, such as pH, calcium carbonate, organic matter by wet oxidation, soil texture (%), and cation exchange capacity (Porta *et al.*, 1986). The soil classification system used was Soil Taxonomy (Soil Survey Staff, 2014), producing a soil map with 26 different soil types. It was correlated with WRB (2014).

All the information related to the soil map and climate, relief, parent material, and landscape factors were recorded in a Soil Geodatabase (SGDB) using ArcInfo ArcGIS software, which was the software used to produce the final soil map.

This Soil Geodatabase (SGDB) plus additional data were essential for the MEDALUS, land capability and RUSLE models application on the study area.

2.3 Reference framework and applied methodology

2.3.1 *MEDALUS model*

2.3.1.1 *Methodology of MEDALUS model*

The MEDALUS methodology is fully described by Kosmas *et al.,* (1999). The Environmental Sensitive Areas Index (ESAI) describes the desertification status of the area on the basis of some 16 aggregated physical and socio-economic parameters (Kosmas *et al*., 1999) (Table 1). The socio-economic data evaluates the interactions of mankind with the environment, but their intangibility makes them difficult to define (Basso *et al*., 2000).

Table 1. Description of the quality indicators and the parameters involved.

Soil quality	Climate quality	Vegetation quality	Management practices quality
Texture Rock fragments Drainage Parent material Soil depth Slope	Rainfall Aridity Aspect	Plant cover Fire risk Erosion protection Resistance to aridity	Intensity of land use in rural areas Pastures and forest areas Managerial policies

One of the particular aspects of the proposed system is that the Environmental Sensitivity classes are not directly linked to an absolute value of sensitivity but are indirectly and relatively related through scores that define different levels of sensitivity, for different parameters, for a particular area (Basso *et al.,* 2000). The value of each parameter is divided into a number of classes, the thresholds of which have been determined empirically from extensive field work during the MEDALUS projects (Kosmas *et al.,* 2003).

Scores (weights), ranging from 1 (best) to 2 (worst), are assigned for each classes of each parameter according to the relevance for the processes of degradation (Table 2). The MEDALUS model is quite flexible providing the option to add or remove parameters in order to the specific environmental conditions or particular aspects of the study area. These parameters need scores assigned to emphasise the characteristics that can be highlighted. Finally, the ESAI is calculated as a composite index that uses the four quality indexes defined within the Medalus model, soil quality index (SQI), climate quality index (CQI), vegetation quality index (VQI), and management quality index (MQI)) which are in turn calculated from the individual parameters.

Once each physical and socio-economic parameter is created, some mathematical algorithms are applied to obtain the four thematic indexes (Figure 3):

$$\text{Index/Indicator}_i = (\text{parameter}_1 \times \text{parameter}_2 \times \text{parameter}_3 \times \ldots \times \text{parameter}_n)^{1/n} \quad (1)$$

Where i (1 to 4) represents the different quality indicator/index, and n represents the number of parameters.

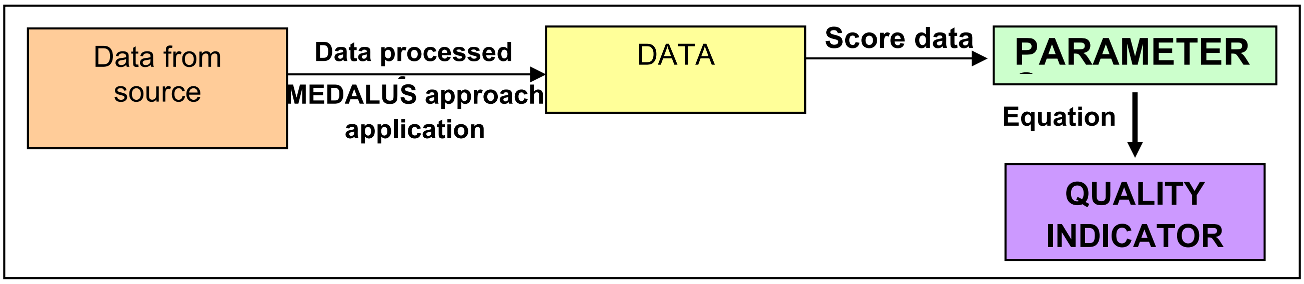

Figure 3. Flowchart showing the process for obtaining the quality index desired. Based on Kosmas *et al.*, 1999.

Figure 4 shows the four indexes combined to calculate ESAI applying the following equation 2.

$$ESAI = (SQI \times CQI \times VQI \times MQI)^{1/4} \qquad (2)$$

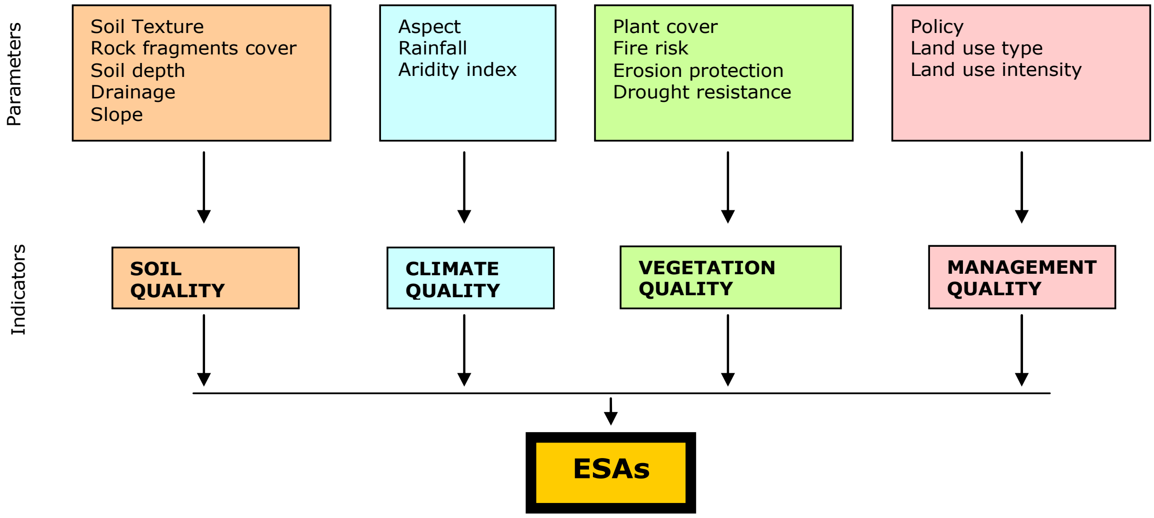

Figure 4. Flowchart showing the process of the development model. Based on Kosmas *et al.*, 1999.

Based on the calculations, MEDALUS model (Kosmas *et al.,* 1999) distinguishes four main classes of the areas threatened by land degradation (ESAI) (Table 2):

- Non-affected areas (ESAI < 1.17)
- Potential (1.23 > ESAI > 1.17): the area is at risk of desertification. Areas under a particular land use or land management change will produce severe problems.
- Fragile areas (1.38 < ESAI < 1.23): Areas where it is easy to appreciate a high unbalance of natural and human activities. Desertification and degradation of the area are taking place.
- Critical areas (ESAI > 1.38): High-intensity use of the area and degraded areas, presenting a threat to the environment of the surrounding areas.

Two maps of environmental sensitivity to land degradation with different legend resolution (4 and 8 classes) were established.

A GIS-based approach was adopted for database development and elaboration of the parameters and indexes/indicators. Initially, the indexes were developed as separate layers (parameters) and they were subsequently assembled in thematic quality layers, each one representing one of the four quality indexes. Depending on the different parameters that produced each index, ArcInfo ArcGIS software was used with the current working set of thematic layers, as it offers good results with gently varying surfaces. Then, each quality index and the final ESAI maps were calculated for the area. These quality indexes characterize the impact of each factor by means of several quality classes (low, medium, high and very high) as illustrated in Table 1 and Table 2.

Table 2. Classification carried out over the range of values of the ESA index (Kosmas *et al.,* 1999) used to build the maps of environmental sensitivity for the Solsonès model area.

Indicator	Quality classes	Description	Range
SQI	1	High quality	< 1.13
	2	Moderate quality	1.13 - 1.45
	3	Low quality	> 1.45
CQI	1	High quality	< 1.15
	2	Moderate quality	1.15 - 1.81
	3	Low quality	> 1. 81
VQI	1	High quality	1 - 1.13
	2	Moderate quality	1.13 - 1.41
	3	Low quality	> 1.41
MQI	1	High quality	1 - 1.25
	2	Moderate quality	1.25 - 1.5
	3	Low quality	> 1.5
ESAI	Critical	C3	> 1.53
		C2	1.41 - 1.53
		C1	1.37 - 1.41
	Fragile	F3	1. 32 - 1.37
		F2	1.26 - 1.32
		F1	1.22 - 1.26
	Potential	P	1.17 - 1.22
	Non - affected	N	< 1.17

2.3.1.2 *Data set selected for MEDALUS model*

A highly accurate search was applied with the aim of obtaining the best input data for the MEDALUS approach but some difficulties were found when we tried to obtain data with the same level of detail, in particular in relation to the socio-economic parameters. Data used comes from several sources (Table 3) and the resolution of this data was available at different scales (Table 1). Some data were developed for the purpose of the parameter, such as slope and aspect, which were obtained from the 15x15 DTM. Soil data were obtained from the detailed soil map of the study area. Climate, land use, land cover were available data online from the Agriculture Department of the Government of Catalonia (DARP).

All data were adapted (Figure 3) in order to develop the adequate parameters, by adopting an approach as illustrated in Figure 4. This approach includes the study of multiple parameters that are associated with information from soils, vegetation, climate and management.

Table 3. Classes, scores of the parameters and data sources for each parameter used for the calculation of soil, vegetation, climate and management quality for the Solsonès model area. Based on Kosmas *et al.*, (1999).

	Parameter	Class	Description	Scores	Data sources	Resolucion/scale	Data type
SQI	Slope (%)	1	< 6	1.0	Slope map (DTM, from the Environmental department. Government of Catalonia).	15 x 15 m	Raster file
		2	6 - 18	1.2			
		3	18 - 35	1.5			
		4	> 35	2.0			
	Parent materials	1	Shale, schist, basic, ultra basic, conglomerates, unconsolidated	1.0	Soil map, field observations.	1:25.000	Vector file Excel, Access PgAdminIII
		2	Limestone, marble, granite, rhyolite, Ignibrite, gneiss, siltstone, sandstone	1.7			
		3	Marl, pyroclastics	2.0			
	Soil texture	1	L, SCL, SL, LS, CL	1.0	Soil map, laboratory analysis.		
		2	SC, SiL SiCL	1.2			
		3	Si, C, SiC	1.6			
		4	S	2.0			
	Soil depth (cm)	1	> 75	1.0	Soil map, field observations.		
		2	30 - 75	2.0			
		3	15 - 30	3.0			
		4	< 15	4.0			
	Drainage	1	Well drained	1.0			
		2	Imperfected drained	1.2			
		3	Poorly drained	2.0			
	Rock fragments cover (%)	1	> 60	1.0			
		2	20 - 60	1.3			
		3	< 20	2.0			
CQI	Rainfall	1	> 650	1.0	Digital climatic atlas of Catalonia. Environmental department. Government of Catalonia.	180 x 180 m	Raster file
		2	280 - 650	1.5			
		3	< 280	2.0			
	Aspect	1	NW - NE	1.0	Aspect map (DTM from the Environmental department. Government of Catalonia).	15 x 15 m	Raster file
		2	SW - SE	2.0			
	Aridity (BGI*)	1	< 50	1.0	Digital climatic atlas of Catalonia. Environmental department. Government of Catalonia.	180 x 180 m	Raster file
		2	50 - 75	1.1			
		3	75 - 100	1.2			
		4	100 - 125	1.4			
		5	125 - 150	1.8			
		6	> 150	2.0			
VQI	Vegetation cover (%)	1	> 40	1.0	Land use of Catalonia in 2002. Environmental department. Government of Catalonia. (Processing of multitemporal data captured by the sensor Thematic Mapper (TM) satellite Landsat in 2002).	30 x 30 m	Raster file
		2	10 - 40	1.8			
		3	< 10	2.0			
	Drought resistance	1	Mixed Mediterranean macchia/evergreen forests, mediterranean macchia	1.0			
		2	Conifers, deciduous, olives	1.2			
		3	Perennial agricultural trees (vines, almonds, ochrand)	1.4			
		4	Perennial grasslands	1.7			
		5	Annual agricultural crops, annual grasslands	2.0			
	Erosion protection	1	Mixed Mediterranean macchia/evergreen forests	1.0			
		2	Mediterranean macchia, pine forests, permanent grasslands, evergreen perennial crops	1.3			
		3	Deciduous forests	1.6			
		4	Deciduous perennial agricultural crops (almonds, orchards)	1.8			
	Fire risk	1	Bare land, perennial agricultural crops, annual agricultural crops (maize, tobacco, sunflower)	1.0			
		2	Annual agricultural crops (cereals, grasslands), deciduous oak, (mixed), mixed Mediterranean, macchia/evergreen forests	1.3			
		3	Mediterranean macchia	1.6			
		4	Pine forests	2.0			

	Parameter	Class	Description	Scores	Data sources	Resolucion/scale	Data type
MQI	Cropland	1	Low land use intensity (LLUI)	1.0	Land cover map of Catalonia. Research, Innovation and Knowledge Transfer in Terrestrial Ecology (CREAF), Catalan Goverment.	1:25.000	Vector file
		2	Medium land use intensity (MLUI)	1.5			
		3	High land use intensity (HLUI)	2.0			
	Pasture	1	ASR< SSR	1.0	Livestock Information system (SIR) of Catalonia. Agriculture department. Government of Catalonia.	1:5.000	Vector file Access data
		2	ASR=SSR to 1.5*SSR	1.5			
		3	ASR>1.5*SSR	2.0			
	Natural areas	1	A/S = 0	1.0	Land cover map and Forest information system of Catalonia (SIBosC). Environmental department of Catalonia goverment.	1:25.000	Raster and Vector file
		2	A/S < 1	1.2			
		3	A/S ≥ 1	2.0			
	Mining areas	1	Adequate	1.0	Mining areas map. Environmental department. Government of Catalonia.	1:5.000	Vector file
		2	Moderate	1.5			
		3	High	2.0			
	Recreation areas	1	A/P < 1	1.0	Field observation and expert knowledge.		
		2	1 ≤ A/P < 2.5	1.5			
		3	A/P ≥ 2.5	2.0			
	Policy	1	Complete: >75% of the area under protection	1.0	Natural heritage map (PEIN) Environmental department. Government of Catalonia.	1:25.000	Vector file
		2	Partial: 25-75% of the area under protection	1.5			
		3	Incomplete: <25% of the area underprotection	2.0			

2.3.2 *Land evaluation models: land capability.*

2.3.2.1 *Methodology for land capability*

Land evaluation is formally defined as 'the assessment of land performance when used for a specified purpose (FAO, 1976). Land evaluation can be a key tool for land use planning, either by individual land users (e.g., farmers), by groups of land users (e.g., cooperatives or villages), or by society as a whole (e.g., as represented by governments). A distinction is made between qualitative evaluation, mainly based on expert judgment, and quantitative evaluation, based on process simulation models (Rossiter, 1994).

The Department of Agriculture of the United States developed a categorical land assessment system. This system categorises according to the actual productivity, without any soil degradation on medium to long term (Klingebiel & Montgomery, 1961), regardless of the applied cultivation and management. This land evaluation method is being currently applied to the soil map of Catalonia 1:25 000 by the Cartographic and Geologic Institute of Catalonia (ICGC).

The application of this evaluation was adapted by the Soil Conservation Service establishing eight land capability classes. This criterion was adapted and modified for the Agriculture Department of the Government of Catalonia (DARP) in 1995 (Table 4). The classes I and II are considered as Prime Farmlands. The first four classes are arable land, in which the limitations on the use and need for conservation measures and careful management increase with class number (Helms 1992). The remaining four classes are not suitable for cropland, but may have uses for pasture, woodland, grazing, wildlife, recreation and other purposes. The capability units are groupings of soils that have common responses to pasture and crop plants under similar systems of farming but require different management (Table 5; FAO, 2007).

Table 4. Criterion for determining land capability class in Catalonia. Source: DARP, (1995) personal communication.

			Arable land				Non arable land			
Parameters / Classes			I	II	III	IV	V	VI	VII	VIII
Rainfall			>600 mm or irrigated	300 – 600 mm or irrigated	300 – 600 mm or irrigated	300 – 600 mm or irrigated	Whatever	Whatever	Whatever	Whatever
Temperature (Papadakis criteria)			Corn to cotton	Warmer than corn	Warmer than corn	Warmer than corn	Whatever	Whatever	Whatever	Whatever
Slope (%)			< 2	< 5	< 10	< 20	< 5	<35	< 50	Whatever
Erosion	Apparent		No erosion or low	Low	Low or moderate	Moderate	No erosion or low	Whatever	Whatever	Whatever
	Risk	Sheet	No erosion or low	Low or moderate	Moderate	Moderate – high	No erosion or low	Moderate – high	High	High
		Other	No erosion	No erosion	Low	Moderate	No erosion	Moderate – high	High	High
Soil depth (cm)			> 120	> 80	> 40	> 30	Whatever	Whatever	Whatever	Whatever
Soil drainage			Well	Moderate-well	Imperfectly	Imperfectly	Whatever	Whatever	Whatever	Whatever
Superficial soil texture			L or SL	L or SL	CL, SiCL, C, L, CL, SiL, CL, S	CL, SiCL, C, L, CL, SiL, CL, S	Whatever	Whatever	Whatever	Whatever
Rock Outcrops (%)			Non rocky	< 2	< 10	< 25	Whatever	Whatever	Whatever	Whatever
Surface Stoniness (%)			Non stony	< 0.1	< 0.1	< 3	Whatever	Whatever	Whatever	Whatever
Soil salinity			< 4 dS/m	< 8 dS/m	8 - 16 dS/m	Whatever	8 - 16 dS/m	Whatever	Whatever	Whatever
Soil sodicity			SAR < 8	SAR < 8	SAR < 16	SAR < 16	Whatever	Whatever	Whatever	Whatever

We applied the DARP criterion (Table 4) to the soil profiles described in the study area, and then the resulting agrological class was assigned to the soil mapping unit where it belonged, of the detailed soil map (1:25.000), hence, we obtained the Agrological Capability map for the study area, according to their ability to support general kinds of land use without degradation or significant off-site effects.

Table 5. Interpretation of land capabilty class according to MAPA (1974).

CLASSES	**Land capability**	Comments
I	Suitable for cultivation. Few restrictions on its use.	
II	Suitable for crops with some limitations	Some simple conservation practices are advisable
III	Suitable for cultivation with strong limitations compared with the previous class	Moderate conservation practices should be applied
IV	Suitable for occasional or very limited cultivation	Requires very careful management
V	Suitable only for pasture and forestry	Do not cultivate. Suitable for forests and pastures
VI	Suitable only for pasture and forestry	Do not cultivate. Suitable for forests and pastures
VII	Not suitable for farming or forestry	Do not cultivate. Suitable for forests and pastures
VIII	Not suitable for crops or pasture and forestry	Do not cultivate.

2.3.2.2 *Data set for Land Capability class*

The model is basically based on soil data, thus most of the data needed is obtained from the detailed soil map developed. These data are slope (%), soil depth (cm), soil drainage class, superficial soil texture, rock outcrops (%), surface stoniness (%), soil salinity, soil sodicity. The slope data were compared verified and corrected with the slope map developed from the 15x15

DTM. Climatic data were meaningless in study area because the area has only 10 km2 and the rainfall and temperature change but not enough according to the criterion of this model.

2.3.3 *Application of the USLE/RUSLE model at detailed scale (1:25.000)*

2.3.3.1 *RUSLE methodology*

In 1985 the US Department of Agriculture (USDA) decided that the Universal Soil Loss Equation (USLE) developed by Wischmeier and Smith (1978) should be revised to incorporate additional research, resulting in a modified version called the Revised USLE - RUSLE (Renard *et al.*, 1994).

RUSLE is a straightforward and empirically based model that has the ability to predict long term average annual rate of soil erosion on slopes using data on rainfall pattern, soil type, topography, crop system and management practices. The equation model (equation 3) is a function of five input factors in raster data format: rainfall erosivity, soil erodibility, slope length and steepness, cover management and support practice. Data sources used for the procedure are listed in Table 6.

The five input factors were stored as raster GIS layers in the ArcInfo ArcGIS software.

The procedure used for applying RUSLE model is the following:

- Determine the rainfall and runoff erosivity - R Factor. We used, only for the study area, data obtained by Angulo-Martínez and Beguería (2009).

- Soil erodibility - K value is determined using soil data taking into account several soil surface parameters such as grain size distribution, organic matter content, permeability and structure. K value was determined using the 37 soil profiles described in the study area. Once the K value was defined for all profiles. The K factor map was calculated using ArcInfo ArcGIS software and applying Inverse Weighted Distance.

- The computation of the Slope length and steepness - LS factor, requires the calculation of other factors such as flow accumulation and slope steepness. The procedure for computation was as follows:

    - using the Spatial Analyst Extension: slope was derived from DEM (15 x 15);
    - using the Hydrological Extension: sinks in the DEM (15 x15) were identified and filled;
    - the filled DEM was used as input to determine the Flow Direction;
    - the Flow Direction was used as an input grid to derive the Flow Accumulation.

- The LS factor was then computed using Raster Calculator from the menu to build an adapted expression for estimating LS, based on flow accumulation and slope steepness (Sims *et al.*, 2003):

$$\mathrm{LS} = (\mathrm{Flow\ accumulation} * \mathrm{Cell\ size}/22.13)^{0.4} * (\sin \mathrm{slope}/0.0896) \qquad (4)$$

- The C-factor represents the effect of soil-disturbing activities, plants, crop sequence and productivity level, soil cover and subsurface bio-mass on soil erosion. It is defined as the ratio of soil loss from land cropped under specific conditions to the corresponding loss from clean-tilled, continuous fallow (Wischmeier and Smith, 1978). The C factor is a dimensionless factor that ranges between 0 and 1, which incorporates cropping and management factors. The C factor map was calculated using ArcInfo ArcGIS software. The layer was created along with its corresponding attribute table and adopting values recommended by Wischmeier and Smith, 1978. This C factor GIS layer was then converted to a grid.

- P factor was selected according to the soil conservation practice used. In our case the factor was 1 for the whole area.

The 5 factors together were multiplied (equation 3) to obtain the soil loss per hectare.

$$A = R \times K \times LS \times C \times P \tag{3}$$

This step was done using ArcInfo ArcGIS software.

The annual soil erosion rate map (in Mg $ha^{-1}y^{-1}$) was generated for a mountainous area, which represents most of the terrain characteristics of the Catalan Pre-Pyrenees.

2.3.3.2 *Data set selected for RUSLE model*

RUSLE (Renard *et al*., 1997) computes the average annual sheet and rill erosion expected on hill slopes by multiplying several factors. Each factor is the numerical estimate of a specific condition that affects the severity of soil erosion at a particular location (Wischmeier and Smith, 1978).

These factors were obtained or calculated using data from different sources and from data at different working scale (Table 6):

Table 6. Data source used for the RUSLE model application

RUSLE Factor	Data information	Data source	Scale
Rainfall and runoff erosivity R-factor	Spatial distribution of the RUSLE R factor in the Ebro Basin	Angulo-Martínez and Beguería, 2009	(Ebro Basin) 1:50,000
Soil erodibility K-factor	Soil erodibility K-factor was determined using inherent soil properties	Field work and soil profiles description.	1:25,000
Slope length and steepness LS-factor	Digital elevation model (DEM) soil profiles from the study area and applying GIS tools	ICGC	15 x 15 m
Cover management C-factor	Land cover map of Catalonia (3rd edition)	Research, Innovation and Knowledge Transfer in Terrestrial Ecology (CREAF), Catalan Government	1:25,000

2.4 *Validation of the models*

The selected models do not share a common view of soil quality and it is very therefore challenging to compare their results. Thus, these models were validated against the available ground trusting exercise. Because of that, three different existing data (timber data, actual erosion and evolution after land abandonment) were used for the validation of the map products obtained through the application of the MEDALUS/RUSLE/Land capability models:

- **Timber (wood) yields:** data were obtained through the collaboration of the Agriculture department of the Catalonia government and the Forest Ownership Centre. Together, they have provided the timber yield data (m^3/ha) for the study area (Figure 5a). Despite it is affected by type of land ownership and socio-economic factors, these values should have a direct relation with soil quality for biomass production.

- **Presence of erosion features: actual mass movements.** The "National Inventory of Soil Erosion - INES (2002-2012)" project was carried out to study the process of soil erosion in Spain (INES, 2007). For every province the following erosion types are inventoried: rill erosion, gully erosion, sheet erosion, river bank erosion, mass movements and wind erosion and mapped at a scale of 1:50,000 (Figure 5b).

  For this validation only the mass movement map was used, which represents a form of actual erosion that was obtained through detailed photointerpretation. This map has a qualitative classification of areas of potential risk to actual mass movements. There are only two main classes affecting the study area (out of five) medium and high class of mass movements. The types of mass movements in this area are mainly rock fall and soil creep (INES, 2007). This information is related to soil quality since unstable soils would qualify for low quality. This map has not been compared to the RUSLE map, since they depict different erosion processes.

- **Type of land abandonment.** Ubalde *et al.* (1999) studied the land use changes in the Ribera Salada (El Solsonès), wherein our study area is, according to different variables (aspect and slope). They used sequential photointerpretation in two periods (1957 and 1992) and mapped the land uses at a scale 1:25,000. These maps were used to determine use change intensities and the final land use. This information indicates the type of the land abandonment; whereby a progressive change is a positive land use change that shifts towards more land cover (in terms of soil protection), and a regressive change is a negative land use change towards a decrease of soil protection (Figure 5c). There were not many land use changes in the study area; however, the progressive changes (21.3%) occurred mostly in middle and upper slopes, and mostly to an increase of land cover. Crops and pastures that have been abandoned and were facing north are more likely to be currently forests. Contrary, in the flat areas remain largely unchanged, remaining uses of crops and meadows and mountain pastures of the northern part of the basin. Regressive changes (3.1%) increase in intensity with decreasing slope and land cover. The relation between these changes and land quality is indirect, since we do not know the starting point (original land quality in 1957) and also because land abandonment in the Pyrenees responds more to societal evolution than to soil quality (i.e. land was not abandoned due to low quality). Nevertheless, progressive changes should indicate higher quality soils (that are able to produce higher biomass if not subjected to stress) and also lower average erosion values (due to higher soil cover).

These data were represented spatially, and there were then spatial intersected with the MEDALUS/RUSLE/Land capability models maps products.

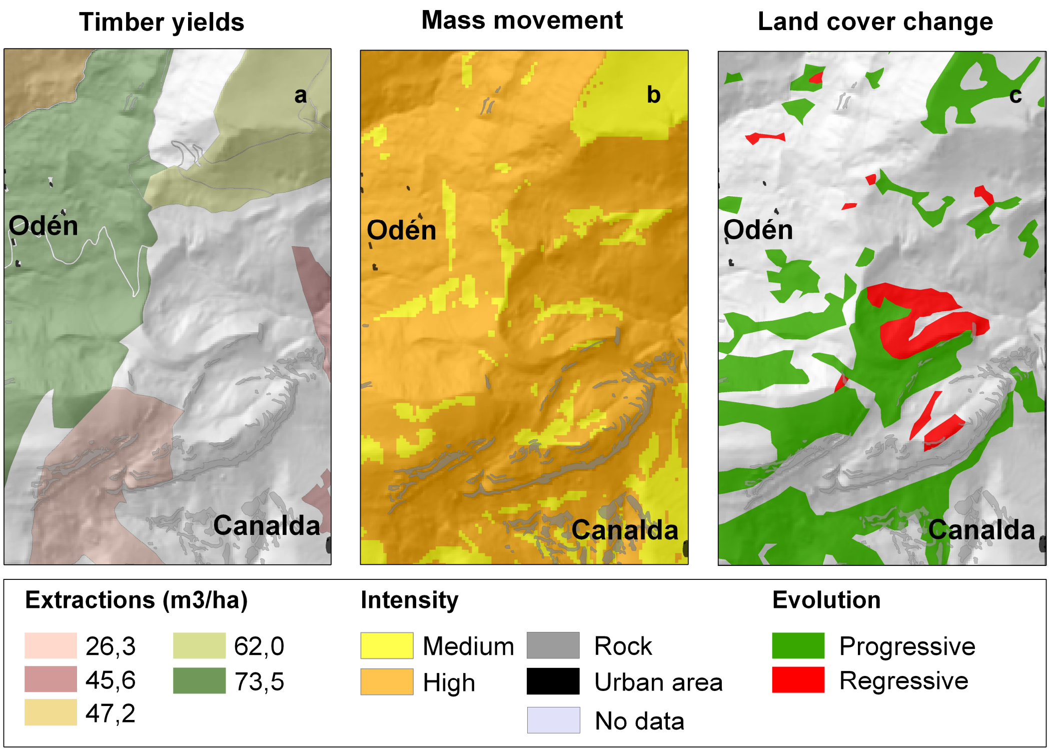

Figure 5. Maps used for the validation of the soil loss (RUSLE) (Mg/ha·year), desertification (MEDALUS) (ESAs class) and Land capability, approaches applied in this thesis.

3 RESULTS

3.1 *MEDALUS model*

The overall results of the elaboration of the five quality indexes/indicators are given at the study area in Figures 6a, 6b, 6c, 6d and Figure 7 for SQI, VQI, CQI, MQI and ESAI, respectively.

3.1.1 *Soil Quality Index*

According to the Soil Quality Index, the majority of soils of the area are classified as having moderate quality (86.3% of the area) with respect to risk of desertification. An 11.5% of the soils have high quality, while the poor quality soils are very restricted (2.09%) (Figure 6a). The moderate and high quality soils are soils mainly with good drainage, balanced texture (loamy, clay loam, silty loam) and enough soil depth. On the opposite side, poor quality is due to steepness, stoniness and landslide risk. The soils of the study area have high content of coarse elements, normally ranging from 20-60%. Most soils are moderately deep (30-75 cm soil depth) and few areas are highly degraded because of shallow soils (< 30 cm depth).

3.1.2 *Climate Quality Index*

As shown in Figure 6b, most of the study area is characterised by moderate climate quality (68.6%) with almost one-third (31.2%) having a high climate quality. The lack of low climate quality is due to the low level of aridity, since the climate is humid to sub-humid that prevents soil moisture deficits for prolonged periods of time, benefitting vegetation development while also reducing the risk of fire.

3.1.3 *Vegetation Quality Index*

According to the Vegetation Quality Index about half (51.7%) of the area is classified as having high quality with the other half (48.3%) showing to be of moderate quality (Figure 6c). This is mainly attributed to the good coverage by natural vegetation adapted to drought conditions. Most of the study area is covered with perennial (pines) trees and some deciduous (oak) trees that protect the soil from rainfall impact therein reducing the risk of erosion.

The vegetation is predominantly made up of pines, bushes, high mountainous pastures and crops. The pine forest has high fire risk but because of the prevailing weather conditions existing in the area we have high and moderate quality classes. The moderate class is related to the slopes facing to east and south and high class is north and west.

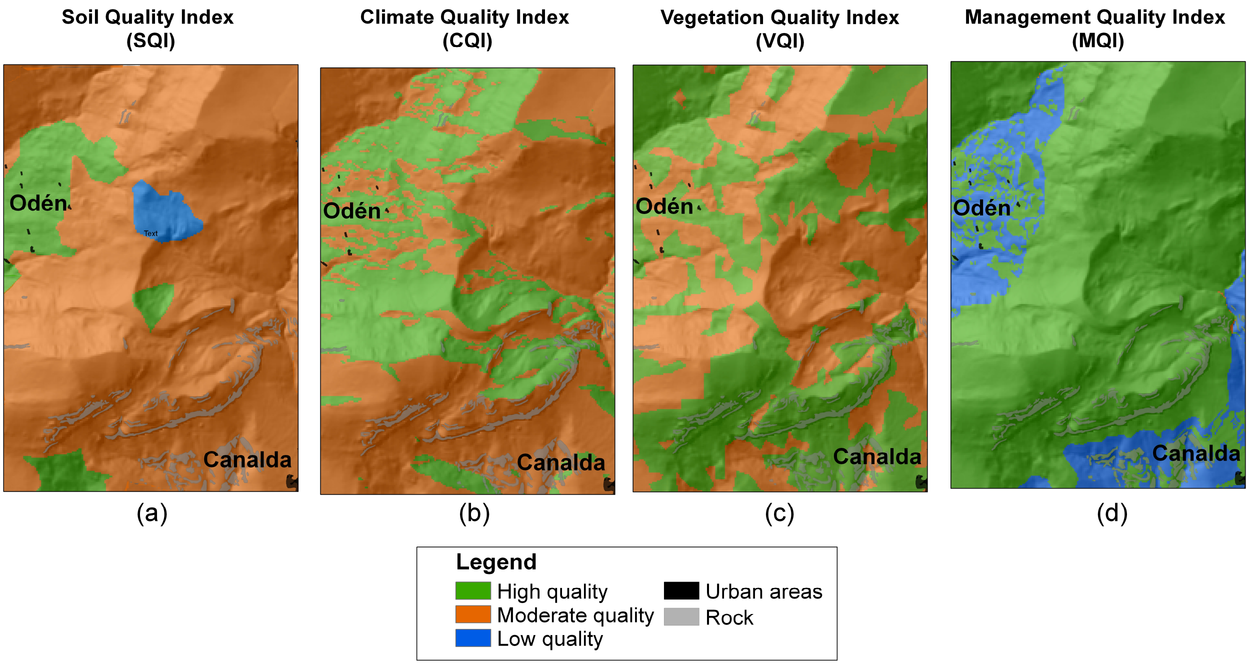

Figure 6. Quality indexes for the study area; (a) SQI, (b) CQI, (c) VQI, (d) MQI.

3.1.4 *Management Quality Index*

The management quality index is represented in the Figure 6d. It shows that 79% of the area is classified as having high quality of management associated with good tree cover. Depending on the particular type of management, land resources are subject to a given degree of stress. The low quality management index (20.9% of the area) is found in areas with very sparse forest (pines), bush and bare soil. These areas with bush are mainly land abandonment where natural vegetation is taking over the place. Land abandonment may lead to a deteriorating or improving phase of the soils. Some of these fields abandonment are on steep slope, hence on terraces. These abandonment terraces could cause big damages to the environment because of they could collapse causing a rapid removal of the soil by the runoff water

Notably, if the areas with bush and scrubs land cover and bare soil areas were overgrazed, the soil erosion could be severe. The overgrazing in areas of natural vegetation causes damage to cleared spaces in the absence of an acceptable vegetation cover.

Figures 6d and 7 show that high sensitivity indices in this model area are mainly related to the management index. The existence of environmental policies which apply to a certain area moderate the anticipated impacts of a given land use type compared to the situation where no such policies are in effect. The area classified as high quality is considered as natural heritage (PEIN), whilst low quality area is not. This protected natural heritage has mainly forest and pasture land uses. The forest in this PEIN area should be considered as well managed, that lead to current forest conditions including logging large trees, fire suppression, and a controlled livestock grazing. The agriculture fields that are in the study area, they are classified as low land use intensity. The potato crops have a minimal management because of the low-input, conservation agriculture techniques applied.

3.1.5 *Environmental Sensitive Area Index*

The results show that the majority of the study area is classified as fragile (61.5 %) followed by critical (17.8%) and non affected (16.8%), while only 3.7% of the area is potentially sensitive (Figure 7). Figure 7 displays the spatial distribution of ESAI over all study areas.

Results showed that plant cover and managerial policies were the most important indicators affecting desertification process.

The non affected areas belong to agricultural fields, producing high quality potatoes. The management quality index collects this information under the land use intensity. These fields are under adequate agriculture practices, with traditional farming management, so they have low land use intensity.

Fragile (F1, F2, F3) (Figure 7) areas are very sensitive to any climate change or any change in the land use, showing up soil degradation. A good correlation exists between management and fragile class, as most of the area corresponds to a high quality management. Some areas classified as fragile are protected areas because they are included in the natural heritage (PEIN). In these areas all activities are limited to traditional agricultural, livestock and forestry activities, being these compatible with the specific objectives of protection according to their scientific, ecological, scenic, cultural, social, educational and recreational values.

The soils in fragile class are soils that have a large range of depths, including depth from 40 to 120 cm. Soil depth in this class is not one of the constraints of fragile soil class degradation because of the recovery of the natural perennial vegetation is medium. Even where soil depth is good, loss of the topsoil is often not conspicuous but nevertheless potentially very damaging. Soils in fragile class are well-drained, mostly forest or mountainous meadows.

The critical areas (C1, C2, C3) (Figure 7) have mostly shallow and very stony soils resulting in a low soil quality. These have poor or moderate vegetation cover, even some areas quite eroded. Some pasture areas have moderate intensity of grazing, which drives to soil degradation and deterioration. There are some hectares with sparse pines forest but it is not included in the natural heritage (PEIN).

These critical areas are under very steep mountain slope; hence they are very sensitive to heavy rains, which promote erosion and degradation. The critical areas additionally include abandoned land or very steep slopes with bare soil (not including outcrops). These areas are found interspersed with the non affected or potential areas.

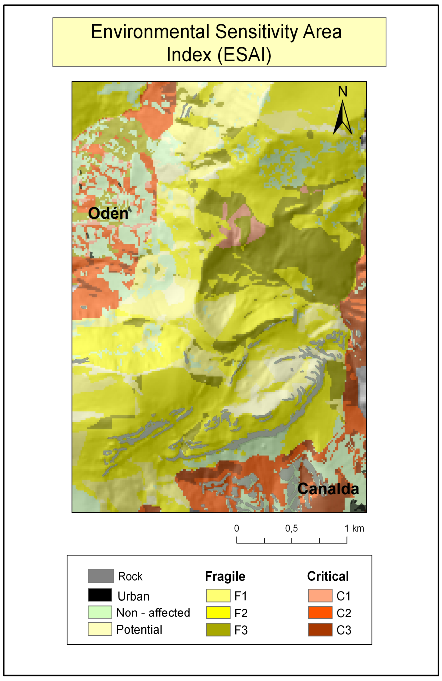

Figure 7. Environmental Sensitivity Areas Index (ESAI).

These results also show that in areas classed as moderate quality, the severity of desertification is still low because of the good management practices. These areas are generally used for agricultural activities.

The developed model attempts to assess and identify the factors affecting desertification. It gives an overall picture of the study area. A more comprehensive description on how the environmental layers are linked to the degradation or desertification phenomena is given in Kosmas *et al.* (1999).

One issue on the model is that the model relies on the relationship between the climate, the physical environment, socio-economic factors, land use and land management, to be able to predict land degradation on the study area. Land degradation is mediated and affected by at least the following: available technology, market mechanisms, historical tradition, friendly farming management, culture, and various economic factors such as subsidies which have not been taken fully into consideration. All these aspects are currently invisible to the model and are not directly present in any of the available data. However, such a model is presently beyond technological feasibility and data availability. Conversely, it is hoped that the missing variables are invisibly present in the data that are used and are thus taken into account implicitly by the neural nets that are applied to model the relationships.

One of the particular aspects of the proposed system is that the desertification severity classes are not absolute values of severity but are relative for a particular area. There is scope for minor improvements in the model whenever more recent data is available or modifying the model by addition of parameters. The selection of the layers is an open process, though only meaningful layers will produce meaningful results.

The best method and model of desertification zonation should be a method that could easily use available data and information, one that can comprehensively assess all the important factors and apply some flexibility in the application of model weights assigned.

Recognizing and determining the relation between desertification intensity and the effective predictive factors can help with quantifying the desertification process. For example, in some areas there may be present wrong agricultural practises supporting desertification, while in other areas using a corrected irrigation system or conservation farming can lead to a decrease in soil degradation.

3.2 <u>*Land Evaluation: Land Capability Classes map*</u>

The dominant land capability classes in the area are VII, and VIII. These classes have 66.5% and 23.2%, respectively over the total extent of the study area (Table 7), and despite that the whole area is defined as ‘do not cultivate’, 10.3% of the area is dedicated to crops and meadows.

The soils with crops are shallow soils or with moderate depth and stony, and they occur on steep mountain slopes. Also, their management is minimal and conservation agriculture techniques are applied. These crops and meadows are located in low altitudes and are classified as VII and VIII land capability classes (Figure 8), however they are classified as non affected areas in the ESAs (Figure 7), because of the management practices.

Table 7. Occurrence of the land capability .

CLASSES	%	Comments
V	10.3	Severe limitations. Do not cultivate. No limitations for pasture, forest or scrubland.
VII	66.5	Do not cultivate. Suitable for forests and pastures
VIII	23.2	Do not cultivate.

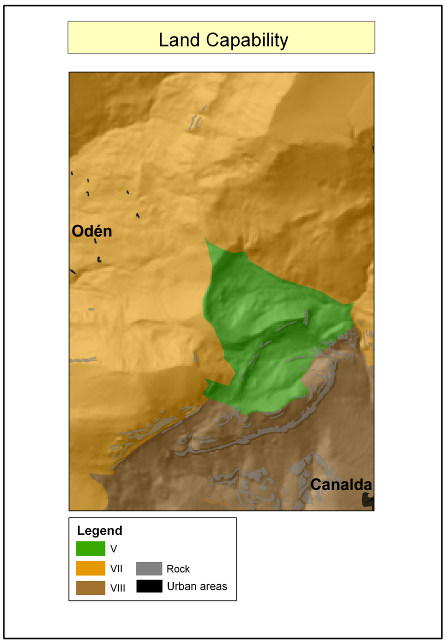

Figure 8. Land capability map of the study area.

Figure 8 shows the spatial distribution of agrological capability classes according to the soil mapping units (Figure 9 and Table 8). These units are defined in the soil survey (Figure 9) and described in detail. In general, the land capability is an indicator for local soil use and management but we should consider it as a relatively permanent, static land characteristic that does not take into account socio-economic factors. Although the system provided a general appraisal, it does not assess capability separately for each kind of land use.

Figure 9 shows the soil map at detailed scale (1:25.000) of the study area and the soil mapping units. Table 8 has the agrological class for each map unit and the limitations of these.

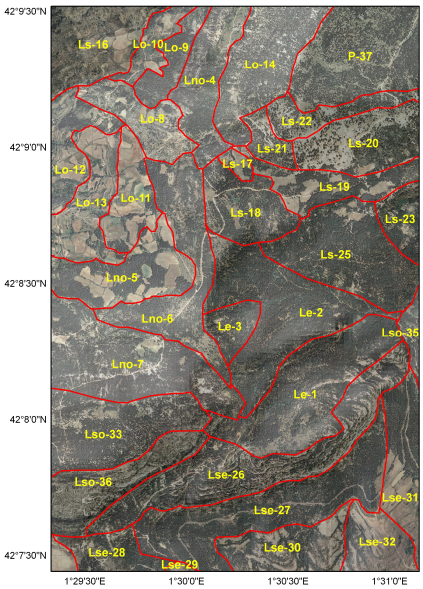

Figure 9. Soil map at detailed scale (1:25,000) from the study area

Table 8. Land capability classes (MAPA, 1974) for each map unit of the soil map.

Mapping unit	Land Capability Classes	Limitations
Le1, Le2, Le3	**V**s	slope
Lno4	**VII**s,d,e	slope, depth, erosion
Lno5, Lno6, Lno7	**VII**s,d,e,t	slope, depth, erosion, texture
Lo8 to Lo15	**VII**s,d,e, ss	slope, depth, erosion, surface stones
Ls16 to Ls25	**VII**s,d,ss	slope, depth, surface stones
Lse26 to Lse32	**VIII**s,d,t,rf,ss	slope, depth, texture, coarse elements, surface stones
Lso33 to Lso36	**VII**s,d,ss	slope, depth, surface stones
P37	**VII**s,s,t,rf,ss	slope, depth, texture, coarse elements, surface stones

The land evaluation model applied is a combination method using qualitative evaluation, essentially based on expert judgment, quantitative evaluation, and broadly based process simulation models. Hence, particular crops (potatoes) and the specific land management practices of the area cannot be considered within the model. This is the case of the potato fields in the study area. This potato fields are classified as VII class, being considered unsuitable for agriculture, but in fact, these fields are very good for potato production owing to good soil characteristics for growth, such as, appropriate stoniness, optimum soil texture, high organic matter content, good drainage, and weather conditions. This crop is based on conservation agriculture management making it specific to the area.

3.3 *RUSLE model application*

3.3.1 *Soil erodibility factor (K)*

Different soil types are naturally resistant or susceptible to more erosion than other soils. This depends on the soil texture, drainage potential, structural integrity, organic content and cohesiveness. Erodibility of soil constitutes its resistance to both detachment and transport. The corresponding K values for the soil types were identified from the soil erodibility nomograph (Wischmeier and Smith, 1978) by considering the particle size, organic matter content and permeability class. The K values in the SI unit system vary between 0 and 0.1 Mg h MJ^{-1} mm^{-1}.

The range of K values in the study area is 0.01 to 0.06 Mg h MJ^{-1} mm^{-1}.The estimated K values vary according to soil texture and soil structure, for the textural groups vary from 0.05 Mg h MJ^{-1} mm^{-1} (gravelly loam with blocky or massive structure), 0.03 Mg h MJ^{-1} mm^{-1} for loam with subangular blocky soil structure and 0.01 Mg h MJ^{-1} mm^{-1} for clay with subangular blocky soil structure.

3.3.2 *Slope length and steepness factor (LS)*

The majority of the study area has LS value less than 10 with a few specific areas only showing values of higher than 20. The study area exhibited slopes steeper than 30%, with many exceeding 60%, and even some approaching 100%. The slope factor indicated by obtaining the maximum score is one of the main factors affecting erosion.

3.3.3 *Cover management factor (C)*

C-factors for cropland in the study area, ranged between 0.2 and 0.9, depending on the land use, 0.2 for pasture, 0.21-0.22 for deciduous and pines forest with more than 20% canopy cover, 0.23 for deciduous and pines forest with less than 20% canopy cover and 0.9 for already eroded areas.

3.3.4 *Average annual soil loss*

We have the highest values (> 200 Mg/ha·year) in streams and canyons. The lowest erosion values (0 – 25 Mg/ha·year) are found in the highest altitudes for which the main land uses are forest and pastures. In slopes with more than 30%, we have the middle values such as 25 – 100 Mg/ha·year (Figure 10).

By removing the most fertile topsoil, erosion reduces soil productivity and, where soils are shallow, may lead to an irreversible loss of natural farmland. Severe erosion is commonly associated with the development of temporary or permanently eroded channels or gullies that can fragment farmland.

The spatial patterning of classified soil erosion risk zones indicates that the areas with high and severe erosion risk are located to the west, northwest and southern slopes of the study area, while the areas with low erosion risk are in the eastern and central parts of the study area.

Unsurprisingly, at the same time the spatial distribution of annual average soil erosion risk map shows high spatial correlation with LS-factor map, and it indicates the role played by topography in controlling soil movement in a watershed.

The RUSLE tends to overestimate erosion rates for Mediterranean conditions, where most of the sediment mobilization takes place during the extremely, intense rainfall of a high- return period (Ibáñez, 1997).

Processing of data for input into the RUSLE required the use of several algorithms, each of which may accentuate the existing errors in data. Because the RUSLE model requires the six input data layers to be multiplied together, the errors inherent in each layer are similarly multiplied, contributing to an even greater error in the derived soil loss values. A critique of the RUSLE method therefore is that it incurs an overestimation of soil loss values.

However, this can be potentially overcome through additional field research that could be conducted to validate the data obtained in the application of the RUSLE model.

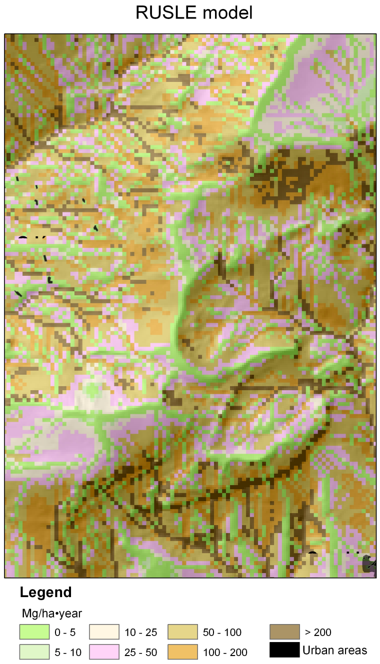

Figure 10. Soil loss (Mg/ha·year) map obtained by application of the RUSLE model.

3.4 *Validation of the maps products*

Figure 11 shows the percentage result of the spatial intersection in GIS of timber yields and mass movements compared with MEDALUS, RUSLE and land capability map products of the models applied. In the case of timber production and mass movements the absolute areal values were compared (Figure 11), while in the case of the land use changes the three situations (no changes, progressive, regressive) were compared using relative values to the 100% surface of each one (Figure 12).

The areas with VII and VIII land capability classes are occupied by forests and pasture, while lower classes are either used for agriculture or marginally exploited. Most of the area with VII class, which it is suitable for forestry and pasture uses, is the class that has the highest timber yields extractions (Figure 11a). The same figure shows that the areas with VIII class produce the lowest yield extractions or have no data (marginally exploited forest). These results are probably a reflection of a good forest management in the area, and show a positive correlation between timber yields and land capability map. Figure 11b show a positive tendency between the ESA classes and the timber yields, that means that the classes with less affection to desertification (Non affected, Potential, Fragile) are those with highest yield extractions. Only 4% of the critical ESA class areas have woody extractions, however, this critical class is dominated by *Pinus nigra,* with >= 20% of land cover, and the clearing due to the exploitation may help to prevent the burn risk.

Good management of the forest provides a habitat for wild animals, thus ensuring both increased biological diversity and additional abundance. MEDALUS model result is positively correlated to the timber yields extractions, also some results for RUSLE agree with the hypothesis of higher yields in the lowest erosion areas (also classified as non affected or potential ESA classes). However, 14% of the surface has high timber yields in areas with ≥ 25 Mg/ha·year of soil loss (Figure 11c). It could due to the fact that timber production can be high on high slopes, which would bias the RUSLE erosion values.

In spite of the fact that the mass movement map has a lower scale resolution than the one applied in this thesis, and that only two classes are represented,, Figure 11d and Figure 11e show similar tendency for medium and high mass movements, versus land capability and ESAs classes. Nevertheless 10% of the V class is classified as high potential of mass movement. Since class V is defined by low slope and high stoniness, this mass movement could be soil creep. In addition, the non affected ESA class occupy 13% and classified as high potential of mass movement, while these areas are mostly agriculture on 10-15% of steepness. Notably, the actual mass movements do not show to be much related to the other indices studied.

Figures 12a and 12b show that areas with present day VIII class, which is classified as fragile or critical in ESAs, had a progressive land use change evolution (increase of land cover), so it would correspond to a theoretical improvement on soil quality from 1957 to 1992. Nevertheless, one must consider that this change corresponds to a shift from traditional agricultural land use or more intensive pastures than today to forest or shrubs, which in some cases has led to a worsening of soil quality (more compaction, higher erosion, Lasanta and Ruiz, 1990; Arnáez and Ortigosa, 1997; Pias, *et al.*, 2014), therefore the ESA is reflecting this worsening of quality due to higher cover in the factor of higher susceptibility to forest fires. Figure 12c represents the relation between the land use changes and soil erosion. Since soil cover is a direct factor in RUSLE, progressive changes have, in average, lower erosion values than the rest.

In summary, the application of MEDALUS model to the area gives good matches with timber production and can be related to the changes in land cover in the second half of 20th century, but the cause-effect relations are not probably enough reflected in the models. RUSLE values are also biased by the original application of the model to agricultural areas; therefore they do not match to higher timber productions that can occur in high slopes. Finally, actual mass movements cannot be properly related to any of the indices, because of low resolution of the original maps.

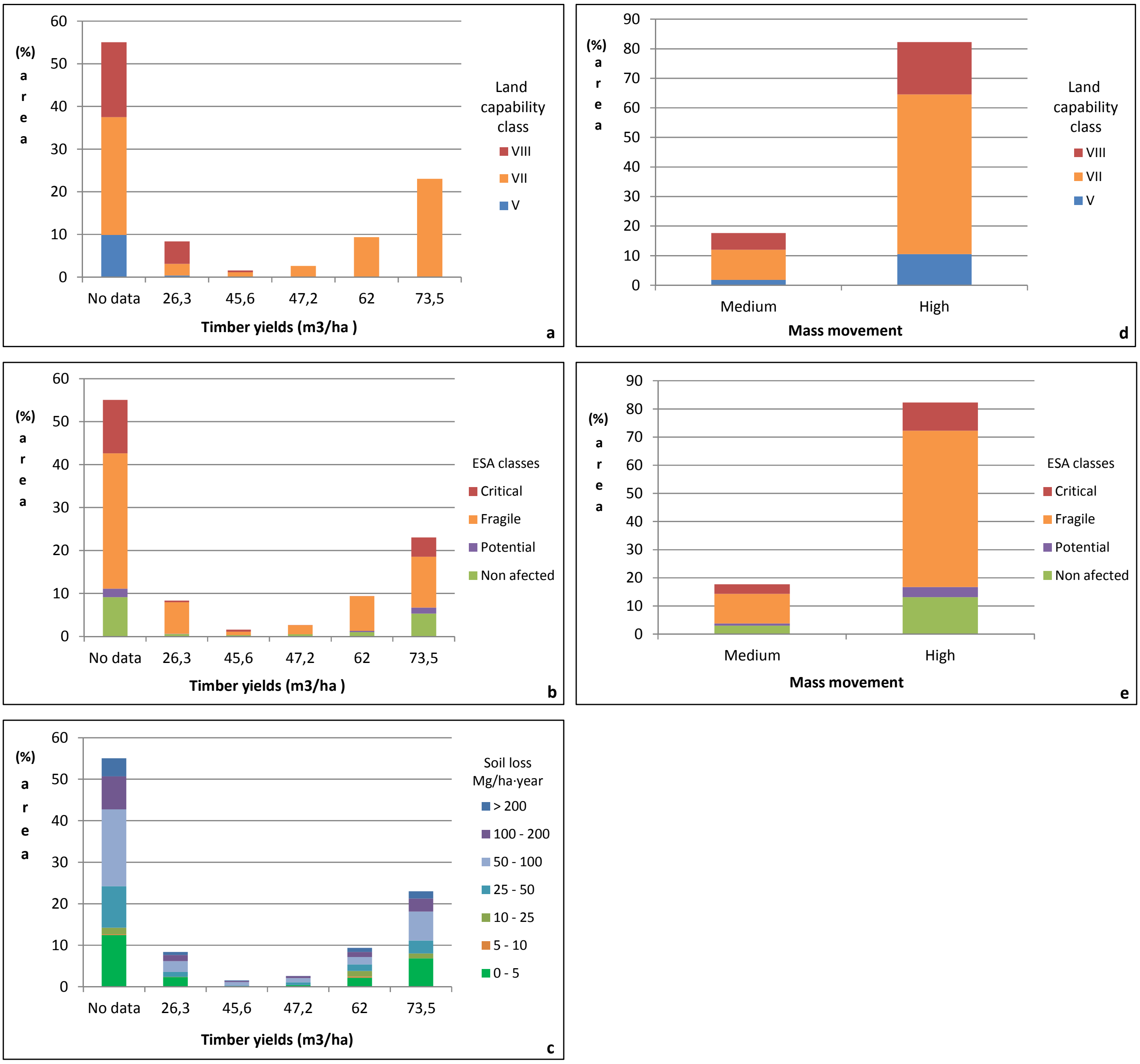

Figure 11. Contingency graph of soil loss (RUSLE) (Mg/ha·year), desertification (MEDALUS) (ESAs class) and Land capability compared with timber yield and mass movements

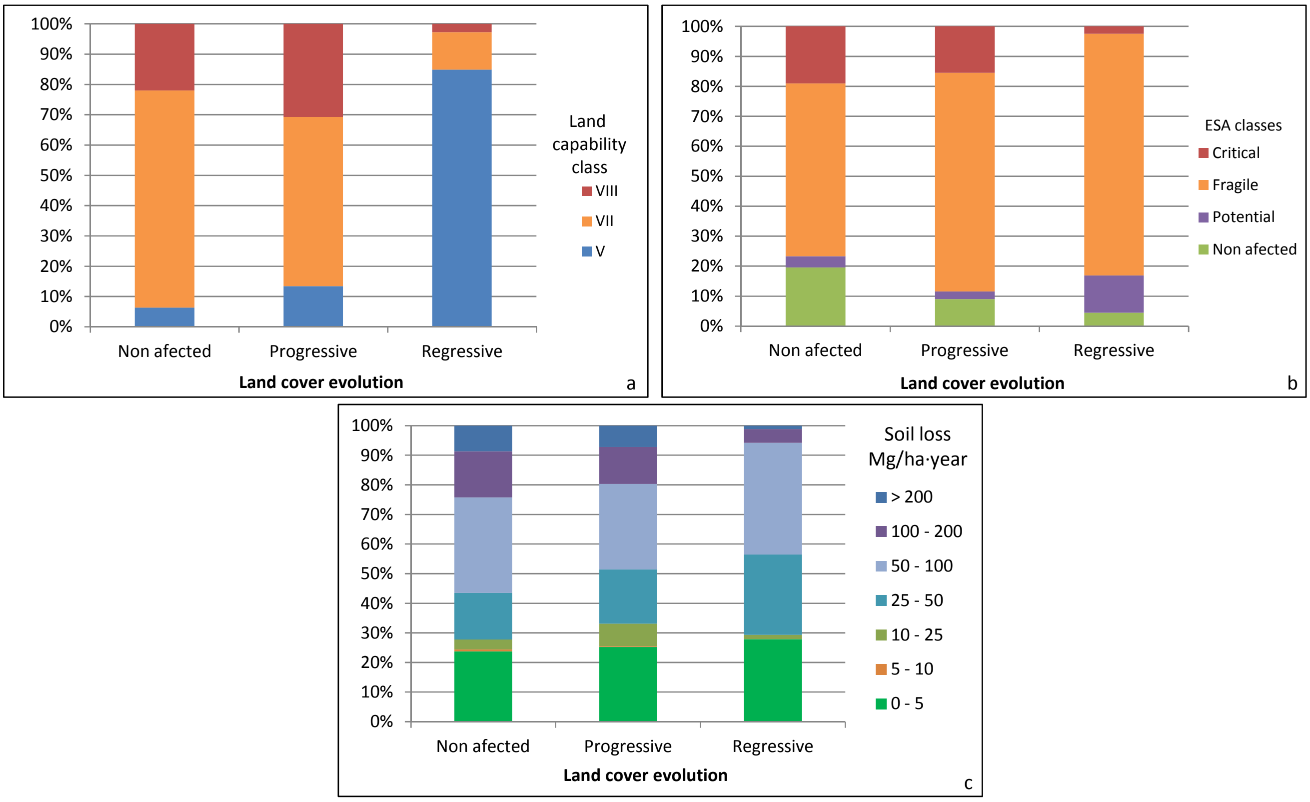

Figura 12. Contingency graph of soil loss (RUSLE) (Mg/ha·year), desertification (MEDALUS) (ESAs class) and Land capability compared with the three types of land cover evolution.

4 DISCUSSION

In order to compare and discuss the different maps contingency diagrams have been used. Each class of one model represents the occurrence of the classes of the other model.

4.1 *MEDALUS model and land capability classes*

The relation between both maps, if any, could be caused by the fact that both systems use soil and topographical information. The 'fragile ESA' class is the most frequently occurring class in all of the land capability, being the most dominant class in VII agrological class. Approximately, 11% of the study area is classified as Class V, which is mainly forestry. The agriculture fields of the study area are classified as VII and VIII classes in land capability.

The agrological class evaluation applies a categorical system instead of suitability. This kind of evaluation does not take into consideration particular land uses adapted to the territory with appropriate management such as the conservation agriculture techniques being applied in the agricultural fields in the study area. These fields are classified as 'non affected' class in the MEDALUS model, but are not taken into account as input factor in the land capability classes.

The areas classified as VII and VIII classes in land capability, and critical as ESAs, generally have sparse forestry cover, comprising mainly of pines found on south-southwest facing slopes. Overall, the main difference between land capability class VII and VIII is the slope. The VII class has a slope of less than 30% whereas VIII has more than 30%.

4.2 *Land capability class and RUSLE model*

The land capability classes are determined by the actual erosion among other factors. The comparison of both maps gives us information about the role of the other land capability parameters in the soil agrological capability. The main classes of soil loss class are 0 – 5 Mg/ha·year and 50 – 100 Mg/ha·year. The results are linked to the altitude. We have lower soil loss values at the highest altitude because the slope in these areas is low (< 8%) and they are covered by dense forest or pastures, while, the areas on the steepest slopes have highest values of soil loss.

Soils in class V have moderate-high erosion hazard (25-50 Mg/ha·year) (50-100 Mg/ha·year) related to the slope. Slopes of 10 - 40% are less affected by erosion than, slopes with more than 40%. As a result, these soils have practical limitations and therefore use is largely confined to pasture, woodland and high canopy cover. Because of these limitations cultivation of the common crops is not feasible but pastures can be improved and benefits of proper management can be expected, and can serve to avoid raising erosion hazard.

Soils in class VII and VIII have a large range of erosion hazard (0-200 Mg/ha·year). The agriculture fields classified as class VII existing in the study area, have slope of 10 - 40%. In these field crops were found low values of erosion (0 – 10 Mg/ha·year) basically related to low slopes (10 – 25 %), and the highest erosion values (100- 200 Mg/ha·year) associated with slopes of more than 40%. Where these soils in class VII are not suited to any of the common cultivated crops; in unusual instances, some soils in this class may be used for special crops under unconventional management practices. Some areas of class VII may need seeding or planting to protect the soil and to prevent damage to adjoining areas (Klingebiel & Montgomery, 1961).

The rest of the VII class and major part of class VIII have forest use and the erosion rates are variable. Class VIII is not classified as natural heritage (PEIN) but it should be considered for this classification. It may be necessary to give protection and management for plant growth to soils and landforms in order to protect other more valuable soils, to control water, or for wildlife or aesthetic reasons (Klingebiel & Montgomery, 1961).

4.3 *MEDALUS and RUSLE model*

Being soil erosion one component of land degradation, the comparison of both maps (MEDALUS and RUSLE) can give information of the weight of soil erosion in land degradation risk. The classes 0-5; 50-100 Mg/ha·year are the most frequent erosion hazard, both with pines and scrub land uses.

Fragile class in MEDALUS model with low-moderate slopes (<8%) has low erosion hazard (0 - 5 Mg/ha·year), however fragile ESAs are well distributed in the study area, and erosion rates in this class vary depending upon the slopes and land uses. In 'potential' and 'fragile' (F1/F2) ESA classes with a slope of 25 – 50% and under bush and scrub land use, reflect lower values of erosion (0 – 10 Mg/ha·year). The potential class is mainly crops with low erosion hazard (0 – 10 Mg/ha·year).

There are fragile (F1/F2) and critical (C2) classes with 10 – 25 Mg/ha·year erosion hazard where the main uses are scrubs and pine land use (*Pinus nigra*). The critical class is not classified as natural heritage (PEIN) and maybe it should be considered. There are fragile (F2/F3) and critical (C3) classes with a range of erosion hazard of 25 – 50 Mg/ha·year, having pine land use (*Pinus sylvestris*) on slopes of 10-30%, whilst on erosion hazard of 50 – 100 Mg/ha·year with same land use, have slopes of 40 -70%.

The areas with highest values of erosion hazard (100 - 200 or more Mg/ha·year), are areas with very sparse forest on very abrupt slope (40 -70% or higher). This is because bare soil is overall the most susceptible to soil erosion (Tóth *et al*., 2008; Virto *et al*., 2015), and here it is compounded by the very abrupt slope.

Some of the parameters used for studying desertification and erosion have a low influence in the variability of the soil degradation index in the area, such as aspect, climate conditions, and soil properties. Therefore, the actions required for mitigation of desertification in environmentally sensitive areas to desertification in the study area are mainly related to soil cover management in order to protect soil from erosion and to those modifying local topography. The optimal management for Potential ESAs and Fragile ESAs, far from setting land aside, should include special management such as conservation agriculture. The management of critical areas should be directed to mitigate erosion, the main degradation process. Some examples of recommendations on forest management practices are appropriate forest regeneration method(s), limitation of grazing areas, prescribed fires for forest fire prevention with conservation criteria, rational contour timber harvest, and the Best Management Practices (BMPs) that apply to each practice. These forestry activities represent a balance between overall natural resource protection and use.

5 CONCLUSIONS

In MEDALUS and RUSLE approaches certain amount of data that is needed, that is not always available with the precision needed at detailed working scales. Therefore it may hinder the applicability of the models or making them less accurate. Moreover, some of the data needed in the model, such as climate data, at detailed scale (1:25,000) are not particularly relevant because the spatial variability is reduced, and thus do not contribute to prioritize treatment areas for erosion risk reduction. The input parameters having the highest influence in the spatial variability of the indices are, unsurprisingly, the physiographic characteristics and soil cover, since they are affecting erosion, which is the main desertification process in the study area, more than other erosion factors as soil type.

The socio-economic factors in MEDALUS model are quantified through scoring. This step has a subjective component that requires a thorough knowledge of the agricultural and land use systems, that can be done at detailed scales, but not when mapping large areas at general scales.

MEDALUS and RUSLE are models based on a combination of parameters and that can easily be implemented within a GIS. The accuracy of the predicted soil loss can be improved, if each parameter is better estimated. Certain aspects of this modelling exercise reflected overestimations of soil loss values associated with a multiplier effect. Although future iterations of this work will seek to improve errors associated with this methodology, at a practical level an overestimation represents less risk from a forward management perspective than an underestimation of soil loss.

Based on the MEDALUS model, almost 80% of the study area was sensitive and affected by desertification, but only 17.7% of the area studied was classified as environmentally critical to desertification, corresponding to areas degraded due to soil management (abandoned land) or extreme topography without vegetation (very steep slope with bare soil). Potential ESA class to desertification may require either protection from erosion and special management such as ongoing conservation agriculture.

The RUSLE and MEDALUS models can assess the extent, intensity and severity of erosion and desertification process in the target area. However, there is scope for improvements as data become available, such as erosion data. Therefore, the MEDALUS model should be flexible, giving the opportunity to add new parameters into the model, such as the slope length and steepness factor, that have more importance when working at detailed scales.

The land capability assessment, on the contrary, is a general appraisal, that uses relatively permanent, static land characteristics that does not take into account socio-economic factors, nor special management practices that could improve land suitability for specific uses. Crops such as potatoes with minimal management and low erosion impact in the study area are classified as VII agrological class, despite these are as 'non affected' ESAs for the MEDALUS model.

The application of MEDALUS model to the area gives good matches with timber production and can be related to the changes in land cover in the second half of 20th century, but the cause-effect relations are not probably enough reflected in the models. RUSLE values are also biased by the original application of the model to agricultural areas; therefore they do not match to higher timber productions that can occur in high slopes. Finally, actual mass movements cannot be properly related to any of the indices, because of low resolution of the original maps.

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Chapter 5

GENERAL DISCUSSION

The Challenge

Through the knowledge generated as a result of the present research, we have sought to propose economic and feasible techniques and models to develop soil quality indicators, representative to the major soil threats affecting the sustainability and multifuncationality of the soils in the study area. We have also taken on the more difficult challenge of trying to make realistic recommendations related to soil degradation specific to each study area.

GENERAL DISCUSSION

Soil quality indicators: are they meaningful?

A good design of a soil monitoring system for soil quality indicators should be able to define the soils status in front to soil threats and soil functioning. The selection of the two different pilot areas was strictly dependent upon the soil degradation processes observed in the study areas and their representativeness in the Mediterranean region. They have shown differences on the soil quality indicators, which was the main goal of this thesis: salinity and sodicity status (Northern side of the Ebro Delta) and SOC stocks, vulnerability to erosion and desertification (Solsonès area). They are considered as priorities of the European Union Thematic Strategy for Soil Protection, the European Commission Directorate - General for the Environment (DG Environment) and the European Environmental Agency (EEA) (Panagos, *et al.*, 2013).

The selection of the soil quality indicators will vary depending on the socio-economic and environmental interests of the area. Moreover, these indicators should be able to express changes over time. For instance, as shown in **Chapter II, the salt content** could be quantified using direct and indirect measurements such as electrical conductivity ($EC_{1:5}$ and ECe) and electromagnetic induction (ECa), respectively, whose 3D patterns consistently changed after a 12 year period.

The proposed European policy in the agricultural sector is placing a high emphasis on soil organic carbon (SOC), as an indicator of soil quality and as a means to offset CO_2 emissions through soil carbon (C) sequestration (Lugato *et al.*, 2013). The results obtained in **Chapter III** show how the **SOC stock** could vary considerably at different depths for the SOC stock calculations, and different land uses. While topsoil SOC content and organic carbon stocks are specifically defined as priority indicators for evaluating the soil status in Europe, several researches have proved that deeper layers in the soil profile are able to store a substantial amount of organic C (Batjes, 1996; Jobbágy and Jackson, 2000; Muñoz-Rojas *et al.,* 2012). In our case, the relative SOC content in the middle 60 cm (20-80 cm) ranges from 49% to 53%, depending on the different land use. Stratification of SOC with depth is common in many natural ecosystems, managed grasslands and forests, and conservation-tilled cropland. However, the temporal change in SOC stock is difficult to assess over a short term period. Muñoz-Rojas *et al.* (2012) argued that at detailed scales, anthropic transformation of ecosystems may strongly affect SOC content, additionally, at local scale, redistribution processes of soil organic matter by water erosion processes following wildfires may be substantial. Our results confirm that not only topsoil SOC stock should be considered as a soil quality indicator, but also SOC stock at deeper layers.

Desertification is a complex concept. It is a consequence of a set of important processes which are active in arid and semiarid environments controlled by multiple factors, such as climatic conditions, vegetation cover, soil properties and human activities. Water erosion is one of those processes that it influences desertification in Mediterranean areas. The soil indicators selected to study vulnerability to **desertification and erosion** in **Chapter IV** are those of the MEDALUS and RUSLE models respectively. They are based on multi-criteria decision analysis (MCDA). MCDA is often suggested as a suitable approach to support decision-making because of its capacity to rank remediation alternatives based on an assessment of criteria associated with the environmental, socio-cultural as well as the economic domains of sustainable development (Kumar and Jhariya, 2015). In addition, **land capacity** using agrological classes by Klingebiel & Montgomery (1961) was applied to the study area as well. This system categorises according to the actual productivity, irrespective of soil degradation on medium to long term and regardless of the applied cultivation and management.

In both study areas the important linkage between the preservation of soil quality on one hand, and the achievements of sustainable agriculture and the conservation management practices on the other, are noted. The results presented in Chapter II showed that in the Ebro Delta, the irrigation water management of the paddy rice had a positive decrease of the soil salt content for a period of 12 years time. However, saline soils are needed to maintain the ecosystem exiting in this area. The study of soil salinity indicators allow both consultants and determining authorities to better predict appropriate design for sustainable practices and management of the hydrologic cycle. Equally, in the Solsonès mountainous area, the crop fields (potatoes) existing in the area have minimal management; that are associated with soil-friendly tillage practices. The result of that are shown in chapter III, where these agricultural fields are classified as class VII for land capacity in slope of 10 - 40%, while MEDALUS model classify them as non affected areas, and with low erosion values (0 – 10 Mg/ha·year) according to RUSLE thanks to this good management. However in Chapter II, these crop fields (potatoes) have the lowest SOC stock in the study area (36.4 Mg C/ha), that, on the other hand, they are much higher than the average levels (15.9 Mg C/ha) found in other crop systems such as intensive and extensive crops in the Southern of Spain (Muñoz-Rojas *et al.* 2012). Hence, these agroecosystems should be considered of high quality as a whole, taking into account the socio-economic factors. Under conservation management (pastureland and forestland) SOC is significantly more stratified with depth than under conventional cropping. This stratification should be considered within as an improvement in soil quality, because several key soil functions are enhanced, including sequestration of C from the atmosphere (Franzluebbers, 2010).

Soil quality indicators: are they useful?

In chapter II, the electromagnetic sensor showed a significant spatial and temporal variability after 12 years. The extent of saline soils declined by 22% in 2007 with respect to 1994-96, while those of sodic soils increased. These results are in line with Herrero and Castañeda (2015) found for a similar environment in paddy fields after 20 years. Regarding the spatial distribution, non-saline soils are found at the highest locations, where they are well drained. There are some saline soils located in the flatter areas near the coast and have moderate coarse or coarse textures. The saline-sodic soils are located in ancient fluvial soils or river lakes. The remaining soil types, except for soils near the water recharge areas of adjacent uplands, are moderately to strongly saline. Overall, spatial and temporal distribution of the salinity indicator was well captured by the method utilised in this study.

In relation to SOC stock, in chapter III, showed that land use, altitude and depth describe part of the variations in SOC at different depths and the SOC stock. Soils under cropland use (62.57 Mg/ha) had less SOC than grazing (88.53 Mg/ha) or forest (116.33 Mg/ha) soils, thus land use is a strong factor affecting SOC distribution. Having in mind that this area has suffered a considerable reduction of croplands in the last 20 years (Ubalde, 1999), afforestation of pasture by pines could increase SOC and would provide protective cover in vulnerable, steep and mountainous areas. Moreover, a conversion of crop land to pasture could cause a substantial C accumulation below 1 m depth. In the study area, more than 50% of SOC on average is stored in the topsoil (Muñoz-Rojas *et al.*, 2012), however, the relative SOC content in the middle 60 cm (20-80 cm), ranges from 49% to 53%, depending on the land use. This finding highlights a greater need for further research, especially in relation to developing, new methods and tools to explore the potential impacts of future climate changes in SOC contents at different soil depths and land use types (Muñoz-Rojas *et al.,* 2013).

The results obtained in Chapter IV, showed that RUSLE and MEDALUS models can assess the extent, intensity and severity of erosion and desertification process in the target area. Based on the MEDALUS model, almost 80% of the study area was sensitive and affected by desertification, but only 17.7% of the area studied was classified as environmentally critical to desertification, corresponding to areas degraded due to soil management (abandoned land) or extreme topography without vegetation (very steep slope with bare soil). Potential ESA class to desertification may require either protection from erosion and special management such as ongoing conservation agriculture. Ibáñez (1997) reported that RUSLE model tends to overestimate erosion rates for Mediterranean conditions, where most of the sediment mobilization takes place during the extremely, intense rainfall of a high - return period. Our results on RUSLE model showed the highest values (> 200 Mg/ha·year) are found in small rivers and canyons, that are very often rock outcrops. The lowest erosion values (0 – 25 Mg/ha·year) are found in the highest altitudes where the main land uses are forest and pastures on relatively flat surfaces. On >30% steepness we found mid-ranging values such as 25 – 100 Mg/ha·year. Both models are based on a combination of parameters that can easily be implemented within a GIS, however, the required amount of data that is needed for applying them is not always available with the precision needed at the detailed working scale. Therefore, it may hinder the applicability of the models or make them less accurate. Certain aspects of this modelling exercise reflected overestimations of soil loss values associated with multiplier effects, nevertheless, at a practical level an overestimation represents less risk from a forward management perspective than an underestimation of soil loss.

The soil quality indices studied are good for diagnoses, however they are still far from determining the best management practices to improve soil quality. This is due to the fact that it is difficult to determine the causal relationships between the factor indices, partly because the factors used in the models for developing the soil quality indices, are often not independent. This fact occurs in the RUSLE and MEDALUS models, for example slope and land cover vegetation; slope and land use or soil type and land use. This is also the case when modelling the SOC using land use and soil depth, where agricultural soils are usually the deepest ones. Contrary to this finding, this was not the case for soil salinity because the index studied is a direct measurement of a property, and the causes of soil salinity can be related to management practices affecting its temporal and spatial variability. The in-built biases that can occur when indices are combined within certain modelling applications must be considered when interpreting the results.

Soil quality indicators: can they be improved?

Several types of mapping procedures were used in our research. The spatial representation of soil salinity and SOC was done using a geostatistical model while a detailed soil map was used for the SOC as well. The results of applying a MCDA for desertification, EROSION and land capacity were class maps. In all cases we found room for improvement.

In Chapter II, soil salinity distribution map was obtained with the use of co-kriging approach. The validations of the co-kriging maps were calculated for a subset of data for the two campaigns (1994-1996 and 2007-2008). Prediction errors were highest in the soil salinity map for the 2007-2008 than for the 1994-1996. The prediction coefficient of determination is highest and very good (R_p^2 = 91.6) for the soil salinity map in 1994-1996, however the accuracy for the soil salinity map in 2007-2008 is good enough (R_p^2 = 67.8), based upon previous papers describing the accuracy assessment of predictive maps, where an acceptable confidence levels of between 50-80% is recommended (Moran and Bui, 2002; Minasny and McBratney, 2007; Simo *et al.,* 2015).

Chapter III showed that satisfactory results were obtained using a detailed soil map for mapping SOC, when it was compared with other digital mapping methods. Regional assessments of SOC usually have to rely on soil surveys not originally designed for assessing SOC stock. Not many countries have available detailed soil maps, with enough data to calculate SOC, as measured bulk density or stone contents. Consequently, only detailed soil maps should be used and addressed for assessing SOC stock. In chapter III, other techniques were considered such as universal kriging for obtaining SOC stock maps, once environmental variables are available in the study area, others than a detailed soil map. The SOC stock predictions made from the detailed soil map or from the geostatistical approaches are validated. This validation of the different mapping approaches and evaluating the uncertainty of these mapping methods allows a comparison between maps. The results of the validation for all the SOC maps showed that the best accuracy for the prediction of SOC found in our study area was the use of the detailed soil map, ranging from 25 to 45.1%, depending upon the depth considered. These validation results are in line to other studies reported by Meersmans *et al.* (2008) and Schulp *et al.* (2013). There are several possibilities that potentially could improve the prediction of SOC stocks, as by intensifying the number of sampling points, or through the application of other approaches such as random forest model, or the use of a higher resolution of the predictor parameters would probably result in a more accurate modelling of SOC.

In chapter IV, a conservative approach is to use MEDALUS and RUSLE to look at trends in desertification and erosion vulnerability at particular locations. The user would thus have an indication of relative changes in desertification and soil loss at a particular site. The absolute values of the estimates thus become less important as the emphasis shifts to trends of degradation or improvement. However, these types of approaches may constrain the spatial extrapolation of site estimates. Sometimes it is necessary to introduce some modifications to the models, in view of the physical and socio-economic characteristics peculiar to the region. Especially, the MEDALUS model, would be more adequate for the studied area if it incorporated two new indices, one related to the land use change that had occurred during the last 20 years in the study area and another related to vulnerability to burning of various species. Also, some changes mainly to the assignment of scores to some of the variables and a different definition of socio-economic type indicators according to those actually occurring. This concern was positive to other authors, as Ladisa *et al.,* (2012), Enne and Zucca (2000). Further studies on RUSLE accuracy using field data are proposed. This field data collection is important for status and trends in erosion to be determined for working at local scale. The number of sample sites used for erosion estimation must be statistically valid in order to minimize variability and that allow for spatial extrapolation with a high level of confidence. The implementation of the RUSLE model will require a commitment of time, technical knowledge, and field application to produce defensible results

Overall, our results confirmed that the study of these indicators proved useful in the determination of soil status in the two study areas. In general, despite its importance for our society, and unlike air and water, there is no EU legislation specifically targeting the protection of soil. Emerging regulatory requirements on soil protection demand a holistic view on soil quality. This view holds that not only contaminant concentration, but also physical, biological and other chemical soil quality aspects should be considered, as it is reported in this thesis. These results of this thesis confirm that there is a clear need for soil protection and there are effective ways to map soil quality, and continued work on focused on soil quality and functions will open new paths for a European Soil Directive to be considered.

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Chapter 6

GENERAL CONCLUSIONS

GENERAL CONCLUSIONS

Based on the research objectives as described in Chapter 1, the following conclusions can be drawn from our study on the model areas:

- The salinisation research done provides a starting point of a methodology that can be used to prevent the risk of soil salinity for land management in Mediterranean conditions. The method used can also reveal line areas linked with discharges of saline waters or saline aquifers.

- The application of a regular grid is a simple design, however is a laborious and time-consuming when is used over a large area. The appropriate sampling strategy must be determined based on achieving adequate precision of spatial information of soil properties. Further research should be conducted to obtain the minimum optimal distance for a better sampling design in the area.

- Sensor-based geospatial EM measurements provide relevant information on within-field variability of soil salinity. Therefore, techniques for rapid determination of soil salinity based on electrical conductivity were assessed and proved to be satisfactory.

- The measurements of parameters such as EC1:5, ECe, ECa could be used as indicators for soil salinity. These allow both consultants and determining authorities to better predict appropriate design for sustainable practices and management of the hydrologic cycle.

- The study shows that there is a spatial and temporal variability of salinity and sodicity after 12 year. Applying this methodology with electromagnetic sensor (ECe-ECh-EM), the extent of saline soils has declined by 22% with respect to 2007 and 1994-96. While soil salinity has declined after 12 years, soil sodicity is increasing to some extent. Comparing the trend of all the splines at depth, graphs show that soil salinity in depth in 1994-1996 was higher than in 2007 and 2008, related mainly to climate, irrigation water management and river flow discharges.

- The salinity maps developed could be used to delineate potential saline areas and to locate additional soil sampling sites for a deeper characterization of saline and/or sodic-saline conditions and the subsequent potential deterioration of soil physical properties.

- Despite the climate change, in this time period of 12 years, there is a decrease of soil salinity in the Ebro delta, probably caused by changes in irrigation management and regulation of the Ebro river through dams. This means that the management of irrigation and the river still has leeway to compensate the possible concentration of salts caused by increased evapotranspiration of the Ebro delta by e.g. a climate change. Further work is recommended in focusing on the effect of rising sea levels, and how may this affects to the evolution of the delta and the soil salinity in this area.

- Land use and altitude describe part of the variations in SOC at different depths in the study area. Soils under cropland use (62.57 Mg/ha) had less SOC than grazing (88.53 Mg/ha) or forest (116.33 Mg/ha) soils, down to 1 m depth. Forest soils are the ones that store more SOC in depth than soils under cropland use.

- Notably, land use is a strong factor affecting SOC distribution in space and in depth, because land use type significantly alters the vertical distribution. Soil depth is important in terms of SOC stock, because in the topsoil is stored 50% of the SOC, however, in the middle 60 cm (20-80 cm), ranges from 49% to 53%, depending on cropland, forest or grazing use. The spatial variability of these stocks in depth is very useful information for the assessment of the soil resilience and soil quality.

- A good management of the cropland such as soil-friendly practices should maintain SOC, however, a conversion of crop land to pasture –which happened in the past in the area- could cause a substantial C accumulation below 1-m depth. Moreover, afforestation of pasture by pines could increase SOC and would provide protective cover in vulnerable, steep and mountainous areas.

- The use of a detailed soil map for mapping SOC shows satisfactory results, when it is compared with other digital mapping methods. Moreover, it illustrates SOC differences between soil mapping units. However, not many countries have available detailed soil maps, and the necessary data to calculate SOC. Consequently, only detailed soil maps should be used for this purpose. We therefore considered other techniques such as universal kriging relevant for obtaining SOC stock maps, when environmental variables are available in the study area, others than a detailed soil map.

- Climate data at detailed scale (1:25,000) are not particularly relevant because the spatial variability is reduced, and thus do not contribute significantly to the prediction. Unsurprisingly, other parameters such as physiographic characteristics and soil cover are relevant to the desertification and erosion process.

- MEDALUS model predicted that 80% of the study area was sensitive and affected by desertification. The Critical ESAs are 17.7% of the area. These are the most degraded areas due to soil management (abandoned land) or extreme topography without vegetation (very steep slope with bare soil). Potential ESA class to desertification may require either protection from erosion and special management such as ongoing conservation agriculture.

- The MEDALUS model can assess the extent, intensity and severity of desertification process in the target area. However, there is scope for improvements. It uses socio-economic factors that are quantified through scoring, which are not always as objective as the terrain and soil factors. It should also be more flexible, giving the opportunity to add new parameters into the model, such as the slope length and steepness factors of the RUSLE that have more importance when working at detailed scales.

- The level of importance of the parameters used for land evaluation depended of the precision at which these were mapped. Unsurprisingly, land use and relief attributes appear to be more relevant than other attributes, such as soil type or economic factors. In part, this is related to the fact that they have been mapped at finer spatial resolution.

- The selection of the right method is less significant than the use of the right data at the proper scale. Therefore, the knowledge and the understanding of the hydrological and soil processes and the use of detailed data representing these processes, is more relevant than the proper method applied.

- The soil quality indices studied are good for diagnoses, however they are still far from determining the best management practices to improve soil quality. This is due to the fact that it is difficult to determine the causal relationships between the factor indices, partly because the factors used in the models for developing the soil quality indices, are often not independent. The in-built biases that can occur when indices are combined within certain modelling applications must be considered when interpreting the results.

- The selection of the soil quality indicators will vary depending on the socio-economic and environmental interests of the area. In both study areas the important linkage between the preservation of soil quality on one hand, and the achievements of sustainable agriculture and the conservation management practices on the other, are noted.

- Overall, our results confirmed that the study of these indicators proved useful in the determination of soil status in the two study areas. The results confirm that when we have to care about soil protection, there are effective ways to map soil quality. Continued work on focused on mapping soil quality and functions will open new paths for a European Soil Directive to be considered.