Agricultural Land Salinization in coastal area Soil and Water Analysis From Landscape to Soil Aggregate

Aplena Elen Bless

THÈSE POUR OBTENIR LE GRADE DE DOCTEUR DE L'UNIVERSITÉ DE MONTPELLIER

En Sciences du Sol

École doctorale GAïA

Unité de recherche UMR LISAH
Laboratoire d'Etude des Interactions entre Sol-Agrosystème- Hydrosystème

Agricultural Land Salinization in Coastal Area : Soil and Water Analysis from Landscape to Soil Aggregate

Présentée par Aplena Elen Siane BLESS
Le 4 juillet 2019

Sous la direction de Stéphane FOLLAIN
et François COLIN

Devant le jury composé de

Catherine HENAULT, Directeur de recherche INRA, UMR Agroécologie - Dijon	Rapporteur
Emmanuelle VAUDOUR, Maître de conférence HDR AgroParisTech, UMR ECOSYS - Grignon	Rapporteur
Maritxu GUIRESSE, Professeur INP-ENSAT, UMR EcoLab - Toulouse	Examinateur
Philippe CATTAN, Chercheur HDR CIRAD, UR GECO – Montpellier	Examinateur
Didier MICHOT, Maître de conférence AgroCampus-Ouest, UMR SAS – Rennes	Examinateur
Stéphane FOLLAIN, Professeur AgroSup Dijon, UMR Agroécologie – Dijon	Directeur de Thèse
François COLIN, Professeur Montpellier SupAgro, UMR LISAH – Montpellier	Co-Directeur de Thèse

UNIVERSITÉ DE MONTPELLIER

Montpellier SupAgro

RESUME

Au niveau mondial, les processus de salinisation des eaux et des sols constituent un processus majeur de dégradation des terres arables, mettant en péril la durabilité des systèmes de production agricole, la qualité de l'environnement ainsi que la qualité de vie humaine (CEC, 2006 ; Rengasamy, 2006B, Szabolcs, 1989). Dans le même temps, les zones sodiques et salines telles que les estuaires, constituent des hot-spot de biodiversité qu'il est indispensable de préserver. Ces milieux abritent de nombreuses interactions eau-sol-plantes, zones cultivées-zones naturelles-zones aménagées et doivent donc être considérés comme des systèmes complexes qu'il est nécessaire d'étudier afin de les préserver. Ici, la préservation est vue comme la capacité à pérenniser les différentes fonctions et services écosystémiques associés à ces environnements.

Le travail de thèse se positionne dans le cadre global de la compréhension de l'évolution des terres agricoles de marges côtières, soumises aux processus de salinisation. L'objectif principal est de contribuer à la production de connaissances sur le fonctionnement des ces systèmes agricoles de marge côtière, en vue de proposer des stratégies de remédiation durables. Pour cela nous avons adopté une démarche en plusieurs étapes, fondée sur l'observation et l'analyse du système à différents niveaux d'organisation : du paysage (périmètre de production) à l'agrégat de sol (élément fonctionnel de base). Dans cette démarche, l'observation au niveau du paysage nous a permis de quantifier la pression saline, d'identifier les sources de sel et de préciser la nature des processus responsables de l'augmentation des concentrations en sel dans la zone racinaire, de fait préjudiciables à la production agricole. Par ailleurs, cette étape nous a permis d'identifier des parcelles de référence, présentant différents usages et propres à traduire différents fonctionnements hydro-pédologiques. Sur ces parcelles, l'analyse morpho-structurale et géochimique fine des profils de sol ainsi que le suivi des niveaux piézométriques nous a permis de tester la robustesse de la stabilité structurale comme indicateur de la qualité des sols ou plus exactement comme indicateur de la capacité des sols à assurer leurs fonctions. Enfin, de manière exploratoire, nous avons confronté, pour des conditions contrastées d'occupation des sols et de salinité, les caractéristiques physico-chimiques et l'activité microbiologique des sols, abordée de manière globale au travers de mesures de respiration associées à différents substrats.

De manière spécifique, nous avons mené notre étude sur la commune de Sérignan (Hérault – Occitanie - France) en aval du bassin du fleuve Orb. Depuis une décennie, les viticulteurs de Sérignan observent une diminution problématique de leurs rendements viticoles ainsi qu'un dépérissement accentué de leurs vignes. Nos travaux ont donc été initiés à partir de l'hypothèse formulée par les viticulteurs de la cave coopérative de Sérignan qui identifiaient « le sel » comme facteur principal de leurs problèmes de production. La salinisation, comme variation de sel dans le système au cours du temps et dans l'espace, est imputée aux changements d'ordre climatiques, direct et indirects, intervenus ces dernières années.

Les premiers travaux, menés en collaboration avec les viticulteurs de Sérignan ont débuté en 2012, soit avant le début de la thèse (2015). Un des premiers chantiers de la thèse fut donc de rassembler et de consolider dans une base de donnée unifiée l'ensemble des résultats acquis sur la période 2012-2015. Cette première action d'inventaire a permis de construire un système d'information géographique (SIG) sur le périmètre de la zone d'étude de Sérignan.

Quantification de la salinité des eaux et des sols à l'échelon du paysage

Le SIG constitué en début de thèse fut le point de départ des travaux à l'échelon du paysage. Dans ces travaux, l'hypothèse de départ était celle proposée par Pisinaras et al. (2010), Chernousenko et al. (2011) et Gkiougkis et al. (2015) : les processus de salinisation sont sous le double contrôle des conditions naturelles (salinisation primaire) et des activités humaines (salinisation secondaire) qui concurrent à la construction du paysage physique ; l'analyse de la dynamique des sels doit donc intégrer l'évolution au cours du temps les composantes naturelles et anthropiques du paysage physique. Les investigations avaient donc pour objectif de caractériser l'état du système à différents temps (analyse chrono-paysagère), afin d'identifier les ruptures d'équilibres, potentiellement responsables de l'augmentation des concentrations en sel dans la zone racinaire.

Pour autoriser une étude sur une étendue spatiale importante, à haute densité d'échantillonnage, il nous fallait adopter des méthodes de caractérisations faciles d'accès, techniquement et financièrement, et autorisant un haut débit d'analyse. Nous avons donc opté pour des méthodes fondées sur la mesure de la conductivité électrique des sols qui mettent à profit la relation de

Richards (1974). Une première étape a consisté à vérifier localement la robustesse de cette relation entre la mesure de conductivité des sols mesurée sur extrait de pâte saturée (EC_{sp}) et la quantité totale d'éléments dissous (parmi lesquels Na^+ et Cl^- sont prépondérants) (Fig 3.2). La deuxième étape a consisté à utiliser la corrélation entre cette mesure standardisée (EC_{sp}) et la mesure plus simple de conductivité sur des dilutions à l'eau, dites 1:5 ($EC_{1/5}$).

Les résultats des prospections de terrain montrent que les valeurs de conductivité électrique actuelle de eaux (EC_w) et des sols ($EC_{1/5}$) sont spatialement structurées, avec des conductivités décroissantes pour des distances d'éloignement croissantes à la mer Méditerranée et au fleuve Orb (Fig. 3.3). Par ailleurs, à cette structuration spatiale se surimpose une stratification des conductivités (EC_w et $EC_{1/5}$) avec la profondeur de sol.

Une observation fine de la distribution spatiale de la conductivité montre que EC_w possède une structuration plus marquée que $EC_{1/5}$. En effet, le niveau de salinité mesuré en différents points et pour différentes qualités des eaux montre une décroissance de la salinité des eaux de mer vers les eaux de la zone saturée selon le gradient suivant :

Mer > exutoire du réseau de drainage > fossés > fleuve Orb > zone saturée du sol.

Aussi, nous avons posé l'hypothèse d'un transfert de sels depuis la mer Méditerranée vers le fleuve Orb (intrusion marine) puis d'une d'intrusion au niveau des berges et d'un mélange avec l'eau de la zone saturée du sol.

Afin de rassembler nos observations et hypothèses dans un formalisme propre à représenter le processus de salinisation, nous avons construit un modèle conceptuel des flux d'eau entre les différents compartiments constitutifs de ce système complexe. (Fig. 6.2). Trois de ces compartiments ont subi des évolutions significatives sur la période 1962-2012 : la structuration et l'occupation du paysage, le fleuve Orb et son régime et le climat.

- Concernant le paysage, la transition principale fut celle d'une augmentation de la surface des parcelles agricoles, responsable d'une diminution drastique de la densité du réseau de drainage. Par le passé, suite aux phases de submersion des parcelles, l'importante densité

du réseau de drainage permettait la collecte et l'évacuation en dehors du système des eaux de percolation et des sels lixiviés, tout en garantissant un abattement substantiel du niveau de la zone saturée. Suite au redimensionnement de 1962, puis de la baisse d'entretien des fossés, la fonction d'exportation des sels s'est amoindrie et a créé les conditions favorables à une remontée des sels i) via une remontée directe de la zone saturée, ii) via une augmentation des remontées capillaires vers la surface du sol, dans la zone racinaire.

- Concernant le fleuve Orb, l'analyse des chroniques de débits montre une diminution des débits minima mensuels (QMNA) sur les 50 dernières années avec une stabilisation au niveau le bas depuis la fin des années 1980. Cette tendance sur les débits minima mensuels est probablement imputable à la diminution de la pluviométrie et plus certainement à l'augmentation croissante puis la régulation des prélèvements humains sur le même temps. Quoi qu'il en soit, ces variations ont été par la suite propices aux intrusions marines dans le fleuve côtier.
- Concernant le climat, l'analyse des chroniques montre une nette variation des températures moyennes annuelles ; cette augmentation étant favorable à une augmentation de l'évapotranspiration de référence. La tendance pour la pluviométrie est moins évidente, même si elle semble être celle d'une diminution de la pluviométrie sur les 50 ans. Sur les 50 dernières années, ces tendances climatiques ont été responsables d'une augmentation significative du déficit hydrique climatique pour les vignes au moment de la période estivale pouvant provoquer un assèchement accentué et d'une remontée par capillarité et accumulation des sels dans le profil de sol et la zone racinaire.

Analyse morpho-structurale, géochimique et dynamique des nappes à l'échelon des parcelles

La stabilité structurale des sols est souvent présentée comme un indicateur robuste d'évaluation de la qualité des sols. Un grand nombre de recherches ont d'ailleurs porté sur la mise en évidence du potentiel d'agrégation (*via* la mesure de la stabilité structurale) en relation aux propriétés intrinsèques des sol potentiellement explicatives de ce potentiel (Le Bissonnais et al., 2018; Regelink et al., 2015; Six et al., 2004; Totsche et al., 2018). Par contre, beaucoup d'entre elles ont été réalisé dans des contextes de sol non dégradés et plus encore sur les horizons de surface. De fait, peu de références sont aujourd'hui disponibles quant à la capacité d'agrégation des sols salins

et sodiques, qui plus est, sur l'ensemble du profil de sol, depuis les horizons organo-minéraux de surface jusqu'aux horizons minéraux profonds. Cette évaluation globale à l'échelon du profil de sol semble pertinente i) pour juger de la capacité globale de percolation de l'eau et de lixiviation des sels, et ii) pour intégrer dans une phase de diagnostic l'ensemble du volume racinaire prospecté par la vigne. L'objectif de cette section était donc de mener une étude détaillée de la stratification des propriétés pédologiques et de la dynamique de la zone saturée à l'échelle des profils de sol, puis d'analyser les relations entre ces propriétés et la stabilité structurale par horizons.

En pratique, nous avons travaillé sur 6 parcelles, variables au regard de leur historique d'occupations ou de leurs mode de conduite : 2 vignes palissées, 2 vignes en gobelets et 2 parcelles en friche. Pour chacune d'elles nous avons opéré sur des fosses pédologiques (ouvertes à la pelle mécanique offrant une surface d'étude de 2,50 m de large sur 1,50 m de profondeur) et effectué un échantillonnage stratifié suivant la nature des horizons pédologiques, depuis la surface jusqu'à 1,40 m de profondeur.

Concernant l'horizonation (Figure 4.2), l'ensemble des fosses pédologiques présente des sols peu différenciées, carbonatés, développés à la faveur d'un matériau alluvial non induré. Dans ces profils, la géométrie des horizons organo-minéraux et sous forte dépendance des différentes profondeurs de travail des sols viticoles. Trois des 6 profils présentent une hydromorphie de profondeur marquée. Les résultats des tests de stabilité structurale indiquent des valeurs de MWD comprises entre 0,28 et 1,10 mm, associées à un MWD moyen de 0,52 mm. Selon la catégorisation proposée par Le Bissonnais (1996, 2016), ces agrégats doivent être considérés comme non stables. Par ailleurs, quelles que soient les fosses considérées, les MWD sont variables en surface (horizons organo-minéraux), puis diminuent en valeur et en variabilité intra-profondeur de la surface vers la profondeur, pour atteindre un minimum de 0,4 mm dans les horizons minéraux (vers 0,80 m). L'analyse corrélative entre ces valeurs de MWD et les paramètres chimiques et agronomiques montre que les stabilités des horizons organo-minéraux semblent plus dépendantes de l'occupation des sols, de la teneur en matière organique et de l'importance de la fraction argileuse alors que vers la profondeur, les valeurs de stabilité deviennent plus corrélées à la teneur en sels, dont l'abondance est fortement liée à la présence de la zone saturée sous-jacente.

Dans ce contexte, il semble pertinent de conseiller des apports ou restitution en matière organique afin de restaurer le potentiel d'agrégation des horizons de surface. Par ailleurs, l'analyse détaillée

des teneurs en Na^+ et Cl^- suggère que les sels sont précipités dans le volume poral, plutôt qu'adsorbés (cas de Na^+) sur le complexe d'échange cationique. Par conséquent, il existe donc un bon potentiel à la « désalinisation », sans nécessité de désorption, fondé sur l'apport d'eau en surface, la mise en solution des sols puis la lixiviation et l'exportation des solutés *via* le réseau de drainage (*Figure 4.7.*).

Facteurs d'agrégation des sols : approche bio-physico-chimique

Les résultats précédents nous informent que la stabilité structurale des sols peut être un indicateur pertinent pour juger de la capacité de remédiation des sols salins de Sérignan. Cependant, une part significative des variabilités observées (MWD) n'est pas expliquée par les paramètres physico-chimiques de base. Une hypothèse probable est que l'approche précédente ne permettait pas d'intégrer le potentiel d'agrégation lié à l'activité microbiologique des sols. C'est pourquoi nous avons engagé en dernière approche une étude prospective visant à qualifier le potentiel de minéralisation des sols de Sérignan.

La capacité d'agrégation des sols est sous forte dépendance des micro-organismes du sol *via* leur activité de décomposition des matières organiques et la production associée de métabolites. L'objectif de ce travail était d'identifier de potentielles variabilités microbiologiques parmi les 6 parcelles étudiées. Là, notre intérêt ne s'est pas porté sur l'identification des organismes et/ou des communautés en présence, mais sur une évaluation globale de la fonction de potentiel de minéralisation des matières organiques à partir de mesures de respiration en conditions contrôlées de laboratoire à l'aide des dispositifs de type MicroResp^{TM}.

Les sols traités, proviennent des 6 fosses pédologiques présentées précédemment. Considérant la dimension exploratoire de ces travaux, nous avons décidé de travailler exclusivement sur des prélèvements de surface (composites entre 0 et 15 cm), collectés au moment des prélèvements pour les analyses physico-chimiques. Dans cette démarche, nous avons exposé les microorganismes à 28 substrats différents :

- 13 carbohydrates (D-mannose, D-mannitol, D-trehalose, L-arabinose, D-xylose, D-sucrose, D-galactose, meso-inositol, D-sorbitol, L- rhamnose, L-arabitol, meso-erythrol, D-glucose,);

- 5 acides carboxyliques (citric acid, malic acid, DL-maleic acid, Na-gluconate, L-glutamic acid sodium);
- 10 acides aminés (L-leucine, L-asparagine, D,L-valine, L-methionine, L-glutamine, L-alanine, N-acetyl-glucosamine, L-serine, D,L-histidine, L-proline).

Nous avons alors mené une démarche d'analyse corrélative entre propriétés physico-chimiques, stabilité structurale et activités microbiologiques potentielles sur les différents substrats.

En premier lieu, nous avons montré que pour les horizons organo-minéraux, les stabilités structurales (MWD) étaient significativement plus élevées en conditions de friches qu'en condition de vignoble (Table 5.1). Dans le même temps, ces conditions de friches présentaient les valeurs les plus élevées pour la conductivité électrique (5.09 dS m^{-1}). Là, une explication possible était celle d'une stabilité plus importante en relation à la teneur en matière organique plus élevée en surface des sols en friches (Table 5.1), ainsi qu'à des teneurs plus élevées en carbone labile, azote total, magnésium et potassium.

Cette explication est cohérente avec l'observation d'une plus grande diversité des matières organiques restituée sur ces espaces et des couvertures végétales dominées par des espèces induisant des rhizodépositions, connues pour leur effet agrégeant (Nguyen, 2003). De plus, certaines études (Mikha and Rice, 2004) suggèrent que la fraction labile du carbone organique et l'azote total augmentent le potentiel de macro-agrégation en particulier pour les sols non travaillés avec restitution de fumures. Enfin, dans les conditions de friches ont montré des teneurs plus faibles en Ca^{2+}, connu pour son pouvoir bloquant sur les minéralisations, et des teneurs plus importantes en Mg^{2+} constituant un facteur de floculation essentiel (Rengasamy et al., 1986).

En conclusion, nous avons proposé plusieurs solutions de remédiation au problème de salinisation dans la zone d'étude à partir d'une liste générale d'actions adaptées aux processus mis à jour. Plusieurs seraient à activer simultanément et concernent les différents acteurs présents. Un premier levier concerne la gestion du fleuve Orb. Puisqu'il constitue l'entrée majeure de sel dans le système, il serait important d'analyser s'il est possible d'augmenter ses débits d'étiages ou d'aménager son lit afin de limiter les intrusions marines. Le second levier consiste à augmenter

l'efficience des pratiques de lutte anti-sels par le raisonnement des submersions et l'amélioration des conditions de drainage (densité et entretien des ouvrages). Le troisième levier concerne la gestion des sols par les agriculteurs. La maîtrise des submersions est première mais contrainte par la disponibilité et le prix de l'eau qu'il conviendrait de négocier avec l'entreprise de distribution BRL. Les pratiques de travail du sol et les logiques d'intervention dans les parcelles doivent être raisonnée afin de limiter le tassement et la compaction des sols néfaste à la percolation et l'élimination des sels. La restauration de la structure du sol afin de garantir un meilleur fonctionnement apparaît impérative dans de nombreuses situations. Enfin, l'étude de l'adaptation variétale en vigne afin de disposer de porte-greffes plus tolérants au sel nous apparaît comme une perspective à investiguer.

Contents

LIST OF ABBREVIATION

Abbreviation (A – Z)	
AEM	Airborne Electromagnetic
AMD	Annual Mean Discharge
AMF	Arbuscular Mycorrhizal Fungi
ASL	Above Sea Level
CEC	Cation Exchange Capacity
DDL	Diffused Double Layer
EC	Electrical Conductivity
$EC_{1/5}$	Soil Electrical Conductivity with soil solution ratio 1:5
EC_a	Apparent Electrical Conductivity
EC_{sp}	Soil Electrical Conductivity of Saturated Paste
EC_w	Water Electrical Conductivity
EM	Electromagnetic
ESI	Exchangeable Sodium Index
ESP	Exchangeable Sodium Percentage
ET0	Evapotranspiration reference
FAO	Food Agricultural Organization
ISO	International Standard Organization
LMD	Low Monthly Discharge
MWD	Mean Weight Diameter
MicroResp™	Microplate-based Respiration System
POC	Particulate Organic Carbon
SAR	Sodium Adsorption Ratio
SDGs	Sustainable Development Goals
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
TDI	Total Dissolved Ions
TDS	Total Dissolved Solids
WRB	World Reference Base
WHC	Water Holding Capacity

LIST OF PARAMETERS

Parameters	Unit
Salinity	
$EC_{1/5}$	dS m^{-1}
EC_{sp}	dS m^{-1}
EC_{w}	dS m^{-1}
ESP	%
SAR	%
Soluble salts in water	
Na^{+}, Ca^{2+}, Mg^{2+}, Cl^{-},HCO_3^{-}, SO_4^{2-}	mmol/L
Soluble salts in soil	
Na^{+}, Ca^{2+}, Mg^{2+}, Cl^{-}, HCO_3^{-} SO_4^{2-}	meq/100 g
Climate and Hydrology	
ET0	mm
Annual Temperature (T)	°C
Annual Rainfall	mm
LMD	m^3s^{-1}
AMD	m^3s^{-1}
Soil Quality	
MWD	mm
SOC	% or g/kg
SOM	% or g/kg
pH	
CEC	meq/100 g or cmol/kg
BD	g/cm-3
Soil texture	%
N-Total	%
Total Carbonate	g/kg
pi	%

LIST OF FIGURES

LIST OF TABLES

DECLARATION

I declare that the thesis contains no materials, which has been accepted for any others degree or diploma in other universities and based on my knowledge, it contains no material previously published or written by other person, except where references have been put in the text.

Aplena Elen Siane BLESS

CHAPTER 1

STATE OF THE ART

1.1. SALT AFFECTED SOILS

Salt affected soil is a general term that used for saline, sodic, and saline-sodic soils. These soils embrace high soluble salts contents appeared in various environmental conditions and possess different morphological, physical, chemical or biological properties (Szabolcs, 1989).
Soil salinization is defined as increasing and accumulation of soluble salt ions (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, SO_4^{2-}, HCO_3^{2-}, Cl^{-} and NO_3^{-}) in the soil or soil water that adversely affect agricultural production, environmental quality and economic welfare (Rengasamy, 2006a; Tóth et al., 2008; William, 1987). Salinity level is determined by the total dissolved salts (TDS in kg l^{-1}) or the electrical conductivity (EC in dS m^{-1}) measured in water-soluble electrolyte in soil (ISO 11265:1994) or in water (ISO 7888:1985).

The US salinity laboratory categorizes saline soil with EC_{sp} (electrical conductivity extracted from saturated paste) value > 4 dS m^{-1}, soil pH value less than 8.5 and exchangeable-sodium percentage (ESP) less than 15%. While for sodic soils, it has EC_{sp} < 4 dS m^{-1}, soil pH > 8.5 and ESP > 15%. Another categorization consists in combination of these two types, which is saline-sodic soil (EC_{sp} > 4 dS m^{-1}, soil pH < 8.5 and ESP >15%) (Rhoades et al., 1999). The ESP and SAR (sodium adsorption ratio) are two important indicators that determined the sodium percentage in soil and water respectively. The exchangeable sodium percentage (ESP) of soil was calculated from ratio between Na^{+} that measured from soil saturation extract and Cation Exchange Capacity (CEC) (U.S. Salinity Laboratory Staff, 1954) as eq.1.

$$ESP = \frac{[Na^{+}]\ (meq/100g)}{CEC\ (meq/100g]} \times 100\ \% \qquad \text{(eq. 1)}$$

- *ESP : Exchangeable Sodium Percentage (%)*
- *$[Na^{+}]$: Sodium that extracted from saturation extract (meq/100 g)*
- *CEC : Cation Exchange Capacity (meq/100 g)*

The sodium adsorption ratio (SAR) was calculated from surface or groundwater samples as eq. 2.

$$SAR = \frac{[Na^+]\ (meq\ L{-}1)}{\sqrt{([Ca^{2+}]\ [meq\ L{-}1]+[Mg^{2+}]\ (meq\ L{-}1))/2}} \qquad \text{(eq. 2)}$$

- *SAR : Sodium Adsorption Rasio (%)*
- *Na^+ : Sodium concentration in water (meq L^{-1})*
- *Ca^{2+} : Calcium concentration in water (meq L^{-1})*
- *Mg^{2+} : Magnesium concentration in water (meq L^{-1})*

Salinization process, the mobility of mixing salt and water in soil, is seasonally variable and complex with the interaction to groundwater. In our context, the groundwater integrates unsaturated zone, where immediately below the land surface, contain water and air in the soil pores. There are two pathways of salinization. Firstly, it appears on natural process called primary salinization. For instance, physical and chemical weathering that transporting salt from parent material, sea water intrusion, fossil salts and atmospheric salts (Herczeg et al., 2001). The second pathway is caused by human intervention called secondary salinization. For instance, salt in irrigation water, salt in fertilizer, application of salts on road during winter time and over use of a water body that lead to decreasing of groundwater and intrusion of sea water. In addition, land salinization would be emphasized by the change of land use and land development (Payen et al, 2016).

Salt fluxes are mainly associate to water fluxes, as water is certainly the main media for transferring salt into the soil. Thus salinity in various water sources (irrigation, aquifer, river) may indirectly control variation of salt stocks per unit of time in soil body. For example, river salinization in five major basins in the arid zone (Colorado, Indus, Murray, Nile and Aral) has influence on the agricultural production in those areas (Smedema, 2010). Salt river absorption, understood as a salt load introduced into river water, determine salinity in the river profile. Water pumping for agricultural use (irrigation), industries and municipality disturbs the river flows, which are the primary factor in river salinity (Smedema, 2010). Salinity of the river coupled with complex and dynamic function of the climate (actual and past), geology of catchment area, distance to sea, topography and vegetation are the key factors (Cañedo-Argüelles et al., 2013). The mixing of fresh groundwater and inherited salty water is one factor of salinization in the aquifer and

riverbank infiltration at the coastal area (Cary et al., 2015; Petelet-Giraud et al., 2016). Salts in groundwater and saturated zone are transferred to soil through capillary rise during the dry season with high evaporation and through increasing of groundwater during wet season. The salts would be crystalized and accumulated on soil surface and sub soil (Rengasamy, 2006a).

Soil salinization is a global problem that severely affects the arid and semi-arid land. According to FAO in 2000, the total salt affected areas are approximately 830 million ha globally (Martinez-Beltran and Manzur, 2005). In Europe particularly costal southern Europe, intensive groundwater abstraction has led to sea water intrusion (Daliakopoulos et al., 2016). Around 3.8 million ha lands in Europe are effected by salts and it generally occurred in the Mediterranean basin (Italy, Spain, Hungary, Portugal, France, Slovakia, Austria) (European Commission, 2006). Based on the map of actual and potential salinization in Europe (Figure 1.1.), soil salinization in these Mediterranean basins are mainly driven by intrusion of sea water (Daliakopoulos et al., 2016).

1.2. SALINIZATION IN COASTAL DELTA

Estuary is generally known as meeting of fresh water (river) and sea water. The definition of estuary by Pritchard (1967) is “a semi-enclosed coastal body of water which has a free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage”. The formation of estuarine wetland ecosystems is often from alluvial river deltas due to water retention and fine nutrient rich sediment transported by different streams on the plane topography. The delta areas are favorable for agricultural land use, settlement, tourism and industry, as it offers magnificent prospect for economic development. In the Mediterranean coastal area, there is a rapid increase of population growth especially in the summer time. This development, however, threat the water sources of Mediterranean basin which mainly originated from alluvial, karstic carbonate or sedimentary aquifers (Aureli et al., 2008). Over use of water sources in the aquifer for irrigation, house hold, tourism and industries lead to water deficit (Custodio, 2002). Moreover, with the typical Mediterranean climate, low rainfall intensity and increasing temperature resulted in less water recharge to the aquifer which favorable for sea water intrusion to the aquifer and river and induced salinization process in water and soil (Wöppelmann G. and Marcos M., 2012).

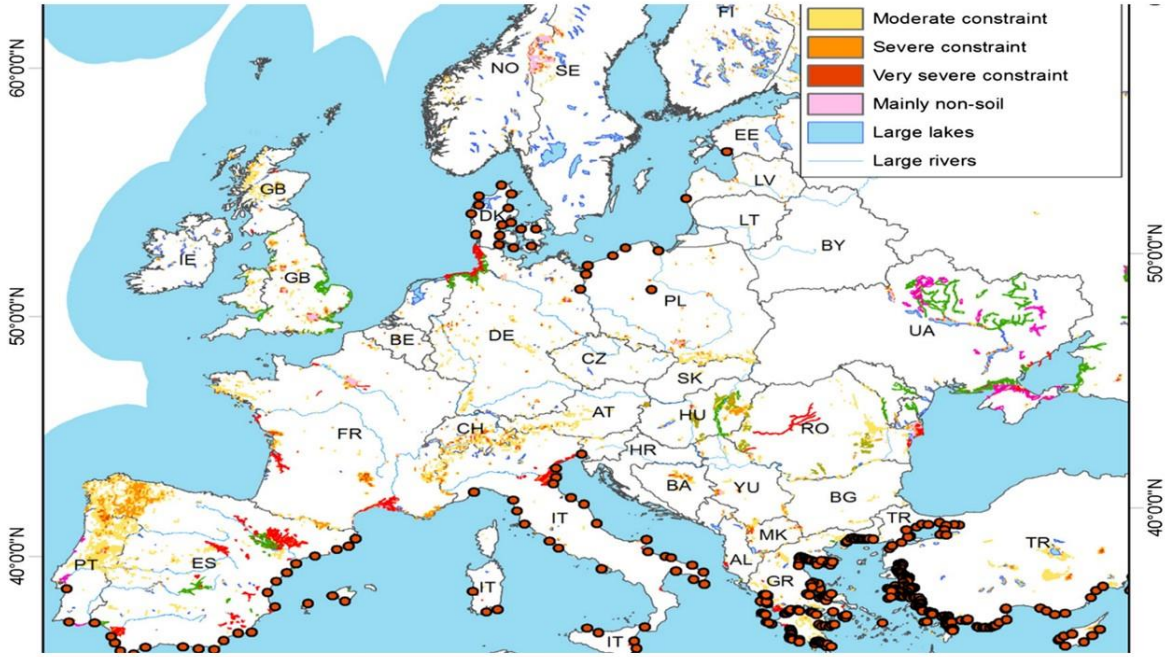


Figure 1.1. Map of Saline and Sodic Soils in the European Union : Status and Potentials in the coastal area of southern Europe. Image use from Daliakopoulos et al., (2016)

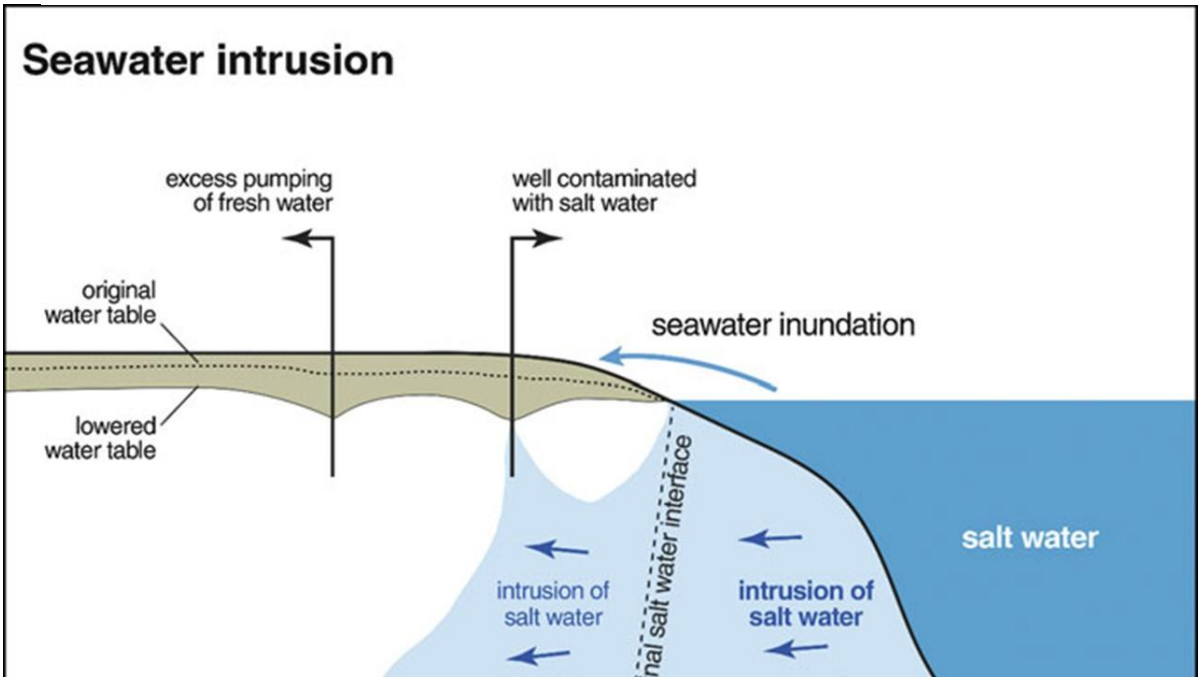


Figure 1.2. Process leading to salinization of fresh groundwater in coastal aquifers due to the intrusion of salt water from the sea. Image use from Green et al., (2016).

At the normal hydraulic equilibrium, the original interface of fresh and saltwater is located just below coastal plain (Figure 1.2.). Yet excessive use of freshwater abstraction and pumping reduces the hydraulic head of inland groundwater, so it will allow the seawater to intrude further inland and salinizing the landscape. Groundwater extraction at rates exceeding up-stream recharge by freshwater allows the interface to progress inland and locally may cause increase upwards and landward flow of saline seawater (Greene et al., 2016).

Salts in the coastal area originated from mixing between actual or fossil saline groundwater and fresh water in the aquifer (Jones et al., 1999), saline marine sediment, salts in river flows, and atmospheric seawater drops on soil surface. Source of salts, transport pathways and geomorphic structures of the coastal area determine distribution and composition of salts in soil profile (Chernousenko et al., 2011). For the Mediterranean coastal aquifer, one or two of the following contribution induces salinization:

- current seawater intrusion and mixing of saline-fresh water (Pulido-Leboeuf et al., 2003; Kouzana et al., 2010);
- contribution of deep saline water (Khaska et al., 2013; Petelet-Giraud et al., 2016);
- water rock interaction that evaporates (Mongelli et al., 2013).

Salt accumulation can occur in sea coast which is the main source of salts mainly connected with the sea, and the other accumulation can also happened in the delta area which the source of salts is not only from the sea but also coupled with salt from the river (Zaidel'man, 1987).

Salinization threats water and soil quality and declines agricultural production in almost all coastal delta around the globe. For instance, salinization in a coastal semi-arid region the Northen Greece, in the eastern Nestos River Delta and Western Greece in south-central Crete (Alexakis et al., 2016; Gkiougkis et al., 2015); also in the coastal region of Netherlands (Raats, 2015). Outside Europe, for instance salinity problem at vineyard in California Central Coast (Hingston and Galbraith, 1990; Battany, 2010) and the coastal area in the south west of Western Australia (Hingston and Galbraith, 1990). In the coastal of artic and pacific region of Russia (Chernousenko et al., 2011); the Nile Delta in Egypt (Kotb et al., 2000), in China at Shuangtai Estuary Wetland (Lin et al.,

2016) and Coastal zone of the Yellow River Delta (Yu et al., 2014), and Mekong Delta in Vietnam (Nguyen and Savenije, 2006).

1.3. IMPACTS OF SALINITY

High concentration of salt will disrupts soil quality and specific soil functions (Tóth et al., 2008). It affects soil structure and other physical soil properties (Pearson and Bauder, 2006). Accumulation of sodium in soil disperses soil aggregate and caused aggregate failure, so decline soil permeability. The colloidal particles from aggregate dispersion would enclosed soil pores, so reduce the hydraulic conductivity and limit the transport of colloidal associated contaminant (Kanti Sen and Khilar, 2006). These conditions favor rainfall runoff and induce soil loss due to soil erosion. Soil salinity and sodicity also reduce soil organic carbon (SOC) contents stocks (Wong et al., 2010) as the SOC pool is generally dependent from the input from the vegetation. Salt affected soil negatively impacts plant health upon SOC stocks in salt affected areas. Saline conditions restrict soil organic matter decomposition and substrate access, because of aggregate flocculation as a result of high concentration of soluble salt. In addition, sodic soils generally contain abundant carbonate content that complicate organic carbon dynamic. Salinity also influence soil microbial biomass and activities inducing change in CO_2 fluxes and nature of nutrients delivery to vegetation (Setia et al., 2013). It was estimated that around 3.47 ton of worldwide soil organic carbon (SOC) per hectare had loss from saline soil (Setia et al., 2013).

Table 1.1. Category of Soil Salinity based on the Electrical Conductivity (EC_{sp}), (Gkiougkis et al., 2015; L. A. Richards, 1954).

Salinity Category	EC_{sp} (dS m^{-1})	Salinity effects on crops
Non-saline	<2.0	Salinity effects are negligible
Slightly saline	2.0-4.0	Yields of very sensitive crops may be restricted
Moderate saline	4.0-8.0	Yields of many crops restricted
Very saline	8.0-16.0	Only tolerant crops yield satisfactory
Extremely saline	>16.0	Only view very tolerant crops yield satisfactory

Soil salinity becomes major issue in agricultural sector, because it is a limiting factor for crop production and treats sustainability of agriculture. The FAO categorized soil salinity as a threat that caused land degradation (Dubois, 2011). Soil salinity provokes harmful impacts on crop plants (Machado and Serralheiro, 2017). It inhibits evapotranspiration due to less osmotic potential of soil solution. Salt affects closure of stomata that obstruct photosynthesis process, thus reduce plant growth and vegetal biomass production, even cause mortality (Shani and Ben-Gal, 2005). Particular salt ions like chloride has function for photosynthesis and regulate stomata guard cell and osmosis process of the plants (Deinlein et al., 2014; Xu et al., 2000). When crops plants fail to maintain chloride in the root, they transfer it to the shoot, making them vulnerable on plant tissue accumulation.

The Table 1.1. shows a general categorization of soil salinity in relation to crops. Vegetables (carrot, lettuce, radish, string bean) are sensible to saline soils, whereas, perennial crops (vine, olive, fig tree) are moderately tolerant to slightly saline soils. Some plants (cotton, rapeseed, sugar beet and coconut), however, are very tolerant to moderate saline soils exhibiting electrical soil conductivity up to 8 dS m^{-1}. The assessment of plant respond towards salinity is based on yield, particularly the threshold value resulted yield loss, and rate of yield loss per increasing dS m^{-1} of soil salinity (Maas and Hoffman, 1977). For example, in every increasing of 1 dS m^{-1} soil electrical conductivity of extracted paste, resulted in reduction of 13%-14% of leaf-stem dry weight and fruit biomass of vine in Arava Israel (Shani and Ben-Gal, 2005). High concentration of Cl^- also inhibits the absorption of others nutrients like calcium and potassium. The threshold for Cl^- in the irrigation water is should be less than 150 mg l^{-1} and the EC_{sp} of soil in the range of 1-3 dS m^{-1} (Xu et al., 1999).

Soil salinity also disrupt non-soil ecological function, such as destruction of infrastructure such as water supply and transport infrastructure (Montanarella, 2007) and also devaluation of agricultural land (Schiefer et al., 2016). It was estimated that the cost of salinity due to agricultural yield lose and infrastructure damage are approximately 158-321 million euro annually in three European countries (Spain, Hungary and Bulgaria) (Montanarella, 2007). In the review paper of Daliakopoulos et al., (2016) about salinization at the European scale, they presumed that soil

salinity not only threat agricultural production, but it could also escalate social-economic conflicts due to deficient of natural resources in some arid and semi-arid areas.

1.4. SOIL SALINITY AND VINEYARD

Vineyard needs a good soil quality condition in order to produce a quality wine. In France, the term "terroir" indicates an area that has a specific soil condition and climate that could produce a distinctive wine quality. For example, the Grand Cru and Premier Cru of de Côte d'Or in Burgund and The Médoc in Bordeaux are well-known places for its wine quality (White, 2003). This indicates that soil be an important factor that determined quality of wine. A good management of soil quality is necessary to sustain vineyard productivity for several reasons. Firstly grape vine is a perennial crop that expected to grow and produce commercially in the same soil for about 40-50 years. Thus it is crucial to build a harmonies relationship between soil and vine. Secondly, soil provides nutrients and water that important for vine productivity, as it affects quantity and quality of grape berries. The last important thing is because most of pests and diseases in grapevine like phylloxera or the spore of downy mildew fungus are associated with soil, as this organisms live in soil (White, 2003). Therefore, in order to achieve sustainability of wine production, we need to maintain physical (structure), chemical and biological condition of soil properties.

Some areas around the globe encounter soil salinity problem that threaten vine production. For example, in Australian vineyard, it has dryland salinization (Rengasamy et al., 2006; Hingston and Galbraith, 1990). Salinity also affects vineyard in Israel (Shani and Ben-Gal, 2005), South Africa (Clercq et al., 2011) and in Europe like Spain (Aragües et al., 2014, Acosta et al., 2011). The accumulation of salts concentration on the root zone of grapevine adversely affects the growth and yields, even cause mortality of wine at very high concentration. High salt concentration restricts water uptake from soil due to high osmotic pressure in the root zone, so the roots work extra to absorb water. The rate of photosynthesis in grapevine reduces by increasing salinity (Downton 1977; Walker et al., 1997). When the chloride concentration in the grapevine leave is up to 150 $mMol.l^{-1}$, it would retain the capacity to recover normal physiological function. It disturbs the stomata opening and increases the resistance to CO_2 diffusion, so interrupt photosynthesis process (Walker et al., 1981). Salts that contain sodium, chloride, and boron have toxic effect on plant

metabolism. The limit of soil electrical conductivity of grapevine is 2 dS m^{-1}, some varieties even have lower threshold value because its productivity start to decline when the EC value is over 1.8 dS m^{-1} (Table 1.2). Based on the export guideline, the sodium chloride concentration in wine should be less than 1000 mg l^{-1} (Biswas et al., 2010).

Table 1.2. Grape vine varieties and its sensitivity on salinity (Bistwas et al., 2010)

Crop Sensitivity	Varieties	EC_{sp} at which yield decline starts
Sensitive to moderately sensitive	-Own roots, (Vitis Vinifera): e.g. [a]Sutana, [b]Shiraz, [b]Chardonnay -Rootstocks : [b]K51-40, [b]3309C, [b]1202C, [b]Kober 5BB, [b]Teleki 5C, [b]SO4	1.8 (dS m^{-1})
Moderately tolerant to tolerant	Rootsocks : e.g. [a]Ramsey, [a]1103 Paulsen, [b]140 Ruggeri, [b]Schwarzmann, [b]Rupestris St. George	3.3 (dS_m^{-1})

[a]Based on the yield response (Walker et al, 2002; Zhang et al., 2002); [b]Based in relative capacity for salt exclusion (Walker et al., 2004; Tregeagle et al., 2006)

1.5. SALINITY MEASUREMENT (WATER & SOIL)

The evolution of salt concentration in different water sources and soil landscape need to be monitored periodically, especially in the salt affected area, in order to predict the extent effect of salt to water and soil quality, as well as plant growth. The water salinity is measured by using a calibrated electrical conductivity meter (EC). The probe of EC meter could be put directly in water, then the monitor will display the value of water electrical conductivity (EC_w). Whereas, for soil, there are different techniques for measuring its salinity, field technique and laboratory technique (Hardie and Doyle, 2012).

1.5.1. Field Technique

The field technique is an indirect measurement of apparent electrical conductivity (EC_a) in soil at multiple locations to generate map of apparent electrical conductivity. The advantage of this

technique is it could cover big area in short period of time at minimum cost. However, it is compulsory to calibrate the result of EC_a measurement with electrical conductivity from saturated soil paste of laboratory analysis, to acquire a reliable data of soil salinity in a certain location. The most popular device of field salinity measurement is electromagnetic induction (EM). This device has transmitting coil to create a magnetic field that penetrates into the soil. The magnetic field creates an electrical current that in turn create a secondary magnetic field, which is received by the meters receiver coil. The ratio of the secondary to primary magnetic field is a linear proportion to the EC_a. Therefore, electrical conductivity of soil and water are represented by the EC_a and some soil properties like salt content, soil porosity, permeability, moisture content, soil temperature and clay content also affect the EC_a. Consequently, the interpretation of salinity using EC_a value in terms of salt content is non-unique and depends on other assumptions about soil properties. Based on some researches in Australia, the reliability of EC_a value for interpret salt content are between 75-90% of EC_a value that could explained by total soluble salt content (Spies and Woodgate, 2003).

The field technique requires access to appropriate electromagnetic devices depending of the scale and depth of the required investigation. Typical EM devices are hand held or vehicle mounted EM38, EM31, down borehole apparatus (EM39) and also airborne electromagnetic (AEM). In topsoil or within root zone, the EC_a is usually determined by EM38 that operates to a maximum depth of 1.5 m, while for subsoil, it is usually measured by EM3, or down borehole EM 39. The EM 38 and EM 31 devices may be operated manually or straddling to a vehicle with a precision GPS to enable rapid paddock of farm scale mapping. Soil salinity interpretation from EC_a maps requires calibration of EC_a values to electrical conductivity ($EC_{1:5}$; EC_{sp}) measured from different locations and soil depths and analyzed in laboratory. In addition, the EM devices may also be mounted behind fixed wing aircrafts or helicopters (AEM). Combined with other devices such as LIDAR, radiometrics or digital elevation models, AEM enables rapid catchment scale mapping of multiple soil attributes and their position on landscape. Calibration of AEM surveys requires considerable calibration data usually from down borehole induction (EM 39) or electrical depth sounding. In order to interpret AEM surveys, we need to use the statistical and image processing software. AEM has advantages over standard EM survey due to the ability to resolve spatial variation in EC_a at discrete depth intervals. Michot et al., (2013) conducted a digital assessment to

analyze the salt dynamic on the landscape scale. They suggest the landscape scale assessment of soil salinity is compulsory for remediating soil salinity problem.

1.5.2. Laboratory Technique

In laboratory, soil salinity can be determined by measuring:
- electrical conductivity (EC) of soil water extracts;
- soluble ion concentrations of soil water;
- mass of total dissolved solids (TDS), or total dissolved ions (TDI).

Electrical conductivity measurement is determination of the salt content from a soil water suspension or a soil water extract. The ability of extracted soil or suspension to conduct electricity between two metal electrodes determines the measurement. When there is a high salt content in the extracted soil or suspension, the more current is conducted between the two electrodes, resulting in higher EC values. Temperature influences electrical conductivity values, by increasing temperature the electrical conductivity will rise. The temperature should be corrected for if not measured at 25 °C.

Soil salinity ($EC_{1/5}$) is generally analyzed by measuring electrical conductivity in soil distilled water suspension with the soil solution ratio is 1:5 (soil:distilled water) following minimum 1 h of end-over-end mixing. The procedure of $EC_{1/5}$ measurement is done without using filtering. Firstly, prepare a 1:5 soil/water suspension. For example, 20 g of air dry soil in to a flask and add 100 ml deionized water. Mixing the solution mechanically in 30 sec at room temperature (25 °C) to dissolve soluble salts and let the soil to settle for minimum 1 h. The conductivity meter need to be calibrated at 25 °C using the standard solution that provided. Then dip the conductivity cell into the supernatant, by moving it slowly up and down without disturbing the settled soil. Note the EC value when the system has stabilized. Rinse the cell with deionized water between samples and remove the excess water with tissue. Finally report $EC_{1/5}$ (dS m^{-1}) at 25 °C.

While the electrical conductivity of saturated paste (EC_{sp}) is determined by extracting liquid from a saturated soil paste using centrifuge or a suction device after allowing time for the soil to

equilibrate. Then measure the conductivity of extracted liquid. The procedure is as follows. Determine soil moisture content of air dried soil by drying at oven at 105 °C. Sieve the soil to 2 mm and weight the sample for about 200-400 g. Add deionized water while mixing the ground soil sample to saturation, which is soil paste glistens, flows slightly when the container is tipped. Then allow the soil to equilibrate for at least 4 h or even more time to 8 h or one overnight if the soil has thickened or not glisten add more distilled water and mix thoroughly. Then move the soil paste to Buchner funnel fitted with highly retentive filter paper and apply vacuum to collect extract until air passes thorough the filter. Extract the soil solution by centrifuge. Store extract at 4 °C until analyzed for EC. Calibrate the EC meter following the manufacture instruction. Finally dip the conductivity cell into supernatant, moving it up and down slightly without disturbing settled soil. Take the reading with cell stationary when the system has stabilized. The general unit of electrical conductivity (EC_{sp}; $EC_{1/5}$) is dS m^{-1} that equals to mS cm^{-1} or 1000 µS cm^{-1}.

Total dissolve salts measurement (TDS) is also another way to determine soil salinity. The TDS measurement is done by evaporating a known filtered volume of water to dryness, then weighing the remaining solid residue. The incomplete filtration of clay platelets and inclusion of non-saline dissolve organic compound in evaporite can also occur, so this approach is likely to be error. There is no exact relationship between EC (µS cm^{-1}) and TDS (ppm) as the form of the salt species present influences conversion values. Yet, in water with a high proportion of sodium chloride, one can estimate TDS (ppm) by multiplying EC (µS cm^{-1}) by 0.5-0.6.
In the current research, we determined the soil salinity with electrical conductivity using laboratory technique particularly $EC_{1/5}$ for soil salinity at landscape scale as this method is appropriate for a big number of soil samples (Chapter III), while EC_{sp} for salinity at field scale (Chapter IV). We also examined water salinity by measuring electrical conductivity directly in the different water sources using EC meter (Chapter III).

As previously stated that one drawback of soil salinity is destruction of soil structure particularly dispersion of aggregate stability. Thus the following text (section 1.6.) will discourse the importance of soil aggregate stability, which is an indicator of soil quality under degraded salt affected soil. It will discuss how soil aggregate formed, stabilized and destabilized. It also discussed factors that influence those processes.

1.6. SOIL QUALITY INDICATOR OF DEGRADED SALINE SOIL

Soil quality is simply defined as "the capacity of soil to function". Three major components in soil quality concept are sustainability of biological productivity, environmental quality, and plant-animal health. There is a need to balance between soil conservation and consumption for different practices such as agricultural production, remediation of waste, urban development, forest, or recreation in order to achieve good soil quality. Karlen et al., (1997) defined soil quality in more detailed 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". Quality of soil is determined by three main key attributes; soil physical, chemical and biological properties. Thus these soil key attributes need to be considered when identifying soil quality.

Soil structural stability is a soil quality indicator for degraded salt affected soil (Le Bissonnais, 2016). Soil structure refers to the arrangement of soil particles (aggregate) and pores space designed between the aggregates (Brady and Weil, 2008). Thus soil structure consists of solid part and the empty space. Soil aggregate is related to the solid part of the soil structure, which varies in size and shape. Macro size aggregate is >250 µm, and the micro-aggregate is below this value (Figure 1.3.). Since soil aggregate consist of different hierarchy of particle sizes and pores sizes that are joint together, so soil aggregate would also be defined as the three dimension organization of matters (particles) and voids.

Soil structure can be in the form of granular, blocky or columnar, whereas unformed aggregate is known as massive (Brady and Weil, 2008). The arrangement of soil aggregates determines size, shape, and position of soil pores, that important for water retention, infiltration and percolation, gaseous exchange, soil organic matter, root penetration, and activities of soil biota. The small aggregate size between 1-0.105 mm and <0.105 mm supports water retention and positively correlated with clay and organic matter content (Boix-Fayos et al., 2001). Thus soil aggregate plays important role for soil function; particularly for biomass production, storage and filtering the water, storage and cycling nutrients, and as a platform for biological activities. Soil aggregate particularly small aggregate is also a good indicator for soil degradation.

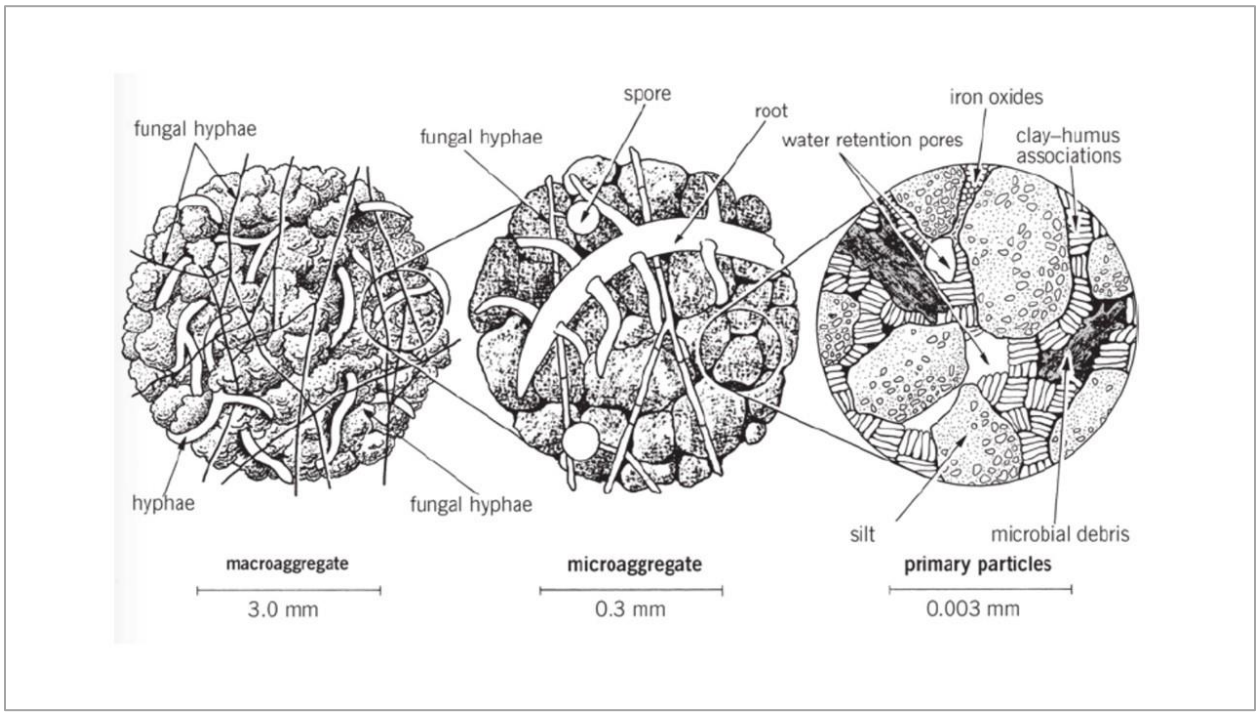


Figure 1.3. The structural hierarchy of aggregates: Macro-aggregates, micro-aggregates and primary particles (Dubbin, 2001)

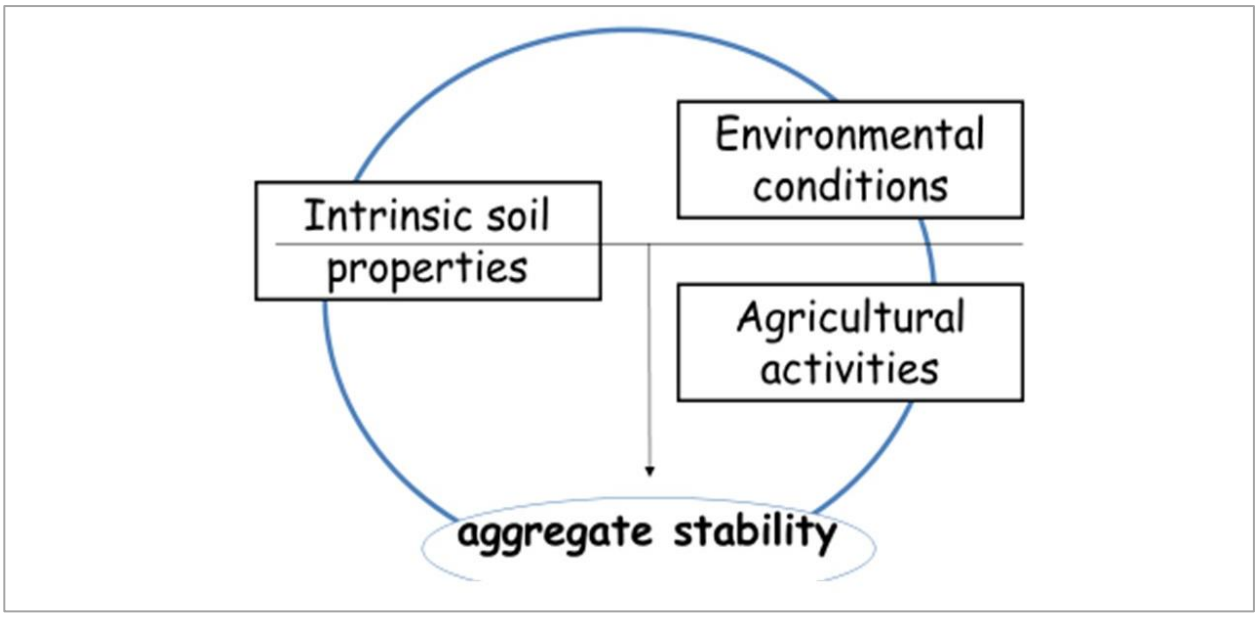


Figure 1.4. Scheme of Soil Aggregate Stability factors; Intrinsic soil properties, Environmental conditions and Agricultural activities (Scheme by author)

1.6.1. Factors Controlling Aggregate Formation and Destruction

The soil aggregates formation and stabilization are simultaneously occurred involving physic-chemical and biological processes. Physical processes like wetting, drying, freezing or thawing affect the aggregate formation. Prior researches postulated hierarchy of soil aggregate from micro to macro aggregate. At first, aggregate form in micro scale, involving physic-chemical soil properties like mineral clay particles, organic matter (Emerson, 1959), and microbial soil compounds. These components flocculate and bind together with the help from cementing agents (carbonate, Fe/Al hydroxide, polysaccharide, glomalin) in order to form micro aggregate that consider as the nucleus of soil aggregate. Each soil crumbs connect and join afterwards by soil biological activities to form macro aggregate. Biological soil properties, such as soil organic matter, plants roots, and microorganism (AMF hypha) play fundamental role in the macro size formation. Later, a new insight of aggregate hierarchy suggests, that the hierarchy of aggregate formation of micro scale was formed within the formation of macro aggregate (Six et al., 2004).

Stability of aggregate is capability of soil to keep the particles attachment under destruction forces. There are four main mechanisms in aggregate stability breakdown: raindrops effect, slaking, shrinking/swelling and physical-chemical dispersion (Le Bissonnais, 2016). The external force associated with kinetic energy of raindrops would mechanically disrupt soil aggregate. It detaches and displaces soil particles (Bradford and Huang, 1992; Le Bissonnais, 2016), particularly under wet soil condition when the aggregate attachment is weaker. The second mechanism of aggregate break down is due to slaking. It is a force associated with entrapped air inside aggregate when occurs at wet condition. A sudden immersed of dry soil in water could block the air inside soil pores and lead to compression of air. Slaking depends on the volume of air that is trapped, the rate of wetting and clay content (Loch, 1994). When the soils reach saturation, the slaking force decrease. Also increasing of clay content reduces the destruction force from slaking (Le Bissonnais, 1996). The third type of aggregate break down is caused by internal force due to different swelling and shrinking of clay mineral under wetting and drying process. This mechanism is quite similar with slaking, yet the disturbance of soil aggregate increases with increasing of clay and result in greater particles compared to the slaking disturbance (Le Bissonnais and Le Souder, 1995). The last one is soil aggregate break down due to physical-chemical dispersion. Less attractive force between colloidal particles during wetting condition leads to dispersion of soil

aggregate. The presence of polyvalent cations induces higher attractive force between the clay particles and thus induces flocculation, while the presence of monovalent leads to dispersion. Therefore, electrolyte concentration of soil solution (EC) affects dispersion (Agassi et al., 1985). Dispersion of soil aggregate results in individual particles of soil and not in micro aggregate. Thus it would be consider as a severe break down of soil aggregate and can influence the other processes (Bresson and Boiffin, 1990).

Soil aggregation and destruction are influenced by different factors, soil properties, microbial activities, litter input, vegetation, abiotic environmental condition (T°, H), and also anthropogenic activities such as agricultural one (Figure 1.4.).

Physical-chemical soil indicators

A multiple interactions between some soil properties affects soil aggregate stability. Soil texture, clay mineralogy, organic matter content, sesquioxide (Al/Fe iron oxide), soil pH, type and concentration of cations, electrical conductivity, and carbonate content are generally known as factors for soil aggregate stabilization (Emerson and Greenland, 1990). The main three properties suggested by Le Bissonnais (1996) are Exchangeable Sodium Percentage (ESP), iron and aluminum oxide (Regelink et al., 2015), and organic matter joint mineral particles.

The ESP value shows concentration of sodium in the exchangeable part of soil mineral. When soil has more sodium, it will induce swelling in clay layer described by diffuse double layer theory (ddl). According to the ddl theory, the superfluous charge of clay mineral are balanced by the counter-ions (mostly cations) and these ions tend to diffuse in the bulk electrolyte solution and being retained close to the surface in order to equilibrate the charge at the surface. The repulsive force would occur between two close planar surfaces if the ddl are overlapped. The ddl is larger with monovalent than with divalent ions, so it means that Na-smectic would has larger swelling than Ca-smetic (Quirk 1952). The slit-shape pores and the overlap pores are the two geometric plate crystal models that explain best the swelling of Na-smectite and the limited swelling of Ca-smectite. When Na^+ substitutes for Ca^{2+}, the attractive pressure declines and smectite starts to swell (Quirk, 2013).

In the rich oxide soils, Fe and Al oxides are important binding agents for stabilizing and aggregating soils (Oades, 1984; Six et al., 2004) particularly at micro-aggregate scale (Igwe and Mbagwu, 1999; Muggler et al., 1999; Wu et al., 2017). The oxides be the binding agent in three main processes:

- it provides surface for organic material adsorption (Oades, 1984);
- oxides have positive charge and will create electrostatic binding with clay minerals that have negative charge (El-Swaify and Emerson, 1975);
- coat of oxides on the surface of minerals forms bridge between primary and secondary particles (Fordham and Norrish, 1983; Muggler et al., 1999; Kitagawa, 1983).

The binding of oxides to minerals reduces the cation exchange capacity particularly for soil containing kaolinites, so promoting the aggregation through electrostatic binding (Dixon, 1989).

Organic matter and clay were correlated to micro-aggregate (<0.1 mm) that affect water retention so could be indicators for soil degradation (Boix-Fayos et al., 2001). Organic matter has double role in (1) stabilizing aggregate to prevent failure and (2) stabilizing suspensions and restricting flocculation of dispersed particle (Quirk, 2013). The stabilization of soil aggregate is a result of interaction between clay minerals, cementing materials and organic matter that called as flocculation plus (Boix-Fayos et al., 2001). The threshold of organic matter content for aggregate stability is around 5-6%. In the arid Mediterranean area, the SOM is lower thus the carbonate would be the cementing agent (Boix-Fayos et al., 2001).

Biological soil indicators (Microorganism)

The activities of soil microbial community play a critical role for soil organic matter decomposition, carbon sequestration and nutrient cycling (Schimel et al., 2007). Thus loss of soil microbial function would disturb its role and be the indicator of declining of soil health and its quality (Chapman et al., 2007). Environmental stress would induce a physiological response from microbial community that can alter ecosystem-level C, energy and nutrient flows. For example, soil salinity reduced the microbial biomass as saline condition is a stressful environment for microorganisms (Yuan et al., 2007). Rietz and Haynes, (2003) also found that salinity and sodicity in agricultural land greatly affected soil microbial and biochemical properties.

Soil microorganisms play important role for soil aggregation, as they are decomposer of soil organic matter that are essential for aggregation (Chotte, 2005). Furthermore, their metabolite products acted as glue for joining soil particles. For example, soil bacteria produce mucilage (polysaccharide) enhancing formation of micro-aggregate (Chenu, 1993; Oades, 1993). Another soil microbial is mycorrhizal fungus. Their metabolism was better for colonizing dry environment like litter surface and pores between aggregate compared to the soil bacteria, especially at non-tillage soil condition. These fungus have hyphae network with plant roots would entangle soil crumbs and build soil macro-aggregate. The hyphae produce a metabolite product, glomaline, acted as glue for sticking the soil particles to join together (Chotte, 2005).

Agricultural practices and environment

The agricultural practices, such as soil management are aimed to improve plant production, by improving soil quality. By improving soil managements practices like right tillage method, mulching, manuring, composting results in increasing soil carbon into the soil, reducing the rate of carbon loss due to decomposition processes and erosion (Figure 1.5.). This soil management affects the stability of soil aggregate and soil structure.

Conventional tillage practices disturb soil aggregate, compact soil and negatively affect communities of plant and soil fauna that are important for soil aggregation by decreasing SOM, CEC, nutrients content, microbial activity and soil fauna activities (Plante and McGill, 2002). Whereas, reduced tillage and no-tillage practice system improved soil aggregate stability by protecting soil organic carbon (Filho et al., 2002) and create macro pore that benefit for water infiltration (Logan et al., 1991). Addition of mulch to soil surface, protect surface ofsoil from mechanical breakdown of rain drops, reduce erosion and evaporation (Layton et al., 1993) as a result it is good for soil structure (Martens, 2000). Amelioration of organic mulching increases amount of SOC pool (Duiker and Lal, 1999) and also maintains soil temperature and moisture status that favor for microbial activities and soil fauna. While application of manure support soil structure, increase mean weight diameter (MWD) and macro-aggregation and reduce the bad effect of slaking, but it might not effective under dissolution and dispersion of soil aggregate (Pare et al., 1999). This because applications of manure increase ionic concentrations like Na^{+}, Mg^{2+} and Ca^{2+}

that would disperse soil aggregate (Haynes and Naidu, 1998; Hao and Chang, 2002). Manured soil tends to have high soil aggregate stability when it is dry and low stability in wet condition.

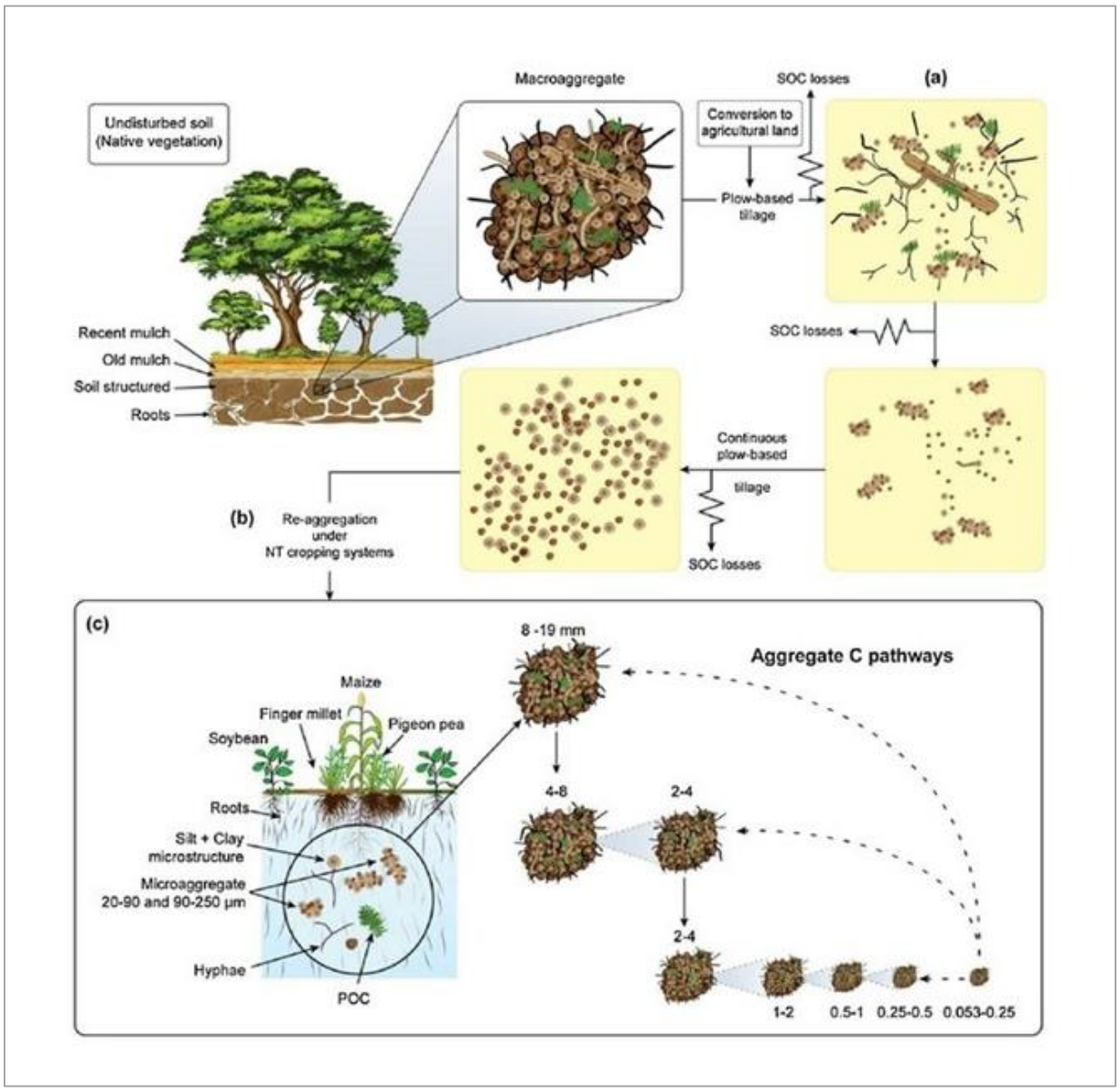


Figure 1.5. "Schematic diagram illustrating the effect of land use conversion on soil aggregation and soil organic carbon (SOC) dynamics. (a) Conversion of native vegetation to conventional plow-based tillage (CT) disrupts soil aggregates, promotes the dispersion of clay particles and silt + clay micro-aggregates, and diminishes the formation of new aggregates, (b) build-up of aggregates by diverse biomass-C inputs under no-tillage (NT), and (c) processes involved in accumulation and redistribution of SOC between aggregates size classes. Aggregate C pathway for soils under Non Tillage cropping systems at the LRV site. Note: SOC losses mean emission of CO_2 and CH_4 by oxidation" Image used from Tivet et al., (2013).

Another type of agricultural practices such as crop management and water management also influence soil aggregate stability. The change of native vegetation to cultivated crop would also influence soil aggregate stability. It increases metabolic CO_2, microbial SOC pool and reduces soil fauna (Saggar et al., 2001). Soil aggregate is also different among different crops, crop rotation and cover crops (Jarecki and Lal, 2003). It has different root structuring, chemical composition that exudate by plant roots that will alter chemical and biological properties of soil (Chan and Heenan, 1996). Another agricultural practice like water management through irrigation also influences soil aggregation. It influences soil water content and soil temperature that has effect on soil microbial activities. The quality of irrigation water also needs to be considered. For instance using salty water for irrigation will increase salt concentration in soil that will disperse soil aggregate.

Environmental conditions like climate and topography also has effect on soil structural stability. For example, temperature influences soil moisture regime and soil temperature that affect activity of soil microbial for decomposing organic material that are important for soil aggregation. High intensity of rainfall will mechanically breakdown soil aggregate under bare soil surface, inducing runoff and soil erosion. Climate could influence cycle of dry-wet and freeze-thawing conditions in soil than influence soil aggregation (Peng et al., 2015). Geographic region, elevation, slope gradient influence vegetation and erosion. Elevation has indirect effect on soil structure, because it influence rate of weathering in soil. Stability of soil aggregate is higher in the north facing slope than south facing slope of Mediterranean soil as difference in microclimate (Boix-Fayos et al., 2001).

1.6.2. Measurement of Soil Aggregate Stability

Various methods of measuring soil aggregate stability have been done since 1930. Le Bissonnais (1996) has listed the different methods that are varies in terms of the sample form and the expression of the results (see Table 1.3.).

Table 1.3. List of different methods for testing aggregate stability (some of these methods include various treatment)

Type of measurement	Form of sample	Expression of the result	Authors
Wet Sieving	3-5 mm	MWD	Yoder (1936)
	< 2 mm	% > 200 µm	Hénin et al., (1958)
	Whole soil	Change in MWD	De Leenheer & De Boodt (1959)
	1-2 mm	% > 250 µm	Kemper & Rosenau (1986); Pojasok & Kay (1990)
	2-3.4 mm	MWD	Churchman & Tate (1987)
	4-5 mm	Time to break down	Low (1967)
Rain drops or rainfall	2-9 mm	MWD	Young (1984)
	5-8 mm	Time to break down	Farres (1987)
	Whole soil	% < 125 µm	Loch (1994)
Ultrasonic dispersion	4-5 mm	Dispersion rate	Edwards & Bremner (1967)
	4-5 mm	Inter-aggregate pore volume	Grieve (1980)
Immersion	3-5 mm	qualitative	Emerson (1967)
Dry sieving	< 4 mm	MWD	Kemper & Chepil (1965)
Fast wetting	3-5 mm	MWD	Le Bissonais (1996)
Slow Wetting	3-5 mm	MWD	Le Bissonais (1996)

In this study, we determined soil aggregate stability by measuring mean weight diameter (MWD) of soil sample with fast wetting method (Le Bissonnais, 1996, 2016). The detail explanation of this method is in chapter 4.

1.7. CONCLUSION

- Soil salinity is a big problem that occurs in coastal zones
- Soil salinity is threatening ecosystems and human activities, in particular agricultural activities.
- The problem must be addressed at the landscape scale in order to consider natural and anthropogenic forcing and at the plot and soil profile scale in order to better understand the processes involved.
- The study area of the Orb Delta was chosen to explore this issue in greater depth combining landscape scale approach and soil quality indicators.

CHAPTER 2

DESCRIPTION OF STUDY SITE

This chapter discusses about geographical and socio-historical agriculture aspects of the study area, Sérignan, downstream of the Orb catchment. Firstly, it discusses the area at the regional scale for its catchment, climate and hydrological conditions, as these components related to water fluxes which influence salinization processes. Then, it locates the study area, and describes its geology, pedology (soil types), and coastal aquifer. It also highlights the land use evolution that related to vineyard issues over a century by providing a timeline description with maps and figures to give a better illustration for each events. At the end, it discusses social-agricultural aspect dealing with agricultural activities and farmer perception about salinity problem in that area.

2.1. THE BOUNDARIES CONDITION

2.1.1. The Orb River Catchment

The geomorphology of Orb river catchment (Figure 2.1.) is well describe by Larue (2008). The river originates at the altitude of 825 m asl, in the southern slopes of the Massif Central (the Larzac plateau) made of Jurassic limestones and dolomites. In the upper valley, the river flows through (i) the Montagne Noire north slope shape in sedimentary (from the Antecambrian to Lower Palaeozoic) and granite units, (ii) between the Montagne Noire axial zone and the Avant-Monts zone according the east-west Orb-Jaur valley ended by plunging gorges. At the Avant-Mont foot, the river crosses the Saint-Chinian ridge formed of sedimentary strata ranging from the Triassic to the Eocene and structured in thrust nappes. In the lower valley to the south, the Languedocian piedmont presents a hilly topography with butte and depression shape in the Miocene formation and where several Quaternary alluvial terraces have developed. The downstream part of the catchment will be discussed in the next section of geology and pedology of study area.

The main tributaries of the Orb River are the Jaur and the Vernazobre River in the upper valley and the Taurou and Lirou Rivers in the lower valley. The catchment is crossed in the south part near Beziers by the Canal du Midi, which relies the Mediterranean Sea to the Atlantic Ocean through Toulouse since the century XVII. At the sea mouth, the catchment area is 1585 km^2, the river length is 134 km, mean elevation and mean slope calculated from DEM are 444 m asl and 12.7°. Natural land use is dominant, near 73% of the total area (with about 55% of total area of forested land mainly in the upper part of the catchment). Meanwhile, about 25% from the total area is cultivated land, which dominated by vineyard. The urban area is about 2% of the total area increasing with the sea shore. The main towns are from upstream to downstream Bedarieux (6,000 inhabitants), Béziers (76,000 inhabitants), and Sérignan-Valras-Plage (11,000 inhabitants).

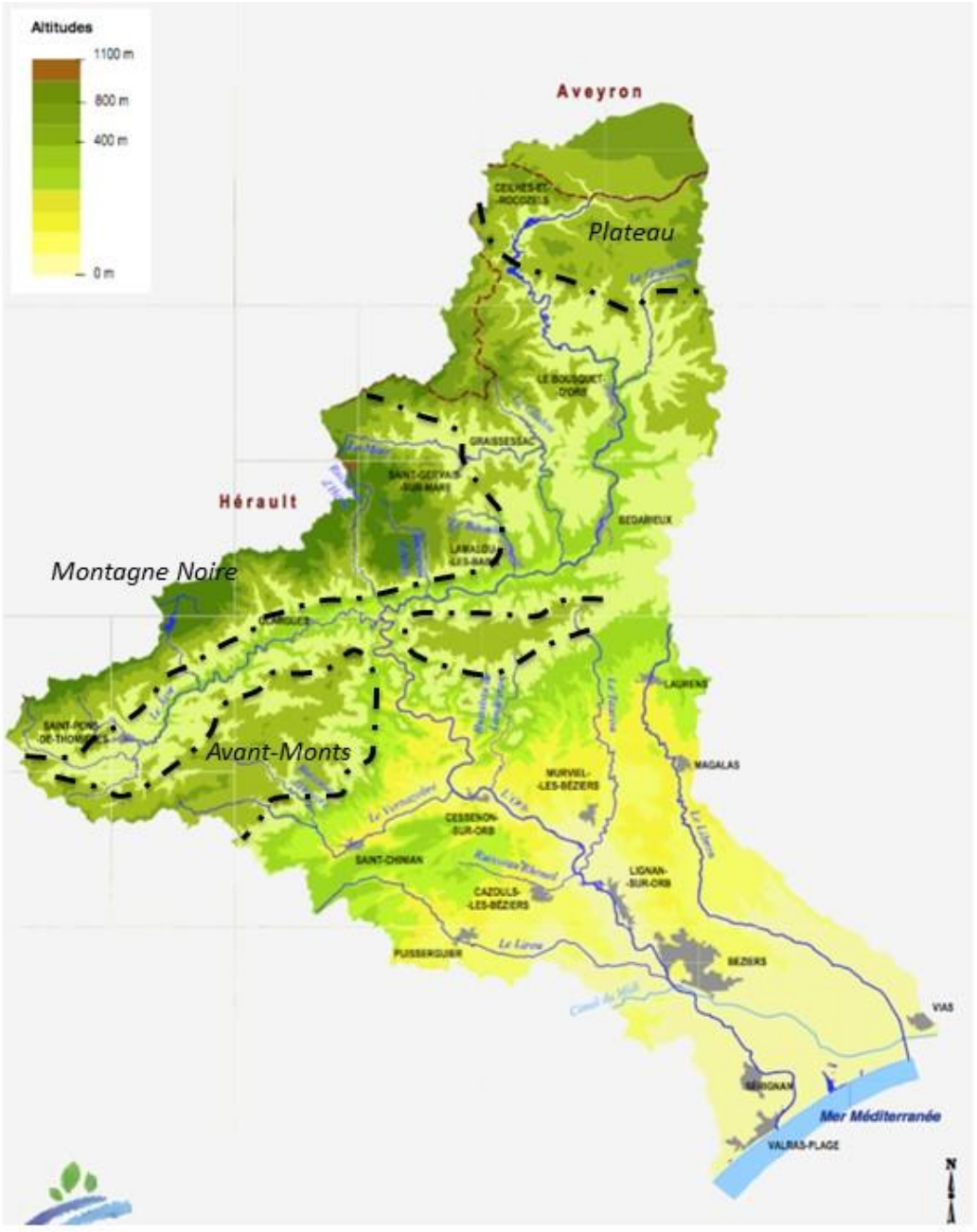


Figure 2.1. Map Of Orb River Catchment Area (from SMVOL, Syndicat Mixte de la Vallée de l'Ord et du Libron).

2.1.2. Climate And Hydrology

According to the Köppen-Geiger climate classification, to northwest upper part of the Orb catchment is Warm Oceanic climate (Cfa) and the main part is Warm Mediterranean climate (Csa). The Ombrothermic diagrams are shown at Figure 2.2. for three locations: upstream (Bedarieux), intermediate (Saint-Chinian) and downstream (Béziers). The wet season occurs from September to February and is shorter in the downstream catchment part. The intermediate season occurs from March to May and dry season from June to August and is more intense in the downstream catchment part.

Considering the Orb River catchment defined at the Béziers outlet, the global water balance calculated from the period of 1960-2004, shows mean annual rainfall equals to 1020 mm, mean annual reference evapotranspiration equals to 950 mm, mean annual discharge equals to 640 mm (Meteo France, Banque Hydro, Lespinas et al., 2014). As noticed by Lespinas et al. (2009), the rainfall-runoff ratio value of 63% is high compared to the surrounding Mediterranean catchments. This is because of groundwater inputs in the catchment upstream part and the influence from human activities, and particularly water transfers with neighbor catchments.

In order to have a better understanding of salinity in the Orb delta zone, it is crucial to give attention to discharge measured at the immediate upstream (Béziers-Tabarka hydrometric station mention by the DREAL French Ministry of the Environment). Annual river discharge is highly variable across years ranging from less than 300 mm to more than 1500 mm. It seems that annual discharge decreases with time but this trend is not statistically significant according to the MannKendall test (p value > 0.1 in Lespinas et al., 2014). Considering monthly discharge, there was an obvious change: before the 80's the Low Monthly Discharge (LMD) were higher and variable, then after the 80's the LMD were lower and quite stable (see Chapter 3). Considering flood occurrence and magnitude, Figure 2.3. shows the Annual Maximum Discharge calculated for 10 consecutive days (AmaxD10d) and associated Return Period (RP) from 1967 to 2016. Since 2005, the maximum flood event magnitude decreased with AMaxD10d less than 150 $m^3.s^{-1}$ and RP less than 2 years. These results indicated that no major flood event occurred during the past decade.

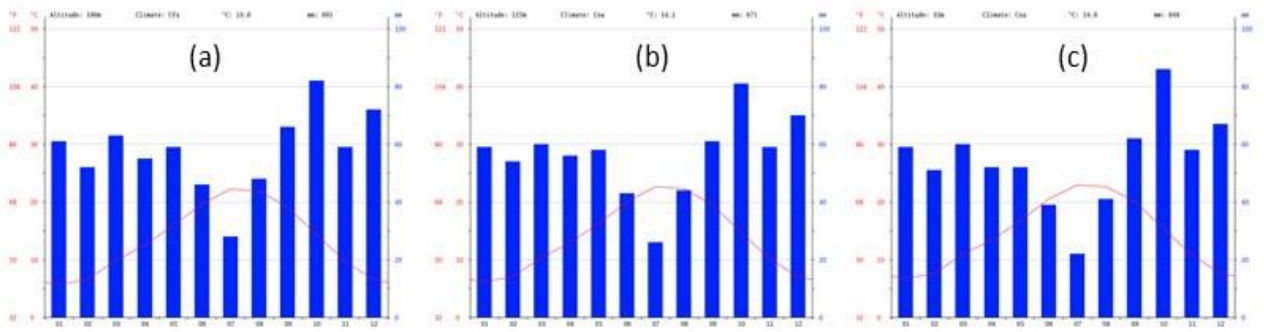


Figure 2.2. Ombrothermic diagram calculated from the period of 1982-2012 at three locations in the Orb catchment (a) Bedarieux, (b) Saint-Chinian, (c) Beziers (https://fr.climate-data.org)

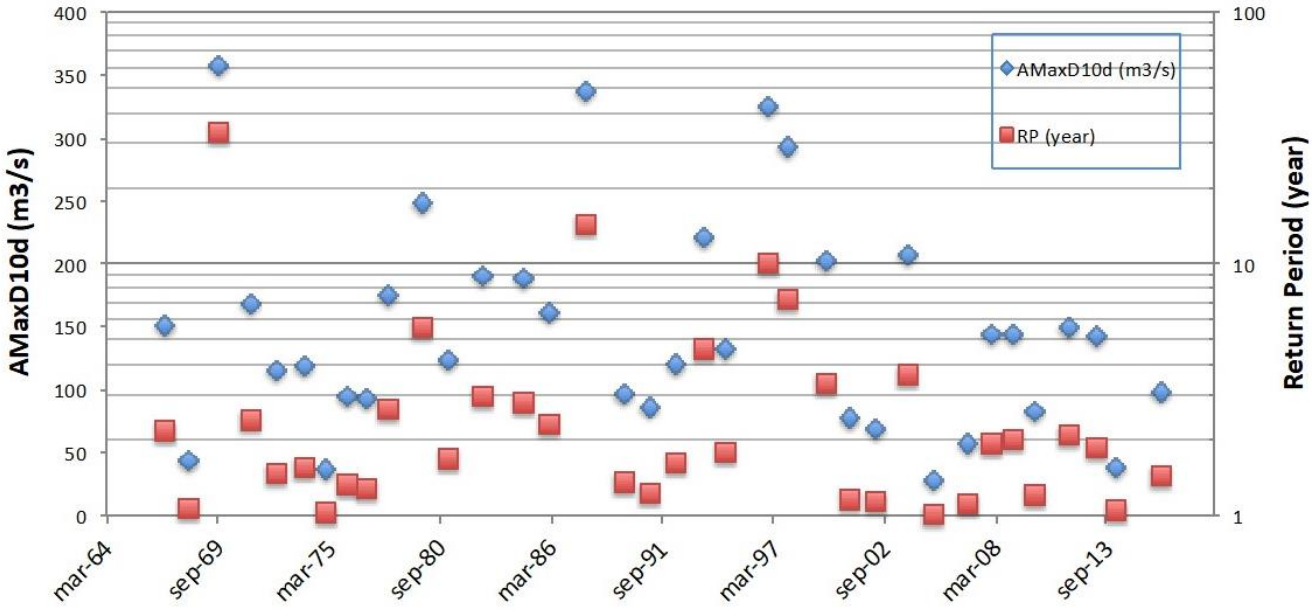


Figure 2.3. Annual Maximum Discharge calculated for 10 consecutive days AMaxD10d and associated Return Period (RP) from 1967 to 2016 at the Beziers-Tabarka Station (Banque HYDRO).

2.1.3. Water Management

Water management issues in the Orb River basin is well described by Girard et al (2015). They note that the population growth rate in the region is the highest in the country (1.6% per year), added to high seasonal variation due to touristic activities in the summer. Water demand concerns urban water supply, agricultural and environmental water demand. Near 15% of agricultural land (more than 60 km^2) are irrigated and half of this irrigation goes to vineyard. This leads to a competition for water consumption in space and time, which expected to increase in the future. Trying to take into account this pressure on the Orb water resources, several infrastructures have been built since the 1960s, most of them involving the BRL Company (Figure 2.4.). The "Mont d'Orb" reservoir (30 Mm^3 of capacity) regulates the Orb River discharge since 1964 to limit flood magnitude and to mitigate the water abstraction particularly at the Réals pumping station where water is forwarded to the Mediterranean coast areas (inside and outside the Orb Catchment). The Orb Watershed Council ("Syndicat Mixte de la Vallée de l'Orb") estimated water inputs from the z,Monts d'Orb resevoir and the withdrawal at the Réals pumping station (for municipality and agricultural use) and in the alluvial aquifer (only for municipality use). From June to September in year 2009, 2010 and 2011, the reservoir released 1 to 3 $m^3.s^{-1}$, water pumping at Réal station was around 0.2 $m^3.s^{-1}$ and water pumping in alluvial aquiver for drinking water was 0.4 $m^{-3}.s^{-1}$. Girard et al., (2015) add that (i) the action plan of the Orb Watershed Council appeals for the improvement of the quantitative management of water resources as a priority and (ii) the river basin management plan has classified the water bodies of the Orb River basin as at risk of not meeting the good status required by the EU-WFD due to a quantitative imbalance in water abstractions.

Figure 2.4. Orb water big infrastructures and main pumping stations (from BRL, 2009)

2.1.4. Sea Water Intrusion

Electrical Conductivity (EC) has been measured in the downstream part of the Orb river. Results stored in a French national database (http://sierm.eaurmc.fr/surveillance/eaux-superficielles/liste-stations.php?donnees=signaletique&codeRegion=&codeDept=34&codeCommune=&bassin=&sousBassinVersant=&coursdeau=ORB&numeroPage=1) are shown in Figure 2.5. At the upper station of Villeneuve, the EC are shown from 1981 to 2013 at a monthly time step. Even if the EC values are low, ranging between 0.15 and 0.9 dS m^{-1}, the time series exhibit seasonal variations: summer values (June-July-August), when Orb discharge is low, the EC values are double and even triple times higher than values measured during the wet season. At the downstream of the Orb river, the EC measurements are available at the Sauvian and Sérignan stations from 1994 April to 1995 February. The results show that during wet season, the EC values are quite similar to those measured at Villeneuve station. During the summer season, the EC values are also increasing but with a higher rate, reaching 4 dS m^{-1}. These results highlight that sea water intrusion (i) leading to an EC increase in the river is happening in the summer season when Orb discharge are low (ii) is decreasing with the distance to the sea mouth but is observable until near 10 km to the upstream.

The EC in the Orb river is also measured by the wine growers (CUMA) using a pumping station near Sérignan city during Spring period (Envilys, 2017). This survey allows them to decide making submersion or not on the right bank of the Orb river. The daily EC measured values are plotted in Figure 2.6. with discharge at Béziers-Tabarka station and the EC at the Villeneuve station. Available data focus on 2005 and 2012 Spring period. At this daily time step, results show that EC is highly variable. Lower values are close to 0.1 dS m^{-1} during flood event, which is near the values measured at the Villeneuve station. Higher EC values are more than 5 dS.m^{-1} during inter-flood period. These results confirm the seawater intrusion in the downstream part of the Orb river when discharge are low, in dry season but also during inter-flood period.

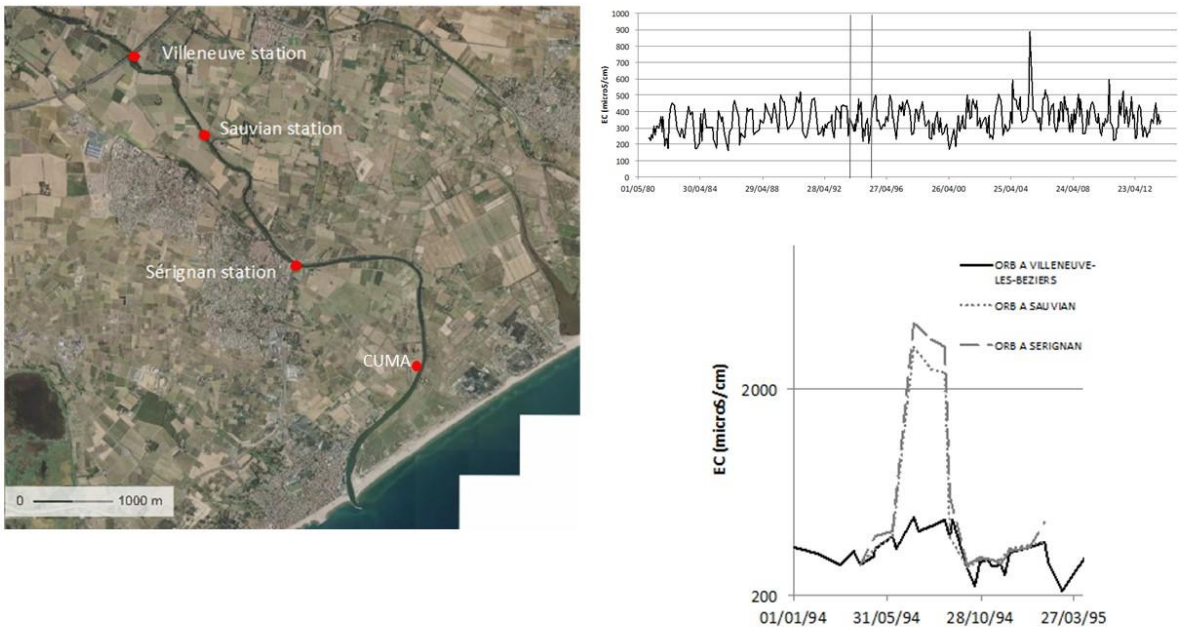


Figure 2.5. Location of electrical conductivity measurements at the Villeneuve station (from 1981 jan. to 2013 dec.), Sauvian and Serignan stations (from 1994 apr. to 1995 feb.).

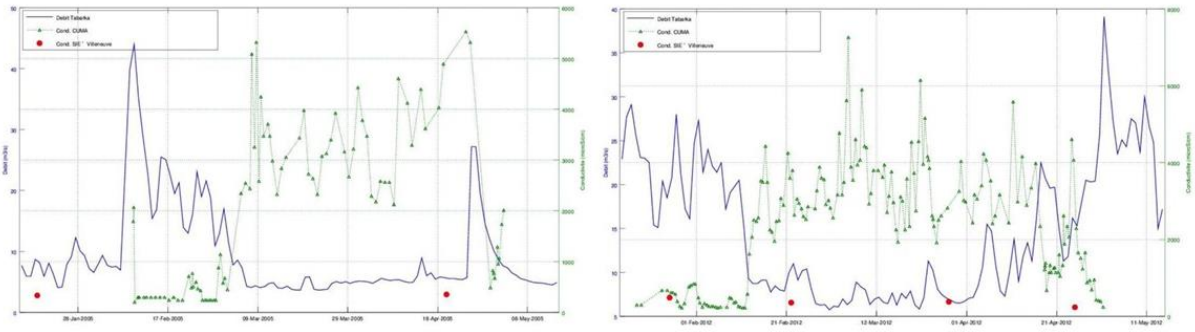


Figure 2.6. Daily EC values downstream Serignan (green dot-line), at the Villeneuve station (red dot), discharge at Beziers-Tabarka (blue solid line): a) 2005 Spring, (b) 2012 Spring.

2.2. GEOPHYSICAL CONDITION OF STUDY AREA

2.2.1. Geographic Position Of Study Area

The study area is in the Sérignan municipality (43°28'N; 3°31'E), Department of Hérault, Occitanie region, south of France. It is located in the Orb River delta connected to the Mediterranean Sea (Figure. 2.7.). The study area of Sérignan is about 2,744 hectares, that consists of four main sectors: the agglomeration of Vieux Sérignan, the natural area near to coastline, the Marina of Sérignan particularly on the right bank of the river Orb, and the alluvium plain with vineyard. The number of population in Sérignan is over 7,000 inhabitants, and this number could be double or even triple in the summer time as increasing number of tourists. The most touristic destination place is Sérignan beach that always packed during summer time as it has a beautiful long beach with white sand. Part of the livelihood of people in Sérignan is agriculture, for its alluvial delta that best for agricultural production particularly vineyard with high yields. The other income also comes from touristic activities because the territory stretch from vineyard plain to the coastal area with dunes and natural places has attracted many visitors. In addition many professionals and companies also looking to settle in this city because of its economic growth, current cultural and sport infrastructure including its historic center. The bordering cities are city of Villeneuve-les-Beziers in the east nearby Portiragnes, Sauvian in the northwest, Vendres in the west and Valras-Plage in the southwest. The city is facing the Mediterranean Sea in the south.

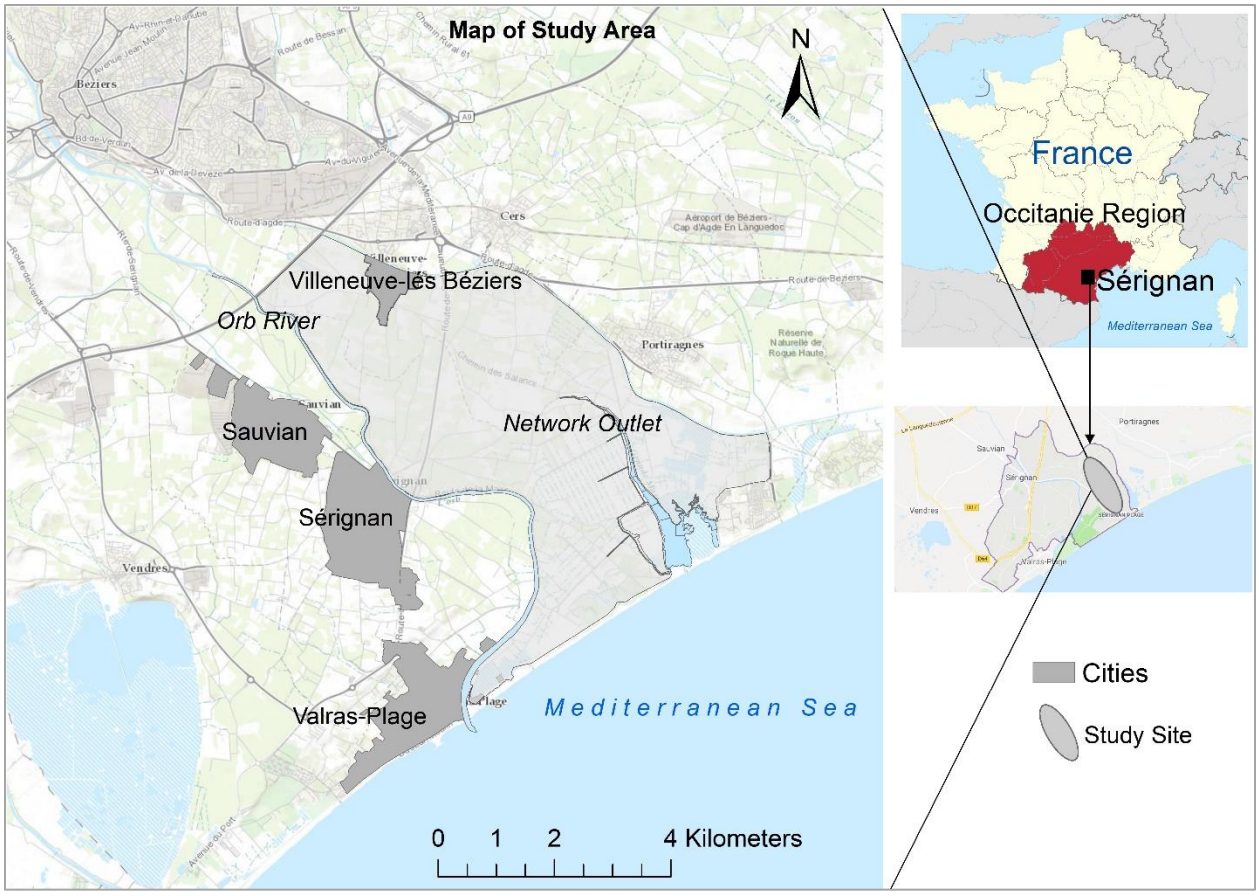


Figure 2.7. Location of the study area: Sérignan, Occitanie Region, South of France.

2.2.2. Geology

All parental material (geological formations) observed in the study area are mention on the geological map of Beziers (1/50 000). A simplified version of geological map is presented in Figure 2.8. The geological formations that observed on Sérignan study site are:

- ***Fza-b***: Quaternary Holocene – alluvial deposits sediments
  It spreads in the Northwestern part and in the center of the site, some fine and undifferentiated alluvial sediments, associated to "Orb", "Herault" and "Libron" rivers.
- ***FLzb***: Holocene – fluviolacustrine sediments and deposits
  It is in the southeastern part of the site and mainly located on the left bankside of the Orb river.
- ***LMz-b***: Holocene – Lagoon deposits
  This formation is located on seashore line.

All these geological formations (*Fza-b, FLzb, LMzb*) are bounded, i) in the northeastern part the "Portiragne" relief, then a structural contact with **pC** formations (Tertiary – Pliocene from continental origin). ii) in the southwestern part by the "Vendre" relief, then a structural contact with **Fw** formation (Quaternary – young superficial formations).

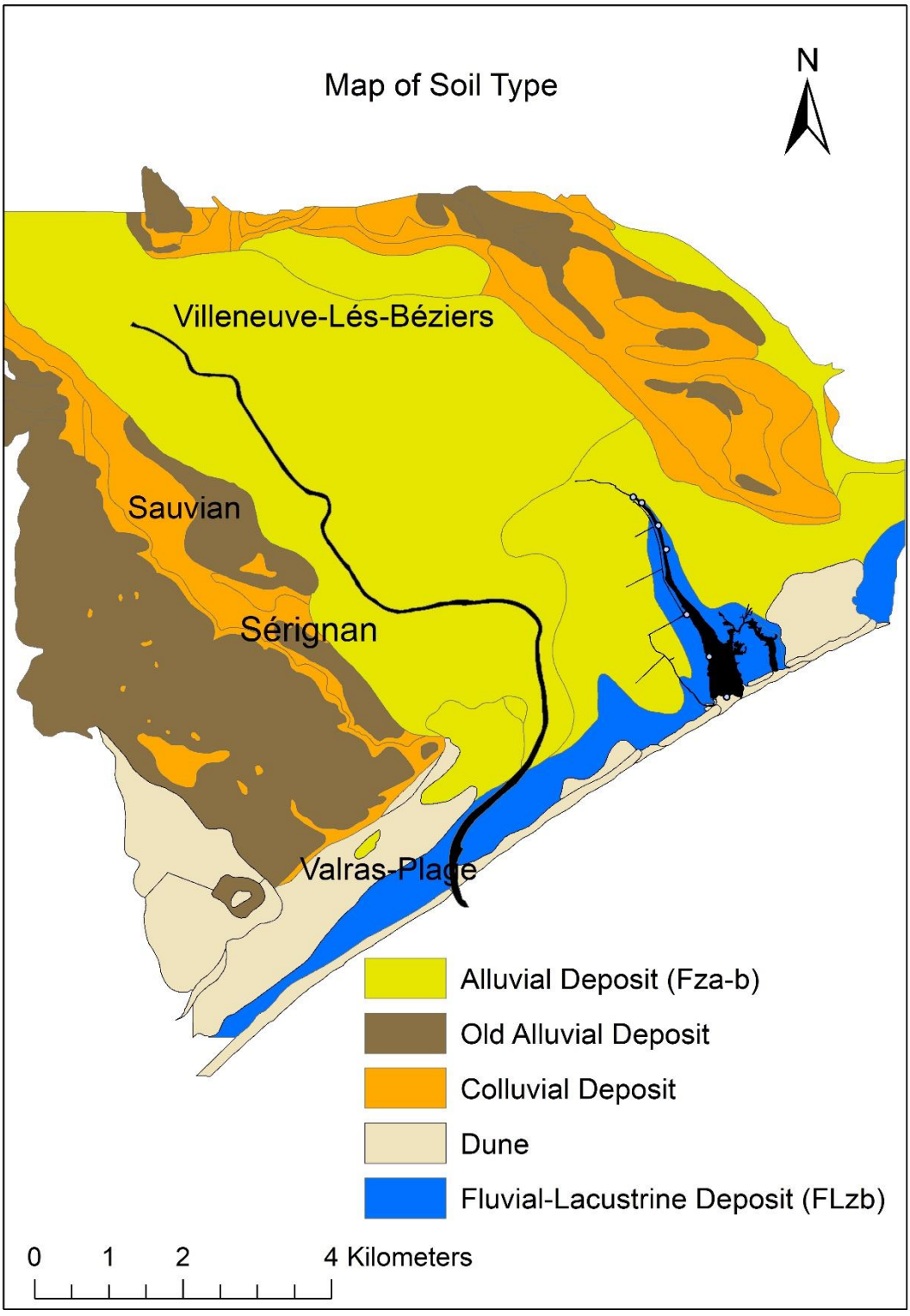


Figure 2.8. Simplified Geological Map was projected based on Geological Map of Béziers (1:50 000)

2.2.3. Soils

On the Sérignan peodological domain, soils are developed by pedogenesis on these geological formations from alluvial and fluviolacustrine origin, and actually constitutive from Valras-Sérignan delta plain system. The soil categorization of soil type based on WRB (2015) are Cambisols and Tidalic Fluvisols. Hereafter some main characteristic of Sérignan pedological cover:

- Soils are developed on quaternary unconsolidated alluvial material. Due to this "young" contribution, we noticed **Sj**-horizon for structural **S**-horizon type.
- Soils are deep (>1.2 m) and locally all mechanical drilling allowed us describe more than 3.00 m depth, except for the location in the vicinity of the "Maïre" place. In this context, the challenge is to bind the limit between soil volume and alluvial formations. This indicated for consideration of non-limited condition of roots development.
- When describing soil profiles, no clear limit between soil horizons and underlying geological formation was found. In this context, soil horizon differentiation is low and mainly based on low content of soil organic matter stratification with soil depth and textural variability with soil depth. High level confidence could be attributed to organo-mineral horizon determination, but medium confidence level had to be attributed to mineral horizon thickness over the depth of 1.2 m.
- In this context of delta plain, a superficial water table is developed in superficial materials. This water table dynamic creates global waterlogged conditions in the soil profile could be described using iron status (Fe^{2+}/F^{3+}) and its colors. Soil profile has two different conditions of waterlogged, temporary (***g*-**horizons) and permanent (***G*-**horizon) conditions.
- When comparing all soil profiles, we notice that waterlogged intensity is quite variable with space. This variability seems clearly linked to water table dynamic, local elevation conditions, network density and efficiency. For most of soil profile, waterlogging (***-g***) was observed within the first meter, preponderantly within structural horizons (***Sjg***) and less frequently within organo-mineral ones (***Ag***). Whatever the location, over 2.00 m and more systematically over 4.00 m depth a ***SG-***horizon was observed. Most of structural horizons are ***Sjg***, ***SjgG*** or ***SjG*** horizons.

- Other parameters for soil horizonation were geometrical ones, due to soil tillage. Because of this generalized tillage on topsoil horizons, all topsoil organo-mineral horizons (***A***-horizons) were noticed as ***LA***-horizons. Most of the time, effect of deep tillage was observed for deeper ***A-***horizons.
- The dominant textural fraction is clearly loamy one. The spatial distribution varies from this loamy texture to more sandy one close to the seashore line, or to more clayed one in the center of the study site. Textural variations with soil depth are locally high for prospection over 2.00 m depth. Main observed textural variations are from loamy soil texture within the first meter to more sandy or more clayey texture with soil depth.
- Soil of the study site have a global geochemical signature inherited from alluvial material and surrounding calcareous rocks, then characterized by high carbonate contents with basic pH value mainly over 8.0-8.5 and a saturation of the cation exchange capacity dominated by Ca^{2+}. In relation to this $CaCO_3$ contex we assessed carbonates in the field with a dilute hydrochloric acid (HCl) solution. As a relation of the effervescence intensity, two types of soil horizons were mentioned: ***-ca***-horizons in case of significant effervescence, and in a limited number of locations ***-ci***-horizons for very slight effervescence. The most soil horizons that mentioned above are ***LAca, Sjca, Sjcag, SjcaG*** and ***SjcagG***.
- Soils of the study site are composed of allochthonous mineral species that originates from rocks of the Orb watershed. The best example is the general observation of micaceous minerals inherited from the "Gorges d'Héric" micaschits.
- For most of the prospected locations, soil profiles present a compacted horizon over 0.40 m depth with the overall thickness of 0.20-0.30 m.
- Soil present electrical conductivity ($EC_{1:5}$) that could be high for soil surface and increasing for deepest soil horizons.
- The current agricultural activities are mainly taking place on Cambisol soil type.

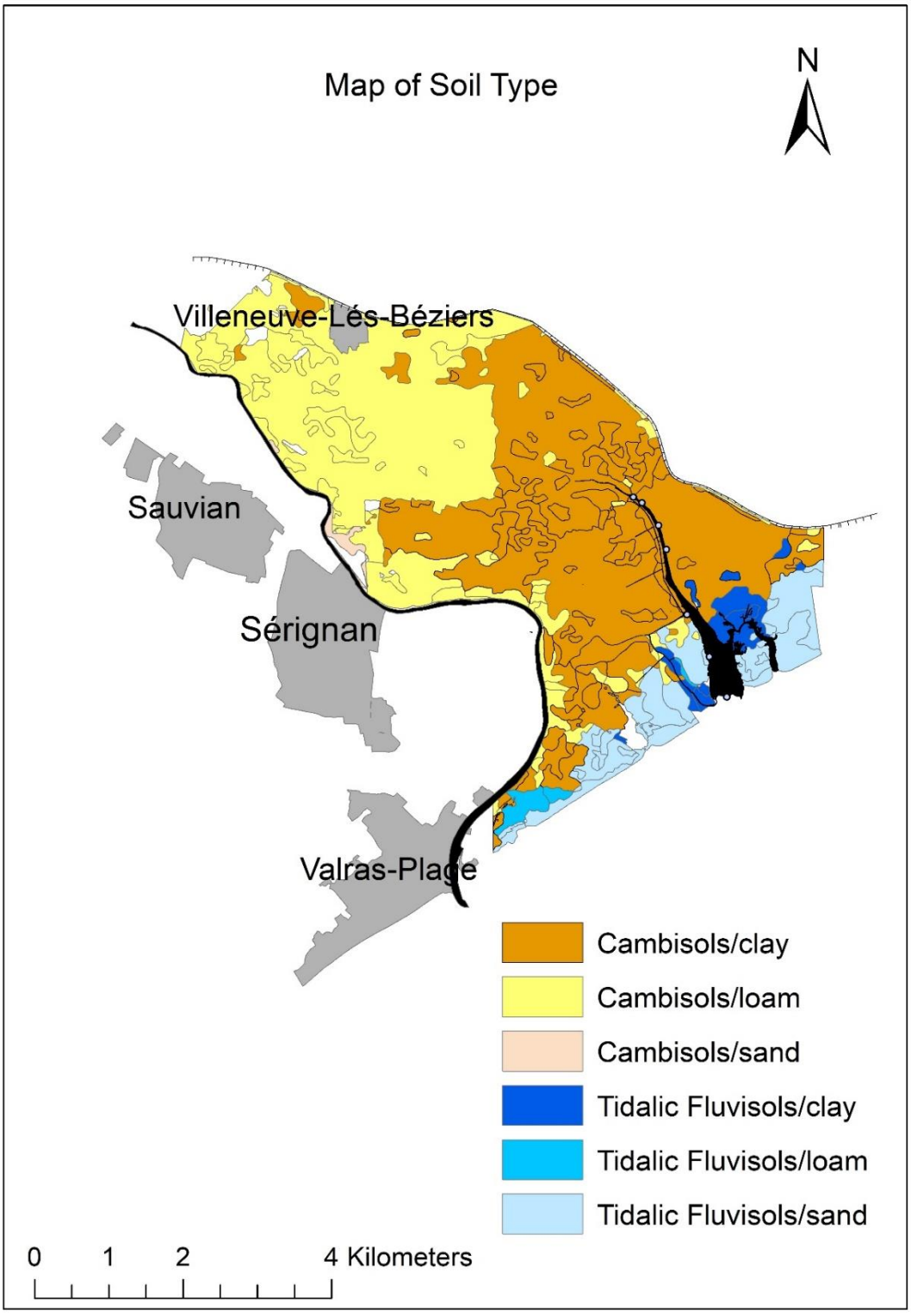


Figure 2.9. Map of soil type (WRB, 2015) and soil texture in the study site, Serignan, South of France.

2.2.4. Groundwater

Sérignan is located in a sedimentary basin of alluvial origin (Orb River) connected to the Mediterranean coastline. In the central part, the sedimentary basin is topographically flat and consists of recent alluvium sediments, whereas high elevations in relation to old alluvium and colluvium sediments characterize basin borders. Near the seashore, fluvial-lacustrine deposits and dune formation substitute alluvial deposits. Elevation ranges from 3 to 6 m.a.s.l. and saturated soil depth is approximately 2–4 m from the soil surface. The Sérignan costal aquifer system can be categorized as a "thick multilayered coastal aquifer system", according to the categorization by Custodio, (2002) (Figure 2.10.). Between the soil surface and -100 m, the system is composed of several small unconfined aquifers. Within this multilayered system, our study focused on the first superficial unconfined aquifer developed in Quaternary and Pliocene materials within the first 10 m. The associated water table that we worked on can be defined as the water-saturated zone of soil cover connected with the Orb River alluvial aquifer. The first confined aquifer is located between -100 and -140 m. This confined aquifer (Pliocene–Piazencian) is composed of sandy materials and is a freshwater source for human use.

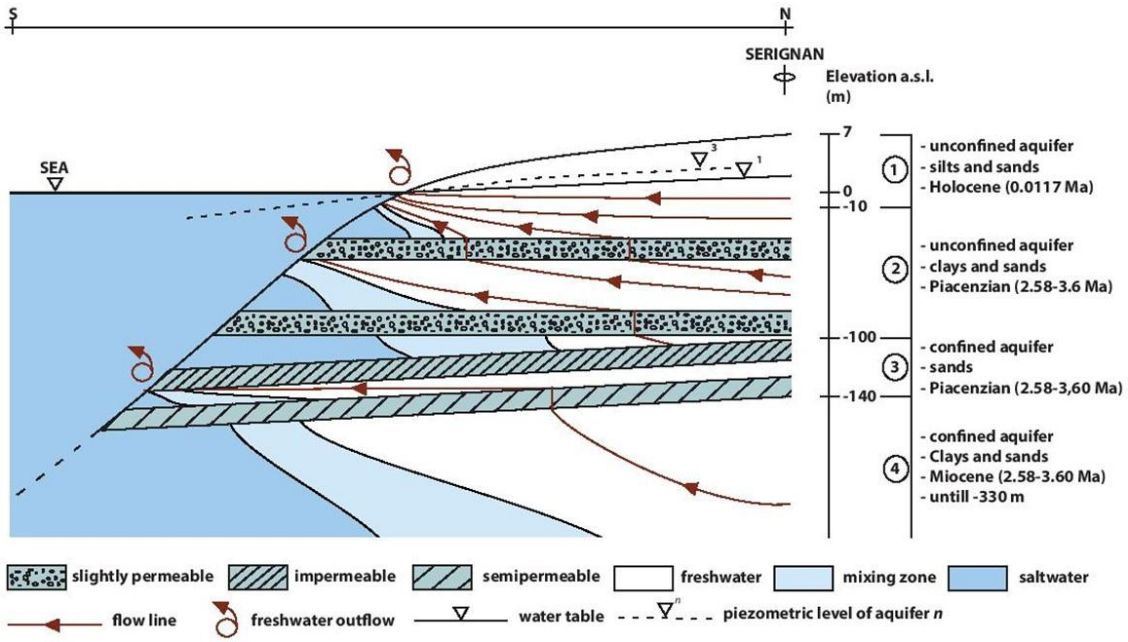


Figure 2.10. Freshwater-saltwater relationship in the thick multilayered coastal aquifer system of Sérignan, France (based on Custodio, 2002).

2.3. HISTORY OF THE REGION AND LAND STRUCTURE EVOLUTION

Over centuries, the Orb River and the Mediterranean sea have shaped the history of the Sérignan. The city was born near the Orb River as attraction of high economic exchanges and cultural activities. A small castle was build and walls for protecting the villages. After the sack of Beziers in 1209, the inhabitant rebuild the village and they become organize to the Consulate, kind of city council from 15th to 18th century. On the bank of the Orb river, the port was commercial until the 18th century, but now is more for touristic activities with more than 300 moorings.

The Roman colonist developed flourishing wine industry that still thriving until now. The development of wine production has influenced the land use/land evolution in Serignan over the century. The timeline presented hereafter describes different events related to vineyard development that influenced the shaping of agricultural land structure in the study area. It is started with the period in 1860's when vineyard encounter a huge crisis, due to an epidemic diseases on its roots that strongly affect agricultural production and social economy aspects in all France including our study area. In the timeline it also describe the introduction of vine rootstock for overcome the diseases, the establishment of winery association in Serignan, the reshaping of land structure in the study area, until the wine welting production in 2000.

Phylloxera Crisis (1863)

An insect (Daktulosphaira vitifoliae), imported from the United States in the 1865-70s, caused the first major crisis in viticulture. Related to aphids, this pest infected the roots of the vine and caused its death in three years. Almost all of the European vineyards were destroyed. In 1879, the overall production of French vineyards fell from between 40 and 70 million hl to 25 million hl. Total area of vineyard in France drop significantly from 2.5 million hectares planted in 1875 to 1.70 million that still remained in 1903. In the south of France, the vineyard left the slopes to be planted in the delta plains, near the coast, since the disease did not affect vineyards planted in sandy soil. The solution was found by grafting local grape varieties onto American naturally resistant rootstocks. In Sérignan, the consequences were the installation of a large number of vineyard plots in the alluvial plain and the coastal zone, whereas this area is roughly described as a wetland in the 18th century Cassini map (Figure 2.11).

Viticulture: between crisis and recovery (1880-1914)

Once the grafting technique was developed, the vineyards settled mainly in the fertile plains, and many small quality vineyards once located on the hillsides disappeared, along with their traditional grape varieties. These very vigorous young vines had high yields. While overall French production was falling, the southern vineyards took advantage of it at first: they marketed a large quantity of lower quality wines produced on the plains. With the reconstitution of vineyards in other regions and the import of wines (in particular from Algeria), there was overproduction. This caused a drop in prices and led to the revolt of the Languedoc winegrowers in 1907 (Figure 2.12). This chronic overproduction was then absorbed by the increase in wine consumption. In particular, in October 1914, the French government, planning a long-term war, decided to allocate wine rations to the soldiers: ½ litres and then ¾ litres per day and per soldier. This wine was made from blends of low-grade wines (Maconnais, Beaujolais or Charentes), with the high-grade production of Languedoc-Roussillon. A large part of the wines of southern France were sent to large regional warehouses and then transported in tank wagons to the warehouses at the rear of the front. A large number of convoys departed from Béziers, proclaimed "Capital of Wine" at that time (Figure 2.13).

Establishment of Sérignan Cooperative Winery (1935).

In 1901, a group of winegrowers from Maraussan, near Sérignan, gathered to vinify and market their wine production. One of the first cooperative winery in France was inaugurated in 1905. In the 1930s, wine cooperation expanded significantly (from 82 cooperatives to 827 between the two world wars), and stabilized in the 1970s. During this period, cooperative winery entered into mass production and played the role of a technical tool by allowing better vinification and protecting winegrowers from the strong variations of the market. In 1935, a handful of winegrowers decided to build the cooperative winery in Sérignan (Figure 2.14). The good results of winemaking and sales led to an increase in the number of cooperators. From 1935 to 1957 the cellar built successive extensions. The production of the Sérignan vineyard grew with the contributions of the winegrowers of the surrounding communes (Valras, Sauvian and Vendres). The cooperative was a way for small producers and employees of large properties to gain independence. In response to customer demand, the Sérignan cooperative began selling directly in 1975 (A. Lupia, pers. comm.). This grouping of winegrowers is reflected in a concerted reflection and incentives to reason the establishment of vines and associated grape varieties. For example, the conversion of Gobelet vine

training types onto Trellised vine (see Figure 2.16). It is also accompanied by collective logic of submersion and drainage.

Landscape and land register reshaping (1962).

The area of vines before 1962 were about 623 ha that spread in almost all of northern part of study area, and small area was located closed to the coastline. These vines had a thriving production for 20 years. Then after 1980, the farmers started to plant cereal that torn out the vineyard. Furthermore the increasing of tourism activities and the establishment of campsite in coastal line had replaced all vineyards near coastal site. Since 2012, about 200 ha of vineyard has been replaced by other land use such as cereals, building/road or abandon for natural land. Thus, total area of vineyard decline from 623 ha in 1960's to 400 ha in 2012 (Figure 2.15).

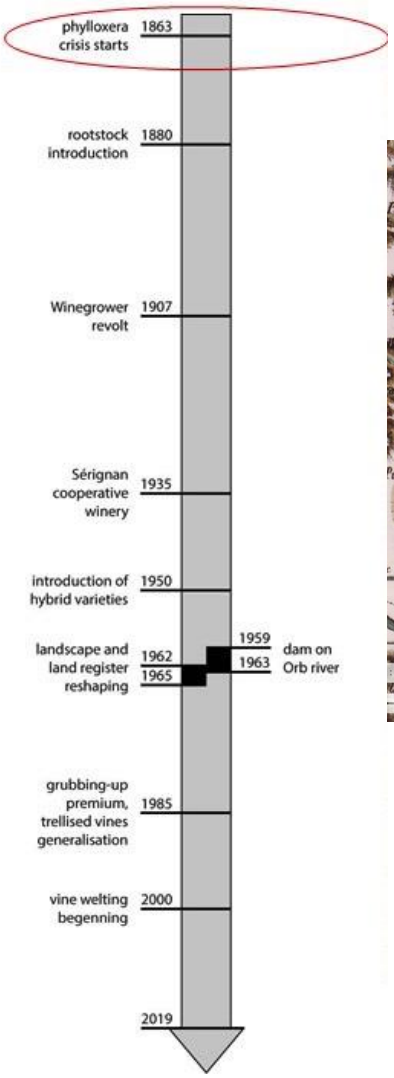


Figure 2.11. Serignan in the century XVIII[th] Century from Casini Map (Source: IGN, Geoportail)

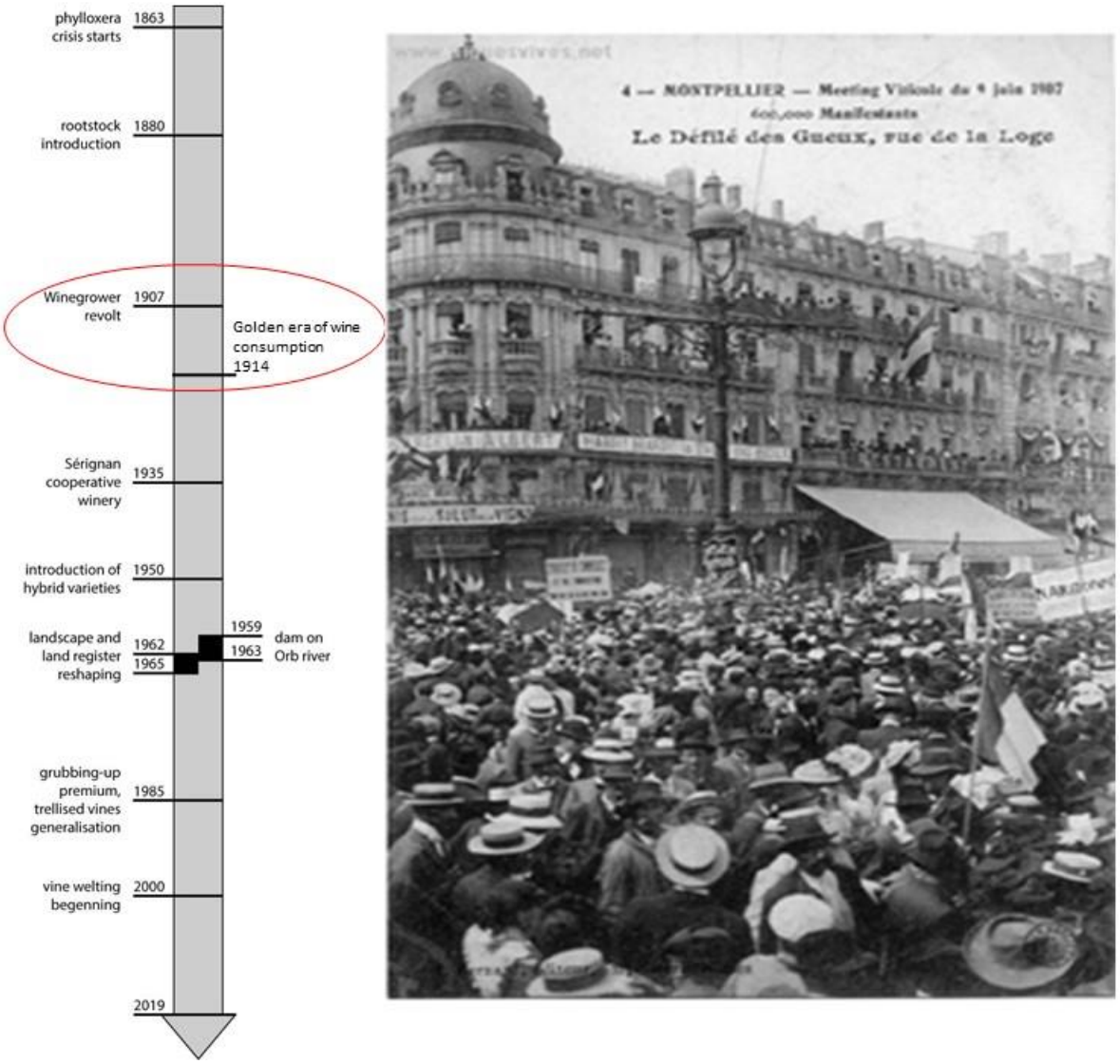


Figure 2.12. Wine grower revolt in Montpellier, 1907 (photo from Wikipedia).

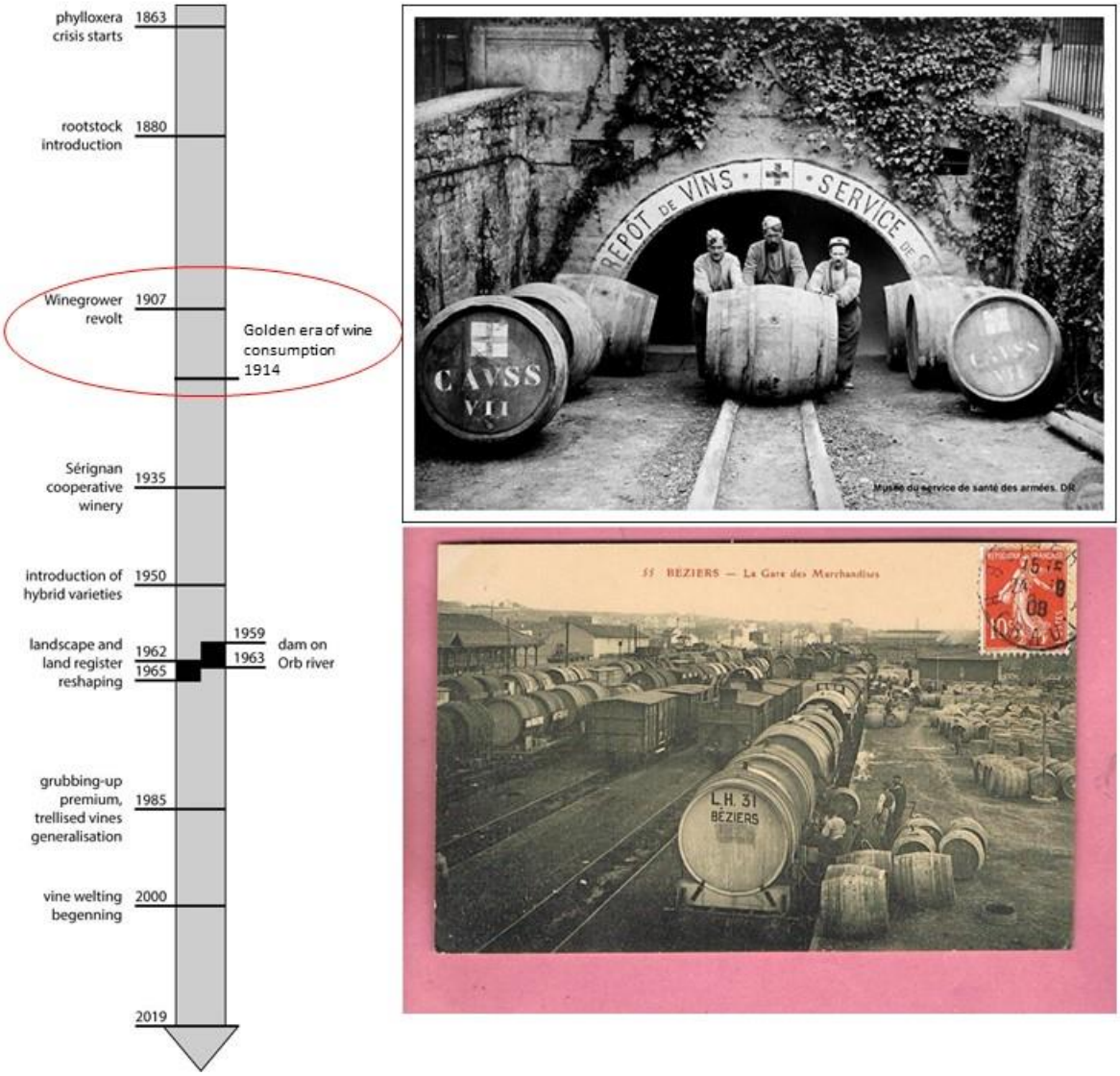


Figure 2.13. Photo of the depot of wine for the soldiers (above); Beziers train station in 1914, supplying wine to the soldiers in the North of France (below). (Photos from Wikipedia)

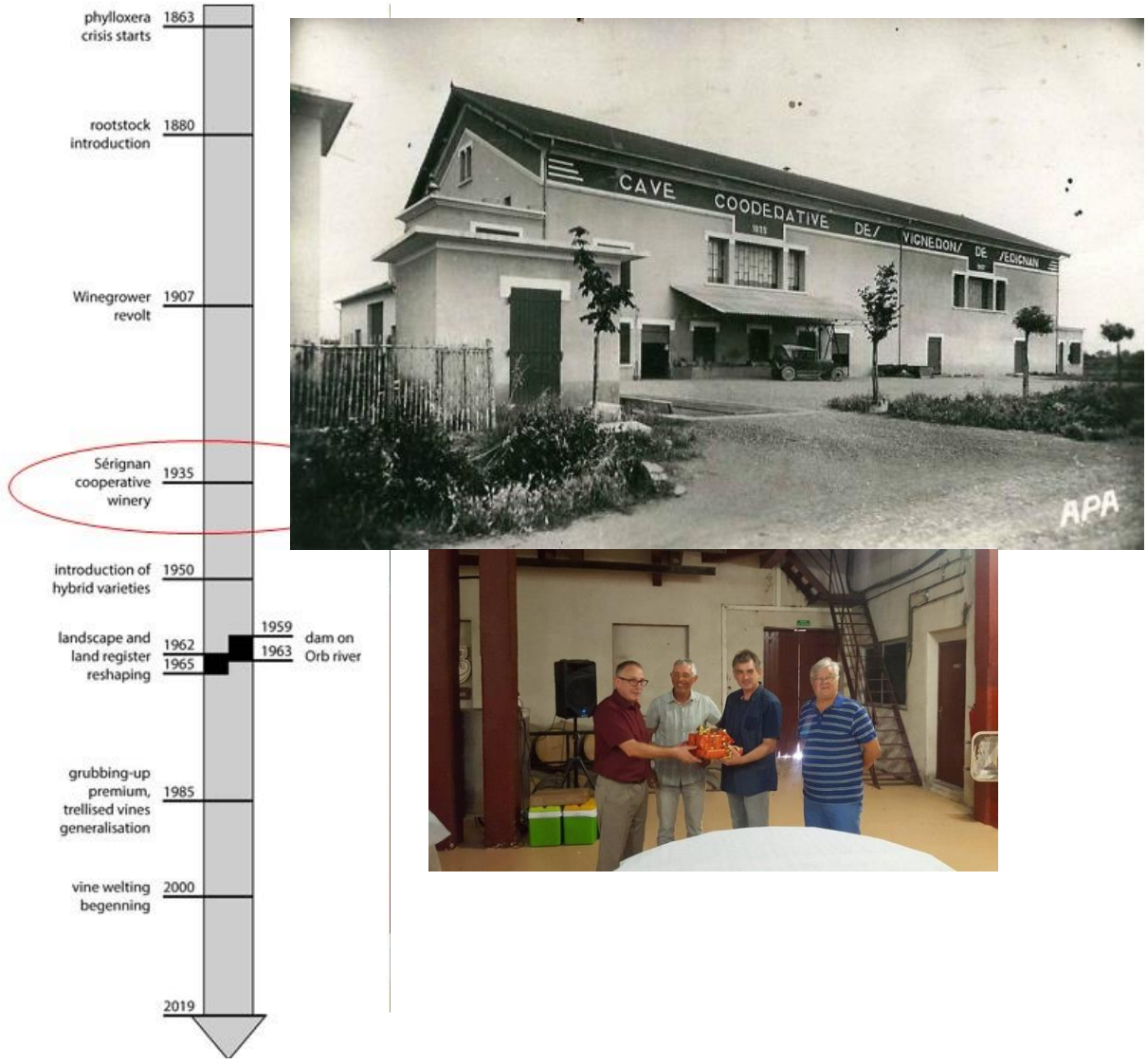


Figure 2.14. Establishment of Sérignan Cooperative Winery in 1935, the current and former cooperative winery presidents.

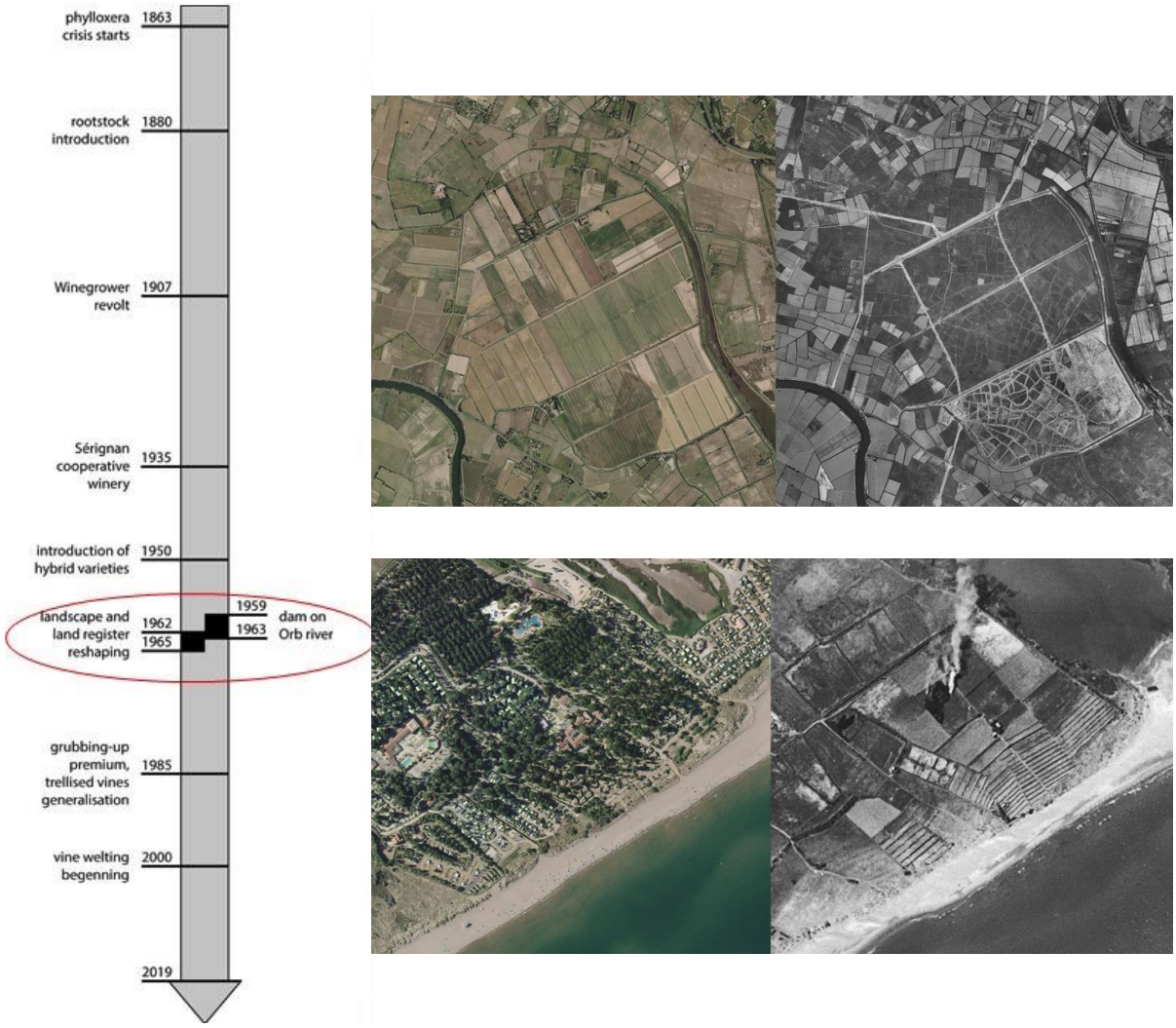


Figure 2.15. Comparison of land structure in recent years (2012) and in the past (1960).

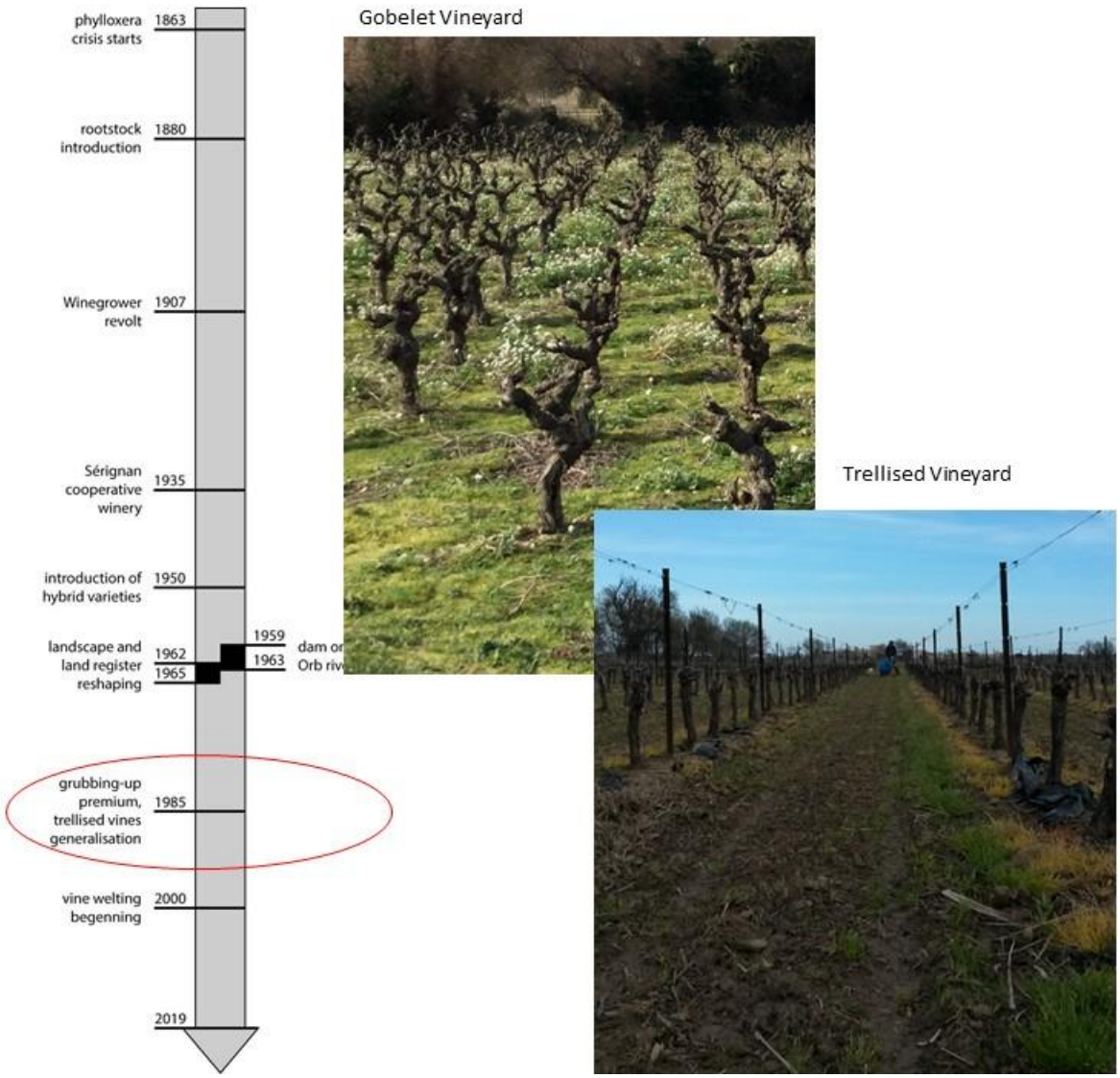


Figure 2.16. Trellised vine and Gobelet vine types. There was a support from government and introduction for new vine plantation system, using a wire in the vine row. This to support mechanization during the harvesting time, as it is simple and faster compared to the conventional style system (Gobelet) that most of the time the harvesting is done manually.

Vines Submersion

The submersion of the vineyard has been started long time ago before the reshaping of the land in 1960's at both banks downstream of the Orb River to prevent the land salinization. It allows the leaching of salts, which are then carried out of the plots by a network of drainage ditches. This submersion can be either of natural origin by overflowing the river during high floods or of anthropogenic origin by water supply. This supply was carried out in the study area, firstly by direct pumping from the Orb River and then, from the 1960s, using the BRL Company's pressurized system from a water intake in the Canal du Midi as shown in Figure 2.17. Faced with high water prices, this practise decreased in the 1990s. In 2012, an agreement between farmers and BRL Company on a moderate water price allowed a return to the practice of submersion.

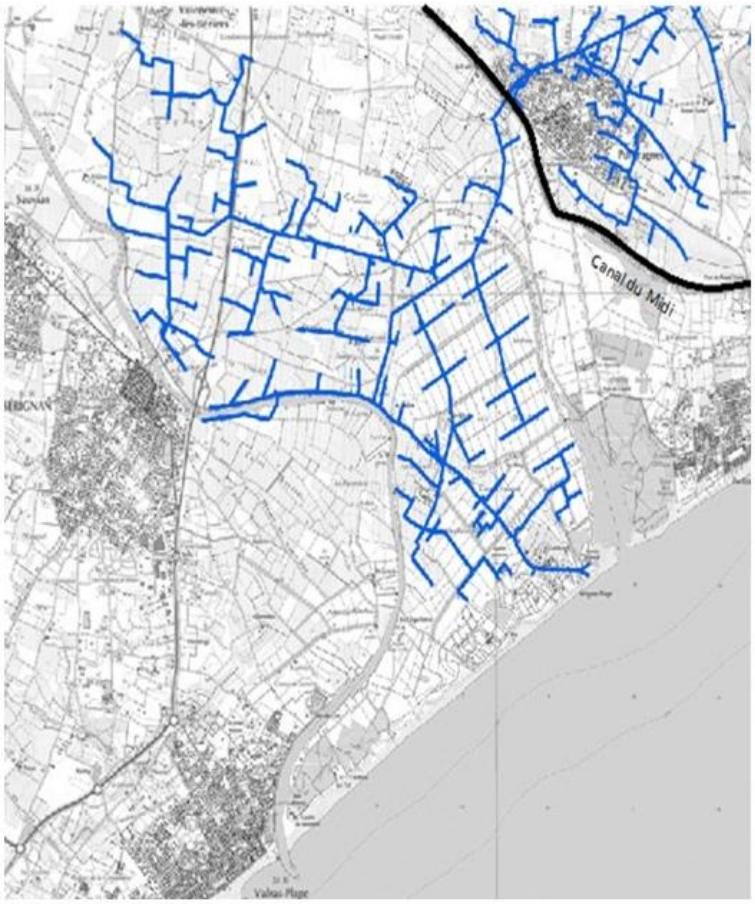


Figure 2.17. Water pressurized network (Source: BRL).

2.4. SOCIAL AND AGRICULTURAL ACTIVITIES

The main livelihoods of inhabitants in Sérignan are agriculture and tourism. Majority of the farmers are winegrower, then least cultivate crops (durum wheat, peas and barley), pasture and arboriculture (fruits and plants). Viticulture is a dominant since the end of the 19th century. The agricultural practices had substantially changed with time as previously shown the timeline.

In 2015/2016 there were about 42 farmers who voluntarily be the respondents for the research conducting by agricultural chamber. The objective was to investigate their perception of the salinization magnitude on the study area. Most of the farmers who participated at the survey own agricultural fields with the total area is about 15 ha. The fields are mainly viticulture and the farmers mostly do the activity as part of the Sérignan winery cooperative. The gross income from their agricultural activities are range from 25,000 to 100,000 euro per year and it mostly come from the viticulture. The farmers are generally over forty years old, and only 20% that considered to be younger than this age. They filled a questioner about their perception of salinity impact on their field. As in the last decade, the farmer of Sérignan has suffered for low agricultural production. They assumed that soil salinity is the triggered of agricultural production reduction. In 2012, the farmer even documented the formation of salt layer in the Orb River bank (Figure 2.18.). This supports their suspicion of soil salinity problem in that area.

The perception was divided in five different classes: not impacted, slight impacted, medium impacted, strongly impacted, and very strong impacted (Table 2.1.). Almost all of the farmers (96%) presumed that their field has been impacted by the salts and only 4% of the respondents alleged that their fields has no salinity problem.

Figure 2.18. Salt layer formation in the Orb River bank. The photo was taken in 2012 by Mr. Reboul

Table 2.1. Farmers' perception about salinity on their field in the Sérignan study area

Farmers' Perception of salinity on their field	Presentation (%)
Not Impacted	3.6
Less Impacted	12.6
Medium Impacted	15.1
Strongly impacted	50.0
Very strong Impacted	18.7

Data from Agricultural Chamber (2015)

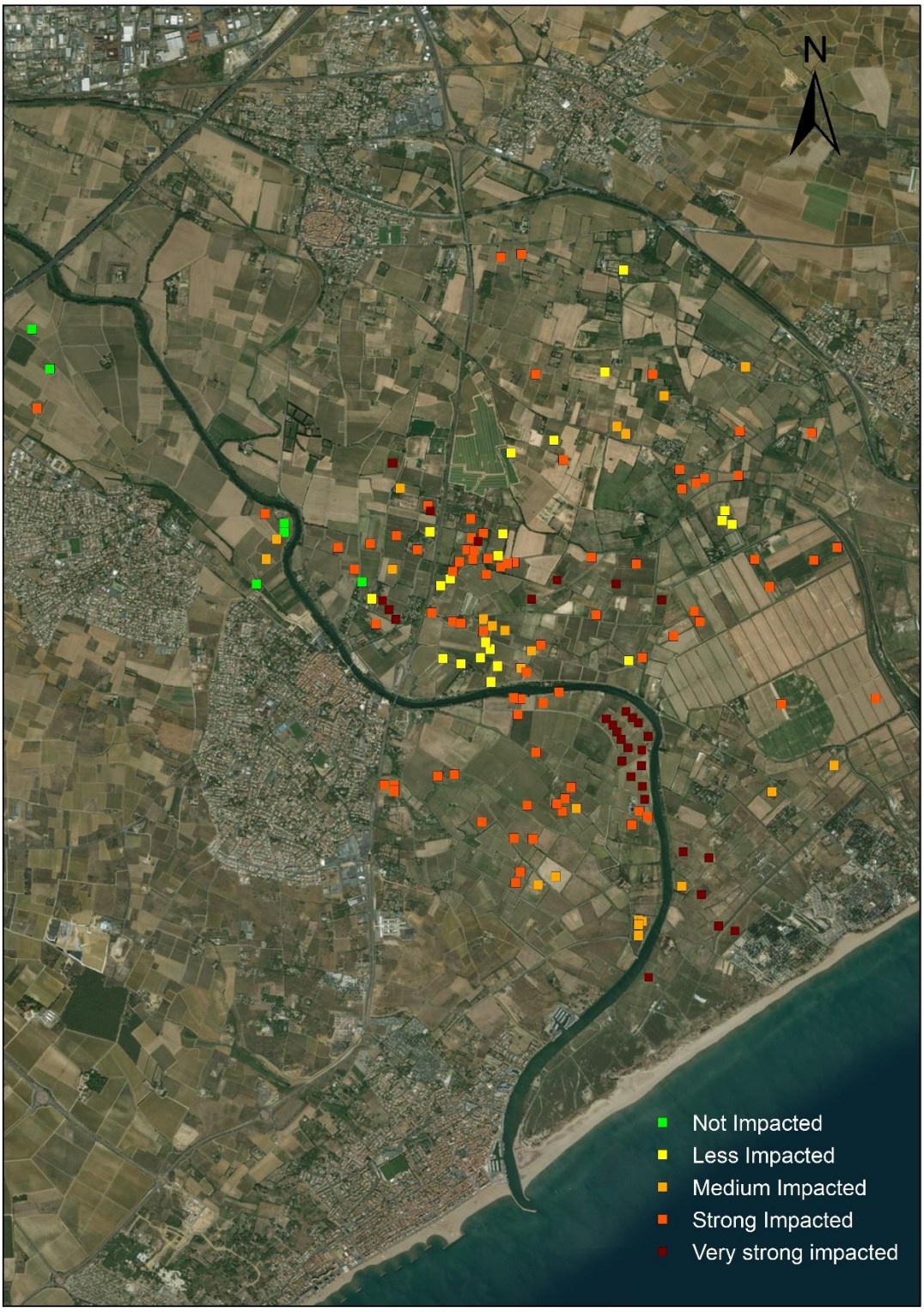


Figure 2.19. Map of farmer's perception of salinity

The perception of farmers about salinity effect on their fields is mapped at Figure 2.19, using data provided by the Agricultural Chamber survey (2015). It shows that farmers, whose fields are located up in the northwest of the area, assumed that their field were lesser impacted by salinity, in contrast with the fields that are located down and in the middle of the area. The majority of fields that are assumed to be strongly impacted by salts were located in left and right banks of Orb River and close to the sea. However, some fields that are assumed to be very strong affected by salts also located in the middle of the study area surrounded by fields considered as middle or less impacted by salts.

This various perceptions about salinity impact on agricultural fields might be due to different vegetation/plants in the fields which can exhibit different response on soil salinity. Different farmers might have different agricultural practices that result in different effects of soil salinity.

2.5. CONCLUSION

The various assumptions of soil salinity problem initiate us to study about actual salinity in that area, which including the driving factor of salinization process at the landscape scale, then study the impact of salinity on soil quality indicator particularly soil aggregate stability under field scale (soil profile) and soil microbial community. This topic will be discussed in the three following chapters (Chapter 3, 4 and 5).

Preliminary Statement

Land salinization management cannot be properly implemented without preliminary knowledge about spatial distribution and evolution of salinization (Michot et al., 2013). Our primary action was to quantify the spatial distribution of soil and water salinity.

This chapter is aimed to identify actual soil and water salinity and the driving factors of salinization at agricultural coastal area by evaluating the evolution of landscape components; land structure, climate and river discharge, that influence water fluxes. In order to simplify the complex system of water fluxes in the coastal area, we built a conceptual model of water fluxes that help us to have a better understanding of salinization process. This chapter has been published in the Journal of Science of the Total Environment 625 (2018) 647–656.

CHAPTER 3

LANDSCAPE EVOLUTION AND AGRICULTURAL LAND SALINIZATION IN COASTAL AREA

ABSTRACT

Soil salinization is a major threat for agricultural lands. Among salt affected lands, coastal area could be considered as highly complex systems, where salinization degradations due to anthropogenic pressure and climate induced changes could significantly alter system functioning. For such complex system, conceptual model can be used as an evaluation tool for the preliminary step in order to identify main evolutions responsible for soil and water salinization. This study was aimed to propose a conceptual model of water fluxes in a coastal area affected by salinity. It can help to identify the relationships between agricultural landscape evolution and actual salinity. First, we drove field investigations from 2012 to 2016, mainly based on both soil ($EC_{1/5}$) and water (EC_w) electrical conductivity prospections. It allowed us to characterize spatial structures for $EC_{1/5}$ and EC_w and to identify river as a preponderant explanatory factor for land salinization. Then, we proposed and used a conceptual model for water fluxes and engage a time analysis (1962-2012) for three of its main constitutive elements; e.g. climate, river and land system. When integrated within the conceptual model framework, it appeared that all constitutive element evolutions since 1962 were responsible of system equilibrium disruption favoring overall salt accumulation in soil rootzone.

Key Words: Coastal Landscape, Conceptual Model, Salinization, Vineyard

3.1. INTRODUCTION

Land resources are being irreversibly lost and degraded due to pressure generated by increasing of human populations, its activities and by changes in climate and land use (EEA, 2000). For agricultural lands, anthropogenic changes are induced by farmers at both field unit and farm scale or by policymakers from farming to administrative divisions (Verburg et al., 2002; Rounsevell et al., 2005; Claessens et al., 2009). Climate-related changes are induced by climate factors as predicted by projections of future climate change (IPCC Core Writing Team et al., 2014).

Within this context of land degradation, soil salinization is a major threat. According to FAO in 2000, around 830 Mha of land worldwide is considered to contain salt-affected soil (Martinez-Beltran and Manzur, 2005). The salt affected soil can be found in all continents apart from Antarctica and occurs in more than 100 countries worldwide (Szabolcs, 1985; Rengasamy, 2006), with the most prominent areas being arid and semiarid climatic zones. In Europe, soil salinity affects about 3.8 Mha of land (Tóth et al., 2008; Rhoades, 1999; JRC, 2012) and is particularly problematic in the coastal areas of southern Europe (Daliakopoulos, et al., 2016). Salinization is an increase in the concentration of water-soluble salts in water and soils. Soluble salts could be of environmental origin (geological, climatic, topographic, and hydrological) or result from inefficient or inappropriate human activities (Shrestha, 2006; Szabolcs, 1992; Daliakopoulos, et al., 2016). Whatever the origin, salinity threatens the sustainability of agriculture by affecting crop production through decreased yields and plant death (Feinerman et al., 1982; Maas and Hoffman, 1977; Li et al., 2012), according to processes summarized by Rengasamy (2010).

In order to preserve land resources and potential crop production, a possible solution to salinization is the promotion of sustainable land management practices that sustainably reduce salinity. To this end, modeling could be an appropriate method for simulating the evolution of soil salinity according to different land management scenarios. Daliakopoulos et al., (2016) and Coletti et al., (2017) have listed models for studying the evolution of salinity in agricultural environments. However, the use of these models presumes the ability of users to produce quantitative data and to have hypotheses on system functioning and complexity. A preliminary step prior to this numerical work could be to use a conceptual model as an evaluation tool, allowing the reduction of complexity (Margoluis et al., 2009).

Estuaries can be defined as areas where salt water from the ocean mixes with fresh water from land drainage (Potter et al., 2010; Whitfield and Elliot, 2011). For centuries, estuarine wetland ecosystems have been valuable to humans. Now, estuaries have become hotspots not only for agricultural land use but also for human settlements, tourism, and industries. Due to this anthropogenic pressure, estuaries are more susceptible to land degradation and ecosystem disturbances. Indeed, estuaries represent highly complex situations, e.g., situations where social, political, economic, cultural, and environmental factors interact (Brechin et al., 2002; Hannah et al., 2002).

One serious ecological problem at the coastal areas is salinization on soil and water. Primary source of salt ions is from natural pathway: tiny seawater drops on the soil surface and from bottom with saline ground water and salt-containing marine sediments. In addition, some amounts of salts are also from river discharge, especially when the river transports the composed salt-bearing rocks. The source of salts, transport pathway and geomorphic structure of estuary determine distribution pattern and composition of salts in the soil profile (Chernousenko et al., 2011). Furthermore, based on the accumulation characterization, salinization in coastal area is divided in two main categories; seacoasts salinization, the main source of salt is connected with the sea. The second is delta areas, salinization under the impact of the sea combined with the input of salts river water. In France, the Orb River estuary can be considered an appropriate example of agricultural land that is currently threatened by salinization and where conceptual models could be helpful in identifying the main factors of land salinization.

Consequently, the objectives of this study were (i) to determine actual soil and water salinity in the Orb estuary, (ii) to determine the landscape evolution, and (iii) to build a conceptual model of water fluxes in a coastal area affected by salinization in order to identify the relationship between landscape evolution and actual salinity.

3.2. MATERIALS AND METHODS

3.2.1. Data Collection and Data Analysis

Soil and water Samples

To estimate soil and water salinity, we used the relationship by Richards (1974) that relies on the electrical conductivity of saturated paste extracts and total dissolved salts. In order to perform a high number of measurements, electrical conductivity with temperature corrections was measured using a Consort K912 probe (Consort bvba, Belgium) with (i) direct measurement for water (EC_w) and (ii) a 1:5 ratio of soil to deionized water for soil ($EC_{1/5}$). Figure 3.1. shows the spatial distribution of soil and water sampling locations.

Soil samples (N=1737) were collected for different soil depths (0.20, 0.50, 0.80, 1.10, and 1.20 m) using a soil auger in two contrasting months: at the end of September (dry season) and the end of March (wet season) from 2012 to 2016. Each auger boring for the soil sampling survey was conducted according to FAO rules for soil description (Jahn et al., 2006). All soil samples were air-dried and measured for $EC_{1/5}$ adapting the protocol of the United States Salinity Laboratory (Rhoades et al., 1999). All of the samples were air dried and measured the electrical conductivity ($EC_{1/5}$) by added 20 gram of dried soil sample to 100 g deionized water then mixed. Incubated for about 20 hours to let all of the suspension down, then measured the solution using conductivity meter afterwards. The method was used as a simple and affordable way, especially when we had big number soil samples. Saturated soil depth was determined by drilling 102 holes using the prospecting kit for geological surveys by Eijkelkamp (The Netherlands) (Fig. 3.1). For each hole, we waited for water level equilibrium before noting saturated soil depth, and measuring EC_w. Additional EC_w measurements were taken in the agricultural fields, Orb River, ditch network, network outlet, and sea.

Additional samples were collected for the determination of classical chemical parameters. The analyzed soil and water parameters are listed in Table 3.1. For soil samples, pedological trenches were dug in order to describe the main soil morphological traits and properly sample soil volume per soil horizon type. For water samples, six piezometers (2.0–4.0 m in depth) distributed across the study area were built and sampled in 2016 during both the wet and dry seasons. In order to

compare our data to local reference values, we collected (i) data of fresh groundwater from the French National Data Base on groundwater (www.ades.eaufrance.fr) and (ii) data from "Villeneuve" and "Raysac." Piezometers. The sodium adsorption ratio (SAR) was calculated from the ratio of Na^+ and the square roots of total Ca^{2+} and Mg^{2+} contents divided by two (Richard et al., 1954).

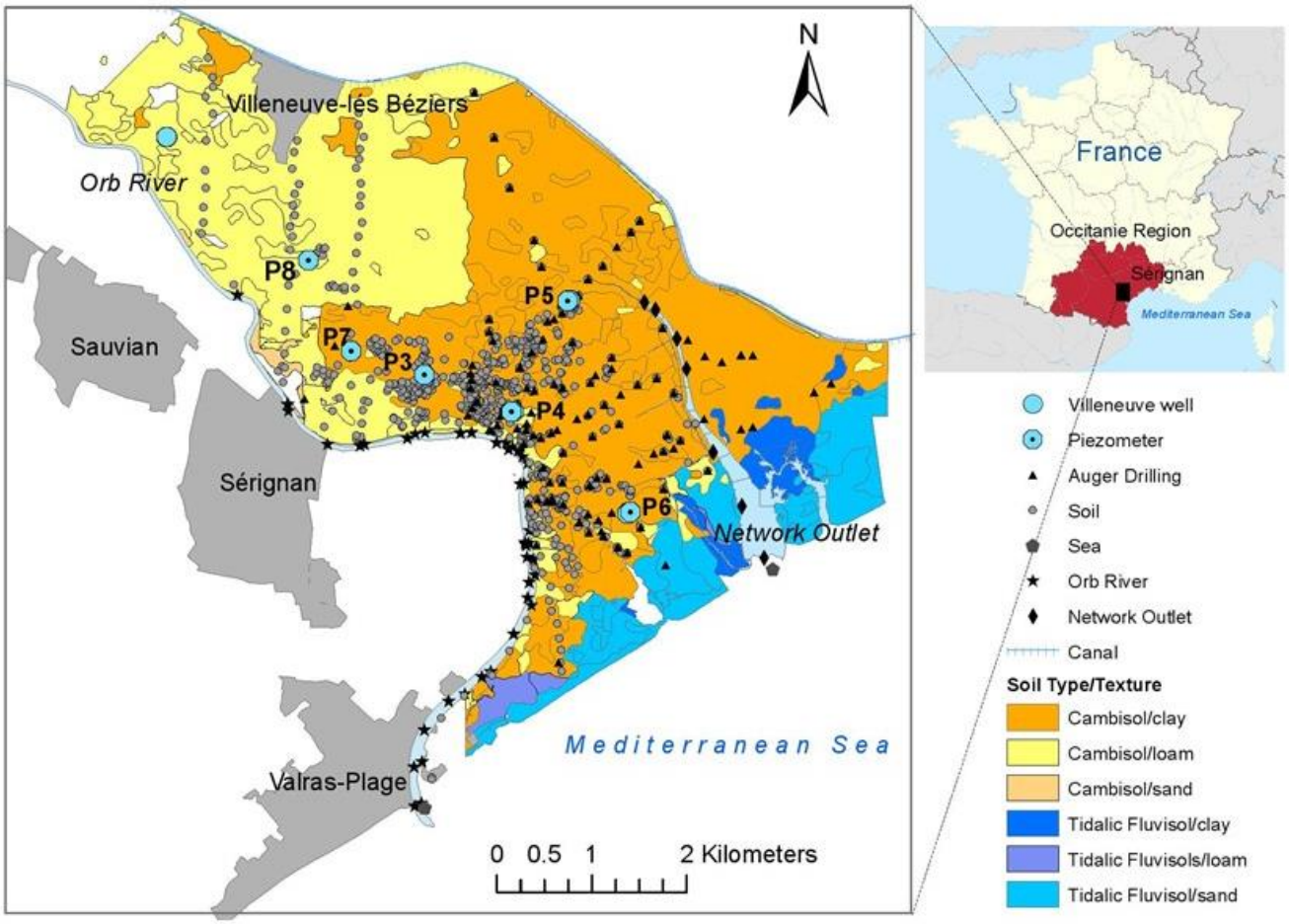


Figure 3.1. Map of soil type and texture in the study area (Sérignan, France) and sampling locations for soil and water (piezometer, auger drilling, Orb River, network outlet, and sea) sampled from year 2012 to 2016.

Data were analyzed statistically via SPSS 22 software. As our data were not normally distributed according to the Kolmogorov–Smirnov test for normality, we used the nonparametric Kruskal–

Wallis test to determine the overall significant difference and the Wilcoxon test to identify particular classes that were significantly different.

Table 3.1. The analyzed soil and water parameters.

	Analyzed parameters/unit	Method	Reference
Soil			
- Auger	$EC_{1/5}$ (dS m^{-1})	Soil electrical conductivity	Rhoades et al., (1999)
- Soil Profile	- pH-H_20, C-Org (%), Texture (%), N (%), C/N, Soluble salts (meq/100 g) (Na^+, Ca^{2+}, Mg^{2+}, Cl^-,HCO_3^-,SO_4^{2-}) - EC_e (dS m^{-1})	-Classical soil analysis -Saturated paste extraction	Pansu and Gautheyrou, (2006)
Water			
- Piezometer	Soluble salts (Na^+, Ca^{2+}, Mg^{2+}, Cl^-,HCO_3^- ,SO_4^{2-}) (meq/L)		Batley, (1989)
- Saturated Zone (Auger Drilling) - Orb River - Ditch Network Network Outlet - Sea	EC_w (dS m^{-1})	Water electrical conductivity	Rhoades et al., (1999)

Landscape evolution data

The data for landscape evolution consist of three main parameters, climate (precipitation, rainfall and evapotranspiration), Orb River discharge and land structure evolutions.

Climatic evolution was studied at regional scale to avoid a potential micro-climate effect associated with one particular weather station. Annual and monthly mean temperature and rainfall time-series were constructed from the extraction of the CRU TS v. 3.24.01 gridded dataset of the Climatic Research Unit (School of Environmental Sciences, University of East Anglia) (Harris et al., 2014). The extracted half-degree gridbox (43°15'N; 3°15'E) is centered near Sérignan and the

time-series concerned the period of 1960–2015. Monthly ET0 was calculated using the formula of Thornthwaite (Thornthwaite, 1948). Annual ET0 was calculated by summing the monthly values. The annual climatic water deficit was calculated as the difference between annual ET0 and the annual rainfall amount.

The Orb River discharges were measured from 1965 to 2015 at Béziers-Tabarka hydrological station (43°22'N; 3°10'E) by DREAL-Languedoc Roussillon (French Ministry of Ecology, Sustainable Development and Energy). The station is located downstream of the catchment, at a distance of 8 km from the coastline. Measured discharges at this station can be considered as a robust estimation of discharge at the final outlet since no main tributaries join the Orb River between this station and the sea. From the time-series, the annual mean discharges (AMD) and the lowest monthly discharges per year (LMD) were calculated for each hydrological year (starting from September 1 of a civil year to August 31 of the next civil year). Climatic and discharge trends were calculated using the 10-year moving average applied to the annual data with R-software.

Landscape structure and land uses are essential components of agricultural systems. A crucial issue was therefore to analyze their evolution in order to identify possible explanatory factors for present soil and water salinization in the study area. The first action was to select two different years using an approach that combines local farmers' perceptions of landscape changes and factual descriptions of landscape design (e.g., aerial photographs from the French National Geographic Institute from 1942 to present). This procedure allowed us to select aerial photographs from 1962, the year prior to governmental coastal land dimensioning programs, and 2012, the year with the highest quality aerial photography and having the same landscape structure as that of 2007–2010 (the period when yield decreases started). The second action was to drive visual recognition and manual digitalization of land uses and landscape structures using ArcGIS software. To validate past land uses and land cover allocations, we conducted additional interviews with elderly farmers.

3.3. RESULTS

3.3.1. Actual Water Salinity

Water geochemistry and electrical conductivity results are presented in Table 3.2. The EC_w measured values from piezometers ranged from 1.43 to 5.74 dS m^{-1} and calculated SAR values ranged from 1.19 to 24.15. Two piezometers, P6 and P8, seemed less affected by salinity from seawater as indicated by their low Na^+ and Cl^- contents. P8 was located in the northern part of the study area, where the distance from the sea is the furthest (5.6 km), and exhibited ionic composition close to that of the "Villeneuve" considered as freshwater reference. The location of P8 contrasted that of P6, which was the nearest to the sea (1.2 km). Piezometers P3, P4, P5, and P7 indicated higher salinity levels: EC_w values ranged from 2.97 to 5.74 dS m^{-1} and SAR values ranged from 5.34 to 24.15. P4 and P7 had the highest Na^+ contents and the highest EC_w values, and the EC_w value of P7 (5.74 dS m^{-1}) indicated that the contents of all cations and anions were high.

Results for EC_w measured at different water sources (sea, river, ditch network, network outlet, and saturated zone) are presented in Table 3.3. The EC_w values ranged from 0.15 to 42.60 dS m^{-1} (excluding seawater). Ranking of mean or median values for EC_w provided the following order from high to low conductivity values: seawater > network outlet > ditch network > river > saturated zone. When comparing all standard deviation values, we noticed that the river was the most variable water source (standard deviation=13.81 dS m^{-1}), followed by the ditch network (10.36 dS m^{-1}) and saturated zone measured during auger drilling (9.08 dS m^{-1}), whereas saturated zones measured by the piezometers had the lowest standard deviation value of around 3.00 dS m^{-1}. The saturated zone from auger drilling had the highest maximum value of 42.60 dS m^{-1}. Categorization of piezometer values in relation to time allowed analysis of the potential seasonal effect. In our measurements, descriptive statistics did not show any significant differences between the two time-series (p-value >0.05).

Table 3.2. Water geochemistry and electrical conductivity in saturated zone of piezometers.

Piezometers	Depth (m)	Ground water analysis (mmol/l)						EC_w (dS m^{-1})	SAR
		Na^+	Ca^{2+}	Mg^{2+}	Cl^-	SO_4^{2-}	HCO_3^-		
P3	2.2	15.96	6.05	11.83	12.32	7.54	13.30	3.46	5.34
P4	2.1	40.48	1.70	3.92	28.81	1.79	13.39	4.72	24.15
P5	1.5	24.52	1.85	3.00	11.92	5.81	12.30	2.97	15.75
P6	1.7	2.83	5.90	5.42	2.60	5.15	6.61	1.43	1.19
P7	3.1	35.74	9.00	12.08	14.55	19.15	17.30	5.74	11.01
P8	4.0	7.30	10.95	11.17	6.89	8.71	10.10	2.95	2.20
E.Rayssac*		0.38	2.74	1.33	0.37	0.64	3.38	0.45	0.27
E.Villaneuf*		2.16	7.25	3.58	2.25	3.11	7.72	1.31	0.93
Sea**		469.57	20.55	106.67	546.61	56.50	1.80	54.75	58.88

**Freshwater; ** Saltwater*

Table 3.3. Water electrical conductivity (EC_w) at different water sources.

	Time	N	Min	Max	Mean	Median	St.dev
				(dS m^{-1})			
Saturated Zone (Piezometer)	Feb/Mar	71	0.53	12.83	4.93	4.13	3.06
	Sep/Oct	50	0.58	12.10	4.82	4.02	3.35
Saturated Zone (Auger Drilling)		101	0.15	42.60	8.00	4.20	9.08
Saturated Zone Depth (m)		114	0.40	4.00		1.80	
Orb River		66	0.28	35.75	8.30	4.23	13.81
Ditch Network		79	2.23	38.17	15.22	14.87	10.36
Network Outlet		12	11.50	26.70	21.04	24.05	5.47
*Sea		2	49.20	60.30	54.75	54.75	5.47

**Reference value; For piezometer the N was total number of measurement inside all 6 piezometers (not number of piezometers).*

3.3.2. Actual Soil Salinity

The correlation between ion concentration (Cl^-, Na^+) and $EC_{1/5}$ shown in Figure 3.2. For the analyzed soil samples (N=16), measured $EC_{1/5}$ values ranged from 0.39 to 1.46 dS m^{-1} and was associated with concentrations measured from saturated paste extracts ranging between 8.3 and 722.1 mg kg^{-1} for Na^+, while between 13.4 and 1086.8 mg kg^{-1} for Cl^-. When fitted with linear models, Na^+ and Cl^- were positively correlated with $EC_{1/5}$. The associated coefficients of determination were 0.84 for $EC_{1/5}$~Na^+ and 0.89 for $EC_{1/5}$~Cl^-.

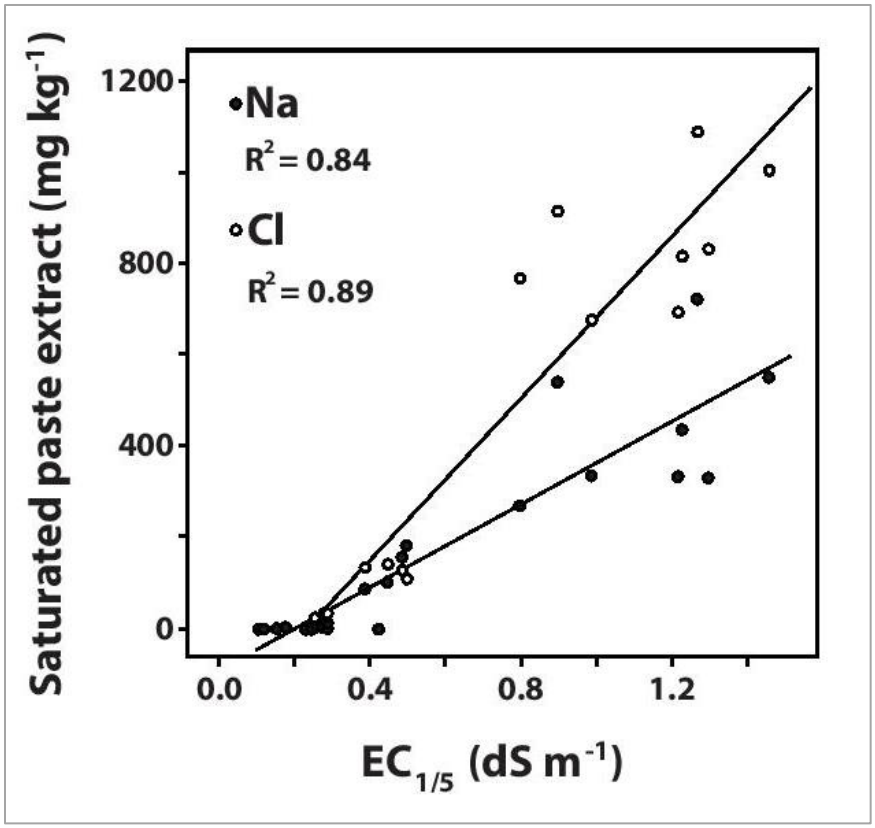


Figure 3.2. Correlation between $EC_{1/5}$ values and the ion concentrations (Na^+, Cl^-) measured from saturated past extracts

In order to investigate the spatial distribution of electrical conductivity, all measured values for both water (from auger drilling) and soil were categorized according to the geographical distance (km) to the river (Fig. 3.3. A1–B1) and sea (Fig. 3.3. A2–B2). The EC_w values categorized according to distance to the river showed a gradual decrease of median values for increasing distance (Fig. 3.3. A1), from 5.18 dS m^{-1} in the vicinity of the river to 1.55 dS m^{-1} for the farthest distance. This trend in median values differed from that observed for variability as informed by inter-quartile values (3rd–1st quartile) and numerical values (maximum–minimum): the 1.0–1.5 km class was associated with higher variability, the 1.5–2.0 km class was associated with higher numerical extend and the >2 km class presents the lowest value regardless of the statistical parameter. For this categorization (EC_w~Orb), statistical tests showed that only extreme distance classes were significantly different from the classes (p-value<0.05). The same representation of EC_w according to distance to sea (Fig. 3.3. A2) offered a different spatial structure: a decrease in median EC_w inter-quartile and numerical values up to 1.5 km, then highest maximum and median

values for intermediate classes of distance from 1.5 to 2.5 km, and a net decrease of median and inter-quartile values for distances of more than 2.5 km. The distribution of electrical conductivity values for soils (Fig. 3.3. B1–B2) did not differ much from that of water, except for the numerical extends of values. Fig. 3.3. B1 presents $EC_{1/5}$ value distributions of significantly different distance classes, namely 0.0–0.5 km and up to 2 km. Once again, the spatial structure of values was less marked when categorized according to distance to coast (Fig. 3.3. B2), and significant differences are noticeable for the 1.5–2.0 and >3.5 km classes.

All $EC_{1/5}$ values were gathered and categorized according to present land use (Table 3.4.). Land uses were listed according to mean $EC_{1/5}$ values. Measurements in both artificial buildings and wetland conditions presented the highest electrical conductivities of 2.06 and 4.58 dS m^{-1}, respectively. Moreover, perennial cultivation systems (arboriculture and viticulture), which are generally located in the northern part of the study area, presented the lowest values, irrespective of the statistical descriptor. Crop, fallow, and pasture land uses had intermediate $EC_{1/5}$ values.

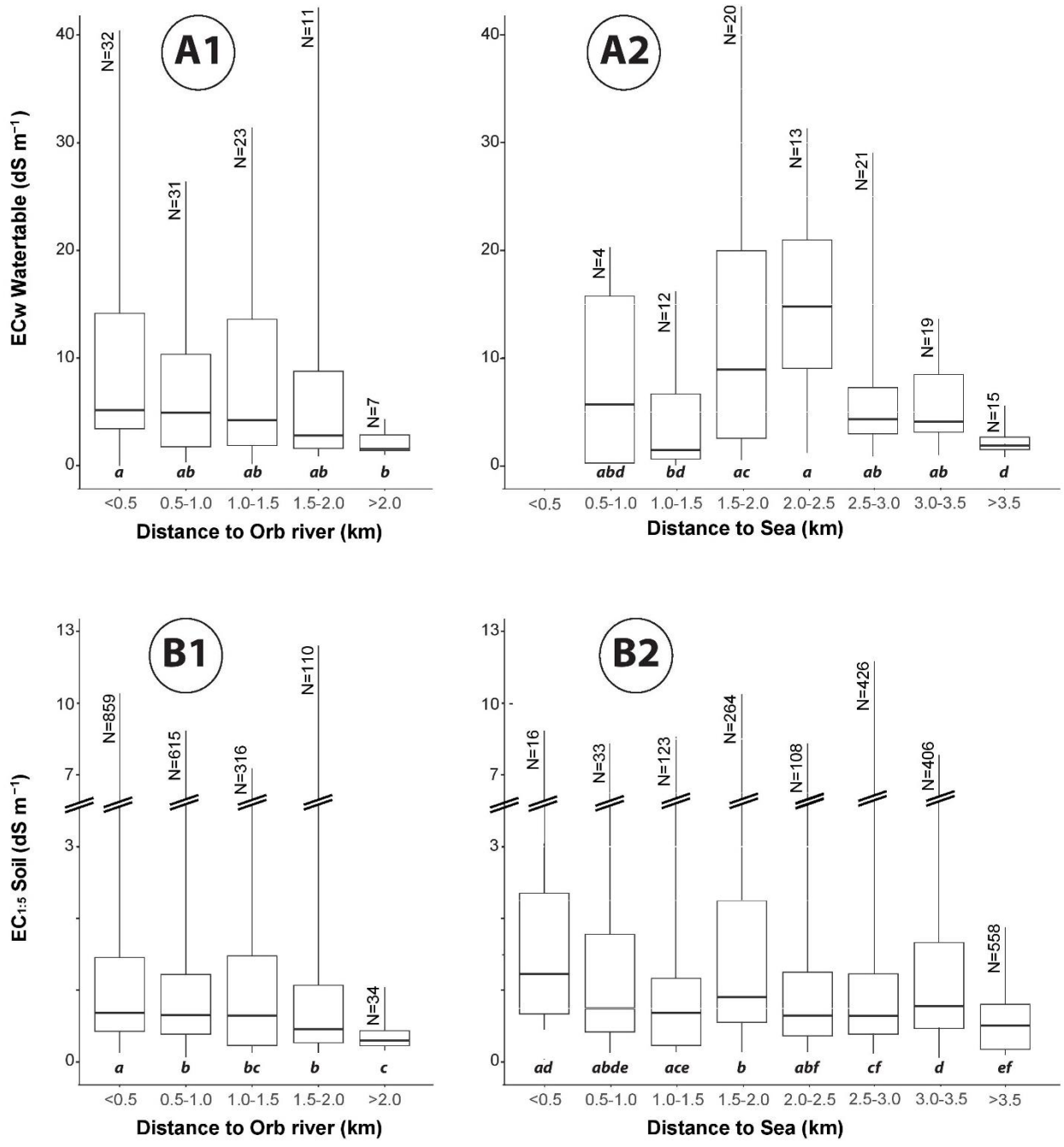


Figure 3.3. Spatial distribution of electrical conductivity measured for both water (from auger drilling) and soil according to the geographical distance (km) to the river (A1-B1) and sea (A2-B2). Different letter means significant difference (p-value <0.05).

3.3.3. Landscape Evolution

Figure 3.4. and Table 3.5. present landscape transitions from 1962 to 2012. Over 50 years, there were substantial changes in both land use and landscape design. From 1962 to 2012, the total number of arable field units decreased significantly, from 1674 to 679 fields, and the associated mean area for vineyards almost doubled, from 0.85 to 1.50 ha. During the same period, the mean size of crop and pasture fields increased by over three times (Table 3.4.). The total area of arable and natural land decreased by about 3 and 29%, respectively. This decrease was contemporaneous with an increase of four times in artificial buildings, mostly located near the coastline. Currently, 21% of vineyard land is new (i.e., where there were no vineyards in 1962) and located in the upper and central parts of the study area, whereas 36% of past vineyard land has now been converted to other land uses close to the river and the ditch network outlet.

Additional results from interviews were related to the traditional landscape design associated with field unit borders, but that could not be recognized on 1962 aerial photographs. Old-field units were systematically bounded by ditch structures, creating a global ditch network for which the ditch network outlet was the outlet connection to the sea. The filling of old ditches created a drastic increase in field area from 1962 to 2012. A detailed analysis of vineyards shows that long-term vineyards (in 1962 and 2012) and new vineyards that were planted between 1962 and 2012 had lower $EC_{1/5}$ values (1.0 and 1.09 dS m^{-1}, respectively) compared to vineyards that were converted (1.3 dS m^{-1}) to other land uses during those periods of time.

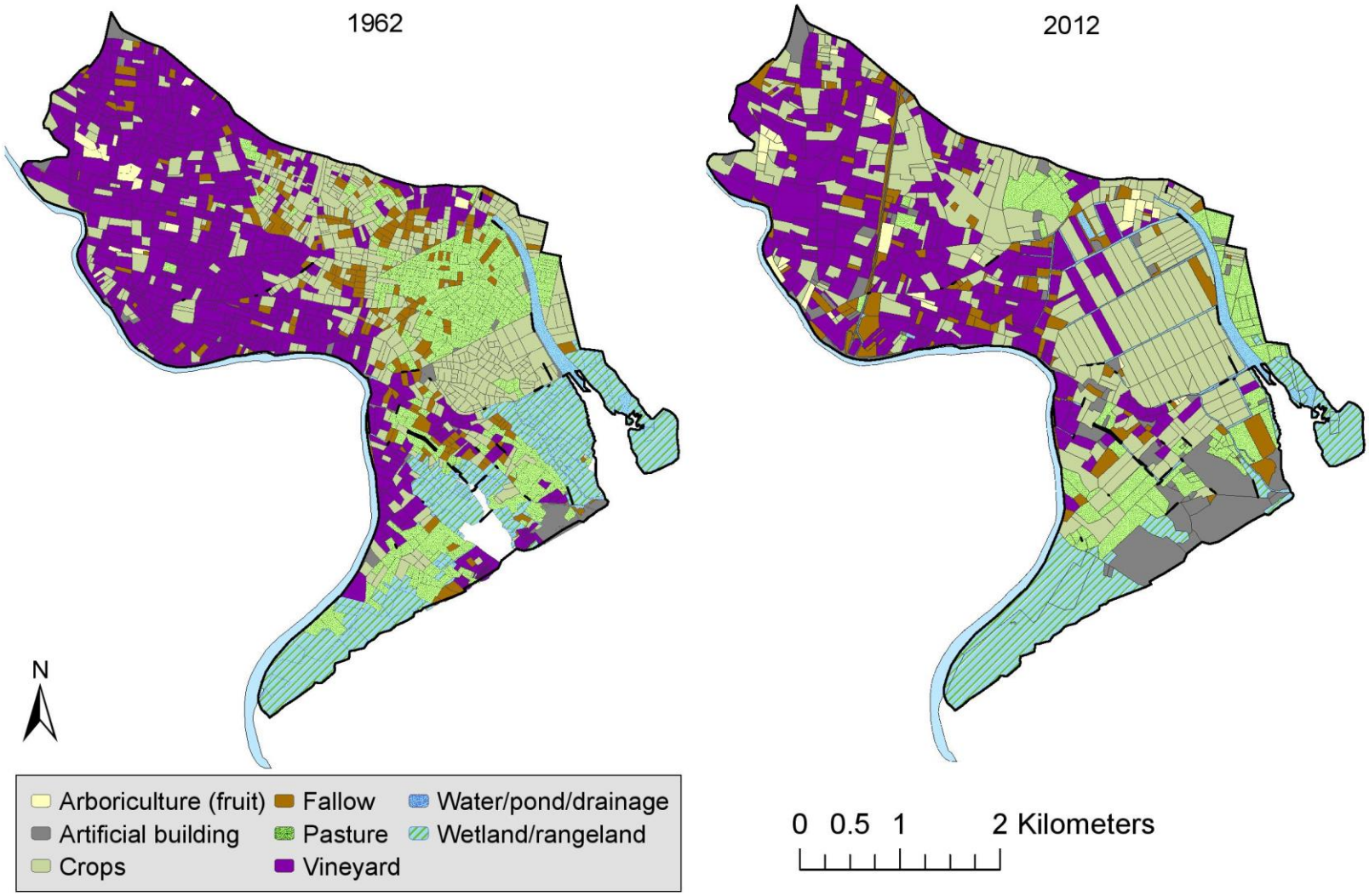


Figure 3.4. Comparison of land use in the 1963 and 2012.

Table 3.4. Soil electrical conductivity ($EC_{1/5}$) according to present land uses

Landuse	N	Min $EC_{1/5}$	Max $EC_{1/5}$	Average $EC_{1/5}$	St.dev
			($dS\ m^{-1}$)		
Arboriculture	27	0.11	3.23	0.87	0.81
Vineyard	1307	0.11	7.97	1.04	1.11
Crop	381	0.11	11.74	1.38	1.73
Fallow	64	0.13	8.91	1.49	1.83
Pasture	46	0.15	8.30	1.66	1.81
Water/pond	8	0.70	3.69	1.42	0.97
Artificial building	14	0.63	5.29	2.06	1.82
Wetland/rangeland	14	0.52	28.1	4.58	7.30

N = number of soil sample

Table 3.5. Statistical parameters of landscape transition from 1962 to 2012

	Number of fields		Min Surface (m^2)		Max Surface (m^2)		Average Surface (m^2)		St.dev Ave.Surface (m^2)		Total Surface (m^2)	
	1962	2012	1962	2012	1962	2012	1962	2012	1962	2012	1962	2012
Arboriculture (fruit)	15	29	2 665	1 242	21 731	35 151	9 760	9 985	5 980	7 490	146 397	289 554
Artificial building	55	103	233	171	79 747	272 749	6 017	15 542	11 625	41 535	330 918	1 600 829
Crops	553	277	594	1 260	66 799	145 402	6 678	20 025	5 816	16 663	3 692 791	5 546 832
Fallow	221	124	299	105	23 032	71 173	5 116	8 721	3 524	11 038	1 130 572	1 081 421
Pasture	370	74	657	1 166	53 624	112 095	6 568	21 566	5 841	24 553	2 430 322	1 595 885
Vineyard	736	299	470	1 547	47 521	142 413	8 473	15 737	6 420	14 529	6 236 035	4 705 483
Water/pond	42	34	78	238	176 423	176 423	6 156	14 434	27 377	30 327	258 539	490 772
Wetland/Rangeland	217	22	385	4 503	240 300	696 433	10 946	76 905	25 041	157 512	2 375 362	1 691 902
Total	2 209	962	5 381	10 232	709 177	1 651 839	59 714	182 915	91 624	305 172	16 600 936	17 002 678

The regional annual means of rainfall, temperature, and ET0 are shown in Figure 3.5. The annual rainfall means ranged from 416 to 1,015 mm with an interannual mean of 655 mm over the 1960–2015 period. The interannual variability was typical under the Mediterranean climate and the trend helped to analyze temporal evolution: the 1960s were more rainy with annual rainfall of around 700 mm, the 1970s showed a transition to the dry years of the 1980s (630 mm per annum), the 1990s were rainy (675 mm per annum) before the driest decade between 2000 and 2010, and since 2010 there has been an increase to around 630 mm per annum.

The mean annual temperature showed an overall increase from 1960 to 2015, with a higher rate at the end of the time-series, and a mean annual temperature of less than 16°C until 2004 and an increase to over 18°C within four years. The consequences of ET0 can be emphasized as follows: Despite a decrease in ET0 between 1960 and 1970, the trend showed a regular increase from 750 to 860 mm by 2005 and stabilizing thereafter. The annual climatic water deficit was highly variable between years with positive and negative values. The extreme years were 1996, when difference between rainfall and ET0 was 240 mm, and 2006 with a difference of 410 mm. The trend indicates an increase during the period; the water deficit more than tripled from around 70 to 230 mm.

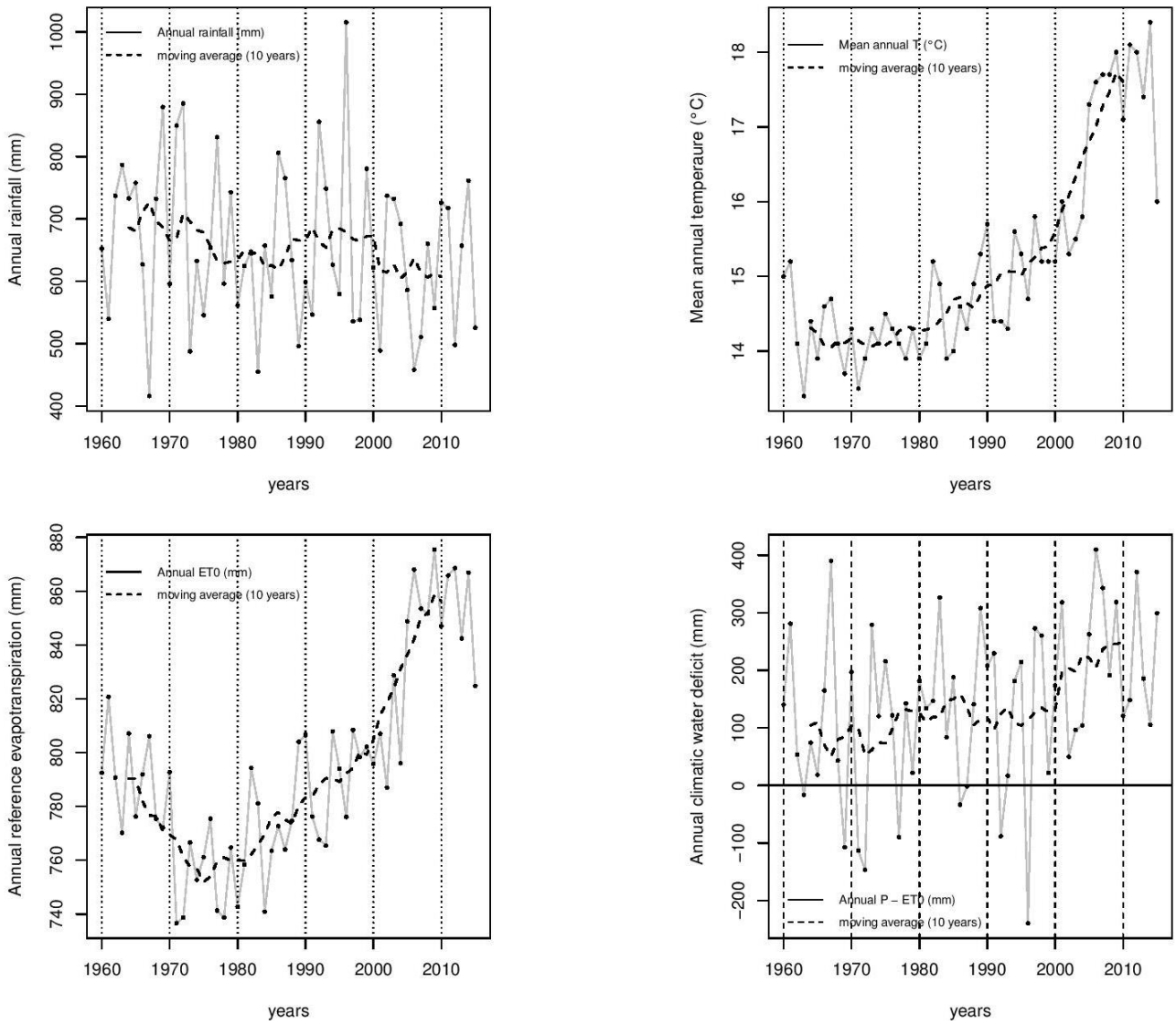


Figure 3.5. The regional annual mean of rainfall, temperature, and reference evapotranspiration (ET0) over the 1960-2015 periods.

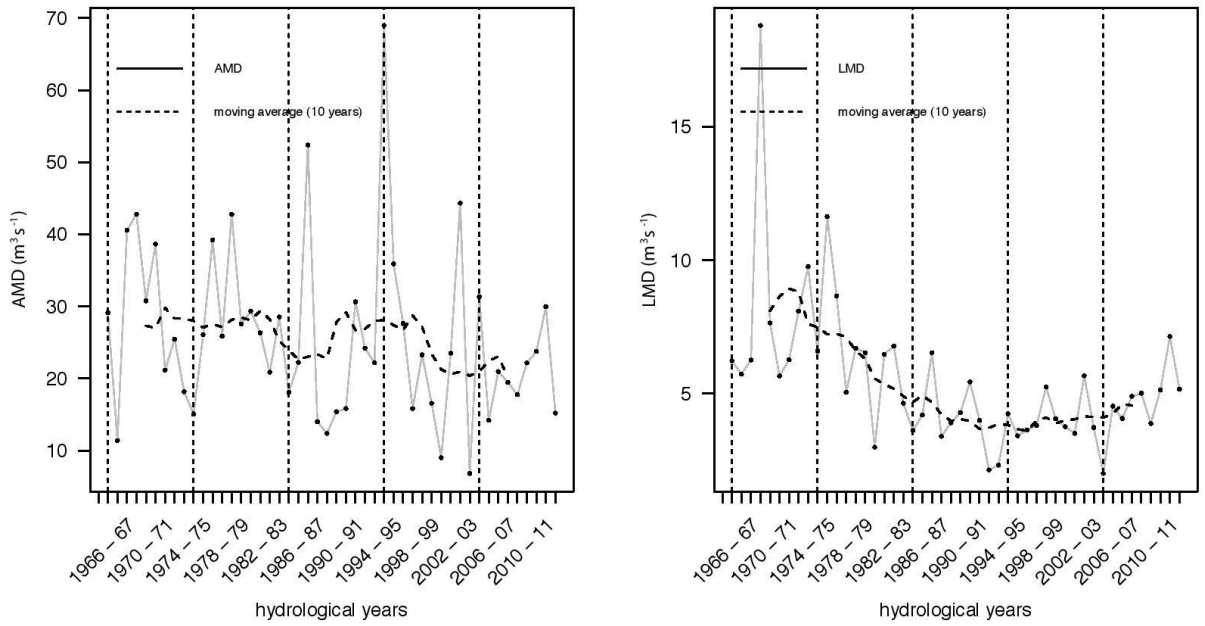


Figure 3.6. The annual mean discharge (AMD) and the lowest monthly discharge (LMD) per year of the Orb River over the 1965-2015 periods at the Béziers-Tabarka station.

The AMD and LMD of the Orb River over the 1965–2015 period at the Béziers-Tabarka station are given in Figure 3.6. The interannual mean discharge value over the period was 25.7 $m^3 s^{-1}$, whereas AMD was highly variable, ranging between 6.8 $m^3 s^{-1}$ in 2004–2005 and 69.0 $m^3 s^{-1}$ in 1995–1996. The trend shows values around 28 $m^3 s^{-1}$ along the time-series with two periods with a lower value of around 20 $m^3 s^{-1}$ at the end of the 1980s and the beginning of the 2000s. However, the time-series of LMD was different: Values were variable and decreased from the 1960s to the mid-1980s, when low flows were fewer and more stable. Three years exhibited particularly low LMD values, namely 1993–1994 (2.1 $m^3 s^{-1}$), 1994–1995 (2.3 $m^3 s^{-1}$), and 2005–2006 (2.0 $m^3 s^{-1}$).

3.4. DISCUSSION

3.4.1. Actual Salinity

Land salinization management cannot be properly implemented without preliminary knowledge about spatial distribution and evolution of salinization (Michot et al., 2013). Our primary action was to quantify the spatial distribution of soil and water salinity. Considering the total area of agricultural land in our study area (170 km^2), we chose electrical conductivity (EC_w and $EC_{1/5}$) rather than classical extract paste measurements that exclude massive data collection. This choice was supported by other studies demonstrating that electrical conductivity is a proper variable for the estimation of salinity (from seawater origin) (Richards, 1974; Rhoades, et al, 1997; Gkiougkis et al., 2015). Locally, geochemical analyses confirmed that the main ions in piezometer saturated zones were Na^+ and Cl^-, even if additional chemical species could contribute to ionic strength (Table 3.2.; Fig. 3.2.).

At the landscape scale, seawater is the main source of Na^+ and Cl^-, but the analysis of the spatial structure of both $EC_{1/5}$ and EC_w (Fig. 3.3) shows that the spatial effect of the river on both soil and saturated zone salinity is greater than that of the sea. The river contains salts and is the most variable water source (standard deviation = 13.81 dS m^{-1}) (Table 3.3.). The observed variability in the river water is linked to river discharges and seawater intrusions over time: depending on river discharge, seawater intrusion could be facilitated. Thus, decreased flow resulted in increased salt concentration (Isidora, 2011). Complementary observations indicated that $EC_{1/5}$ values increased with increasing soil depth, then for higher proximity to saturated zone where salt accumulated as suggested by Greene et al., (2016).

Based on these observations, we assumed that underlying salt influxes were (i) from the sea to the river through seawater intrusion occurring during low flow periods, (ii) from the river to the saturated soil zone through river water infiltration at the embankments, and (iii) from the deep saturated zone to the root zone through waterlogging variations during the wet season and potential capillarity increases in the dry season. However, in order to determine the landscape salt budget, it is necessary to discuss potential salt outflows. For this, we focused on the ditch network, whose primary function is water and salt exportation as described by De Louw P.G.B. et al., (2011; 2013).

The collected water sources were (i) water that percolates within the soil profile, (ii) water from the saturated zone, and (iii) excess runoff water that cannot infiltrate. Runoff water could be considered negligible in our context as field borders are often shaped to maximize within-field water infiltration (water harvesting technique). Origins and residence time of collected water in the inefficient network explain EC_w values associated with ditch network and network outlet: when collected, water in the percolation and saturated zones has similar conductivities when compared to that observed in the soil body, but the salts will concentrate due to evaporation. Higher mean and median values for the network outlet are linked to its geographical position, which is connected to the coastline and subjected to potential direct seawater contributions as reviewed by Daliakopoulos et al., (2016).

3.4.2. Conceptual Model and Landscape Evolution

Previous findings and hypotheses allowed us to construct a preliminary version of the conceptual model proposed in Figure. 3.7. by combining main landscape constitutive elements: (i) the sea, the main source of salts, (ii) the river, the main source of fresh water and a source of saline water, (iii) the atmosphere, source and sink of fresh water, (iv) the land system, composed of three sub-systems (land use and cover, which modifies local water budget; soil, which is the main support for agricultural production and the reserve for plant roots uptakes; and the water table or saturated zone, which is the main plant available water reserve), and (v) humans, whose actions modify all constitutive elements. All of these elements interact through processes mentioned in the conceptual model (Fig. 3.7.), for which we assumed that the system is in equilibrium. When applied to our study, the first order assumption was to consider that old conditions (pre-1962) allowed the reaching of the equilibrium state favorable for wine production by maintaining salinity at a sustainable level. Subsequently, in order to explain actual salinization, we used landscape evolution analysis to identify developments responsible for the system disequilibrium as proposed by Kingwell and John (2007) and Payen et al., (2016).

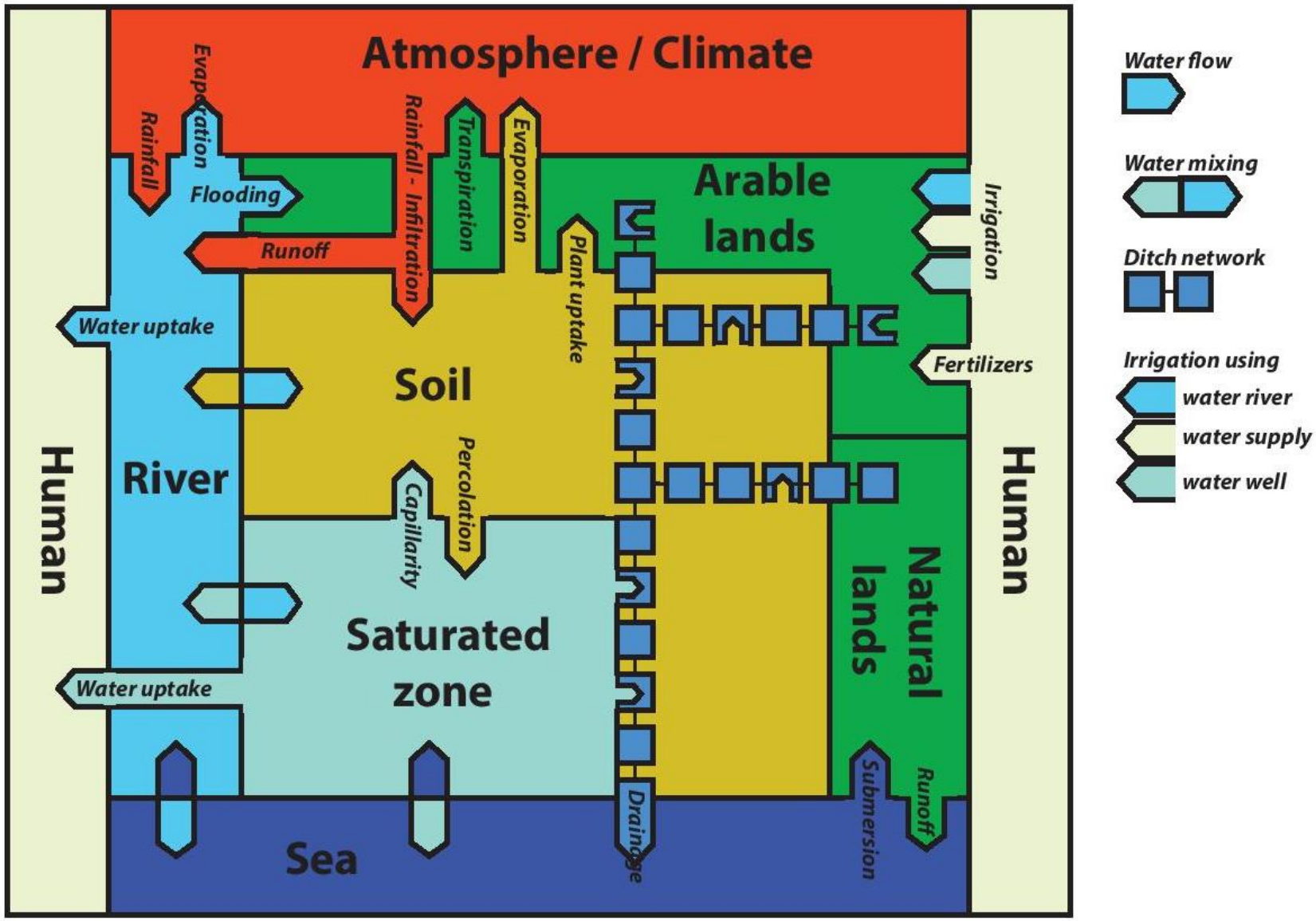


Figure 3.7. Conceptual model of water fluxes in a coastal area affected by salts.

Among the changes cited by farmers is the evolution of landscape structure and the decrease in annual rainfall and river discharge. This disclosure by farmers demonstrated that for them the main issue in soil salinity management is the total available amount of fresh water for salt leaching from the soil root zone to the ditch network. Our aim was to conduct this time-series analysis for the period of 1960–2015 for each of the following conceptual model constitutive elements: atmosphere, river, and land system.

For atmosphere, the ET0 significantly increased over the 1960–2015 period. Time evolution for rainfall was less clear but nonetheless indicated a slight decrease with high interannual variability. Over time, the combination of these two trends for ET0 and rainfall has led to the multiplication of the climatic water deficit by a factor of 3 over the last 50 years (Fig 3.5.). When considering the seasonality of the Mediterranean climate, the consequences of such water deficit during the dry season include the following: (i) increased evaporation, leading to salt precipitation within the soil profile or on the soil surface, (ii) increased plant water uptake, leading to dryer conditions in the root zone, (iii) increased capillary forces acting in the soil body, favoring water and salt transfer in soil profiles from saturated to non-saturated zones (i.e., from the deeper soil horizon to the root zone) (Metternicht and Zinck, 2008), and (iv) decreased plant water uptake, as under SAS conditions in the root zone, plants are incapable of efficient water uptake (Rengasamy, 2010). Subsequently, the increased water stress for plants may lead to their mortality.

For rivers, analysis of AMD revealed no clear trend over time (Fig 3.6.). The evolution of annual water flow in the river could not be interpreted in terms of salt pressure on the study area. However, despite interannual hydrological variability, LMD values have been low and stable since the mid-1980s. The stable LMD value is close to the French legal value of 2.5 m^3 s^{-1}. One realistic explanation of such a value is the regulation by water users of the discharges along the river over the past 30 years. Whatever the explanation might be, the consequence is increased seawater intrusion into the river.

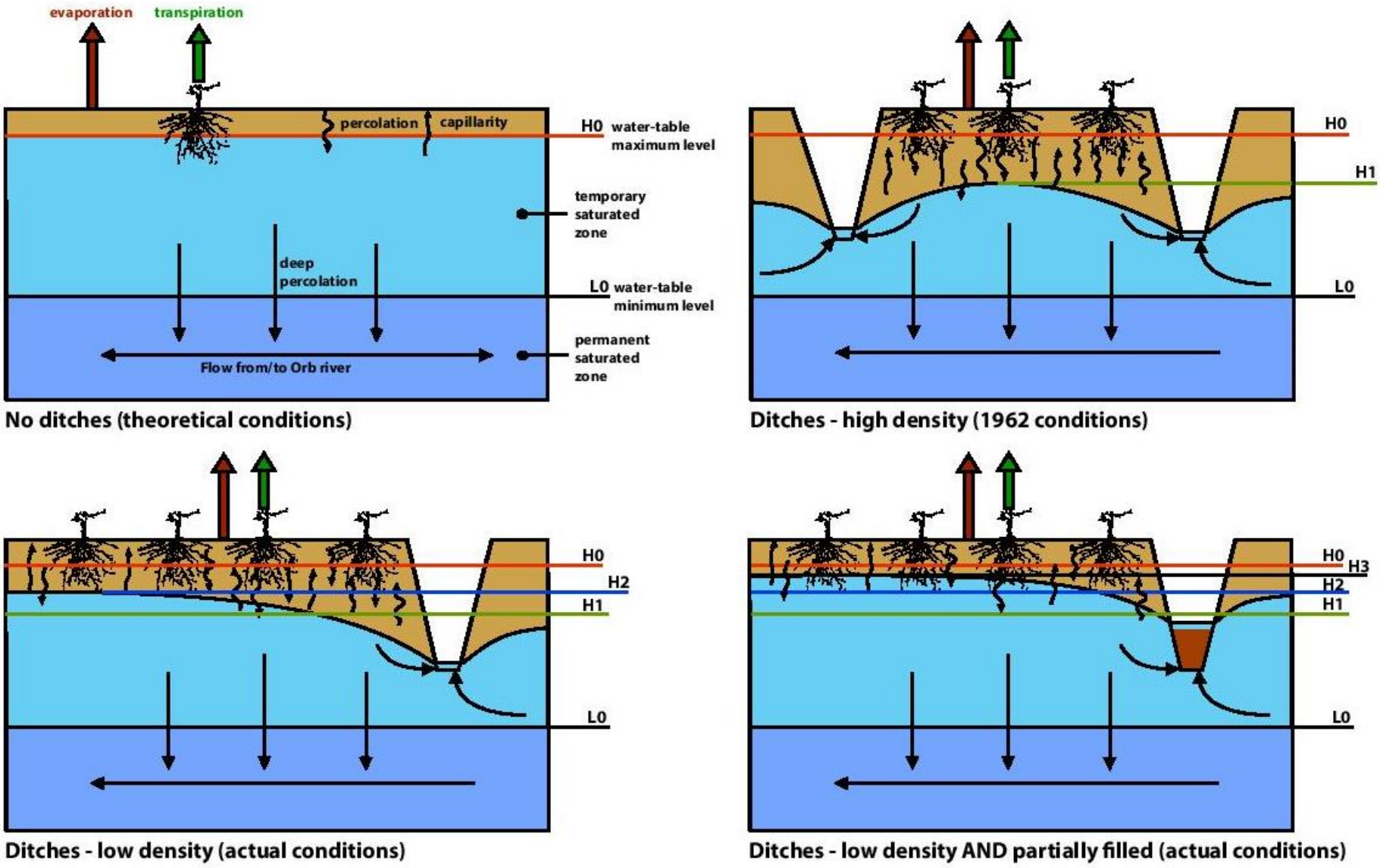


Figure 3.8. Interaction between the ditch network and process controlling water and salt availability in the root zone for different land system conditions.

For the land system, global landscape transition analysis demonstrates that land use and land dimensioning strongly evolved over the past 50 years. This landscape evolution was responsible for water table level evolution, as identified by Wu et al., (2014) as an important factor influencing soil salinization in alluvial plain. Figure 3.8. illustrates the interactions between the ditch network and processes controlling water and salt availability in the root zone for different land system conditions. In this representation, the main processes are (i) evaporation and plant transpiration inducing decreased water content in the root zone, (ii) water infiltration from the soil surface to topsoil, (iii) capillary forces that create water and salt transfer from the saturated zone to topsoil horizons, (iv) percolation of water and dissolved salts (leaching) from the root zone to deeper soil horizons and the saturated zone, (v) water exchange and mixing in the saturated zone or, more generally, groundwater in the alluvial compartment and the associated river, and (vi) drainage of the saturated zone by the ditch network. For the first set of land system conditions, corresponding to a theoretical condition without the ditch network, the maximum water table level rises to near the soil surface (H0). Here, under saline groundwater conditions the consequence is salinization of the whole soil profile. The second set of land system conditions corresponds to the high density of ditch structures as observed prior to 1962. Here, ditch density leads to a maximum reduction of the water table to the maximum H1 level. The third condition indicates the evolution of drainage density that occurred between the 1960s and 2012, the effect of which was to induce less water table reduction when compared to the previous set of conditions, with a reduction in the maximum water table level at the H2 elevation. The last condition is equal to the third but integrates the maintenance default of the drainage network responsible for ditch filling and decreasing the overall network efficiency for water table control, with a maximum H3 level for the saturated zone. For this last set of land system conditions, the increase of the H3 level leads to waterlogging and salt accumulation in the root zone.

Another dimension of the land system is the evolution of land use type allocation in space and time (Fig. 3.4.). The overall transition was to a net decrease in total area of both vineyards and pastures, whereas the total area for crops and building increased. With time, most of the vineyard located close to the coastline was replaced by residential buildings. Historically, this substitution might be due to lower productivity of wine in the area as a result of severe damage by salinization, then a distribution of cultivated plants due to the tolerance to soil salinity as observed by Bas et al., (2017) in coastal cropping systems. It also might be due to economic pressure induced by increasing

numbers of tourism activities over the past 50 years. The tourism dimension was responsible for the increased number of residential buildings in the coastal area and reduced total area of wetland and rangeland, which collected saline water from fields, creating a buffer area between the coastline and arable lands. Mechanization was as a major driver for landscape change: mechanization has triggered increasing size of field units and the associated transition from horses to engines has rendered pastures useless.

When viewed together, the abovementioned developments have been responsible for the disruption of the system equilibrium, favoring overall salt accumulation in the soil root zone. The main driving factor of water and soil salinization in this area was mainly influxes of seawater through the river, which intruded into the saturated zone and entered the soil through the seasonal variation in waterlogging and capillarity. Natural and anthropogenic evolution such as land use, rainfall, temperature, river discharge, and agricultural practices, which are noticeable at the landscape scale, impacted the salinization process. Mechanization practices have led to the reshaping of agricultural fields from 1960 to 2012 and caused the modification of the drainage channel network and reduced efficiency of drainage function to drain saline water from the field. Declining rainfall and increasing temperatures over the last decade have directly impacted local water balance in the study area and probably changed the intensity of capillary saline water fluxes. The hydrological regime of the river played a central role, since the flood events and pumping are sources of field submersion with fresh water, and low flow magnitudes defined seawater intrusion into the river and associated alluvial groundwater. Thus, the solution may lie in overall river catchment water policy and local adaptation involving land use repartitioning, soil management, submersion and drainage practices.

3.5. CONCLUSION

Investigations of both soil and water electrical conductivity of agricultural lands in Sérignan allowed us to characterize spatial structures of soil and water salinity affecting vine production and to identify the river as a preponderant factor in land salinization. We subsequently proposed a conceptual model for water fluxes in the coastal agricultural area. Main landscape elements (sea, river, atmosphere, land use and cover, soil, water table, and humans) and processes constitute this simple model. This model, when coupled with time analyses of climate, river, and land systems since 1960, helped us to understand the main evolutionary processes responsible for the disruption of system equilibrium, favoring overall salt accumulation in the soil root zone: (i) a decrease of freshwater influx due to river discharge evolution, (ii) an increase of freshwater outflow due to climate evolution, (iii) an increase in saline water influx due to seawater intrusions into the river, and (iv) a decrease in saline water outflow due to ditch network evolution.

Preliminary Statement

Soil salinity cause soil degradation and disturb soil function, as it disperse soil particles and then reduce soil aggregate stability. Consequently, this particle dispersion is often associated to global decrease of voids volume and connectivity, then a decrease of soil hydrodynamic properties (soil hydraulic conductivity and water infiltration potential).

In order to remediate soil salinity we need to identify soil indicators that can be easily measured for determining the salt effect on the breakdown of soil aggregate and the chance to leach down the salt from soil layers.

In this this chapter, we determined soil aggregation factors, soil properties (physico-chemical) and different agricultural practices based on the different land uses. Afterward, we discuss soil aggregate stability as the indicator of soil functioning and soil remediation for salt affected soil at soil profile scale from topsoil to subsoil in different land uses.

Note : this CHAPTER FOUR will be the second paper of the PhD. It explains why we adopted an article like style for this chapter writing and structuration. In addition, some redundant element could be found with the CHAPTER THREE, which is the first paper already published.

CHAPTER 4.

SOIL AGGREGATE STABILITY AS SOIL FUNCTION INDICATOR ON SALT AFFECTED SOIL: A SOIL PROFILE ANALYSIS

ABSTRACT

Soil aggregate stability is an indicator of soil quality and soil degradation. The objective of this study was to identify soil aggregate stability and its aggregation factors of intrinsic soil properties along the soil profile of degraded agricultural salt affected area in the coastal area. We conducted field investigations by collecting soil and groundwater samples from six locations that are different in land use and its agricultural management (2 trellised vineyards, 2 gobelet vineyards, 2 fallow fields). The soil samplings were from different soil depth (0.0-1.4 m), at superficial tillage horizon (LA-horizon), organo-mineral horizon (A-horizon), and young structural horizon (SJp-horizon). Fast wetting was the method to measure the mean weight diameter (MWD) for soil aggregate stability. Results showed that the MWD value of this area was ranging from 0.28 mm to 1.10 mm with the mean was 0.52 mm (unstable soil aggregate). MWD values decreased by increasing of soil depth. By the way, structural horizon (>80 cm) generally had very unstable soil aggregate with a MWD value around 0.4 mm. The aggregate stability of topsoil horizons is more variable and was more influenced by agricultural practices (land use), soil organic content and clay content. While at deep horizon other intrinsic soil properties including salinity, should be added to explain overall low stability. The ESP values were less than 15% in all fields, except F1-1-1 at young structural horizon (>90 cm). Even though the ESP values were low (2-12 %), it had low soil aggregate stability. The results indicate that salt (Na-Cl) might be precipitated in the vadose (pore) zone instead of exchangeable soil part. In order to restore the soil quality, priority seems to be dealing with problem of compaction and efficient salt lixiviation.

Key Words: Soil aggregate stability, Soil profile depth, Land Use, Salinity, Agriculture

4.1. INTRODUCTION

Soil salinization is one of major global soil degradation process that threatens global agricultural sustainability. Salinization is an increase of soil soluble salts concentration affecting agricultural yield, environmental quality and human welfare (Rengasamy, 2006b) that could have a natural or anthropogenic origin. According to FAO in 2000, around 830 Mha of soil worldwide are considered to be salt-affected in almost all continents (Australia, America, Europe, Asia, and Africa) (Martinez-Beltran and Manzur, 2005). In the European Mediterranean, it is more prominent in the coastal area (Daliakopoulos et al., 2016). Then, in order to achieve global sustainable goals it is crucial to build good practices strategy to reduce soil salinity effects and or to restore soil quality (then soil capacity to function).

Soil aggregate stability could be considered as a proper soil quality indicator as defined by Doran & Parkin (1996) while it depends on several soil properties and environmental biotic factors. Soil aggregate stability is defined as the ability of soil to keep its particles attachment under mechanical and physicochemical stresses (Le Bissonnais, 2016). In soil aggregate analysis soil aggregate are often categorized, based on their size: macro aggregate (>250 µm) and micro aggregate (<250 µm) (Oades, 1984, Edwards and Bremner, 1967). Overall stability of aggregates depends on the agglomeration of mineral particle (silicate) with organic and inorganic substance materials that affect aggregate formation and stabilization; formation and stabilization of aggregate determine soil aggregation (Allison, 1968). The interrelationships of physical, chemical and biological reaction of soil properties exist under formation and destruction of soil aggregate (Kay and Angers, 2002). Soil aggregates regulate distribution size of soil pores that supporting soil water infiltration, aeration, movement of soil organisms and carbon sequestration. Thus soil aggregate plays important role for water and nutrient cycling (Six et al., 2004).

Soil aggregate stability could be considered as a proper indicator of soil to functioning and to be remediated in salt affected soil conditions. In those conditions soil organic carbon (SOC) stocks decrease (Setia et al., 2013; Wong et al., 2010), whereas in non-salty conditions, SOC enhance clays flocculation and favors bonds formation with clay particles and polyvalent cations (Tisdall and Oades 1982; Six et al. 2004; Abiven et al. 2009). Additionally, high concentration of sodium in exchangeable part of soil would provoke clay dispersion that destabilizes soil structure (Pearson

and Bauder, 2006). Moreover, soil dispersion would reduce efficiency of surface water submersion practices dedicated to create soil solute leaching out of the soil-rooting zone.

Associated to soil aggregate stability, Exchangeable Sodium Percentage (ESP) is widely accepted as relevant indicator for evaluating soil aggregate dispersion. The threshold be more than 15% (Quirk, 2013) would result in soil dispersion with the condition of low soil electrolyte conductivity. However, a different result from Crescimanno et al. (1995) suggested that there is no threshold of ESP, as soil aggregate destabilization could appear at ESP range of 2-5 %. Furthermore, Odeh and Onus (2008) mapped spatial distribution of salt affected soils and predicted the risk area of soil structure dispersion in semi-arid region of NSW Australia using aggregate stability score (ASWAT). They suggested that ESI (Exchangeable Sodium Index), which is incorporation of Electrolyte conductivity (EC) and ESP, has stronger correlation with mechanical dispersion score compare to only using ESP alone. They suggested that aggregate dispersion not only determined by salinity but other factors like soil pH and clay mineralogy (Odeh and Onus, 2008). The interaction between many different factors; soil properties, environmental condition, soil management and plant influence, determined the complex dynamics of aggregation (Bronick and Lal, 2005).

Soil aggregate stability is considered as a proper soil quality indicator. Previous studies have investigated the relation between aggregate stability and intrinsic soil properties as aggregation factors (Le Bissonnais et al., 2018; Regelink et al., 2015; Six et al., 2004; Totsche et al., 2018). But these studies were generally conducted under non-degraded soil conditions at topsoil layers. However, little is known about destabilization of soil aggregate along the soil trench e.g. with soil depth, particularly in salt affected conditions. Thus the objective of this study was to assess soil aggregate stability at different soil depths for different land uses, and to investigate the correlation between aggregate stability and classical aggregation factors.

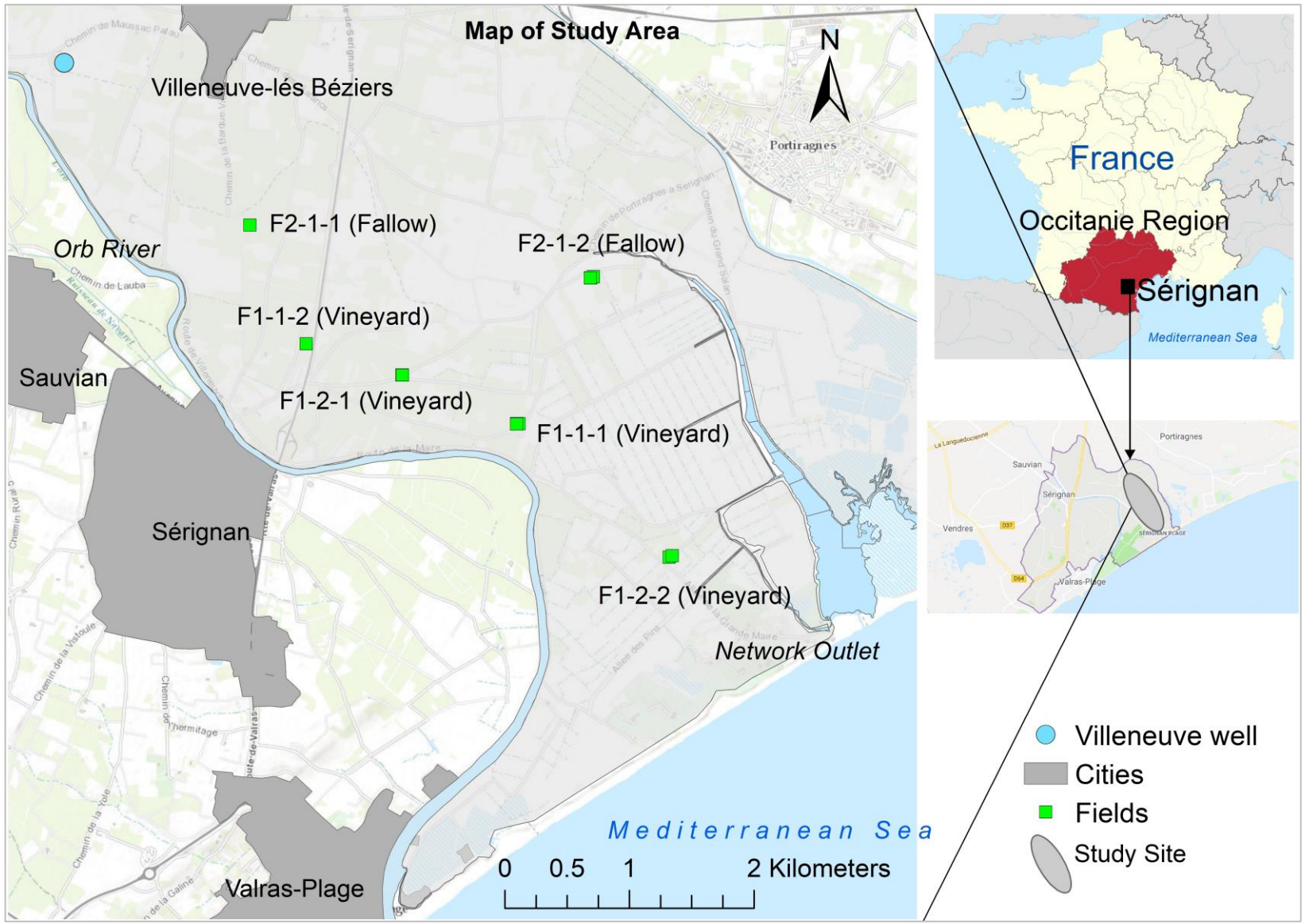


Figure 4.1. Sampling site of six different locations of soil trenches

4.2. MATERIALS AND METHODS

4.2.1. Data Collection And Measurement

The determinations were carried out in February 2017, on six field units located on the Serignan study site within the same soil type condition: Calcaric Cambisols (WRB, 2006). Those fields units were selected in order to represent the local diversity of vineyards:

- F1-1-1 and F1-1-2 fields are trellised vines, a modern system with vines tied up to metal wires to maintain rows. The age of these vineyards is about 30 years old and its soil texture varies between clay and loam;

- F1-2-1 and F1-2-2 fields are Gobelet-trained vines, an old traditional system without metal wires. The age of these vineyards is at least 50 years old;

- F2-1-1 and F2-1-2 fields are fallow fields where vines were grubbing-up three and ten years ago respectively.

For each field unit, we conducted a field investigation based on both pedological trenches description and piezometer monitoring. For pedological trenches, soil description and classification was done following the guidelines for soil description (FAO, 2006). Then, the bulk density (core method) was determined by sampling triplicate undisturbed soil cores (95 cm^3) horizontally in each layer and soil samples were collected at each horizon, corresponding to soil depth ranging from 0.0 to 1.5 m.

Soil samples were divided to allow both aggregate stability and classical soil test determinations afterwards. For soil aggregate stability measurements, sub-samples were air dried (room temperature) for 3-5 days and sieved to 3-5 mm size. Fast wetting method of Le Bissonnais, (1996) ISO 10930 (2012) was the method to measure mean weight diameter (MWD) for aggregate stability. About 5 g per soil samples were soaked in deionized water for 10 minutes for the three replications, then wet sieving (50 µm) with ethanol and dried oven at 40°C for 24 h. Next, the samples were dried sieving using different diameters of soil sieve (<2.00 mm, 1.00-2.00 mm, 0.50-1.00 mm, 0.25-0.50 mm, 0.10-0.25 mm, 0.05-0.10 mm, <0.05 mm). We collected and weighted the mass of each fragment size, while for the least fraction (<0.05 mm) it is estimated as the difference between total initial mass and the sum of the other six fractions. The aggregate stability for each breakdown mechanism is expressed using the mean weight diameter (MWD), which is the sum of the mass fraction of soil remaining on each sieve after sieving multiplied by the mean aperture of the adjacent mesh (Calculated MWDs range between 25 µm and 3.5 mm using the set of six sieves).

For classical soil-test determinations, we followed the method by Pansu and Gautheyrou, (2007). The relevant measured parameters were: soil texture (pipet method), pH-H_2O, soil organic matter (SOM) (dry combustion of C-Organic), soil electrical conductivity of saturated paste extraction (EC_{sp}), CEC, Fe/Al Oxide and carbonate content. Soil determinations were done by an official certified lab (Cirad soil lab – Montpellier, France).

Complementary to soil analysis, water samples were collected in piezometers in order to analyze soluble ions in saturated zone. Piezometers are located in the vicinity of selected field units (5 meters far) and are between 4m depth. Water determinations were done by an official certified lab (Eurofins IPL Nord SAS, France). These samples were compared to those from a well located in

the Orb alluvial aquifer at Villeneuve station using data from the ADES national database (BSS code: 10401X0255/MAUSSA).

4.2.2. Data Analysis

All soil properties were analyzed statistically. We did ANNOVA to see difference of soil properties between different horizons; and the difference of MWD and EC_{sp} between different land uses. The data were analyzed with SPSS 22 and R software. In addition, ESP values were calculated following Richard (1954). The ESP values were obtained from ratio between Na^{+} in saturation extraction (meq/100g) and cation exchange capacity (CEC) from the soil analysis.

4.3. RESULTS

4.3.1. Soil Morphology

The description of the six soil profiles for each soil horizon is presented in Figure 4.2. There was a significant specificity for all six field units, which was the overall effect of soil tillage on soil horizonation: whatever the soil type, the soil profile is influenced by soil tillage with the A-horizon in the first 0.4 m (deep tillage), the deep A-horizon at depths between 0.4 m and 0.6–0.7 m. There are two main soil horizon types which organo mineral horizon (A) and young structural horizon (SJp). In addition, at the deeper horizon (> 0.7 m) we found temporally waterlogged horizon (SJpg) on structural layer at the fields of F1-2-1, F1-2-2 and F2-1-2. A generalized compacted layer is observed at the depth of 0.4 m to 0.6 m with high associated bulk density, over 1.5 g cm^{-3}.

The main soil texture class in almost all soil profile horizons is silt loam (then not reported on Figure 4.2). The silt fraction is over than 50% for all soil depths, exception from the field F1-2-2 presenting a layer of sandy texture at the depth of 0.7 m. The soil color is primarily hue 7.5 YR4/2 with chroma ranging from 4/2 to 4/3 for brown color. The shape of soil structure is mainly sub angular blocky, except for filed F2-1-1 showing a platy shape for the deeper depth (> 0.9 m). The soils contain high carbonate content as it was sparkling when react with hydro chloric acid solution. On the soil surface there were low stoniness (< 2% of the soil surface) and covered by grass in almost fields, except for field F1-1-1, where the soil surface was crack when it was dry and swelling when there was a heavy rain.

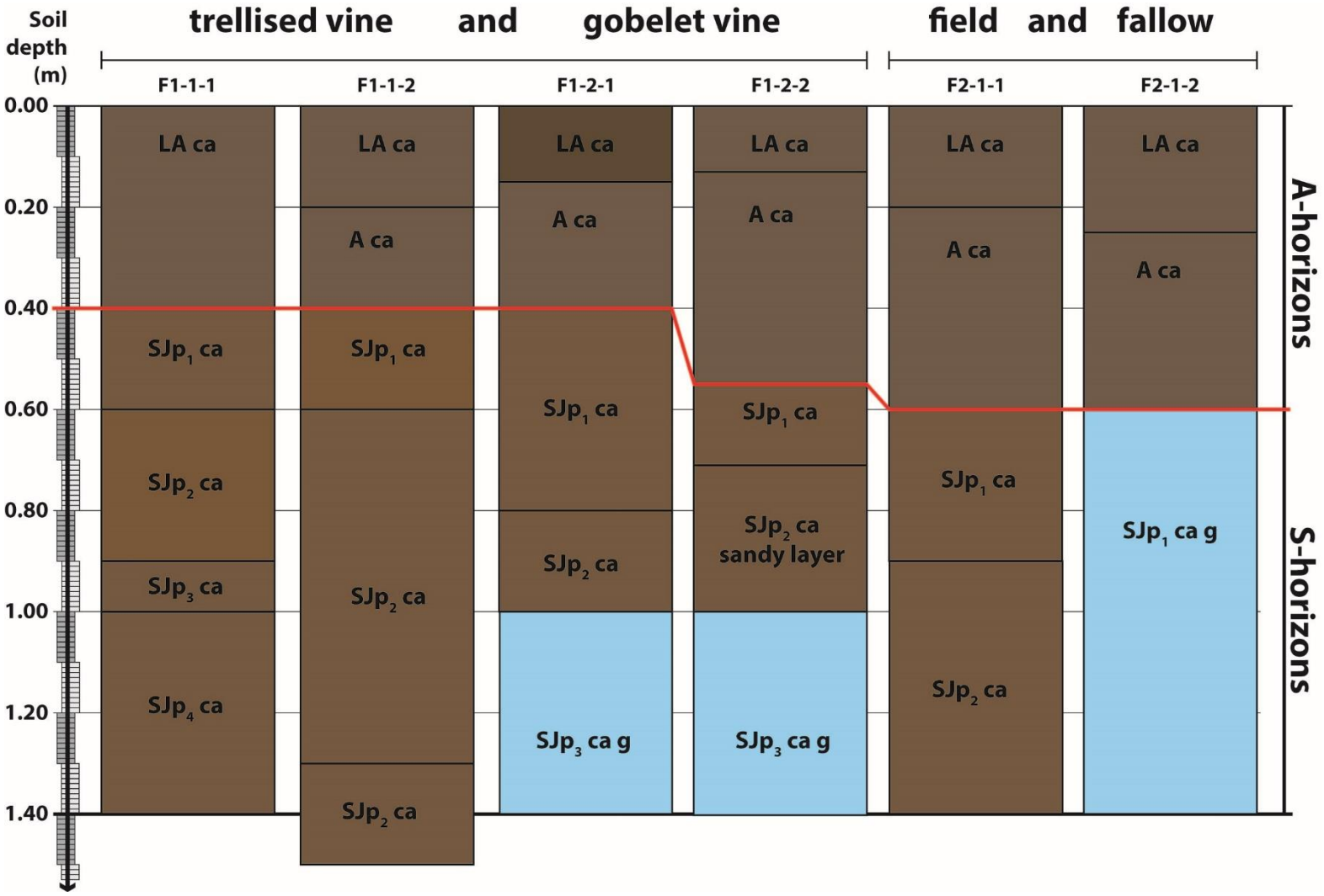


Figure 4.2. Soil profile description of six different locations (LA = Superficial Tillage Horizon; A = Organo mineral horizon; SJp = Young Structural Horizon at deep soil profile; ca = horizon with carbonate content; g = temporally waterlogged horizon.

4.3.2. Soil Physio-Chemical Properties

Measured pH value within the twenty first-centimeters range from 7.96 to 8.87 and reach maximum value for deep SJp-horizon. Total carbonate content follow the same general trend with soil depth as pH values one, with a median value of 6.50 $mg.kg^{-1}$ (Table 4.1 & Table 4.2).

In all soil profile, we noticed high soil organic carbon content when compared to mean topsoil organic carbon of the vineyard at the Herault Department (Salome et al., 2014 reported 1.32%). Moreover, SOC stocks were locally important within the first meter (around 30 kg $C.m^{-2}$) when integrating i) the SOC mainly content over the value of 1% even for SJp-horizons and ii) values for soil bulk density, mostly over the value of 1.4 $g.cm^{-3}$.

Cation exchange capacity (CEC) values present a low variability with mean value for tilled organo-mineral horizons (LA-Horizons) of 11.81 meq/100 g, considered as a local mean value for topsoil horizons. CEC values decrease with soil depth in association with SOC decrease. When considering the saturation of the CEC, measurements show that both Ca^{2+} and Na^{+} are the main elements, and Calcium has the maximum value (10.96 meq/100 g) much more higher than Sodium (max value 0.51 meq/100 g). Sodium as represented by ESP values, shows a wide range of values, the highest value is especially measured at the deeper depth of field F1-1-1. This field could be considered as sodic soil particularly at structural horizon as the ESP value is over 15%.

Table 4.1. Statistical analysis of physical and chemical soil properties.

	MWD (mm)	SOM (%)	C-Org (%)	EC_{sp} (dS m^{-1})	ESP (%)	CEC	Ca	Na	Cl	Clay (%)	BD ($g.cm^{-3}$)	pH (H_2O)	Total Carbonate ($mg.kg^{-1}$)
							(meq/100 g)						
Number	22	22	22	22	22	22	22	22	22	22	21	22	22
Mean	0.52	2.78	1.61	2.98	5.72	10.60	0.55	0.60	0.69	15.72	1.52	8.30	6.93
St. Dev.	0.18	0.95	0.56	2.87	8.16	2.21	0.40	0.82	1.01	3.30	0.12	0.21	1.86
Min	0.28	0.40	0.23	0.49	0.27	2.41	0.11	0.03	0.01	5.60	1.28	7.96	5.30
Q1	0.38	2.23	1.23	0.83	0.53	10.32	0.23	0.06	0.06	14.05	1.46	8.21	5.77
Median	0.49	2.94	1.71	2.02	2.47	11.15	0.41	0.18	0.20	16.40	1.52	8.33	6.50
Q3	0.64	3.54	2.07	3.75	7.71	11.72	0.78	0.89	0.81	17.98	1.60	8.44	7.34
Max	1.10	4.00	2.32	9.11	35.48	12.85	1.75	3.14	3.11	21.30	1.76	8.87	13.94

Table 4.2. Horizonation with the physical and chemical soil analysis

Field	Hor	Depth (m)	MWD (mm)	EC_{sp} dS m^{-1}	ESP (%)	SWC (%)	BD (g cm^{-3})	SOC (%)	Saturated paste extracted (meq/100 g)			CEC	Total Carbonate (%)	pH H_2O	Soil Texture (%)		
									Ca	Na	Cl				Clay	Silt	Sand
TrellisedVine																	
F1-1-1	LA	0.00-0.40	0.66	7.02	9.17	17.38	1.49	2.25	1.7	1.18	2.18	12.85	5.70	8.04	21.3	59.4	19.3
F1-1-1	SJp_1	0.40-0.60	0.71	7.38	11.7	19.71	1.48	1.77	1.1	1.45	1.97	12.35	6.50	8.19	18.3	60.5	21.2
F1-1-1	SJp_2	0.60-0.90	0.55	8.01	12.8	23.75	1.51	1.81	0.9	1.44	2.37	11.25	5.90	8.26	17.8	69.5	12.7
F1-1-1	SJp_3	0.90-1.00	0.60	9.11	19.4	26.27	1.40	1.71	0.7	2.39	2.86	12.35	5.70	8.45	16.2	70.9	12.9
F1-1-1	SJp_4	>1.00	0.38	8.92	35.4	25.70	1.46	1.67	0.6	3.14	3.10	8.85	6.10	8.58	12.7	64.3	23.0
F1-1-2	LA	0.00-0.20	0.68	0.85	0.53	14.61	1.64	2.32	0.2	0.06	0.07	11.50	5.70	8.41	19.5	53.4	27.1
F1-1-2	A	0.20-0.40	0.39	2.34	3.16	14.12	1.57	2.29	0.5	0.39	0.37	12.25	6.50	8.26	17.4	53.2	29.4
F1-1-2	SJp_1	0.40-0.60	0.54	1.64	3.92	13.32	1.57	1.78	0.5	0.45	0.38	11.55	6.10	8.34	14.6	52.7	32.7
F1-1-2	SJp_2	0.60-1.20	0.34	2.66	6.08	17.04	1.50	1.71	0.3	0.69	0.35	11.35	6.70	8.51	14.7	61.5	23.8
F1-1-2	SJp_3	>1.20	0.42	2.34	7.22	21.06	1.52	1.44	0.1	0.79	0.29	11.05	7.00	8.87	16.6	63.3	20.1
GobletVine																	
F1-2-1	LA	0.00-0.15	0.28	2.50	0.27	32.90	1.28	2.31	0.8	0.03	0.01	12.60	6.22	7.96	17.9	57.3	24.8
F1-2-1	A	0.15-0.40	0.35	0.69	0.30	24.21	1.57	1.65	0.2	0.03	0.02	9.70	7.19	8.32	17.4	55.6	27.0
F1-2-1	SJp_1	0.40-0.80	0.49	0.89	0.48	31.77	1.41	1.45	0.3	0.05	0.07	10.60	6.51	8.30	12.2	54.8	33.0
F1-2-1	SJp_2	>0.80	0.34	1.92	2.21			1.24	0.6	0.24	0.42	10.70	8.03	8.38	12.1	69.9	18.0
F1-2-2	LA	0.00-0.23	0.48	2.13	1.48	26.08	1.41	2.02	0.7	0.16	0.07	10.80	8.04	8.16	18.0	52.8	29.2
F1-2-2	A	0.23-0.52	0.47	0.79	1.06	25.18	1.53	1.20	0.2	0.12	0.10	11.30	7.79	8.43	14.6	70.7	14.7
F1-2-2	SJp_1	0.52-0.71	0.77	1.22	2.74	19.78	1.67	0.58	0.3	0.21	0.25	7.60	9.51	8.48	13.6	43.6	42.8
F1-2-2	SJp_2	>0.71	0.36	0.95	5.45	47.52	1.60	0.23	0.2	0.13	0.15	2.41	13.94	8.76	5.6	6.8	87.6
Fallow																	
F2-1-1	LA	0.00-0.20	1.10	2.38	0.38	23.63	1.31	2.08	0.8	0.04	0.03	11.30	5.79	8.04	18.5	53.9	27.6
F2-1-1	A	0.20-0.60	0.64	0.80	0.57	17.56	1.64	1.53	0.2	0.06	0.08	10.50	5.66	8.30	18.6	56.2	25.2
F2-1-1	SJp_1	0.60-0.91	0.49	0.49	0.52	19.46	1.62	0.97	0.1	0.05	0.04	10.50	6.59	8.39	14.0	51.3	34.7
F2-1-1	SJp_2	>0.91	0.40	0.53	0.84	17.48	1.76	1.03	0.1	0.08	0.02	9.80	6.10	8.44	14.2	55.7	30.1
F2-1-2	LA	0.00-0.15	1.24														
F2-1-2	A	0.15-0.40	0.83														
F2-1-2	SJp_1	0.40-0.60	1.16														

* *A (Organo-Mineral Horizon), SJp (Mineral Horizon), L (Superficial tillage); MWD (Mean Weight Diameter); EC (Electrical Conductivity); ESP (Exchangeable Sodium Percentage); SWC (Soil Water Content); CEC (Cation Exchange Capacity); SOM (Soil Organic Matter). For the F2-1-2 we only analyzed the MWD.*

4.3.3. Soil Salinity

Soil salinity was measured from electrical conductivity of soil extracted saturate paste (EC_{sp}). Measured values range from 0.49 to 9.11 dS m^{-1}, for a mean value of 2.98 dS m^{-1}. When categorized according to the field units, EC_{sp} values, show no uniform behavior with soil depth: i) whatever the horizon type, highest values (7.02 dS m^{-1}<EC_{sp}< 9.11 dS m^{-1}) are observed for F1-1-1, ii) the lowest value for LA-horizon (0.85 dS m^{-1}) is noticed for F1-1-2, iii) similar structures are observed for F1-2-1, F2-2 and F3-1 with mean absolute values for LA-horizon and lower values for underlying soil horizons. Most of previous researches find a clear relation between soil salinity as characterized by EC_{sp} and ESP. In our dataset, such relation is not so clearly obvious. The only relevant observation was that samples having an mean ESP value over 15% and those having mean EC_{sp} value was over 8.00 dS m^{-1} as observed for the field F1-1-1. Two exceptions for high EC_{sp} value associated to very high ESP value (35.48%) was also observed for F1-1-1 especially at the deep SJp-horizons affected by waterlogging. For the others soil samples, ESP values are always below the threshold value of 15 % proposed by (Quirk, 2003) and not clearly correlated with EC_{sp}. Complementary to EC_{sp} and ESP, an interesting measurement is the ionic salts (Na^+ saturated and Cl^- saturated) in soil extracted with water. As shown in Figure 4.3., there were a positive linear correlations between EC_{sp} and salt ions of Na^+ (R^2 = 0.88) and Cl^- (R^2 = 0.90).

4.3.4. Groundwater Salinity

The salt ions were also measured in the saturated zone sampled from six piezometers (4 m depth) that placed near soil trench in all fields. Piper diagram shows the type of water that measured in the piezometers (Figure 4.4). The two piezometers located in F1-2-2 and F2-1-1, had Mg-HCO_3 or Ca-HCO_3 that similar to freshwater sampled from the Villeneuve well which represents groundwater from the Orb alluvial aquifer in the upstream of the study area. Piezometers in the fields F1-1-1, F1-1-2, F1-2-1 and F2-1-2 had Na-Cl type that was the closer to the seawater type and the overall geochemical signature seems to indicate a mix between seawater and fresh groundwater.

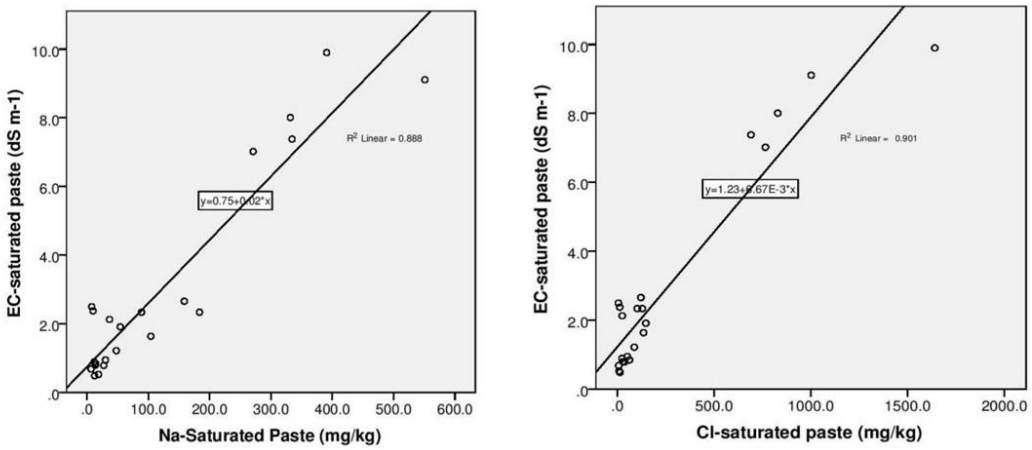


Figure 4.3. Correlation between EC_{sp} and ionic salts (Na saturated and Cl saturated) in soil

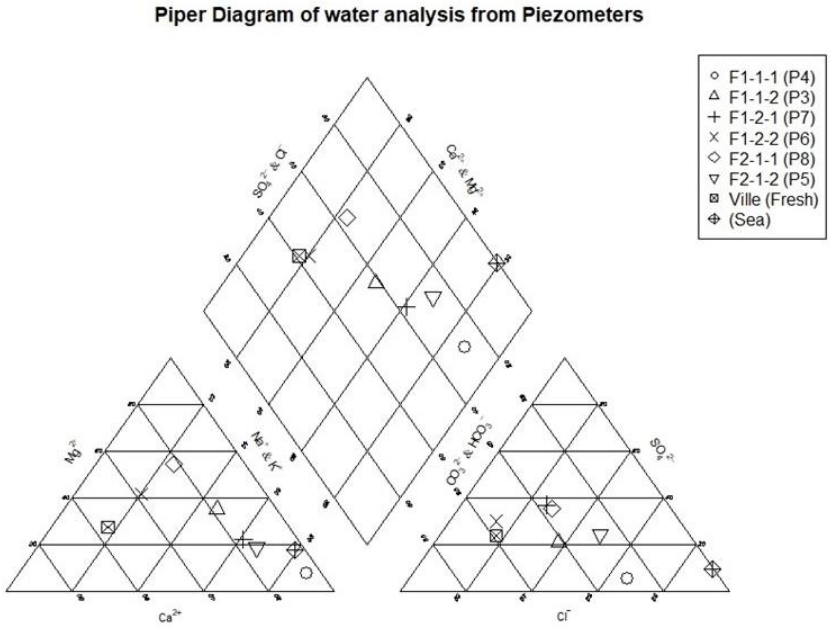


Figure 4.4. Piper diagram of salt ions type measured from water sampled in piezometers.

4.3.5. Soil Aggregate Stability

Soil aggregate stability was determined by measuring mean weight diameter (MWD) of soil samples and range of stability were based on Le Bissonnais (1996). The mean MWD value was around 0.52 mm and the median was 0.49 mm that closed to the threshold value (0.40 mm) of very unstable aggregate. The overall variability within soil samples, as characterized by standard deviation of MWD, is very low (St. Dev.=0.2 mm), for a numerical extend ranging from 0.28 to 1.10 mm (Table 4.2).

The MWD was varies within field units (Fig. 4.4). The global trend of MWD value decreased with increasing soil depth. The MWD at the topsoil horizon (0.0-0.4 m) was higher (median 0.7 mm) compared to the SJp-horizon (median=0.5 mm).

Among field units, both fallow field (F2-1-1 and F2-1-2) possessed high MWD particularly at 0.0-0.4 m depth, then followed by F1-1-1 and F1-1-2 vineyard and the last was the F1-2-2 and F1-2-1 vineyard (Fig 4.5). At the 0.4 m depth, the fallow lands had medium stable aggregate with MWD from 1.1-1.3 mm, while the vineyards, both trellised and gobelet vineyard fields possessed unstable and very unstable aggregate respectively with the MWD range from 0.3-0.7 mm.

The variability of aggregate stability was high in the upper layer, from 0.0-0.4 m depth. When goes down to the deeper depth (0.6-0.8 m), the stability of aggregate were unstable in almost all fields and even very unstable at the depth over 0.9 m. Thus at the depth over 0.9 m the soil aggregate stability was at MWD threshold (0.4 mm) of very unstable soil aggregate.

The box plot of aggregate stability and salinity, for different land use (Figure 4.6.) shows that fallow land had significantly high aggregate stability compared to vineyards. In fallow conditions, lands abandon also shows lower salinity content (> 0.4 m depth) compared to vineyard.

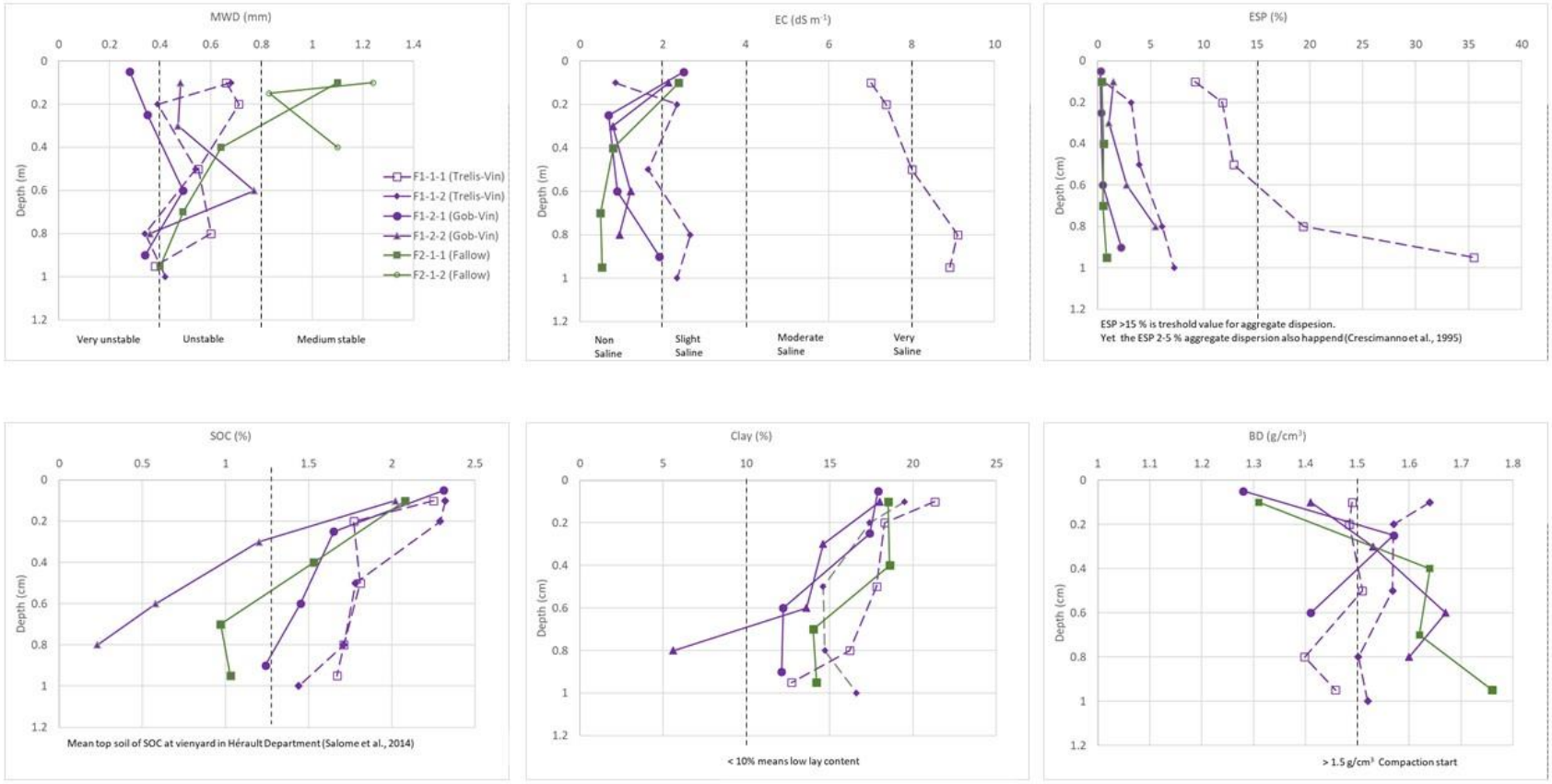


Figure 4.5. Mean weight diameter (MWD) and others intrinsic soil properties for soil aggregation at different soil depths and land uses.

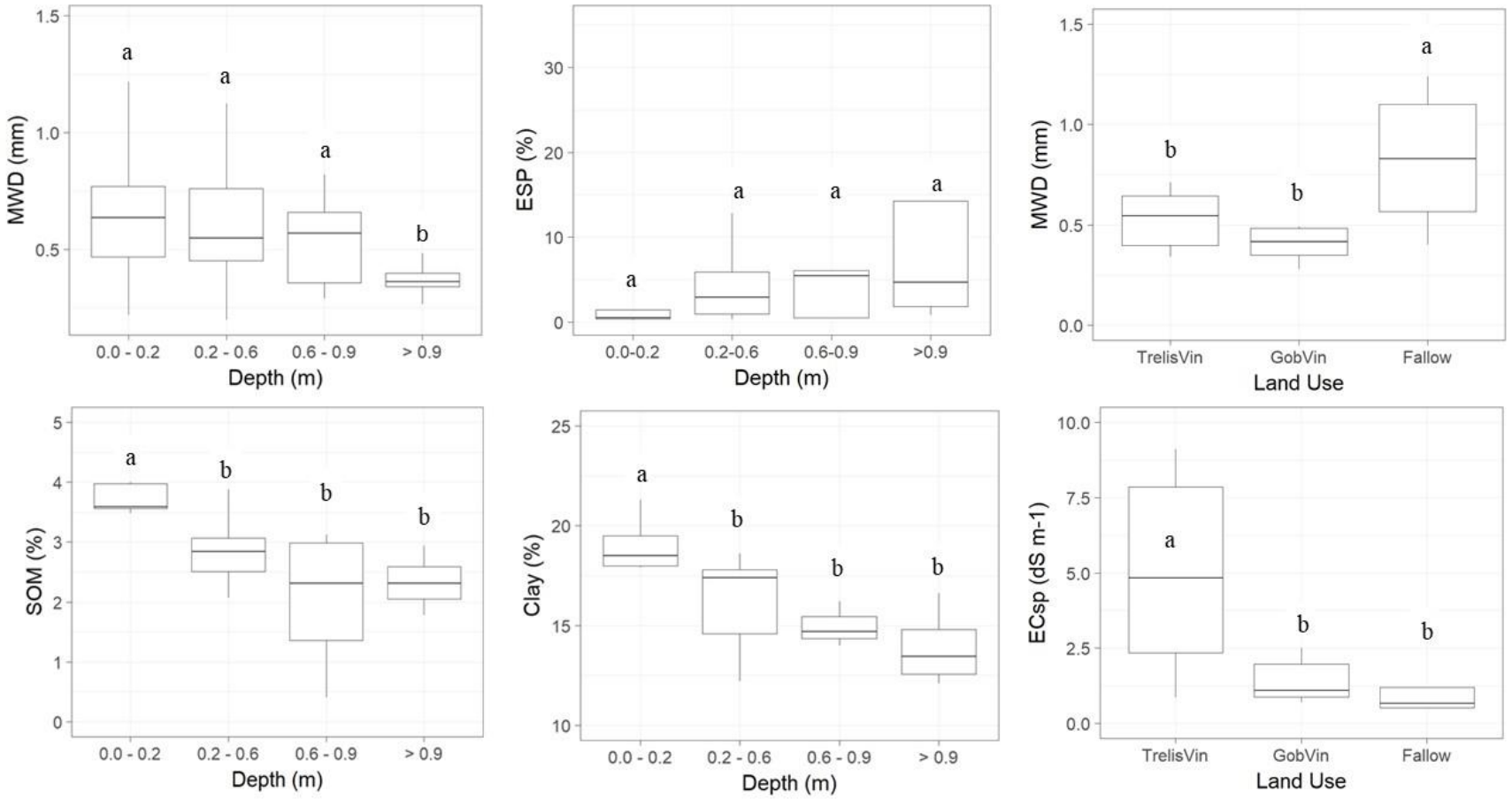


Figure 4.6. Boxplot of MWD, SOM, Clay, ESP and EC_{sp} at different depths and land uses. Different letter means significantly difference (LSD test, p value < 0.05).

4.4. DISCUSSION

The aim of this study was to determine soil aggregate stability along soil trenches under various conditions including agricultural salt affected soils. Current results indicated that the aggregate stability was unstable at topsoil which is superficial tillage horizon (LA-horizon), and very unstable at structural horizon particularly over 0.8 m soil depth (Sjp_3). The difference of the stability was shown from the low MWD value that range between 0.28 and 1.10 mm with the mean value was 0.52 mm (Table 4.1.). Soil organic matter and clay content were significantly higher at topsoil horizon compared to sub soil (Figure 4.6). It is widely known that SOM and clay play important role as agent of soil aggregation on the soil surface (Le Bissonnais et al., 2018; Le Bissonnais et al., 2007; Bronick and Lal, 2005), as both properties form organo-mineral association (Chenu and Stotzky, 2002; Lehmann et al., 2007; Kögel-Knabner et al., 2008) or mineral-organic association (Kleber et al., 2011) that considered as the building block for micro-aggregate formation (Totsche et al., 2018). Thus, this results is in agreement with previous finding that soil aggregate stability in the first top horizon was determined by soil properties particularly SOM and clay.

We also assumed that aggregate stability also determined by other external factors such as agricultural practices of different land uses. The fallow land use had significantly higher MWD compared to both vineyards (Figure 4.6). Since different land use had different agricultural practices, such as tillage and weed control practice. As we identified, the superficial tillage horizon is present in all soil trenches. Yet at fallow land use, there would be less mechanically activity of tillage, thus it has high stability of soil aggregate (Pagliai et al., 2004). While at vineyard, deep tillage activity usually will be more pronounce especially the field that are older than 50 years. It is in agreement with Le Bissonnais et al. (2007), that fallow grassy land has largest aggregate stability compared to young and old weeded vineyard. Previous study also indicated that conventional tillage (high tillage practice) negatively affect soil aggregate stability compared to reduced tillage (Du et al., 2013), as reduced tillage is good to preserve soil organic carbon (Garcia-Franco et al., 2015). At fallow land, we also assume that it might have more soil biological activities supporting soil organic decomposition and soil aggregate formation (Doran, 1980).

Mineral horizons were more susceptible for aggregate destruction and compaction since it had lower organic matter and clay content than top soil (Figure 4.6.). Furthermore, main limitation of agricultural production in this area is salinity, which is originally from groundwater (Bless et al., 2018). Soil salinity in the investigated fields was mainly categorized as moderate saline according to Richards classification (1954) with the median EC_{sp} was 2.3 dS m^{-1} at LA-horizon and 1.7 dS m^{-1} SJp-horizon. The illustration of increasing groundwater that brings soluble salt along soil trench is shown in figure 4.7. The first picture illustrates an actual condition of soil trench of agricultural area, where the mineral horizon was quite compacted and the saturated zone contains salty water. The increasing of groundwater depth is sequenced in time. During wet season, the salty groundwater level increases and reaches the root zone. Whereas in the dry season, the groundwater level decrease, and salinity will rise through capillarity processes. Then whatever the season, in this Mediterranean soil context, salts accumulate in soil profile. Moreover, the compaction layer in the SJp-horizon would reduce the water percolation that is valuable to leach down accumulated salts from the root zone to very deep horizons. Therefore, one priority is to restore infiltration and percolation potential at soil profile scale. This could be done by a mechanical action in order to breakdown the thick compacted layer.

Another issue to restore soil quality could be found in ESP value. Most of the ESP values were lower than threshold value of ESP, 15% (Quirk, 2013) except field F1-1-1, but still had impact on aggregate stability particularly at deeper soil. This finding is in agreement with Crescimanno et al., (1995), who mentioned that destabilization of soil aggregate also happened at very low ESP (2-5%). Moreover the majority of the sodium is not presence in the soil exchangeable part. The salt ions (Na-Cl) might be present and crystalized in the vadose zone during dry period and they would be diluted during the wet period. Therefore, it is possible to leach down the salt ions from soil surface to saturated zone without addition of calcium to replace the sodium in exchangeable soil part. In this case, the quality of water and ditch network management is essential to allow the soluble salt out from the agricultural system.

Other soil properties of soil aggregation like aluminum, iron, bulk density and soil pH also play role. Our results showed that this soil had lower Al and Fe content both are less than 0.02 mg/kg, thus it could not support for soil aggregation and result in the low of MWD value. Since metal oxide had positive relation with aggregate stability. They coagulate with humid acid by covering

surface of metal oxide in order to form micro scale aggregate (Regelink et al., 2015). Other study of Duiker et al., (2003), also suggested that precipitation Al/Fe-Oxide or Al/Fe-hydroxide becomes composite building units for small micro-aggregate <20 m. While for soil with high bulk density, especially silty clay loam texture has threshold 1.5 $g.cm^{-3}$. Over this value caused restriction for root penetration because of compaction. Current result shows that the mean BD at SJp-horizon was around 1.53 $g.cm^{-3}$ thus the restriction of root development might occur. Soil pH is related with concentration of solubility of metal oxides (Al and Fe). Generally, at the pH > 7.0, the solubility of Fe and Al is very low. The soil pH of our sample was around 8.0 (Table 2), so it might be responsible for the low concentration of Al and Fe in soil that influence aggregate stability.

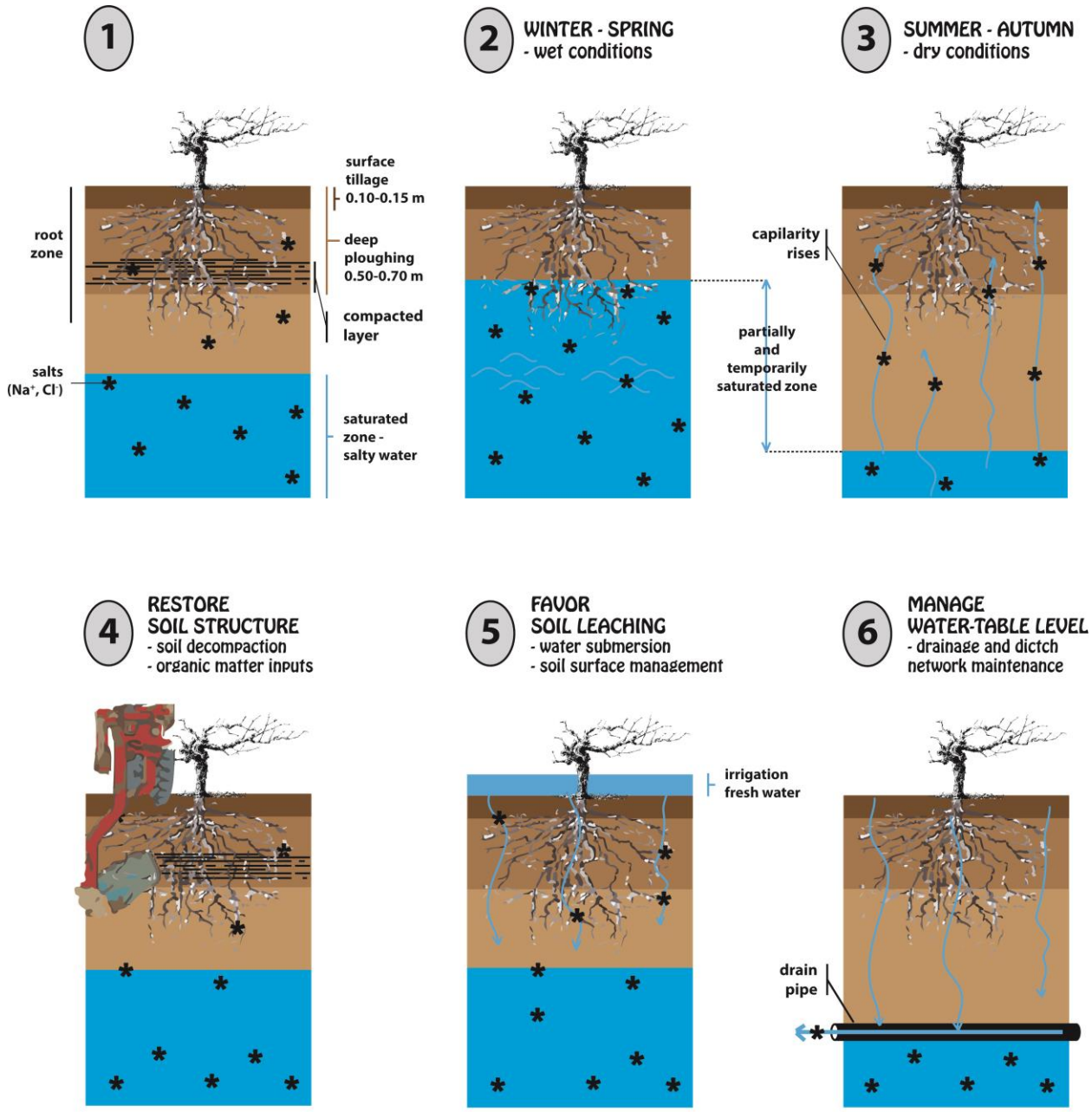


Figure 4.7. Illustration of salty groundwater dynamic along the soil trench in agricultural area.

4.5. CONCLUSION

This study assessed soil aggregate stability in various salty conditions by measurements at different soil depths for different land uses, and investigating the relationships between aggregate stability and known aggregation factors.

Soil aggregate stability was unstable and very unstable at organo-mineral horizon and mineral horizon respectively. The destabilization of soil aggregate at the top horizon is probably related with soil agricultural practice and also properties (SOM and clay content). Meanwhile, destabilization of soil aggregate for subsoil horizons was related with intrinsic soil properties particularly soil salinity. In order to stabilize the aggregate, reduce tillage practice are recommended with input of organic matter. The results indicate that salt (Na-Cl) might be precipitated in the vadose (pore) zone instead of exchangeable soil part except for the field F1-1-1. Therefore there is a possibility to remediate the soil salinity by leaching down the salts with good water quality on submersion. In addition the dynamic of soil aggregate stability it not only determine by physical and chemical soil properties, but also soil biological properties. Since soil biological activities determine decomposition rate of soil organic matter that considers as one important aggregation factors. Thus, it is important to determine soil microbial activities in order to examine the relation of influence of soil microbial activities on aggregate stability. The next chapter (Chapter V) will discuss about soil microbial communities and soil aggregate stability particularly on topsoil.

CHAPTER 5

MICROBIAL CONTROL ON SOIL AGGREGATE STABILITY UNDER SALT AFFECTED CONDITIONS

Abstract

Soil aggregate stability is an indicator for degraded and remediated soil. The dynamics of soil aggregates are influenced by activities of soil microorganism, as they continually decomposed organic materials and synthesized new microbial metabolites. The objective of this study was to identify microbial activity according to stability of soil aggregates originating from vineyards affected by salt that closed to coastal area. We conducted field investigation by collecting soil samples from nine different locations (3 young vineyards, 2 old vineyards, 2 fallows, 2 natural) of soil surface in fields that are affected by salinity. We sampled the soil from soil surface (0-15 cm). We determined soil microbial activities and soil aggregate stability of these samples. Microbial respiration activities were determined with MicroResp method while aggregate stability was determined by fast wetting method and measured its mean weight diameter (MWD). Soil aggregates showed significantly higher stability (2-4 fold) under natural land use (2.16 mm) compared to agricultural field, which are vineyard (0.9 mm) and fallow vineyard (0.54 mm). The stability was limited by microbial respiratory activities induced by amino acid substrates. Soil organic matter content especially in the labile pool, N-total and magnesium were more related to soil aggregation.

Keywords: Soil Microbial Respiratory activities, Soil Aggregate Stability, Soil Salinity

5.1. INTRODUCTION

In salt affected environments, understanding factors affecting soil aggregate stability is crucial in order to properly manage salt leaching and drive sustainable agricultural production. Dynamics of

soil aggregates are actively impacted by several factors and among them is activity of soil microorganism, as they continually decomposed organic materials and synthesized new microbial metabolites responsible for soil aggregation evolution with time (Baldock, 2002). At the same time, soil aggregates constitutes habitats and supports biodiversity of soil. But when speaking about soil aggregates and biodiversity, one has to distinguish size and density-fractionated while micro-organisms colonization preference depends on size and density-fractionated of aggregates. Bacteria and microbial activities mostly exist and play a role in micro-aggregate formation. Bacteria are associated predominantly to the clay fraction and organic matter (Fansler et al., 2005; Neumann et al., 2013; Ranjard and Richaume, 2001; Saviozzi et al., 2007;) while, fungi are more abundant in coarse sand fraction and take part to tangle small micro-aggregate and stabilize macro-aggregates (Kandeler et al., 2000; Kihara et al., 2012). Soil microorganisms are found in association with or near mineral surface to create conditions for interactions between soil mineral and soil organic particles. This organo-mineral association is considered as the structural unit of soil aggregate formation and as nanoparticulate fractions of the micro-aggregate (Totsche et al., 2018). The arrangement and composition of micro-aggregate together with soil properties such as mineral type, metal oxides, OM, pH, water availability, oxygen and nutrients influence bacterial colonization and its community composition and activities (Vogel et al., 2014, Carson et al., 2009); Davinic et al., 2012; Babin et al., 2013). Thus, soil microbes are considered as the architect for construction of soil structure and managing soil surrounding environment (Young and Crawford, 2004).

In salt affected environments, the activities of soil microorganism could decline, as salinity is a stressor for soil microbial communities (Baumann and Marschner, 2011). Salinity reduces matric and osmotic potentials of soil solution, thus microorganisms need more energy in order to obtain water from soil matrix. Additionally, declining of matric potential caused detrimental effect on soil microbes, because it limits the diffusion of substrates used by microbes to synthesize osmolytes useful for maintaining cell water content. High salt concentration in soil solution leads to ionic toxicity and imbalance of nutrient uptake thus it causes a significant reduce of microbial activities, biomass and community structure (Chowdhury et al., 2011; Setia and Marschner, 2013). Even if soil respiration seems mainly influenced by salt contents, Setia et al., (2011), found that cumulative CO_2 emission under salt-contaminated soils, from irrigation and groundwater, also depend on contents of particulate organic carbon (POC), humus-C and clay.

The aim of our study is to identify preponderant factors governing soil aggregation and among them soil microbial activity for different land uses and salt-affected conditions in a coastal agricultural landscape. The overall operational objective is to evaluate the efficiency of soil aggregate stability as a functional indicator for salt leaching in the soil profile, then as a remediation indicator.

Soil aggregation depends on several biotic and abiotic factors, and among them, SOM degradation controlled by microbial organism, quite depending on biotic conditions.

Microbial activity influences decomposition of soil organic compound that is important for soil aggregation on topsoil horizon. In order to achieve the goal, we determined soil microbial respiratory activities and soil aggregate stability at different land uses of soil affected by salts.

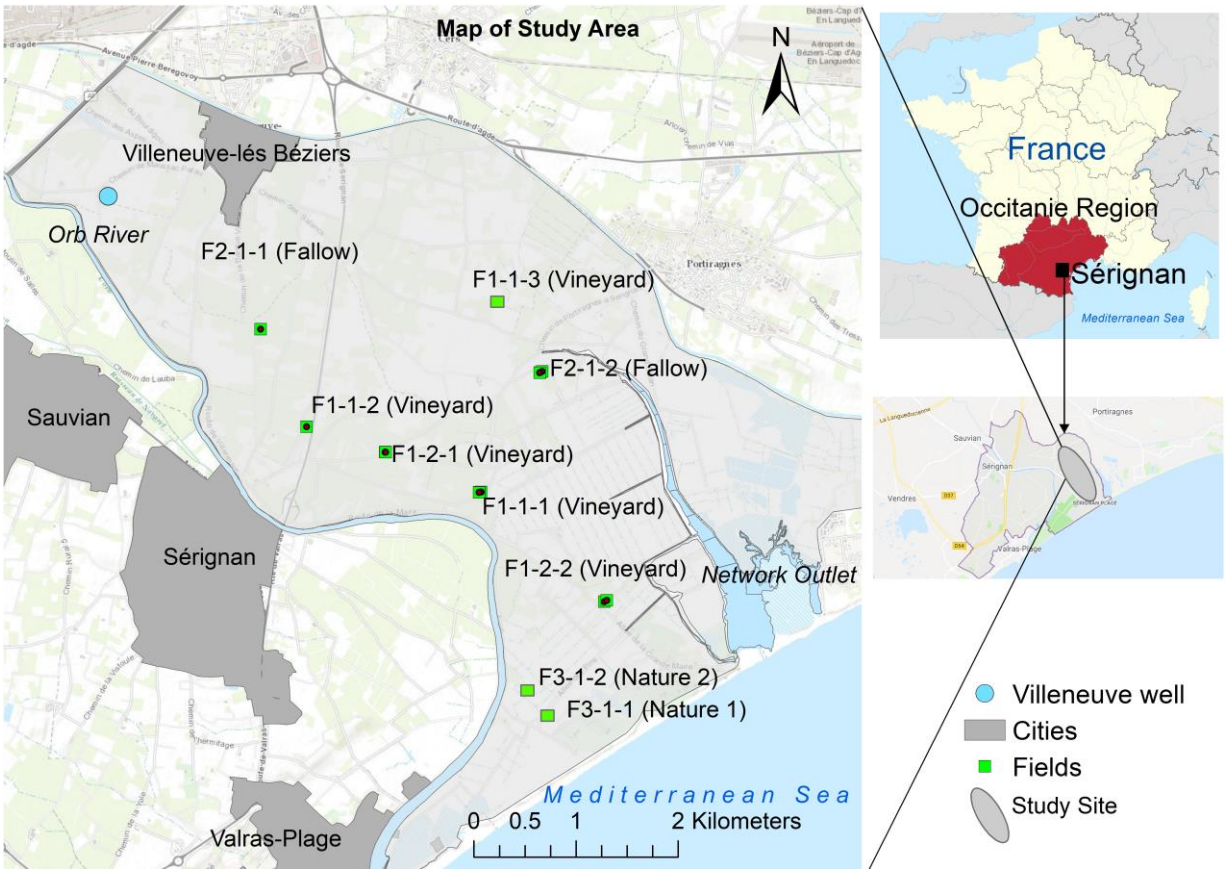


Figure 5.1. Map of study area with the 9 studied locations: trellised young vineyard (F1-1-1; F1-1-2;F1-1-3), gobelet old vineyard, fallow (F2-1-1 and F2-1-2) fields and natural sites (F3-1-1; F3-1-2).

Figure 5.2. Photographs of the fields (i) Trellised vineyard; (ii) Gobelet vineyard; (iii) Fallow and; (iv) Natural field

5.2. MATERIALS AND METHODS

5.2.1. Data collection and measurement

Soil sampling of Aggregate Stability

All soil samples were collected in dry season (June 2018) in nine different fields, affected by salinity (Figure 5.1.). The nine selected fields are different in land use and agricultural management (Figure 5.2):

- 5 vineyards that are different in cultivated systems:
  - 3 with wire (trellised vineyard) (F1-1-1; F1-1-2; F1-1-3)
  - 2 without wire allowing foliage free (gobelet) (F1-2-1; F1-2-2).
- 2 fallow fields (F2-1-1; F2-1-2)
- 2 natural fields (F3-1-1, F3-1-2).

Whatever the land use, all samples were collected on the soil surface and in the inter rows particularly for vineyard. We sampled soil bulk that were kept in plastic box to eliminate the breakdown of soil aggregate and brought to the lab for further analysis. Soils were air dried and sieved to 3-5 mm size for soil aggregate stability measurement with fast wetting method (Le Bissonnais, 1996). For each measurement of three replications, we soaked 5 g of soil samples in deionized water for 10 minutes. Then soil suspensions were wet sieved (50 µm) with ethanol and each sieved fraction was dried oven at 40°C for 24. Next, the samples were dried sieved using different sieving diameters (2 mm, 1 mm, 0.5 mm, 0.25 mm, 0.1 mm and 0.05 mm). The sieved samples of each diameter were weighted then we used the data to calculate the mean weight diameter (MWD_{fast}) (Le Bissonnais, 1996).

In situ catabolic potential (ISCPs) of the soil microbial community

We collected soil from surface (0.00-0.15 m) of the same nine different locations previously cited (Figure 5.1) to measure the ISCPs of the soil microbial community. With triplicated sampling for each location, a total of 27 soil samples were measured for soil microbial activity. The soils were kept in cold storage (5°C) until the analysis. The analysis was based on substrate induced method with Microresp^{Tm} performed in 96-well microtiter plates (Bérard et al., 2015; Boudiaf et al., 2013;

Campbell et al., 2003; Chapman et al., 2007; Dieng et al., 2014). Prior the analysis, we determined soil water content and water holding capacity (WHC) of each soil sample in order to calculate the added volume of sterile distilled water to reactivate the microbial activity during a 3-days incubation time. A colorimetric detection plate was prepared by filling 150 µl of the indicator gel containing cresol red (25 µg ml^{-1}), KCl (300 mM) and $NaHCO_3$ (5 mM) in 1% purified-agar (Oxoid) into a flat bottom-well plate (Thermo Scientific Nunc, Illkirch, France). Then the detection plate was placed in sealed jar containing soda lime for CO_2 absorption and water to prevent dehydration of the gel. The colored gel was stored for several days until it was stable based on absorbance value (coefficient of variation <5%) then used for determining basal respiratory activity.

The 2 mm sieved soil samples were delivered into the 96 deep-wells of soil microplates (±400 mg per well), then added the sterile distillate water which was adjusted to 30% of final WHC and incubated for 3 days in the dark at 25°C under humid atmosphere to avoid extra soil evaporation. During the incubation, we kept checking soil humidity by weighting the microplate each day, if there was a decline of soil microplate mass more than 5 mg, sterile water was added to adjust to 30% of soil WHC. Each soil sample was put in a single microplate. After the pre-incubation, we added different stock solutions (28 substrates and water) into the soil microplate following its recommended map in triplicate for each substrate. The selected substrates are:

- 13 carbohydrates (D-mannose, D-mannitol, D-trehalose, L-arabinose, D-xylose, D-sucrose, D-galactose, meso-inositol, D-sorbitol, L- rhamnose, L-arabitol, meso-erythrol, D-glucose,);
- 5 carboxylic acids (citric acid, malic acid, DL-maleic acid, Na-gluconate, L-glutamic acid sodium);
- 10 amino acids (L-leucine, L-asparagine, D,L-valine, L-methionine, L-glutamine, L-alanine, N-acetyl-glucosamine, L-serine, D,L-histidine, L-proline).

Sterile distillate water was also added in the 6 wells at the center of soil microplate to determine basal respiration. The soil microplate was sealed with MicroResp™ rubber seals that have holes in the middle for CO_2 movement, and added the detection plate on the top of it. The combined plates were incubated for 6 h in the dark at 28°C afterwards. The absorbance of the detection plate was measured before the substrate spiking (t_0) and after the incubation (t_6) with a Tecan infinite M200 Plate Absorbance reader at 570 nm.

The absorbance values at t_0 were subtracted from values at t_6 to obtain absolute respiratory activity of each given well. Then each individual respiration value was subtracted to the average value of basal respiration. To minimize the bias of respiration response due to difference of soil microbial biomass between soil origins, the average value of the triplicate of each substrate was divided by the sum of all the mean substrate specific respiratory activity as following the standardization procedure (Boudiaf et al., 2013). The final values (*pi*) represent a relative measure of the contribution of a substrate to the activity of all substrates.

5.2.2. Soil organic matter and other chemical analyses.

Measurements were performed by the Celesta Laboratory (France; www.celesta-lab.fr). Total soil organic carbon (SOC) and total nitrogen (TN) were determined by the Anne method based on K-dichromate oxidation (AFNOR standard NF ISO 14235) and by the total Kjeldahl nitrogen (TN) method (AFNOR standard NF ISO 11261), respectively (Aubert, 1978). Particulate or labile OM (>50 mm) and mineral-associated (stable) OM (<50 mm) were separated by sieving after mechanical dispersion of the soil by agitation in water with glass beads (Balesdent et al. 1988). Soil texture was determined by Robinson's pipette method, pH (water) was measured in 1:5 soil deionised water suspension, and cation exchange capacity (CEC) by the Metson method. Soil exchangeable cations (Ca, Mg, K and Na) were analyzed by atomic absorption spectrometry after ammonium acetate (1 M) extraction and available Phosphate using the Joret-Hébert method based on soil extraction of 4 g of soil agitated with 100 mL of oxalate ammonium solution (0.1 M). Salinity was measured with electro conductivity meter for $EC_{1/5}$.

5.2.3. Data analysis

The data were analyzed statically with analysis of variance (ANOVA) to compare microbial respiration activities, organic matters, major and micro-elements as a function of soil land use or sampled sites. The significance of the difference between means was evaluated for one way ANOVA Tukey HSD, $p<0.05$ as the criterion of significance using SPSS 22. In order to see the relation between aggregate stability and potential aggregation factors of measured soil properties, we did a linear regression analysis.

Table 5.1. Physical and chemical soil properties at different land uses

Fields	MWD (mm)	$EC_{1/5}$ (dS m^{-1})	pH	Clay (%)	C-Org	Labile SOM	Stabil SOM	N-Total	P	K	Na	Ca	Mg	C/N	CEC cmol/kg	$CaCO_3$
					(g/kg)											
Trelised-vine																
F1-1-1	0.58	0.51	8.3	19.1	1.33	0.7	1.6	1.2	0.02	0.17	0.05	8.58	0.26	11.5	8.7	6.8
F1-1-2	1.12	0.38	8.2	17.5	1.89	1.4	1.9	1.7	0.03	0.14	0.02	8.55	0.32	11.4	9.9	6.2
F1-1-3	1.53	0.34	8.3	20.4	1.00	0.5	1.2	1.2	0.02	0.19	0.09	8.82	0.36	8.3	9.3	7.2
Average	*1.08[a]*	*0.41*	*8.3*	*19.0*	*1.41*	*0.9*	*1.6*	*1.4*	*0.02*	*0.17*	*0.05*	*8.65*	*0.31*	*10.4*	*9.3*	*6.7*
Gobelet-vine																
F1-2-1	0.69	0.39	8.2	18.3	1.48	1.1	1.5	1.5	0.07	0.18	0.03	8.16	0.32	9.9	9.5	6.5
F1-2-2	0.72	0.39	8.2	15.0	1.69	1.0	1.9	1.6	0.06	0.13	0.02	7.95	0.23	10.5	8.8	9.1
Average	*0.70[a]*	*0.39*	*8.2*	*16.6*	*1.6*	*1.05*	*1.7*	*1.5*	*0.06*	*0.15*	*0.02*	*8.05*	*0.27*	*10.2*	*9.1*	*7.8*
Fallow																
F2-1-1	0.40	0.40	8.3	17.7	1.37	0.9	1.4	1.3	0.03	0.12	0.01	7.80	0.24	10.5	9.1	6.4
F2-1-2	0.68	0.39	8.6	20.8	1.06	0.4	1.4	1.2	0.01	0.05	0.21	8.49	0.50	8.8	9.6	7.4
Average	*0.54[a]*	*0.39*	*8.4*	*19.2*	*1.21*	*0.6*	*1.4*	*1.2*	*0.02*	*0.08*	*0.11*	*8.14*	*0.37*	*9.6*	*9.3*	*6.9*
Nature																
F3-1-1	1.19	3.76	9.0	8.5	1.01	0.9	0.8	0.9	0.01	0.16	0.63	2.68	0.48	10.4	5.7	1.2
F3-1-2	3.12	6.43	8.4	22.4	2.38	1.8	2.3	2.3	0.02	0.25	2.31	6.52	1.11	10.5	12.1	7.2
Average	*2.15[b]*	*5.09*	*8.7*	*15.4*	*1.69*	*1.3*	*1.5*	*1.6*	*0.01*	*0.20*	*1.47*	*4.60*	*0.79*	*10.4*	*8.9*	*4.2*
All Average	*1.11*	*1.44*	*8.4*	*17.7*	*1.47*	*0.9*	*1.6*	*1.4*	*0.03*	*0.15*	*0.37*	*7.51*	*0.42*	*10.2*	*9.2*	*6.4*
Min	*0.40*	*0.34*	*8.2*	*8.5*	*1.00*	*0.4*	*0.8*	*0.9*	*0.01*	*0.05*	*0.01*	*2.68*	*0.23*	*8.3*	*5.7*	*1.2*
Median	*0.72*	*0.39*	*8.3*	*18.3*	*1.37*	*0.9*	*1.5*	*1.3*	*0.02*	*0.16*	*0.05*	*8.16*	*0.32*	*10.5*	*9.3*	*6.8*
Max	*3.12*	*6.43*	*9.0*	*22.4*	*2.38*	*1.8*	*2.3*	*2.3*	*0.07*	*0.25*	*2.31*	*8.82*	*1.11*	*11.5*	*12.1*	*9.1*
Std.Dev.	*0.83*	*2.18*	*0.3*	*4.1*	*0.46*	*0.4*	*0.4*	*0.4*	*0.02*	*0.06*	*0.75*	*1.93*	*0.27*	*1.1*	*1.7*	*2.1*

5.3. RESULTS

5.3.1. Chemical soil properties

Statistical analysis of soil aggregate stability (MWD) and chemical soil properties is presented in table 5.1. The sampled soils are significantly alkaline ones, with pH value ranging from 8.20 to 9.00, in agreement with carbonates content ($CaC0_3$ value). Associated loamy soil textures are characterised by a clay fraction varying from 8.48 to 22.37% and soil organic carbon ranging from 1.0 to 2.4 g kg^{-1}. In relation to previously commented soil characteristics, measured cation exchange capacity are low to moderate (mean=9.2 cmol kg^{-1}; Std.Dev=1.57 cmol kg^{-1}).

The MWD values range from 0.40 to 3.12 mm, then from unstable (<0.4 mm MWD <0.8 mm), medium stable (<0.8 mmMWD<1.2 mm), stable (1.2 mm MWD <2.0 mm) to very stable (MWD>2.0 mm), soil aggregate stability classes according to the categorisation proposed by Le Bissonnais (1996). We noticed that half of soil samples present a MWD value under 0.72 mm (median value). Associated salinity measurements as characterised by electrical conductivity ($EC_{1/5}$) showed a large range from 0.34 dS m^{-1} to 6.43 dS m^{-1}.

Soil aggregate stability (MWD) were significantly different at different land uses. The mean MWD value at natural land use was twice (2.16 mm) the vineyard ones (Trelised and Gobelet) (0.95 mm) and four times the fallow land use and was the lowest (0.54 mm). The aggregate stability at natural site is categorized as stable aggregate, while medium stable and unstable soil aggregate goes for vineyard and fallow land use respectively.

We notice that highest MWD value (nature) was observed for conditions having highest EC, SOC, SOM labile, N, K, Ca and Mg values. The labile soil organic matter was twice higher (1.35 g kg^{-1}) compared to fallow land that possesses the lowest stability. However, soil salinity ($EC_{1/5}$) was significantly higher (5.09 dS m^{-1}) and very saline at natural land compared to the rest of land uses. When testing Pearson correlation between MWD value and other chemical properties, it appears that the aggregate stability at soil surface was more related to magnesium (R^2=0.91), total N-contents (R^2=0.69) and SOM in the labile pool (R^2=0.64) (Table 5.2). These three soil properties, soil organic matter in labile pool, N-total and magnesium had the highest values at natural site in comparison to the other land uses.

In addition, other soil properties that were different between land uses, was phosphorous (P). The available P was higher (0.62 g kg^{-1}) at Gobelet vineyard compared to other land use types. The natural land use had the lowest P content (0.16 g kg^{-1}) then followed by fallow land use (0.18 g kg^{-1}) and trellised vineyard (0.22 g kg^{-1}). While Ca was twice higher in vineyard and fallow land compared to natural land use.

Table 5.2. Matrix of Pearson Correlation of soil properties

		MWD	$EC_{1/5}$	Clay	CaCO3	SOM	Labile SOM	Stable SOM	COrg	NTotal	CN	CEC	pH	P	K
MWD	Pearson Correlation														
	Sig. (2-tailed)														
	N														
$EC_{1/5}$	Pearson Correlation	.849**													
	Sig. (2-tailed)	.004													
	N	9													
Clay	Pearson Correlation	.317	-.044												
	Sig. (2-tailed)	.405	.911												
	N	9	9												
CaCO3	Pearson Correlation	-.014	-.351	.717*											
	Sig. (2-tailed)	.972	.355	.030											
	N	9	9	9											
SOM	Pearson Correlation	.596	.490	.339	.352										
	Sig. (2-tailed)	.090	.181	.372	.353										
	N	9	9	9	9										
Labile SOM	Pearson Correlation	.636	.632	.075	-.002	.922**									
	Sig. (2-tailed)	.065	.068	.849	.995	.000									
	N	9	9	9	9	9									
Stable SOM	Pearson Correlation	.458	.258	.562	.658	.922**	.703*								
	Sig. (2-tailed)	.215	.503	.115	.054	.000	.035								
	N	9	9	9	9	9	9								
COrg	Pearson Correlation	.601	.497	.342	.355	1.000	.922**	.922**							
	Sig. (2-tailed)	.087	.173	.367	.349	.000	.000	.000							
	N	9	9	9	9	9	9	9							
NTotal	Pearson Correlation	.694*	.518	.473	.450	.957**	.861**	.907**	.960**						
	Sig. (2-tailed)	.038	.153	.198	.225	.000	.003	.001	.000						
	N	9	9	9	9	9	9	9	9						
CN	Pearson Correlation	-.063	.148	-.270	-.149	.514	.543	.400	.508	.248					
	Sig. (2-tailed)	.872	.704	.482	.702	.157	.131	.286	.163	.519					
	N	9	9	9	9	9	9	9	9	9					
CEC	Pearson Correlation	.527	.198	.882**	.696*	.715*	.497	.829**	.718*	.823**	-.052				
	Sig. (2-tailed)	.145	.609	.002	.037	.030	.173	.006	.030	.006	.895				
	N	9	9	9	9	9	9	9	9	9	9				
pH	Pearson Correlation	.116	.465	-.568	-.791*	-.436	-.212	-.613	-.434	-.438	-.157	-			
	Sig. (2-tailed)	.767	.207	.111	.011	.241	.585	.079	.243	.238	.687	.084			
	N	9	9	9	9	9	9	9	9	9	9	9			
P	Pearson Correlation	-.113	.209	-	-.667*	-.201	.107	-.465	-.198	-.255	.125	-	.495		
	Sig. (2-tailed)	.773	.589	.002	.050	.604	.785	.207	.609	.507	.749	.034	.175		
	N	9	9	9	9	9	9	9	9	9	9	9	9		
K	Pearson Correlation	.722*	.613	.151	-.087	.486	.599	.319	.493	.505	.188	.295	-.147	.118	
	Sig. (2-tailed)	.028	.079	.698	.825	.184	.088	.404	.177	.166	.628	.442	.707	.762	

	N	9	9	9	9	9	9	9	9	9	9	9	9	9						
Mg	Pearson Correlation	.910**	.904*	.333	-.068	.534	.565	.412	.541	.638	-.081	.515	.301	-.130	.512					
	Sig. (2-tailed)	.001	.001	.381	.863	.138	.113	.270	.132	.064	.835	.156	.431	.738	.159					
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9					
Ca	Pearson Correlation	-.266	-.658	.744*	.839**	.099	-.210	.409	.095	.152	-.153	.578	-.838**	-.767*	-.205	-.345				
	Sig. (2-tailed)	.489	.054	.022	.005	.800	.588	.275	.808	.696	.694	.103	.005	.016	.596	.364				
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9				
Na	Pearson Correlation	.914**	.957*	.237	-.095	.606	.656	.451	.614	.674*	.070	.462	.272	-.044	.611	.974**	-.419			
	Sig. (2-tailed)	.001	.000	.539	.808	.084	.055	.223	.078	.046	.857	.210	.479	.911	.080	.000	.262			
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9			
Carbohydrate	Pearson Correlation	-.250	-.398	.441	.713*	.454	.205	.620	.451	.429	.220	.563	-.701*	-.534	-.383	-.212	.665	-.213		
	Sig. (2-tailed)	.516	.289	.235	.031	.220	.596	.075	.224	.249	.569	.114	.035	.139	.309	.583	.050	.582		
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9		
Carboxyl	Pearson Correlation	-.431	-.374	.073	.152	-.587	-.673*	-.391	-.577	-.551	-.230	-	-.066	-.068	-.017	-.402	.215	-.374	-.260	
	Sig. (2-tailed)	.247	.321	.852	.696	.097	.047	.298	.104	.124	.551	.433	.866	.863	.966	.284	.579	.321	.499	
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	
Amino Acid	Pearson Correlation	.422	.552	-.482	-.792*	-.243	.046	-.488	-.243	-.231	-.139	-	.747*	.575	.400	.372	-	.362	-	-.115
	Sig. (2-tailed)	.258	.123	.189	.011	.529	.906	.183	.529	.550	.722	.207	.021	.105	.285	.324	.016	.338	.000	.769
	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9

***. Correlation is significant at the 0.01 level (2-tailed).*

**. Correlation is significant at the 0.05 level (2-tailed).*

5.3.2. Microbial respiratory activities and soil aggregate stability

Figure 5.4. presents a categorisation of *pi* values (microbial relative respiration) according to the land-use types. Due to representation needs, *pi* value are coupled per substrate type e.g. carbohydrate, carboxyl acid and amino acid. Among all land-use types, the mean value of respiratory activities were significantly higher under carboxyl acids (4.3%) and carbohydrates (3.9%), while the lowest mean *pi* value was amino acid substrate group (2.6%). For a considered land use, we notice significant difference of *pi* value between the substrates, especially for amino acid one, whereas no significant difference was detected between the land uses. Specific substrates of carbohydrate which significantly influenced the respiratory activity were the D-sucrose with median value of *pi* was 7.8% and D-glucose (*pi*=6.5%), while for carboxyl acids group, it was malic acid (*pi*=6.7%), and the lowest *pi* value was under amino acid carbon source, D,L-valine (*pi*=1.7%) (Figure 5.6). When comparing the highest *pi* value of three substrates (malic, D-sucrose and D-glucose) based on the different land use, we observed that the highest microbial activities were on carbon source from D-sucrose (9.5%) particularly at fallow land use (Figure 5.5). The microbial activities on D-sucrose and malic acid were significantly higher at vineyard and fallow land use when compared to the natural land use.

When categorized according to soil aggregate stability classes, respiratory activities of microbial community were not significantly different for both carbohydrate and carboxyl acid, whereas under amino acid carbon source, the respiratory activity was significantly lower in the unstable soil aggregate group. The plot of microbial relative respirations under these three different carbon sources (carbohydrate, carboxyl, amino acid) showed a weak relationship with soil aggregate stability (R^2<0.43). The CO_2 release particularly from carbohydrate substrate by microbial activities was more related to soil properties like soil pH (R^2=0.71), carbonate content (R^2=0.70) and stable SOM (R^2=0.62).

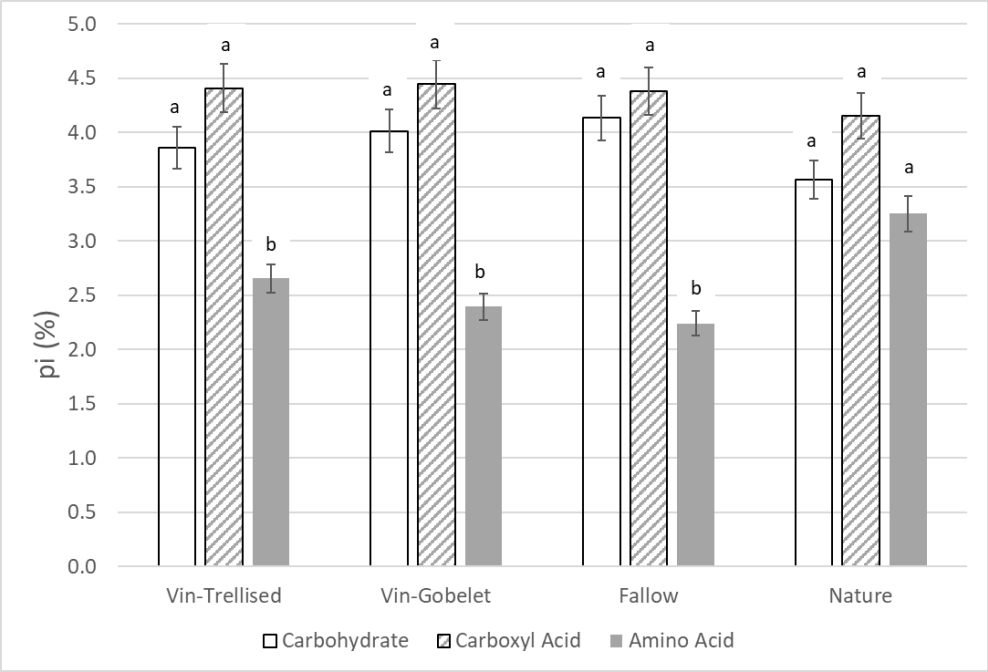


Figure 5.3. Microbial respiratory activities of different carbon substrates at different land uses. (Different letter means significantly different, p<0.05).

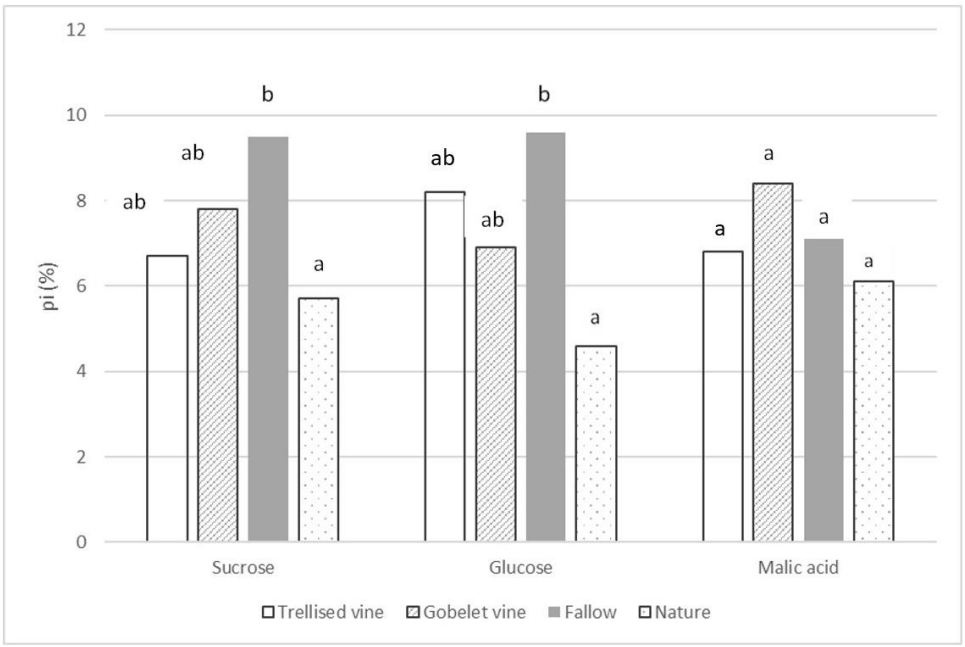


Figure 5.4. Microbial respiratory activities of three substrates that have highest pi at different land uses. (Different letter means significantly different, p<0.05).

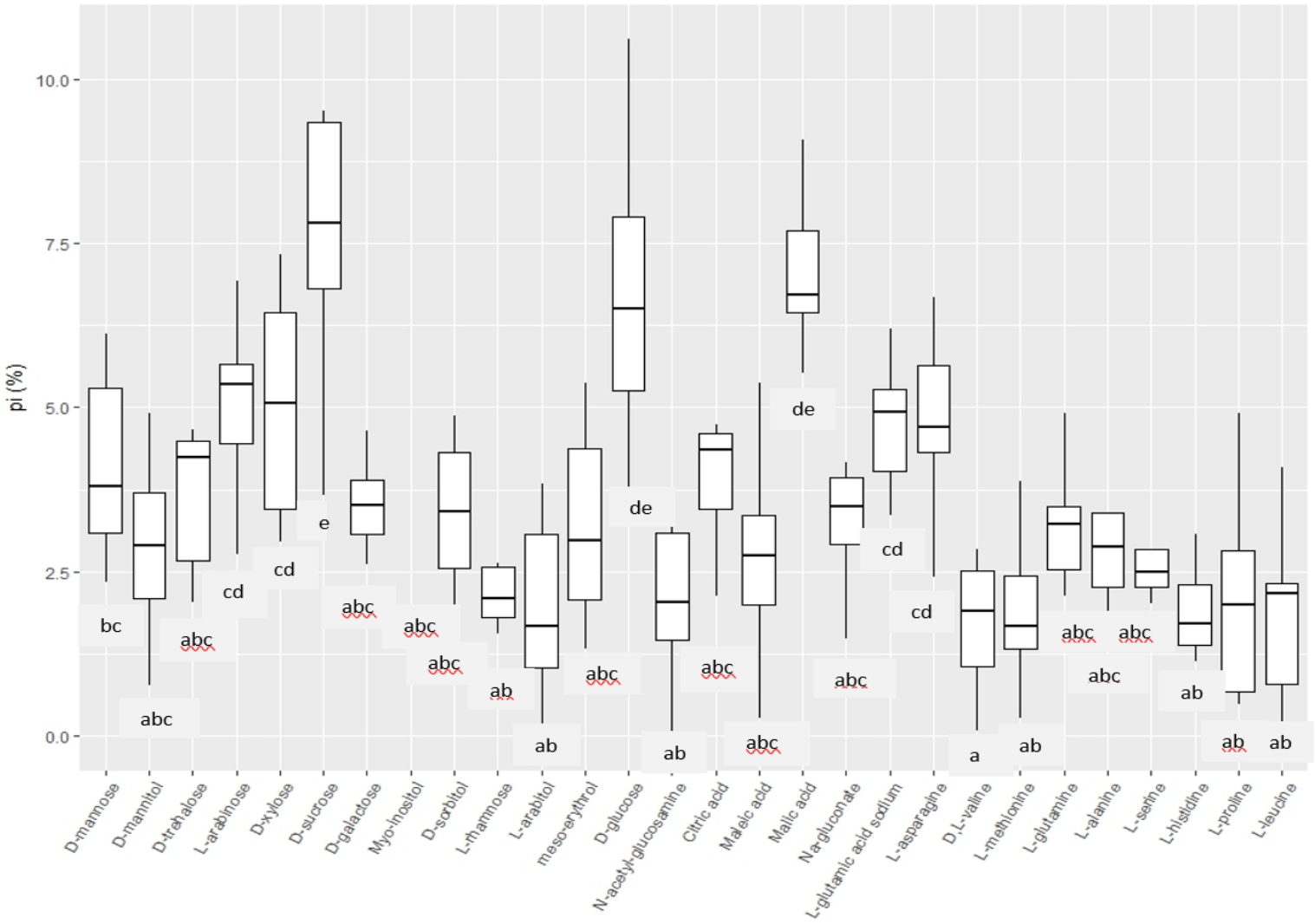


Figure 5.5. Boxplot of microbial respiratory activity under twenty-eight different substrates. (Mean values with a same letter are no significantly different, $p<0.05$).

Table 5.3. Statistical result of microbial respiratory activities (pi %) under 28 different substrates

Susbtrates	Number	Min	Mean	Median	Max	St.Dev
D Mannose	9	2.34	4.07	3.81	6.12	1.29
D Mannitol	9	0.77	2.89	2.90	4.91	1.48
D Trehalose	9	2.03	3.65	4.24	4.67	1.02
L Arabinose	9	2.76	5.04	5.36	6.93	1.33
D Xylose	9	2.97	5.02	5.07	7.32	1.55
Sucrose	9	3.67	7.43	7.80	9.51	2.09
D Galactose	9	2.60	3.50	3.51	4.64	0.61
Myo-inositol	9	2.24	3.36	3.39	4.54	0.87
D(-) Sorbitol	9	1.86	3.41	3.42	4.87	1.08
L(+) Rhamnose	9	0.43	2.19	2.10	4.45	1.07
L(-) Arabitol	9	0.00	2.08	1.68	3.85	1.40
Meso-Erythrol	9	1.33	3.14	2.99	5.37	1.46
D(+) Glucose	9	3.62	6.71	6.51	10.60	2.24
Citric acid	9	0.12	3.58	4.36	4.73	1.54
Maléic acid	9	0.27	3.01	2.74	5.71	1.71
Malic acid	9	5.52	7.08	6.71	9.08	1.07
Na-Gluconate	9	1.48	3.30	3.49	4.17	0.84
N-acetyl Glucosamine	9	0.00	1.99	2.04	3.59	1.25
L-glutamic acid sodium	9	3.36	4.80	4.93	6.19	1.00
L-Asparagine	9	2.43	4.82	4.71	6.68	1.23
D,L-Valine	9	0.09	1.72	1.90	2.84	0.94
L-Methionine	9	0.28	2.06	1.68	4.51	1.34
L-Glutamine	9	2.14	3.21	3.23	4.92	0.88
L-Alanine	9	1.66	3.21	2.88	5.55	1.39
L-Serine	9	2.02	2.84	2.50	5.07	1.02
L-Histidine	9	1.13	1.86	1.71	3.08	0.65
L-Proline	9	0.48	2.02	1.99	4.92	1.44
L-Leucine	9	0.00	2.01	2.17	4.09	1.40

5.4. DISCUSSION

The aim of this study was to identify preponderant factors governing soil aggregation and among them soil microbial activity for different land uses and salt-affected conditions.

We showed that soil aggregate stability was significantly higher at natural land use compared to both vineyard and fallow land use (Table 5.1). In the same time, this natural land also showed a very high mean electrical conductivity (5.09 dS m-1) and is categorized as very saline as closed to the seashore and Orb river (Bless, 2018).

Some possible explanatory factors for this highest soil stability in natural conditions could reside in the highest values of soil carbon, labile soil organic matter, total nitrogen, magnesium and potassium when compared to values observed for other land uses (Table 5.1).

Explanations for this relation could be find in the land use itself rather than in the geographical proximity to seashore line. Here Natural lands are the place for manure of squirrels or horses that might be source for soil organic matter. The surface soil of this area was also covered by forb and grass that produced rhizodeposition increasing soil aggregation (Nguyen, 2003). This is in agreement with Mikha and Rice (2004), suggested that labile C and total N increase the formation of macro aggregates particularly at the no-tillage field with manure application. In addition, higher concentrations of magnesium salts in soil would support the flocculation of soil (Rengasamy et al., 1986) that is important for soil aggregation.

Additionally, whatever the input quantity of organic matter, in natural conditions lowest content of Ca++ is more favourable to organic matter decomposition, while Ca++ had to be considered as an inhibitor for SOM degradation by microorganisms.

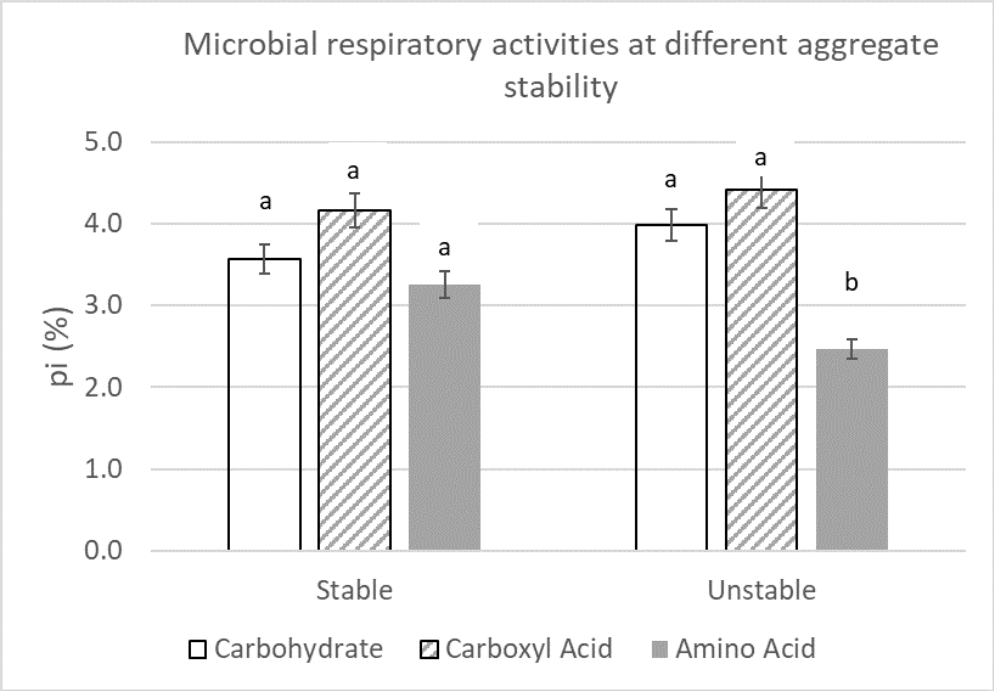


Figure 5.6. Boxplot of microbial respiratory activity under different soil aggregate stability. Different letters mean significantly difference ($p<0.05$).

When determining overall microbial activity through the microbial relative respirations (*pi*) between fields, only significant difference among carbon source of amino acids were found. The microbial activities on amino acid substrate were significantly higher at natural sites compared to fallow and vineyard fields (Figure 5.4.). Furthermore, considering the unstable soil aggregate group, the microbial respiratory activities were significantly lower for amino acid substrates particularly valine, leucine and histidine, while the substrate group of carbohydrates and carboxyl acids showed no difference in both stable and unstable soil aggregates. Microbial activities under amino acid group were not substantially different between very saline and slight saline soil. Thus, we assumed that associated micro-organisms have an important role on stabilization of soil aggregate even in saline condition. This is supported by Bartlett and Doner, (1988) that show positive charges originated from amino acids are stabilized in soils having negative charge and supporting aggregation. Furthermore, amino acid fraction comprises high amount of the total organic C that could remain stable until a long period of time under marine sediments (Stevenson, 1973; Allen et al., 1973). In addition, in natural field the available P concentration in soil was significantly lower and had the lowest value. This P deficiency condition would stimulate exudation of low molecular organic acids from plant roots (Duffner et al., 2012), that could be a

source of organic carbon for soil microbial activity and soil aggregation. The plotting of microbial respiratory against MWD and other soil properties showed that MWD had weak relation with respiratory activities compared to the soil properties. For example, soil organic matter at stabile pool and total carbonate of soil were positively related with microbial respiratory activities under carbohydrates substrate. Thus, it could be assumed that aggregate stability is indirectly related to microbial respiratory activities.

5.5. CONCLUSIONS

The mean weight diameter (MWD) of natural land use was significantly higher (2.16 mm) and had stable soil aggregation compared to other studied land uses, although this land use had very saline soil with the highest soil electrical conductivity ($EC_{1/5}$=5.09 dS m^{-1}). Microbial communities were more active under sucrose, glucose, malic acid substrates induction. We assumed that amino acid substrate group be the limiting factor of stability of soil aggregates. Furthermore, soil aggregate stability on soil surface was more related to labile soil organic matter, total N- and magnesium.

CHAPTER 6

GENERAL DISCUSSION, CONCLUSION AND SUGGESTION FOR FUTURE WORK

Soil salinity has been notify as one of major global soil degradation problem threatens agricultural sustainability, environmental quality and human welfare (CEC, 2006; Rengasamy, 2006b). For some authors, high concentration of salt is not only hazardous for soil fertility but also for water purity, and for plants and animals and human life (Szabolcs, 1989). But the reality is more complex, as salty environments has high biodiversity and then complex system to preserve. Then the open question for our societies is on our overall capacity to design sustainable system, aiming to match several ecosystem services.

Within this complex system perspective, the main objective of the study was to contribute to knowledge production of salt affected soil functioning and potential remediation strategies in an agricultural coastal area landscape. To achieve this goal, we first quantified soil salinization pressure at the landscape scale and then we conducted a soil profile scale analysis to study the overall potential of soil aggregate stability, as a global soil quality indicator. Then a global research, from landscape scale to soil aggregate dimension.

Our research was driven at Serignan city, South of France. Since in the last decade, farmers of Sérignan had suffered from reduction of their agricultural production (a drastic yield decrease since at least 10 years). Farmers presumed that their agricultural fields have been impacted by salt. Due to this local context, research activities by MSc students started in 2012, based on a bottom-up initiative. Then, the first PhD action in 2015 was to consolidate, analyze and complete available data in order to engage the landscape scale analysis. In this analysis, the first assumption was to consider the purpose by Pisinaras et al. (2010), Chernousenko et al. (2011) and Gkiougkis et al., (2015): soil salinization is generated by natural and human activities, and the evolution of natural landscape components (climate change, sea intrusion, rock weathering, river discharge and water balance) or evolution of human activities (intensive agricultural practices, irrigation, land use and management changes) could favor salinization processes by disrupting landscape equilibrium.

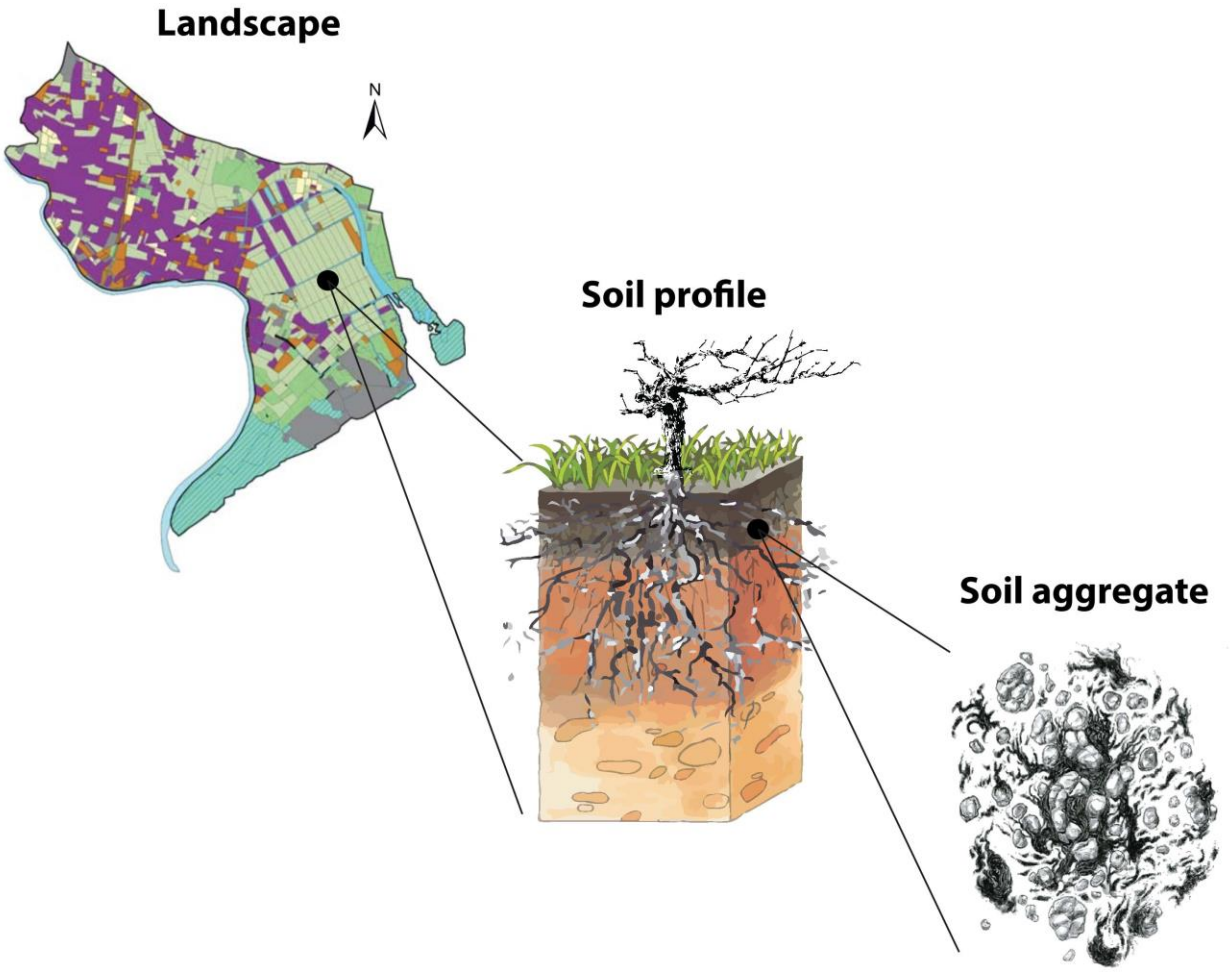


Figure 6.1. Multi scale dimensions of the research

This chapter gives a discussion about our main findings about actual soil/water salinity and the salinization process at the landscape scale in the first section (section 6.1). The second section (section 6.2) continues the discussion with impact of salt on soil quality particularly on aggregate stability and aggregation factors at the soil profile scale. The last section (section 6.3) discusses about remediation activities that need to be consider in order to diminish the harmful effect of soil salinity on agricultural coastal area, particularly in our study area.

6.1. SALINIZATION AT THE LANDSCAPE SCALE

In order to investigate soil and water salinity at the landscape scale, then to perform a high number of measurements, we used the relationship by Richards (1974) that relies on the electrical conductivity of saturated paste extracts and total dissolved salts. The strength of this correlation was tested for our case study. The plot of EC_{sp} against ionic salts in saturated paste extracted (Fig. 4.3.) showed that the main salt ions of soil and water salinity were Natrium (Na^+) and Chloride (Cl^-).

Our field survey and spatial analysis demonstrates that actual electrical conductivity in water and soil was stratified according to the spatial distance to sea and Orb River (chapter III: for increasing distance from the sea and Orb River, electrical conductivity for both soil ($EC_{1/5}$) and saturated zone (EC_w) decreased. Moreover, this spatial distribution for electrical conductivity is also stratified with soil depth: for increasing soil depth, soil and saturated zone conductivity increased.

When comparing soil and water spatial structure, it appeared that EC_w was better stratified than soil $EC_{1/5}$ with decreasing salinity level of different water sources from sea to saturated zone (sea > network outlet > ditch network > Orb River > saturated zone).
Thus, we assume that salt water was transferred through Orb River intrusion into saturated zone through groundwater, and is the source of soluble salt in fields.

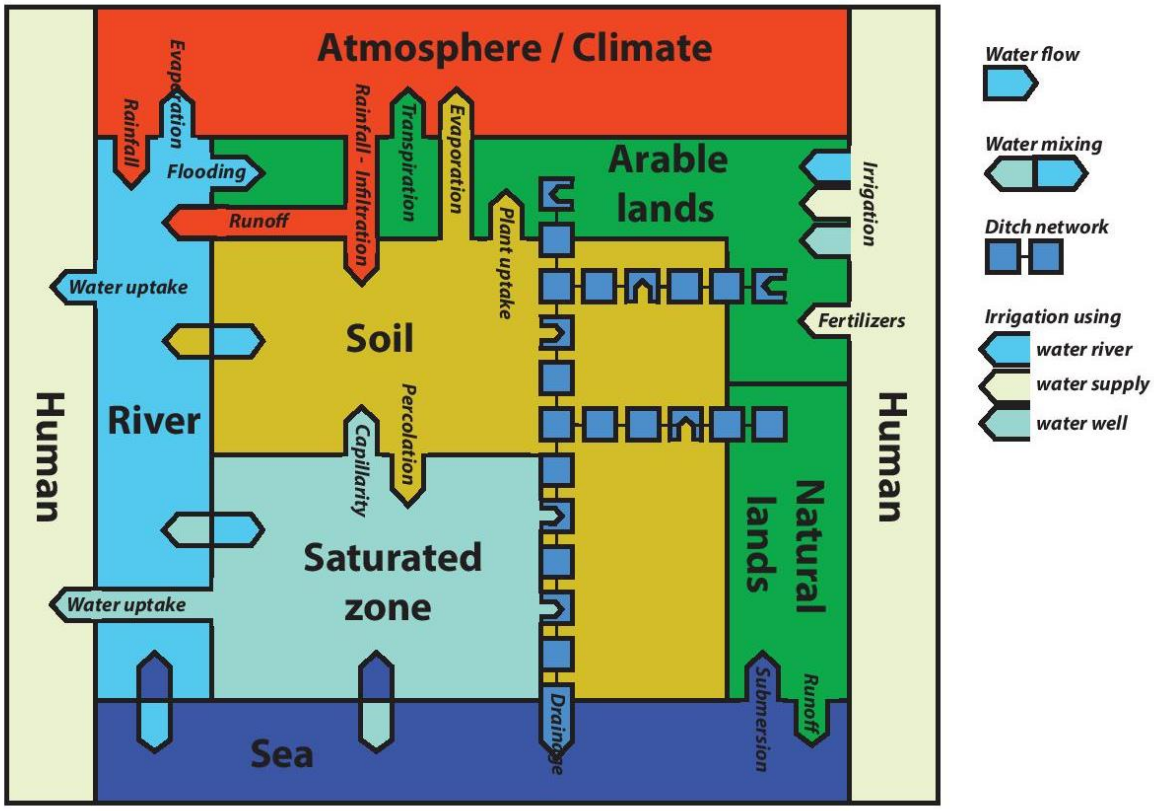


Figure 6.2. Conceptual model of water fluxes in a coastal area affected by salts.

In order to understand the salinization process in a complex system of agricultural coastal area, we built a conceptual model of water fluxes to identify the landscape's elements that responsible for actual salinization in soil and water (Figure 6.2). Our result indicated three main landscape components that had changed over fifty years from 1962 to 2012, and be the driving factors for soil salinity.

The first element is land structure change. In last fifty years, since 1960 to 2012 there was a huge evolution of land structure, particularly change in size and total number of field in Sérignan. Number of agricultural fields reduced significantly from 1895 fields in 1962 to 803 in 2012, while the size of the agricultural field was double from 0.7 ha to 1.5 ha over the last fifty years. The significant change of land structure had modified drainage network system in that area. We showed that in the past, there were high densities of ditch network that allowing the groundwater/saturated zone at lower level. Due to the landscaping dimensioning, the ditch network was disturbed so

raised the groundwater to the new level that might be near to the vine root zone in place. Thus the distance of soluble salt in the ground water to the root zone and topsoil horizon would be shorter.

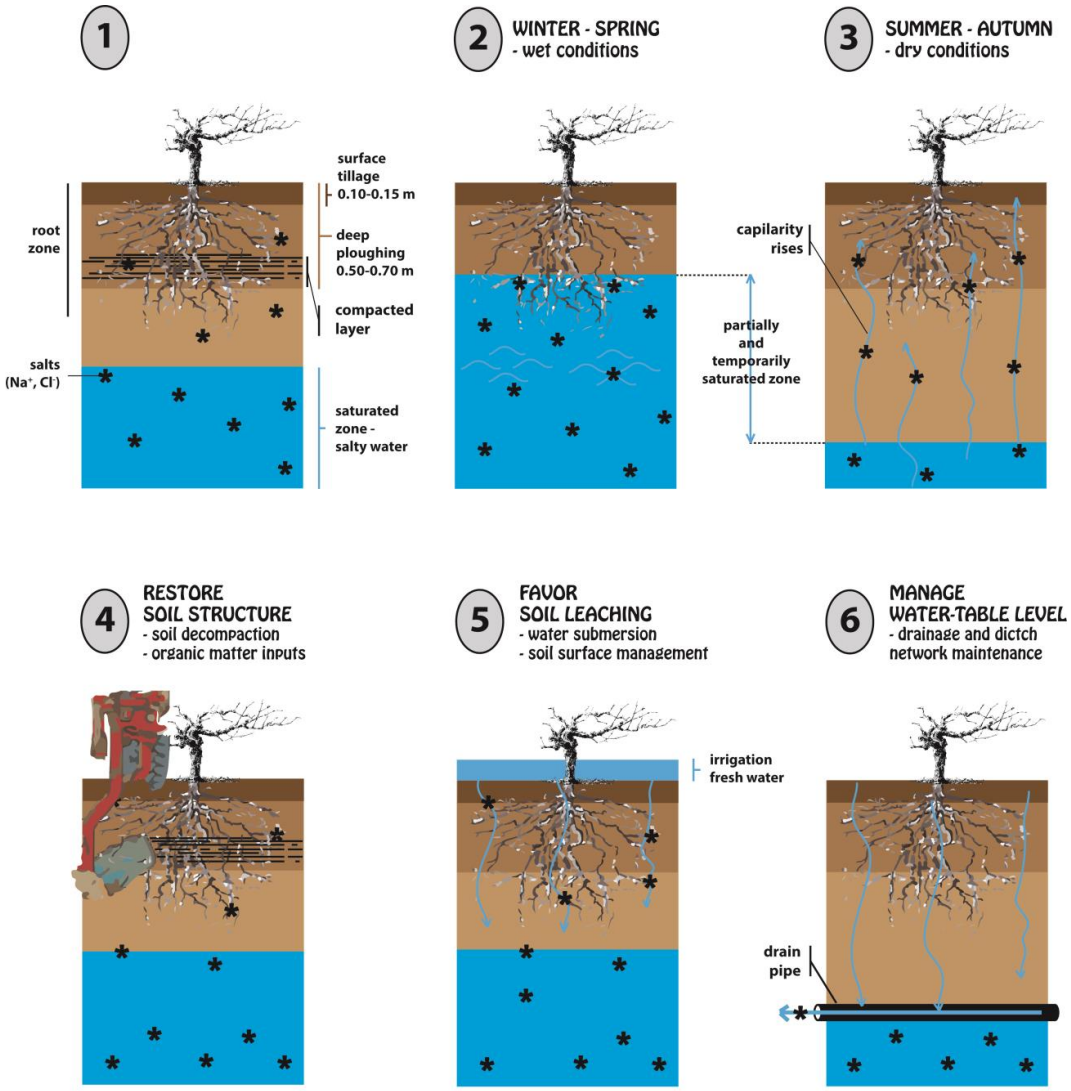


Figure 6.3. Illustration of dynamic of salty groundwater dynamic along the soil profile in agricultural area

In this actual context (Figure 6.3.) salty groundwater would be transfer during wet season through rising of groundwater and through capillary rise during the dry period.

The second component of landscape that had been changed was Orb River discharge. The low monthly discharge (LMD) of Orb River was reduced steadily and the value was smaller over the last fifty years. This reduction might due to less fresh water input of rainfall and increasing of

water abstraction activities along the river. The reduction of Orb River discharge would induce seawater intrusion to the river.

The third element is climate change particularly change in temperature and rainfall amount. In the last fifty years the mean annual temperature was increased significantly thus stimulated evapotranspiration. While, time evolution for rainfall was less clear but nonetheless indicated a slight decrease with high inter-annual variability. Over time, the combination of these climatic factors, reference evapotranspiration and rainfall, has led to the multiplication of the climatic water deficit by a factor of 3 over the last 50 years. When considering the seasonality of the Mediterranean climate, the consequences of such water deficit during the dry season include the following:

(i) increased evaporation, leading to salt precipitation within the soil profile or on the soil surface,

(ii) increased plant water uptake, leading to dryer conditions in the root zone,

(iii) increased capillary forces acting in the soil body, favoring water and salt transfer in soil profiles from saturated to non-saturated zones (i.e., from the deeper soil horizon to the root zone) (Metternicht and Zinck, 2008),

(iv) decreased plant water uptake, as under salt affected soil conditions in the root zone, plants are incapable of efficient water uptake.

Section abstract: The spatial structure investigation of actual salinity in soil and water in Sérignan was induced by Orb River as a preponderant factor. We subsequently developed a conceptual model of water fluxes in the coastal area in order to understand soil salinization process at the landscape scale of agricultural coastal area. We identified the main evolutionary process responsible for system disruption and favoring accumulation of salt in the root zone: (i) a decrease of freshwater influx due to river discharge evolution, (ii) an increase of freshwater outflow due to climate evolution, (iii) an increase in saline water influx due to seawater intrusions into the river, and (iv) a decrease in saline water outflow due to ditch network evolution.

6.2. SALINITY IMPACT ON FIELD SCALE

The fields of our study area were categorized from non-saline, slight saline until very saline condition, with electrical conductivity of saturated paste extraction ranged 0.9 to 10 dS m^{-1}. The main ionic salts in the location were sodium (Na^+) and chloride (Cl^-). Structural stability of soil could be dispersed by high concentration of sodium in soil exchangeable, known as exchangeable sodium percentage (ESP) (Quirk, 2013). The current result shows that the ESP values were ranged from 0.38% to 7.14%, except field F1-1-1 that closed (±500 m) to Orb River, the ESP values at subsoil (>0.9 m) were substantially high, 19.4% - 35.5%. This ESP value was higher than the threshold of ESP (>15%) which caused soil dispersion (Quirk, 2013). Thus we could assume that the sodium concentration in almost all fields might not locate in soil exchangeable part, instead of soluble part of soil. This could be seen from high concentration of total Na^+ from saturated extraction paste, ranged from (6.8 – 588 $mg.kg^{-1}$). The sodium might be crystalized in vadose zone during summer time and diluted in soluble soil during wet season, so it has high electrical conductivity. However this assumption was an exception for field F1-1-1, the sodium might had caused aggregate dispersion at the deeper depth. Moreover, concentration of chloride (Cl^-) was also high, ranged from 4.7-1640 $mg.kg^{-1}$ (Grattan, 2002). High concentration of salts (sodium) threat soil quality and become toxic for plants.

The risk of salt on soil quality can be examined by evaluating soil structure (aggregate stability) that considered as an indicator of soil quality under degraded and remediated soil (Le Bissonnais, 2016). We examined soil aggregate stability and its aggregation factors at the field scale along the soil trench for different land uses. The results show that soil aggregate stability is significantly different within different soil depth. Soil aggregate stability was very unstable (MWD <0.4 mm) at the deepest depth (>0.9 m) compared to the upper depth from top soil to less <90 cm were unstable (MWD 0.4-0.8 mm). Thus we could assume that this area has low soil quality due to low aggregate stability.

The factors of soil aggregations are determined by different soil properties (physical, chemical and biological), agricultural practices and also environmental conditions. Soil organic carbon, clay and metal hydroxide are physico-chemical properties that considered as building block and cementing

agents for micro-aggregation, while microorganism, plant roots play role for macro-aggregation. While disaggregation of soil is resulted from agricultural activity, such as high tillage activity (conventional tillage), and environmental condition like drivers of aggravating salinity problem, as high sodium would disperse soil aggregate. We assume that this difference of soil aggregate stability was mainly due to different soil physico-chemical properties, particularly soil organic carbon (SOC) and clay content. These two soil properties are significantly higher at topsoil positions. Therefore, soil stability at the top soil was determined by soil organic carbon and clay content. Biological soil factors also plays role in soil aggregation. We assumed that the presence of micro-organism capable of degrading amino acid substrate was a limiting factor for soil aggregation in salt affected soil, as this microbial community was significantly low under unstable soil aggregate group.

In addition, our results indicated that soil aggregate stability are significantly different between different land use. The stability of aggregate at abandon land of vineyard and natural land were medium stable and very stable respectively compared to vineyard. Thus, land use influences soil aggregate stability, as different land use apply different agricultural practices such as tillage practices. Natural land use will had no tillage practice, so there is less destruction of big macro-aggregate to small macro-aggregate or to micro-aggregate. Also the microbial activities and soil fauna that support soil aggregation would be more abundant in natural land use compared to agricultural one. Destruction of big aggregate to small aggregate will be vulnerable for soil organic carbon content. Since SOC is not protected inside small soil pores so can be easily accesses and decomposed by soil fauna, or it can be washed away due to runoff or leaching. Therefore soil carbon content might be less under field that apply tillage practice and result in low soil aggregate stability. Tillage practice also would cause soil compaction due to the pressure form the machine to soil particularly on soil surface. Thus we assumed that soil quality on soil surface at the field scale is mainly determined by soil management (tillage practice) and soil properties (soil carbon and clay). While at the deeper depth (>0.9 m), soil structural stability were influenced by soluble salts.

Vine is sensitive to the toxicity of sodium, chloride and boron. The highest concentration of Cl was at the trellised vineyard (F1-1-1) that near Orb River. Visual observation also indicated that

this vineyard showed high mortality of vine, that might be due to high concentration of chloride. Extreme accumulation of sodium and chlorine concentration in root causes mortality due to breakdown of salinity tolerance. The salts elements are translocated from root then accumulated in stems and leaves. Salt affects closure of stomata resulting in the obstruction of photosynthesis process, thus reducing plant growth and biomass production. For quality of wine, the organization of Vine and Wine Office set 60 mg l^{-1} as the upper limit for sodium (Na^{+}) concentration in wine (White, Robert, 2003). While the threshold of Chloride, for economic production of vine was about 20 mM of Cl in irrigation water (Shani and Ben-Gal, 2005) or around 700 ppm Cl^{-} (Grattan, 2002). Our results show that average Cl^{-} concentration in the groundwater was about 13 mM, with the highest concentration was 31.6 mM in piezometer in F1-1-1, thus this field was severely affected by salts.

6.3. REMEDIATION OF SALT AFFECTED SOIL

Approximately more than 800 million hectares of soils globally are salt-affected, with a range of soils defined as saline, acidic–saline, alkaline–saline, acidic saline–sodic, saline–sodic, alkaline saline–sodic, sodic, acidic–sodic and alkaline–sodic. The types of salinity based on soil and groundwater processes are groundwater-associated salinity (dryland salinity), transient salinity (dry saline land) and irrigation salinity. Soil processes in the field determine the interactions between root-zone environments and plant responses to increase osmotic pressure or specific ion concentrations. Soil water dynamics, soil structural stability, solubility of compounds in relation to pH and pE and nutrient and water movement all play vital roles in the selection and development of plants tolerant to salinity (Rengasamy, 2010). Subsequently, the increased water stress for plants may lead to mortality.

Adaptation strategies to remediate water and soil salinity for agricultural purposed will not be identical for all agricultural areas. As salinity problem happened under different range of hydrological and physiographical conditions, soil type, rainfall regime and socio-economic settings. Mashali (1999) suggests that managing agricultural salt affected soil depends on water availability, climatic condition, type of crop and availability of resources. Management practices

and human aspects also need to be subject to actions for remediating agricultural salt affected soil. For management practice it is related with hydraulic, physical, chemical and biological aspect of soil and water. While for human aspects, it needs to be considered about management in socio-economic aspect including farmers involvement, management of organization, institution, policy, and environment. Example of remediation actives that can be executed in agricultural salt affected soil are listed in table 5.1. These remediations strategy can be considered as conventional and innovative one.

Table 6.1. List of remediation activities of salt affected soil (Gkiougkis, 2015; Mashali, 1999).

Conventional remediation	Innovative remediation
1. Control of ground water table - Surface drainage canal - Pump boreholes	1.Irrigation with treated waste water
2. Improvement of soil drainage potentials - Deep plowing - Underground pipe network - Drill through the soils' hard pan	2.Bio drainage
3. Use of calcium for leaching out the exchangeable sodium - Gypsum - $CaCl_2$	3. Remediation with fungi
4. Planting salts tolerant cultivars	4. Remediation with suitable plants
5. Prevention of salts and sodium accumulation - Freshwater transport for irrigation - Chose suitable method for irrigation - Rainfall collection to flush saline soil	5. Automated irrigation control
6. Compost and other organic amendment to improve soil quality	6. Use of genetically modified plants

Based on our investigation of salinization process and its relation to soil characteristic, climatic, hydrological process and evolution of agricultural management practices, we recommend some

remediation strategies for agricultural salt affected soil in Sérignan, South of France. The remediations solutions are based on the driving factor of salinization on landscape scale and field scale evaluation.

Orb river has been justify as main driven factor of salinization process at the landscape scale. Thus improving management of Orb River flow is indispensable. The management is related to water abstraction of Orb River. This activity needs to be concerned and should complied the regulation, as Orb River flows influence intrusion of sea water during dry season low flow period. The flows from the upper dam also need to be considered. Another driving factor at the landscape scale is drain network. It is important to improve the management of drainage system in that area, by increasing drainage density, its depth and its efficiency. This activity is to level down the salty ground water from the root zone. Improving drainage system also allows the salty ground water to be drained out from the agricultural system. Different stake holder are involving for this remediation strategies, like farmers, farmers' cooperative, agricultural chamber, municipality, and researchers. Improving coordination and cooperation between each stake holders is the key to accomplish the remediation of saline soil at the landscape scale.

At the field scale, the remediation strategies for salt affected soil is by applying water submersion at the field. The aim is to leach down the salts from surface and rhizosphere to the ground water. Water can dilute the sodium and leach it down, as most of the sodium is not in the soil exchangeable part but in soil soluble part and crystalized in pore zones. Thus, no need to add calcium for replacing the sodium by using Gypsum. Yet, we need to consider for the quality of water that been used for field submersion. Some farmers in Sérignan, has applied this strategies (Figure 5.2) in collaboration with BRL (water company). Addition of soil organic matter (compost, green manure) is essential in order to improve soil quality (soil aggregate stability) at the field scale. Since the results indicate soil aggregate was not stable at vineyard compared to fallow and natural land use. Tillage practice also need to be measured, as it is influence soil horizonation and the aggregate stability from top soil until 60 cm depth. Reduced tillage practiced would be a better option. Furthermore, for plant remediation in saline soil, we suggest adaptation of cropping system as soil phytoremediation strategies. By cultivating permanent crop in less saline soils and arable crops closed to saline zone (Bas et al., 2017). Farmers also could using rootstocks of vineyard that

are moderately sensitive in saline soil. For example, 140Ru, Schwarzmann, Ramsey (Fort and Walker, 2011) dan rootstock 1103 Paulsen (Hanana et al., 2015).

Figure 6.4. Field submersion of vineyard in Springs time, Sérignan South Of France.

6.4. GENERAL CONCLUSION

Sustainability of agricultural production in coastal area is threaten by soil salinity problem. This soil degradation induced by natural changes and intensive agricultural activities. In this study we found that soil salinization was induced by intrusion of sea water through Orb River and was worsen by disruption network ditches due to land use change. The actual soil salinity in some location had severely impact soil quality and vine growth. Ionic salts were not adsorb in soil exchangeable part thus it can be leach down with water submersion. Soil quality was low, this is indicated by unstable soil aggregate in soil profile analysis particularly on vineyard. The remediation action for soil salinity problem in this area is by improving management of Orb River flow and increasing drainage density. At the soil profile, we suggested a field submersion. While for soil quality improvement, reduced tillage practiced and addition of soil organic matter. As some

farmers in Sérignan has applied field submersion, we recommended further study for evaluating this remediation practices on soil quality.

REFERENCE

Adhikari, K., Hartemink, A.E., 2016. Linking soils to ecosystem services—A global review. Geoderma 262, 101–111.

Agassi, M., Morin, J., Shainberg, I., 1985. Effect of Raindrop Impact Energy and Water Salinity on Infiltration Rates of Sodic Soils 1. Soil Sci. Soc. Am. J. 49, 186–190.

Alexakis, D.D., Daliakopoulos, I.N., Panagea, I.S., Tsanis, I.K., 2016. Assessing Soil Salinity with the use of WorldView-2 Hyperspectral Images in Timpaki, Crete. Presented at the EGU General Assembly Conference Abstracts, p. 15418.

Aureli, A., Ganoulis, J., Margat, J., 2008. Groundwater resources in the Mediterranean region: importance, uses and sharing.

Babin, D., Ding, G.-C., Pronk, G.J., Heister, K., Kögel-Knabner, I., Smalla, K., 2013. Metal oxides, clay minerals and charcoal determine the composition of microbial communities in matured artificial soils and their response to phenanthrene. FEMS Microbiol. Ecol. 86, 3–14.

Baldock, J., 2002. Interactions of organic materials and microorganisms with minerals in the stabilization of soil structure. John Wiley & Sons, Ltd: Chichester, West Sussex, UK.

Balesdent J, Wagner G H and Mariotti A. 1988 Soil organic matter turnover in long term field experiments as revealed by carbon-13 natural abundance. Soil Sci. Soc. Am. J. 52, 118–124.

Banque HYDRO: http://hydro.eaufrance.fr/

Bartlett, J., Doner, H., 1988. Decomposition of lysine and leucine in soil aggregates: adsorption and compartmentalization. Soil Biol. Biochem. 20, 755–759.

Barriere, J. (Dir.), Malacarne, J., Mazier, J., 1973. Carte des sols au 1/5000e – sérignan 10_24_m–la grande maïre 10_24_n 1,2,3,4 – les orpellières 9_24_b 1,2,3. Caractéristques principales et classement pédogénétique. B. Rh. L. - SAGEM – SOLEM – 685 Route d'Arles 30001, Nîmes, France

Bas Niñerola V., Navarro-Pedreño, J., Lucas, I.G., Pastor, I.M., Vidal, M.M.J., 2017. Geostatistical assessment of soil salinity and cropping systems used as soil phytoremediation strategy. Remediat. Polluted Soil – Part 1 174, 53-58. doi:10.1016/j.gexplo.2016.06.008

Batley, G.E., 1989. Trace element speciation analytical methods and problems. CRC Press.

Bless, A.E., Colin, F., Crabit, A., Devaux, N., Philippon, O., Follain, S., 2018. Landscape evolution and agricultural land salinization in coastal area: A conceptual model. Sci. Total Environ. 625, 647–656.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A., Soriano-Soto, M., 2001. Influence of soil properties on the aggregation of some Mediterranean soils and the use of aggregate size and stability as land degradation indicators. Catena 44, 47–67.

Boudiaf, I., Baudoin, E., Sanguin, H., Beddiar, A., Thioulouse, J., Galiana, A., Prin, Y., Le Roux, C., Lebrun, M., Duponnois, R., 2013. The exotic legume tree species, Acacia mearnsii, alters microbial soil functionalities and the early development of a native tree species, Quercus suber, in North Africa. Soil Biol. Biochem. 65, 172–179.

Bouma, J., McBratney, A., 2013. Framing soils as an actor when dealing with wicked environmental problems. Geoderma 200, 130–139.

Bradford, J., Huang, C., 1992. Mechanisms of crust formation: physical components. Soil Crusting Chem. Phys. Process. 55–72.

Brady, N.C., Weil, R.R., 2008. Soil water: Characteristics and behavior. Nat. Prop. Soils Prentice Hall N. J. 177–217.

Bresson, L.-M., Boiffin, J., 1990. Morphological characterization of soil crust development stages on an experimental field. Geoderma 47, 301–325.

Brechin, S. R., Wilshusen, P. R., Fortwangler, C. L., & West, P. C., 2002. Beyond the square wheel: Toward a more comprehensive understanding of biodiversity conservation as social and political process. Society and Natural Resources, 15(1), 41–65

Bronick, C.J., Lal, R., 2005. Soil structure and management: a review. Geoderma 124, 3–22.

Campbell, C.D., Chapman, S.J., Cameron, C.M., Davidson, M.S., Potts, J.M., 2003. A rapid microtiter plate method to measure carbon dioxide evolved from carbon substrate amendments so as to determine the physiological profiles of soil microbial communities by using whole soil. Appl. Environ. Microbiol. 69, 3593–3599.

Carson, J.K., Campbell, L., Rooney, D., Clipson, N., Gleeson, D.B., 2009. Minerals in soil select distinct bacterial communities in their microhabitats. FEMS Microbiol. Ecol. 67, 381–388.

Cary, L., Petelet-Giraud, E., Bertrand, G., Kloppmann, W., Aquilina, L., Martins, V., Hirata, R., Montenegro, S., Pauwels, H., Chatton, E., Franzen, M., Aurouet, A., 2015. Origins and

processes of groundwater salinization in the urban coastal aquifers of Recife (Pernambuco, Brazil): A multi-isotope approach. Sci. Total Environ. 530–531, 411–429. https://doi.org/10.1016/j.scitotenv.2015.05.015

Castro Filho, C., Lourenço, A., de F. Guimarães, M., Fonseca, I.C.B., 2002. Aggregate stability under different soil management systems in a red latosol in the state of Parana, Brazil. Soil Tillage Res. 65, 45–51. https://doi.org/10.1016/S0167-1987(01)00275-6

Cave cooperative de viticulteurs de Sérignan, 2016. Analyses de terre des viticulteurs de Sérignan. Documents internes.

Chan, K., Heenan, D., 1996. The influence of crop rotation on soil structure and soil physical properties under conventional tillage. Soil Tillage Res. 37, 113–125.

Chapman, S.J., Campbell, C.D., Artz, R.R., 2007. Assessing CLPPs using MicroResp™. J. Soils Sediments 7, 406–410.

Chenu, C., 1993. Clay—or sand—polysaccharide associations as models for the interface between micro-organisms and soil: water related properties and microstructure, in: Soil Structure/Soil Biota Interrelationships. Elsevier, pp. 143–156.

Chenu, C., Stotzky, G., 2002. Interactions between microorganisms and soil particles: an overview. John Wiley and Sons, USA.

Chernousenko, G., Kalinina, N., Khitrov, N., Pankova, E., Rukhovich, D., Yamnova, I., Novikova, A., 2011. Quantification of the areas of saline and solonetzic soils in the Ural Federal Region of the Russian Federation. Eurasian Soil Sci. 44, 367.

Chotte, J.-L., 2005. Importance of microorganisms for soil aggregation, in: Microorganisms in Soils: Roles in Genesis and Functions. Springer, pp. 107–119.

Chowdhury, N., Marschner, P., Burns, R., 2011. Response of microbial activity and community structure to decreasing soil osmotic and matric potential. Plant Soil 344, 241–254. https://doi.org/10.1007/s11104-011-0743-9

Claessens, L., Schoorl, J.M., Verburg, P.H., Geraedts, L., Veldkamp, A., 2009. Modelling inter-actions and feedback mechanisms between land use change and landscape processes. Agriculture, Ecosystems & Environment 129, 157–170.

Coletti, J.Z., Vogwill, R., Hipsey, M.R., 2017. Water management can reinforce plant competition in salt-affected semi-arid wetlands. J. Hydrol. 552, 121-140. doi:10.1016/j.jhydrol.2017.05.002.

Commission of the European Communities (CEC), 2006. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions. Thematic Strategy for Soil Protection. Brussels.

Crescimanno, G., Iovino, M., Provenzano, G., 1995. Influence of salinity and sodicity on soil structural and hydraulic characteristics. Soil Sci. Soc. Am. J. 59, 1701–1708.

Custodio, E., 2002. Coastal Aquifers as Important Natural Hydrogeological Structures. Groundwater and Human Development.

Daliakopoulos, I.N., Tsanis, I.K., Koutroulis, A., Kourgialas, N.N., Varouchakis, A.E., Karatzas, G.P., Ritsema, C.J., 2016. The threat of soil salinity: A European scale review. Sci. Total Environ. 573, 727–739. https://doi.org/10.1016/j.scitotenv.2016.08.177

Davinic, M., Fultz, L.M., Acosta-Martinez, V., Calderón, F.J., Cox, S.B., Dowd, S.E., Allen, V.G., Zak, J.C., Moore-Kucera, J., 2012. Pyrosequencing and mid-infrared spectroscopy reveal distinct aggregate stratification of soil bacterial communities and organic matter composition. Soil Biol. Biochem. 46, 63–72.

Deinlein, U., Stephan, A.B., Horie, T., Luo, W., Xu, G., Schroeder, J.I., 2014. Plant salt-tolerance mechanisms. Trends Plant Sci. 19, 371–379. https://doi.org/10.1016/j.tplants.2014.02.001

de Louw, P.G.B., Eeman, S., Essink .G.H.P., Vermue, E., Post, V.E.A., 2013. Rainwater lens dynamics and mixing between infiltrating rainwater and upward saline groundwater seepage beneath a tile-drained agricultural field. J. Hydrol 501, 133-145. doi:10.1016/j.jhydrol.2013.07.026

de Louw, P.G.B., Eeman, S., Siemon, B., Voortman, B.R., Gunnink, J., van Baaren, E. S., Essink, O.G.H.P., 2011. Shallow rainwater lenses in deltaic areas with saline seepage. Hydro.Earth Syst. Sci. 15, 3659-3678.

Doran, J.W., 1980. Soil Microbial and Biochemical Changes Associated with Reduced Tillage1. Soil Sci. Soc. Am. J. 44, 765–771. https://doi.org/10.2136/sssaj1980.03615995004400040022x

DU, Z., REN, T., HU, C., ZHANG, Q., Blanco-Canqui, H., 2013. Soil aggregate stability and aggregate-associated carbon under different tillage systems in the North China Plain. J. Integr. Agric. 12, 2114–2123.

Dubois, O., 2011. The state of the world's land and water resources for food and agriculture: managing systems at risk. Earthscan.

Duffner, A., Hoffland, E., Temminghoff, E.J., 2012. Bioavailability of zinc and phosphorus in calcareous soils as affected by citrate exudation. Plant Soil 361, 165–175.

Duiker, S., Lal, R., 1999. Crop residue and tillage effects on carbon sequestration in a Luvisol in central Ohio. Soil Tillage Res. 52, 73–81.

Duiker, S.W., E. Rhoton, F., Torrent, J., E. Smeck, N., Lal, R., 2003. Iron (Hydr)Oxide Crystallinity Effects on Soil Aggregation.

Edwards, A.P., Bremner, J., 1967. MICROAGGREGATES IN SOILS 1. J. Soil Sci. 18, 64–73.

EEA, 2000. Down to earth: soil degradation and sustainable development in Europe. A challenge for the 21th century. United Nation Convention Fourth Conference of the Parties Bonn, 19 December 2000.

El-Swaify, S.A., Emerson, W., 1975. Changes in the Physical Properties of Soil Clays Due to Precipitated Aluminum and Iron Hydroxides: I. Swelling and Aggregate Stability After Drying 1. Soil Sci. Soc. Am. J. 39, 1056–1063.

Emerson, W., 1959. The structure of soil crumbs. Eur. J. Soil Sci. 10, 235–244.

Emerson, W., Greenland, D., 1990. Soil aggregates—formation and stability, in: Soil Colloids and Their Associations in Aggregates. Springer, pp. 485–511.

Fansler, S.J., Smith, J.L., Bolton, H., Bailey, V.L., 2005. Distribution of two C cycle enzymes in soil aggregates of a prairie chronosequence. Biol. Fertil. Soils 42, 17–23.

Feinerman, E., Yaron, D., Bielorai, H., 1982. Linear crop response functions to soil salinity with a threshold salinity level. Water Resour. Res. 18, 101-106.

Fordham, A., Norrish, K., 1983. The nature of soil particles particularly those reacting with arsenate in a series of chemically treated samples. Soil Res. 21, 455–477.

Fort, K., Walker, A., 2011. Breeding salt tolerant rootstocks. FPS Grape Program Newsl. 10, 9–11.

Garcia-Franco, N., Albaladejo, J., Almagro, M., Martínez-Mena, M., 2015. Beneficial effects of reduced tillage and green manure on soil aggregation and stabilization of organic carbon in a Mediterranean agroecosystem. Soil Tillage Res. 153, 66–75. https://doi.org/10.1016/j.still.2015.05.010

Girard C., Pulido-Velazqueza M., Rinaudob J.D., Pagéc C., Caballerob Y., 2015. Integrating top–down and bottom–up approaches to design global change adaptation at the river basin scale. Global Environmental Change 34, 132-146.

Gkiougkis, I., Kallioras, A., Pliakas, F., Pechtelidis, A., Diamantis, V., Diamantis, I., Ziogas, A., Dafnis, I., 2015. Assessment of soil salinization at the eastern Nestos River Delta, NE Greece. Catena 128, 238–251.

Grattan, S., 2002. Irrigation water salinity and crop production. UCANR Publications.

Greene, R.,Timms, W., Rengasamy, P., Arshad, M., Cresswell, R., 2016. Soil and aquifer salinization: Toward an integrated Approach for salinity management in of groundwater. Integrated Groundwater Management. doi:10.1007/978-3-319-23576-9_15.

Hanana, M., Hamrouni, L., Hamed, K., Abdelly, C., 2015. Influence of the Rootstock/Scion Combination on the Grapevines Behavior under Salt Stress. J. Plant Biochem. Physiol.

Hannah, L., Midgley, G. F., Lovejoy, T., Bond, W. J., Bush, M., Lovelett, J. C., et al., 2002. Conservation of biodiversity in a changing climate. Conservation Biology, 16(1), 264–268.

Harris, I., Jones, P.D., Osborn, T.J. and Lister, D.H. (2014), Updated high-resolution grids of monthly climatic observations – the CRU TS3.10 Dataset. Int. J. Climatol., 34: 623–642. doi: 10.1002/joc.3711.

Hao, X., Chang, C., 2002. Effect of 25 annual cattle manure applications on soluble and exchangeable cations in soil. Soil Sci. 167, 126–134.

Hardie, M., Doyle, R., 2012. Measuring soil salinity, in: Plant Salt Tolerance. Springer, pp. 415–425.

Hartemink, A.E., McBratney, A., 2008. A soil science renaissance. Geoderma 148, 123–129. https://doi.org/10.1016/j.geoderma.2008.10.006

Haynes, R.J., Naidu, R., 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. Nutr. Cycl. Agroecosystems 51, 123–137.

Herczeg, A.L., Dogramaci, S.S., Leaney, F.W.J., 2001. Origin of dissolved salts in a large, semi-arid groundwater system: Murray Basin, Australia. Mar. Freshw. Res. 52, 41–52.

Hingston, F., Galbraith, J., 1990. Salt impaction at near-coastal locations and possible effects on grapevines in southwest of Western Australia. Agric. Ecosyst. Environ. 32, 199–212.

Igwe, C., Mbagwu, J., 1999. Chemical and mineralogical properties of soils in southeastern Nigeria in relation to aggregate stability. Geoderma 92, 111–123.

IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

Isidora, D., 2011. River salt loads as influenced by Irrigation development in the Bardenas Irrigation Scheme (Spain). Proceeding of Global forum in salinization.

Jahn, R., Blume, H.P., Asio, V. B., Spaargaren, O., and Schad, P., 2006. Guildlines for soil description. FAO.

Jarecki, M.K., Lal, R., 2003. Crop management for soil carbon sequestration. Crit. Rev. Plant Sci. 22, 471–502.

JRC. 2012. The State of Soil In Europe. JRC Reference Reports. European Environment Agency. European Union.

Kandeler, E., Tscherko, D., Bruce, K.D., Stemmer, M., Hobbs, P.J., Bardgett, R.D., Amelung, W., 2000. Structure and function of the soil microbial community in microhabitats of a heavy metal polluted soil. Biol. Fertil. Soils 32, 390–400. https://doi.org/10.1007/s003740000268

Kanti Sen, T., Khilar, K.C., 2006. Review on subsurface colloids and colloid-associated contaminant transport in saturated porous media. Adv. Colloid Interface Sci. 119, 71–96. https://doi.org/10.1016/j.cis.2005.09.001

Karlen, D. L., M. J. Mausbach, J. W. Doran, R. G. Cline, R. F. Harris, and G. E. Schuman. 1997. Soil Quality: A Concept, Definition, and Framework for Evaluation (A Guest Editorial). Soil Sci. Soc. Am. J. 61:4-10. doi:10.2136/sssaj1997.03615995006100010001x

Keesstra, S.D., Bouma, J., Wallinga, J., Tittonell, P., Smith, P., Cerdà, A., Montanarella, L., Quinton, J.N., Pachepsky, Y., van der Putten, W.H., 2016. The significance of soils and soil science towards realization of the United Nations Sustainable Development Goals. Soil 2, 111.

Khaska, M., Le Gal La Salle, C., Lancelot, J., team, A., Mohamad, A., Verdoux, P., Noret, A., Simler, R., 2013. Origin of groundwater salinity (current seawater vs. saline deep water) in a coastal karst aquifer based on Sr and Cl isotopes. Case study of the La Clape massif (southern France). Appl. Geochem. 37, 212–227. https://doi.org/10.1016/j.apgeochem.2013.07.006

Kihara, J., Martius, C., Bationo, A., Thuita, M., Lesueur, D., Herrmann, L., Amelung, W., Vlek, P.L.G., 2012. Soil aggregation and total diversity of bacteria and fungi in various tillage

systems of sub-humid and semi-arid Kenya. Appl. Soil Ecol. 58, 12–20. https://doi.org/10.1016/j.apsoil.2012.03.004

Kingwell, R. and John, M., 2007. The influence of farm landscape shape on the impact and management of dryland salinity. Agricultural Water Management 89, 29-38.

Kitagawa, Y., 1983. Goethite and hematite in some solls from the amazon region. Soil Sci. Plant Nutr. 29, 209–217.

Kleber, M., Nico, P.S., Plante, A., Filley, T., Kramer, M., Swanston, C., Sollins, P., 2011. Old and stable soil organic matter is not necessarily chemically recalcitrant: implications for modeling concepts and temperature sensitivity. Glob. Change Biol. 17, 1097–1107.

Kögel-Knabner, I., Guggenberger, G., Kleber, M., Kandeler, E., Kalbitz, K., Scheu, S., Eusterhues, K., Leinweber, P., 2008. Organo-mineral associations in temperate soils: Integrating biology, mineralogy, and organic matter chemistry. J. Plant Nutr. Soil Sci. 171, 61–82.

Kotb, T.H., Watanabe, T., Ogino, Y., Tanji, K.K., 2000. Soil salinization in the Nile Delta and related policy issues in Egypt. Agric. Water Manag. 43, 239–261.

Kouzana, L., Benassi, R., Ben mammou, A., Sfar felfoul, M., 2010. Geophysical and hydrochemical study of the seawater intrusion in Mediterranean semi arid zones. Case of the Korba coastal aquifer (Cap-Bon, Tunisia). J. Afr. Earth Sci. 58, 242–254. https://doi.org/10.1016/j.jafrearsci.2010.03.005

Lagacherie, P., Álvaro-Fuentes, J., Annabi, M., Bernoux, M., Bouarfa, S., Douaoui, A., Grünberger, O., Hammani, A., Montanarella, L., Mrabet, R., Sabir, M., Raclot, D., 2018. Managing Mediterranean soil resources under global change: expected trends and mitigation strategies. Reg. Environ. Change 18, 663–675. https://doi.org/10.1007/s10113-017-1239-9

Larue Jean Piere, 2008. Effects of tectonics and lithology on long profiles of 16 rivers of the southern Central Massif border between the Aude and the Orb (France). Geomorphology 93, 343-367.

Layton, J., Skidmore, E., Thompson, C., 1993. Winter-associated changes in dry-soil aggregation as influenced by management. Soil Sci. Soc. Am. J. 57, 1568–1572.

Le Bissonnais, Y., Blavet, D., De Noni, G., Laurent, J., Asseline, J., Chenu, C., 2007. Erodibility of Mediterranean vineyard soils: relevant aggregate stability methods and significant soil variables. Eur. J. Soil Sci. 58, 188–195.

Le Bissonnais, Y., Le Souder, C., 1995. Mesurer la stabilité structurale des sols pour évaluer leur sensibilité à la battance et à l'érosion. Etude Gest. Sols 2, 43–56.

Le Bissonnais, Y., Prieto, I., Roumet, C., Nespoulous, J., Metayer, J., Huon, S., Villatoro, M., Stokes, A., 2018. Soil aggregate stability in Mediterranean and tropical agro-ecosystems: effect of plant roots and soil characteristics. Plant Soil 424, 303–317.

Le Bissonnais Y., 2016. Aggregate stability and assessment of soil crustability and erodibility: I. Theory and methodology. Eur. J. Soil Sci. 67, 11–21. https://doi.org/10.1111/ejss.4_12311

Lehmann, J., Kinyangi, J., Solomon, D., 2007. Organic matter stabilization in soil microaggregates: implications from spatial heterogeneity of organic carbon contents and carbon forms. Biogeochemistry 85, 45–57.

Lespinas F., Ludwig W., Heussner S., 2014. Hydrological and climatic uncertainties associated with modeling the impact of climate change on water resources of small Mediterranean coastal rivers. Journal of Hydrology 511, 403-422.

Lespinas, F., Ludwig, W., Heussner, S., 2009. Impact of recent climate change on the hydrology of coastal Mediterranean rivers in Southern France. Clim. Change. http://dx.doi.org/10.1007/s10584-009-9668-1

Li, J., Pu, L., Zhu, M., and Zhang, R. 2012. The present situation and hot issues in the salt-affected soil research, Acta Geographica Sinica, 67, 1233-1245, 2012.

Lin, Q., Ishikawa, T., Akoh, R., Yang, F., Zhang, S., 2016. Soil salinity reduction by river water irrigation in a reed field: A case study in Shuangtai Estuary Wetland, Northeast China.

Loch, R., 1994. A method for measuring aggregate water stability of dryland soils with relevance to surface seal development. Soil Res. 32, 687–700.

Logan, T., Lal, R., Dick, W., 1991. Tillage systems and soil properties in North America. Soil Tillage Res. 20, 241–270.

Maas, E.V., Hoffman, G.J., 1977. Crop salt tolerance\-current assessment. J. Irrig. Drain. Div. 103, 115–134.

Margoluis, R., Stem, C., Salafsky, N., Brown, M., 2009. Using conceptual models as a planning and evaluation tool in conservation. Evaluation and Program Planning 32(2), 138-147.

Machado, R.M.A., Serralheiro, R.P., 2017. Soil salinity: Effect on vegetable crop growth. Management practices to prevent and mitigate soil salinization. Horticulturae 3, 30.

Martens, D.A., 2000. Plant residue biochemistry regulates soil carbon cycling and carbon sequestration. Soil Biol. Biochem. 32, 361–369.

Martinez-Beltran, J., Manzur, C.L., 2005. Overview of salinity problems in the world and FAO strategies to address the problem. Proc. Int. Salin. Forum Riverside Calif.

Metternicht, G., Zinck, A., 2008. Remote sensing of soil salinization: Impact on land management. CRC Press.

Météo France: http://climatheque.meteo.fr

Michot, D., Walter, C., Adam, I., Guéro, Y., 2013. Digital assessment of soil-salinity dynamics after a major flood in the Niger River valley. Geoderma 207–208, 193–204.

Mikha, M.M., Rice, C.W., 2004. Tillage and Manure Effects on Soil and Aggregate-Associated Carbon and Nitrogen Contribution No. 03-86-J of Kansas Agric. Exp. Stn. Soil Sci. Soc. Am. J. 68, 809–816. https://doi.org/10.2136/sssaj2004.8090

Mongelli, G., Monni, S., Oggiano, G., Paternoster, M., Sinis, R., 2013. Tracing groundwater salinization processes in coastal aquifers: a hydrogeochemical and isotopic approach in the Na-Cl brackish waters of northwestern Sardinia, Italy. Hydrol Earth Syst Sci 17 2917–2928. https://doi.org/10.5194/hess-17-2917-2013

Montanarella, L., 2007. Trends in land degradation in Europe. Clim. Land Degrad. 83–104.

Muggler, C.C., van Griethuysen, C., Buurman, P., Pape, T., 1999. Aggregation, organic matter, and iron oxide morphology in Oxisols from Minas Gerais, Brazil. Soil Sci. 164, 759–770.

Neumann, D., Heuer, A., Hemkemeyer, M., Martens, R., Tebbe, C.C., 2013. Response of microbial communities to long-term fertilization depends on their microhabitat. FEMS Microbiol. Ecol. 86, 71–84.

Nguyen, A.D., Savenije, H.H., 2006. Salt intrusion in multi-channel estuaries: a case study in the Mekong Delta, Vietnam. Hydrol. Earth Syst. Sci. 10, 743–754. https://doi.org/10.5194/hess-10-743-2006

Oades, J.M., 1984. Soil organic matter and structural stability: mechanisms and implications for management. Plant Soil 76, 319–337.

Odeh, I.O., Onus, A., 2008. Spatial analysis of soil salinity and soil structural stability in a semiarid region of New South Wales, Australia. Environ. Manage. 42, 265.

Pagliai, M., Vignozzi, N., Pellegrini, S., 2004. Soil structure and the effect of management practices. Soil Tillage Res. 79, 131–143.

Pansu, M. and Gautheyrou, J., 2006. Handbook of soil analysis. Mineralogical, Organic and Inorganic Methods. Springer Berlin Heidelberg New York.

Pare, T., Dinel, H., Moulin, A., Townley-Smith, L., 1999. Organic matter quality and structural stability of a Black Chernozemic soil under different manure and tillage practices. Geoderma 91, 311–326.

Payen, S., Basset-Mens, C., Núñez, M., Follain, S., Grünberger, O., Marlet, S., Perret, S., Roux, P., 2016. Salinisation impacts in life cycle assessment : a review of challenges and options towards their consistent integration. Int. J. Life Cycle Assess. 21, 577-594.

Pearson, K.E., Bauder, J.W., 2006. The basics of salinity and sodicity effects on soil physical properties. MSU Ext. Water Qual. Program.

Petelet-Giraud, E., Négrel, P., Aunay, B., Ladouche, B., Bailly-Comte, V., Guerrot, C., Flehoc, C., Pezard, P., Lofi, J., Dörfliger, N., 2016. Coastal groundwater salinization: Focus on the vertical variability in a multi-layered aquifer through a multi-isotope fingerprinting (Roussillon Basin, France). Sci. Total Environ. 566–567, 398–415. https://doi.org/10.1016/j.scitotenv.2016.05.016

Pisinaras, V., Tsihrintzis, V.A., Petalas, C., Ouzounis, K., 2010. Soil salinization in the agricultural lands of Rhodope District, northeastern Greece. Environ. Monit. Assess. 166, 79–94. https://doi.org/10.1007/s10661-009-0986-6

Plante, A.F., McGill, W.B., 2002. Soil aggregate dynamics and the retention of organic matter in laboratory-incubated soil with differing simulated tillage frequencies. Soil Tillage Res. 66, 79–92. https://doi.org/10.1016/S0167-1987(02)00015-6

Potter, I. C., Chuwen, B.M., Hoeksema, S.D., Elliot, M., 2010. The concept of an estuary: A definition that incorporates systems which can become closed to the ocean and hypersaline. Estuar. Coast. Shelf Sci. 87, 497-500. doi:10.1016/j.ecss.2010.01.021.

Pritchard, D.W., 1967. What is an estuary: a physical viewpoint. Am. Assoc. Adv. Sci. 83 3–5.

Quirk, J., 2013. Landmark Papers: No. 2. Quirk, JP & Schofield, RK 1955. The effect of electrolyte concentration on soil permeability. Journal of Soil Science, 6, 163–178. Eur. J. Soil Sci. 64, 2–2.

Raats, P.A., 2015. Salinity management in the coastal region of the Netherlands: A historical perspective. Agric. Water Manag. 157, 12–30.

Raynaud, X., Nunan, N., 2014. Spatial ecology of bacteria at the microscale in soil. PLoS One 9, e87217.

Regelink, I.C., Stoof, C.R., Rousseva, S., Weng, L., Lair, G.J., Kram, P., Nikolaidis, N.P., Kercheva, M., Banwart, S., Comans, R.N., 2015. Linkages between aggregate formation, porosity and soil chemical properties. Geoderma 247, 24–37.

Rengasamy, P., 2010. Soil processes affecting crop production in salt-affected soils. Funct. Plant Biol. 37, 613–620.

Rengasamy, P., 2006a. World salinization with emphasis on Australia. J. Exp. Bot. 57, 1017–1023. https://doi.org/10.1093/jxb/erj108

Rengasamy, P., Greene, R., Ford, G., 1986. Influence of magnesium on aggregate stability in sodic red-brown earths. Soil Res. 24, 229–237.

Rhoades, J., Chanduvi, F., Lesch, S, 1999. Soil salinity assessment: Methods and interpretation of electrical conductivity measurements. Food & Agriculture Org.

Rhoades, J., 1997. Salinity : Electrical conductivity and total dissolved solids. Methods Soil Anal. Part 3-Chemical Methods 417-435.

Richards, L.A., 1974. Diagnostico y rehabilitacion de suelos salinos y alkalios. Limusa, Mexico.

Richard, L.A., 1954. Diagnosis and improvement of saline and alkaline soils. Agricultural Handbook No.60. US Department of Agriculture. https://www.ars.usda.gov/pacific-west-area/riverside-ca/us-salinity-laboratory/docs/handbook-no-60/ (accessed 26.27.2017).

Rietz, D., Haynes, R., 2003. Effects of irrigation-induced salinity and sodicity on soil microbial activity. Soil Biol. Biochem. 35, 845–854.

Rounsevell, M.D.A., Ewert, F., Reginster, I., Leemans, R., Carter, T.R., 2005. Future scenarios of European agricultural land use: II. Projecting changes in cropland and grassland. Agriculture, Ecosystems & Environment 107, 117–135.

Saggar, S., Yeates, G., Shepherd, T., 2001. Cultivation effects on soil biological properties, microfauna and organic matter dynamics in Eutric Gleysol and Gleyic Luvisol soils in New Zealand. Soil Tillage Res. 58, 55–68.

Salome, C., Coll, P., Lardo, E., Villenave, C., Blanchart, E., Hinsinger, P., Marsden, C., Le Cadre, E., 2014. Relevance of use-invariant soil properties to assess soil quality of vulnerable ecosystems: The case of Mediterranean vineyards. Ecol. Indic. 43, 83–93. https://doi.org/10.1016/j.ecolind.2014.02.016

Saviozzi, A., Cardelli, R., Labbaci, L., Levi Minzi, R., Riffaldi, R., 2007. Selected enzyme activities in particle-size fractions from an organically and conventionally managed soil. Fresenius Environ. Bull. 16, 1195–1200.

Schiefer, J., Lair, G.J., Blum, W.E.H., 2016. Potential and limits of land and soil for sustainable intensification of European agriculture. Agric. Ecosyst. Environ. 230, 283–293. https://doi.org/10.1016/j.agee.2016.06.021

Schimel, J., Balser, T.C., Wallenstein, M., 2007. Microbial stress-response physiology and its implications for ecosystem function. Ecology 88, 1386–1394.

Setia, R., Gottschalk, P., Smith, P., Marschner, P., Baldock, J., Setia, D., Smith, J., 2013. Soil salinity decreases global soil organic carbon stocks. Sci. Total Environ. 465, 267–272.

Setia, R., Marschner, P., 2013. Carbon mineralization in saline soils as affected by residue composition and water potential. Biol. Fertil. Soils 49, 71–77. https://doi.org/10.1007/s00374-012-0698-x

Setia, R., Marschner, P., Baldock, J., Chittleborough, D., Verma, V., 2011. Relationships between carbon dioxide emission and soil properties in salt-affected landscapes. Soil Biol. Biochem. 43, 667–674. https://doi.org/10.1016/j.soilbio.2010.12.004

Shani, U., Ben-Gal, A., 2005. Long-term response of grapevines to salinity: osmotic effects and ion toxicity. Am. J. Enol. Vitic. 56, 148–154.

Shrestha, R.P., 2006. Relating soil electrical conductivity to remote sensing and other soil properties for assessing soil salinity in northeast Thailand. Land Degradation and Development 17, 677–689.

Six, J., Bossuyt, H., Degryze, S., Denef, K., 2004. A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. Soil Tillage Res. 79, 7–31.

Szabolcs, I., 1989. Salt Affected Soils. CRC Press.

Szabolcs, I., 1992. Salinization of soil and water and its relation to desertification. Desertification Bulletin 21, 27–32.

Szabolcs, I., 1985. Salt affected soils as a world problem. The reclamation of salt-affected soils. Proceedings of an International Symposium. Jinan, China, pp. 30-47.

Thornthwaite, C.W., 1948. An Approach toward a Rational Classification of Climate. Geogr. Rev.38, 55-94. doi:10.2307/210739.

Tivet, F., de Moraes Sa, J.C., Lal, R., Briedis, C., Borszowskei, P.R., dos Santos, J.B., Farias, A., Eurich, G., da Cruz Hartman, D., Junior, M.N., 2013. Aggregate C depletion by plowing and its restoration by diverse biomass-C inputs under no-till in sub-tropical and tropical regions of Brazil. Soil Tillage Res. 126, 203–218.

Tisdall, J.M., Oades, J., 1982. Organic matter and water-stable aggregates in soils. J. Soil Sci. 33, 141–163.

Tóth, G., Montanarella, L., Rusco, E., 2008. Threats to soil quality in Europe. JRC Scientific and Technical Report. EUR 23438-Scientific and technical research series. Off. Publi. Eur. Communities Luxemb. 61-74.

Totsche, K.U., Amelung, W., Gerzabek, M.H., Guggenberger, G., Klumpp, E., Knief, C., Lehndorff, E., Mikutta, R., Peth, S., Prechtel, A., 2018. Microaggregates in soils. J. Plant Nutr. Soil Sci. 181, 104–136.

Verburg, P.H., Soepboer, W., Veldkamp, A., Limpiada, R., Espaldon, V., 2002. Modelling the spatial dynamics of regional land use: the CLUE-S model. Environmental Management 30, 391–405.

Vogel, C., Babin, D., Pronk, G.J., Heister, K., Smalla, K., Kögel-Knabner, I., 2014. Establishment of macro-aggregates and organic matter turnover by microbial communities in long-term incubated artificial soils. Soil Biol. Biochem. 79, 57–67. https://doi.org/10.1016/j.soilbio.2014.07.012

Whitfield, A. and Elliot, M., 2011. Ecosystem and Biotic Classifications of Estuaries and Coasts, in: Treatise on Estuarine and Coastal Science. Academic Press, Waltham, pp. 99-124.

White, Robert, 2003. Soil for Fine Wines. Oxford University Press.

Wong, V.N.L., Greene, R.S.B., Dalal, R.C., Murphy, B.W., 2010. Soil carbon dynamics in saline and sodic soils: a review. Soil Use Manag. 26, 2–11. https://doi.org/10.1111/j.1475-2743.2009.00251.x

Wöppelmann G., Marcos M., 2012. Coastal sea level rise in southern Europe and the nonclimate contribution of vertical land motion. J. Geophys. Res. Oceans 117. https://doi.org/10.1029/2011JC007469

WRB, 2015. World reference base for soil resources 2014, International soil classification system for naming soils and creating legends for soil maps. Food and Agriculture Organization of the United Nations, Rome, Italy.

Wu, X., Wei, Y., Wang, J., Wang, D., She, L., Wang, J., Cai, C., 2017. Effects of soil physicochemical properties on aggregate stability along a weathering gradient. CATENA 156, 205–215. https://doi.org/10.1016/j.catena.2017.04.017

Wu, J., Li, P., Qian H., Fan Y., 2014. Assessment of soil salinization based on a low-cost method and its influencing factors in a semi-arid agricultural area, northwest China. Environ. Earth Sci. 71:3465-3475. doi: 10.1007/s12665-013-2736-x.

Xu, G., Magen, H., Tarchitzky, J., Kafkafi, U., 1999. Advances in chloride nutrition of plants, in: Advances in Agronomy. Elsevier, pp. 97–150.

Young, I.M., Crawford, J.W., 2004. Interactions and self-organization in the soil-microbe complex. Science 304, 1634–1637.

Yu, J., Li, Y., Han, G., Zhou, D., Fu, Y., Guan, B., Wang, G., Ning, K., Wu, H., Wang, J., 2014. The spatial distribution characteristics of soil salinity in coastal zone of the Yellow River Delta. Environ. Earth Sci. 72, 589–599. https://doi.org/10.1007/s12665-013-2980-0

Yuan, B.-C., Li, Z.-Z., Liu, H., Gao, M., Zhang, Y.-Y., 2007. Microbial biomass and activity in salt affected soils under arid conditions. Appl. Soil Ecol. 35, 319–328.

ANNEX 1a. Soil analysis for 2 fields (Data of Chapter 3 and 4)

Dossier : 1605-0014

CIRAD - US Analyses TA B-49/01 34398
Montpellier Cedex 5

Client : MONTPELLIER SUPAGRO

Interlocuteur client : Stéphane Follain

Analyse de sols de Sérignan

N° Labo :			US1605-00287	US1605-00288	US1605-00289	US1605-00290	US1605-00291	US1605-00292	US1605-00293	US1605-00294	US1605-00295
Code Echantillon :			F1-1-1	F1-1-1	F1-1-1	F1-1-1	F1-1-1	F1-1-2	F1-1-2	F1-1-2	Figue/F1-1-2
Identifiant 2 :			H1	H2	H3	H4	H5	H1	H2	H3	H4
Identifiant 3 :			0-40	40-60	60-90	90-100	>100 (100-130)	0-40	40-60	60-130	130-160
(03) Mesure du pH (extraction à l'eau)	**(1) pH**		8.04	8.19	8.26	8.45	8.58	8.26	8.34	8.51	8.87
(05) Mesure de la matière organique par combustion sèche (Dumas)	**(1) MO**	%	3.88	3.06	3.12	2.94	2.88	3.95	3.07	2.94	2.48
	(2) Corg	%	2.25	1.77	1.81	1.71	1.67	2.29	1.78	1.71	1.44
	(3) N	‰	1.20	0.84	0.74	0.65	0.64	1.01	0.71	0.63	0.61
	(4) C/N		18.71	21.22	24.46	26.08	26.13	22.63	25.08	27.30	23.68
(06) Granulométrie 5 fractions (sans décarbonatation)	**(1) Argiles**	%	18.3	17.8	16.2	12.7	14.8	17.4	14.6	14.7	16.6
	(2) Limons fins	%	37.8	44.2	44.6	34.7	41.9	34.1	30.3	35.5	35.6
	(3) Limons grossiers	%	22.7	25.3	26.2	29.6	28.0	19.2	22.4	26.0	27.6
	(4) Sables fins	%	20.2	12.5	12.7	22.7	15.1	27.1	28.5	21.8	19.5
	(5) Sables grossiers	%	1.0	0.2	0.2	0.3	0.2	2.3	4.2	2.0	0.6
(08) Dosage par colorimétrie du P assimilable (Olsen)	**(1) P**	mg/kg	18,6	8	6,8	8,6	9,2	7	5,8	6,8	7,4

(10) Dosage ICP-AES du complexe absorbant (Co(NH3)6Cl3)	**(1) Ca**	me/100g	8,91	NA	8,54	6,86	5,59	9,36	8,85	7,51	5,59
	(2) Mg	me/100g	1,46	NA	2,59	2,52	2,34	1,27	1,44	2,49	3,12
	(3) K	me/100g	0,36	NA	0,06	0,09	0,08	0,05	0,04	0,02	<0,01
	(4) Na	me/100g	0,00	NA	0,09	0,03	<0,01	0,14	0,17	0,26	0,24
	(5) Al	me/100g	<0,01	NA	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01
	(6) Mn	me/100g	<0,01	NA	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01
	(7) H	me/100g	<0,01	NA	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01
	(8) S(Ca,Mg, K,Na)	me/100g	10,73	NA	11,27	9,50	8,02	10,83	10,50	10,27	8,95
	(9) CEC	me/100g	12,85	NA	12,35	11,25	8,85	12,25	11,55	11,35	11,05
	(10) TS	%	83,53	NA	91,27	84,45	90,58	88,38	90,91	90,46	80,98
	(11) pHCo		8,26	NA	8,35	8,41	8,46	8,30	8,32	8,25	8,37
(17) Mesure de la Conductivité Electrique après extraction pâte saturée	**(1) CE**	µS/cm	7.02ms/cm	7.38	8.01	9.11	8.92	2.34	1646µS/cm	2.66ms/cm	2.34
(19) Dosage ICP-AES du Calcium soluble à l'eau	**(1) Ca**	mg/kg	417,21	262,21	156,94	140,60	267,77	187,92	144,13	90,74	68,04
(19) Dosage ICP-AES du Calcium total après extraction pâte saturée	**(1) Ca**	mg/kg	350,64	225,70	193,47	151,71	133,36	118,13	101,71	61,07	21,14
(20) Dosage ICP-AES du Magnésium soluble à l'eau	**(1) Mg**	mg/kg	76,77	72,14	59,39	61,90	109,87	47,58	41,04	51,93	84,16
(20) Dosage ICP-AES du Magnésium total après extraction pâte saturée	**(1) Mg**	mg/kg	62,16	56,81	60,17	59,82	61,98	17,66	19,05	23,51	16,23
(21) Dosage ICP-AES du Potassium soluble à l'eau	**(1) K**	mg/kg	71,05	18,42	24,92	24,57	20,45	50,40	37,36	42,12	84,14
(21) Dosage ICP-AES du Potassium total après extraction pâte saturée	**(1) K**	mg/kg	17,66	1,34	1,24	1,23	1,77	1,56	1,51	0,70	0,66

(22) Dosage ICP-AES du Sodium après extraction pâte saturée	**(1) Na**	mg/kg	271,00	334,42	331,64	551,02	722,11	88,90	104,19	158,64	183,58
(22) Dosage ICP-AES du Sodium soluble à l'eau	**(1) Na**	mg/kg	368,93	596,49	768,18	908,67	1148,35	159,40	180,17	281,59	395,62
(23) Dosage ICP-AES du Fer total après extraction pâte saturée	**(1) Fe**	mg/kg	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
(24) Dosage ICP-AES de l'Aluminium total après extraction pâte saturée	**(1) Al**	mg/kg	0,09	0,05	0,04	0,03	0,03	0,04	0,02	0,01	0,02
(25) Dosage ICP-AES du Manganèse total après extraction pâte saturée	**(1) Mn**	mg/kg	0,08	0,02	0,02	0,01	0,02	0,02	0,02	0,01	0,01
(26) Dosage ICP-AES du Soufre total après extraction pâte saturée	**(1) S**	mg/kg	169,69	142,70	124,01	121,96	132,48	91,80	93,60	100,56	78,63
(27) Dosage ICP-AES du Phosphore total après extraction pâte saturée	**(1) P**	mg/kg	0,05	0,03	0,02	0,07	0,00	0,06	0,06	0,00	0,00
(28) Dosage ICP-AES du Silicium total après extraction pâte saturée	**(1) Si**	mg/kg	1,54	0,85	0,73	0,69	0,74	1,39	1,25	0,99	0,87
(30) Dosage par colorimétrie de l'azote nitreux après extraction pâte saturée	**(1) N-NO2**	mg/kg	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)	<LD (0.04)
(31) Dosage par colorimétrie de l'azote nitrique après extraction pâte saturée	**(1) N-NO3**	mg/kg	45.4	1.8	18.4	15.3	15.0	3.2	2.6	3.6	2.2
(32) Dosage par colorimétrie de l'azote ammoniacal après extraction pâte saturée	**(1) N-NH4**	mg/kg	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)	<LD (0.1)
(33) Dosage par colorimétrie du Chlore après extraction pâte saturée	**(1) Cl**	mg/kg	764.3	689.9	828.3	1001.1	1086.8	128.3	134.8	121.2	102.7

(34) Mesure des bicarbonates par titrimétrie après extraction pâte saturée	**(1) HCO3**	mg/kg	639,99	404,00	410,75	444,03	489,34	607,07	706,78	568,76	674,84
(35) Carbonates totaux (calcaire total)	**(1) CT**	%	5.7	6.5	5.9	5.7	6.1	6.5	6.1	6.7	7.0
(35) Mesure des carbonates par titrimétrie après extraction pâte saturée	**(1) CO3**	mg/kg	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00

ANNEX 1b. Soil Analysis of 3 fields (Data of Chapter 4)

Dossier : 1712-0009

CIRAD - US Analyses TA B-49/01 34398 Montpellier Cedex 5

Client : INRA

Interlocuteur client : Stéphane Follain

Analyse de sols de Sérignan

N° Labo :			US1712-00397	US1712-00398	US1712-00399	US1712-00400	US1712-00401	US1712-00402	US1712-00403	US1712-00404	US1712-00405	US1712-00406	US1712-00407	US1712-00408
Code Echantillon :			F1-2-1	F1-2-1	F1-2-1	F1-2-1	F2-1-1	F2-1-1	F2-1-1	F2-1-1	F1-2-2	F1-2-2	F1-2-2	F1-2-2
Identifiant 2 :			H1	H2	H3	H4	H1	H2	H3	H4	H1	H2	H3	H4
Identifiant 3 :			0-15	15-40	40-80	>80	0-20	20-60	60-91	>91	0-23	23-52	52-71	>71
(03) Mesure du pH (extraction à l'eau)	**(1) pH**		7,96	8,32	8,30	8,38	8,04	8,30	8,39	8,44	8,16	8,43	8,48	8,76
(05) Mesure de la matière organique par combustion sèche (Dumas)	**(1) MO**	%	3,98	2,84	2,51	2,14	3,59	2,64	1,68	1,78	3,48	2,07	1,01	0,40
	(2) Corg	%	2,31	1,65	1,45	1,24	2,08	1,53	0,97	1,03	2,02	1,20	0,58	0,23
	(3) N	‰	1,58	0,91	0,75	0,75	1,50	0,92	0,64	0,62	1,35	0,84	0,60	0,19
	(4) C/N		14,57	18,12	19,27	16,57	13,83	16,64	15,24	16,75	14,94	14,36	9,68	12,08
(06) Granulométrie 5 fractions (sans décarbonatation)	**(1) Argiles**	%	17.9	17.4	12.2	12.1	18.5	18.6	14.0	14.2	18.0	14.6	13.6	5.6
	(2) Limons fins	%	35.8	33.6	25.0	43.5	32.7	34.6	29.3	30.6	35.1	50.5	34.3	4.7
	(3) Limons grossiers	%	21.5	21.9	29.8	26.5	21.2	21.5	22.1	25.2	17.7	20.1	9.3	2.1
	(4) Sables fins	%	22.5	24.4	32.3	16.6	25.6	23.4	32.1	28.5	22.1	11.4	20.6	32.4
	(5) Sables grossiers	%	2.3	2.6	0.7	1.4	1.9	1.9	2.6	1.6	7.1	3.3	22.2	55.2
(08) Dosage par colorimétrie du P assimilable (Olsen)	**(1) P**	mg/kg	46.86	7.09	6.49	12.74	23.96	11.17	6.38	5.13	8.91	3.28	3.00	6.54
(10) Dosage ICP-AES du complexe	**(1) Ca**	me/100g	10.96	9.84	9.91	9.79	10.06	9.87	9.34	8.98	9.73	10.17	6.87	2.10
	(2) Mg	me/100g	0.95	1.14	1.26	1.89	0.93	1.35	1.18	1.36	1.10	1.81	1.60	0.46

absorbant (Co(NH3)6Cl3)	**(3) K**	me/100g	0.77	0.15	<0.01	<0.01	0.60	<0.01	<0.01	<0.01	0.31	<0.01	<0.01	<0.01
	(4) Na	me/100g	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	(5) Al	me/100g	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	(6) Mn	me/100g	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	(7) H	me/100g	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	(8) S(Ca,Mg,K,Na)	me/100g	12.68	11.13	11.17	11.68	11.60	11.23	10.52	10.33	11.14	11.97	8.47	2.56
	(9) CEC	me/100g	12.60	9.70	10.60	10.70	11.30	10.50	10.50	9.80	10.80	11.30	7.60	2.41
	(10) TS	%	100.62	114.76	105.39	109.14	102.62	106.91	100.18	105.44	103.18	105.96	111.49	106.35
	(11) pHCo		7.806	7.954	7.967	8.084	7.905	8.013	7.977	8.02	7.964	7.997	8.17	8.476
(17) Mesure de la Conductivité Electrique après extraction pâte saturée	**(1) CE**	µS/cm	2500	687	893	1915	2380	800	494	530	2130	792	1223	948
(19) Dosage ICP-AES du Calcium soluble à l'eau	**(1) Ca**	mg/kg	307.4	186.9	216.4	277.5	285.7	185.2	189.4	168.8	270.4	185.9	169.6	134.6
(19) Dosage ICP-AES du Calcium total après extraction pâte saturée	**(1) Ca**	mg/kg	176.97	50.02	62.68	123.76	172.00	51.43	35.38	32.39	150.48	46.38	62.15	40.59
(20) Dosage ICP-AES du Magnésium soluble à l'eau	**(1) Mg**	mg/kg	195.6	51.0	21.4	16.1	135.6	33.0	32.6	26.5	84.8	24.9	50.6	12.3
(20) Dosage ICP-AES du Magnésium total après extraction pâte saturée	**(1) Mg**	mg/kg	31.88	7.17	8.64	22.14	25.64	7.85	4.70	5.48	22.30	8.16	17.60	11.83
(21) Dosage ICP-AES du Potassium soluble à l'eau	**(1) K**	mg/kg	58.7	34.4	31.8	46.7	49.7	37.7	36.0	36.7	50.4	41.1	66.0	26.3
(21) Dosage ICP-AES du Potassium total après extraction pâte saturée	**(1) K**	mg/kg	41.04	2.26	<0.01	0.379	24.61	0.405	<0.01	<0.01	11.05	<0.01	<0.01	<0.01

(22) Dosage ICP-AES du Sodium après extraction pâte saturée	**(1) Na**	mg/kg	7.84	6.77	11.66	54.34	9.83	13.67	12.48	19.01	36.82	27.43	47.82	30.21
(22) Dosage ICP-AES du Sodium soluble à l'eau	**(1) Na**	mg/kg	15.4	19.2	30.1	161.9	109.7	227.4	36.5	49.5	55.3	72.4	78.6	34.9
(23) Dosage ICP-AES du Fer total après extraction pâte saturée	**(1) Fe**	mg/kg	0.286	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
(24) Dosage ICP-AES de l'Aluminium total après extraction pâte saturée	**(1) Al**	mg/kg	0.049	<0.01	<0.01	0.016	0.026	<0.01	<0.01	<0.01	0.043	<0.01	<0.01	<0.01
(25) Dosage ICP-AES du Manganèse total après extraction pâte saturée	**(1) Mn**	mg/kg	<0.01	<0.01	<0.01	0.017	<0.01	<0.01	<0.01	<0.01	<0.01	0.013	0.024	<0.01
(26) Dosage ICP-AES du Soufre total après extraction pâte saturée	**(1) S**	mg/kg	15.22	8.58	12.53	62.50	15.83	12.03	11.84	23.20	19.67	18.72	39.31	24.01
(27) Dosage ICP-AES du Phosphore total après extraction pâte saturée	**(1) P**	mg/kg	0.086	<0.01	<0.01	0.010	0.025	0.038	<0.01	0.016	0.051	<0.01	0.046	<0.01
(28) Dosage ICP-AES du Silicium total après extraction pâte saturée	**(1) Si**	mg/kg	4.81	1.99	1.27	1.10	2.85	1.96	1.22	1.26	2.15	1.01	0.688	0.509
(30) Dosage par colorimétrie de l'azote nitreux après extraction pâte saturée	**(1) N-NO2**	mg/kg	2.81	0.021 <LD(0.03 mg/kg)	0.0168 <LD(0.03 mg/kg)	0.0168 <LD(0.03 mg/kg)	0.13	0.0168 <LD(0.03 mg/kg)	4.04	0.016 <LD(0.03 mg/kg)	2.88	0.08	0.64	0.016 <LD(0.03 mg/kg)
(31) Dosage par colorimétrie de l'azote nitrique après	**(1) N-NO3**	mg/kg	143.64	25.62	26.88	10.67	126.42	19.49	5.96	2.36	106.80	9.00	1.40	0.80

extraction pâte saturée														
(32) Dosage par colorimétrie de l'azote ammoniacal après extraction pâte saturée	**(1) N-NH4**	mg/kg	0.0294 <LD(0.08 mg/kg)	-0.0084 <LD(0.08 mg/kg)	-0.0168 <LD(0.08 mg/kg)	0.30	-0.0084 <LD(0.08 mg/kg)	-0.0168 <LD(0.08 mg/kg)	0.02 <LD(0.08 mg/kg)	-0.016 <LD(0.08 mg/kg)	-0.016 <LD(0.08 mg/kg)	-0.02 <LD(0.08 mg/kg)	0.22	-0.024 <LD(0.08mg/kg)
(33) Dosage par colorimétrie du Chlore après extraction pâte saturée	**(1) Cl**	mg/kg	4.67	6.62	23.26	146.75	11.52	27.84	12.92	8.66	24.86	36.52	86.80	51.12
(34) Mesure des bicarbonates par titrimétrie après extraction pâte saturée	**(1) HCO3**	mg/kg	68.383	60.219	59.109	56.083	61.874	61.553	45.311	46.577	51.623	55.961	39.284	29.708
(35) Carbonates totaux (calcaire total)	**(1) CT**	%	6,22	7,19	6,51	8,03	5,79	5,66	6,59	6,10	8,04	7,79	9,51	13,94
(35) Mesure des carbonates par titrimétrie après extraction pâte saturée	**(1) CO3**	mg/kg	0	0	0	0	0	0	0	0	0	0	0	0

ANNEX 2. Measurement of Mean Weight Diameter (MWD) (Data of Chapter 4)

				Initial	Soil weight of each diameter size							Calculation (mm)							MWD	Mean
No	Samp. Date	Simbol	Hor.	3-5 mm	> 2	1-2	0.5-1	0.2-0.5	0.1-0.2	0.05-0.1	<0.05	> 2	1-2	0.5-1	0.2-0.5	0.1-0.2	0.05-0.1	< 0.05	(mm)	
1	3-Feb-16	F1-1-1,0-40 /I	H1	5.05	0.59	0.19	0.13	0.48	1.84	1.35	0.47	11.74	3.74	2.59	9.54	36.40	26.69	9.29	0.60	0.71
	3-Feb-16	II		5.06	0.68	0.20	0.13	0.42	2.36	0.79	0.48	13.34	3.87	2.59	8.36	46.61	15.66	9.57	0.66	
	3-Feb-16	III		5.05	1.02	0.25	0.10	0.24	1.65	1.42	0.39	20.14	4.85	1.96	4.65	32.60	28.12	7.66	0.88	
2	3-Feb-16	F1-1-1,40-60 /I	H2	5.05	0.46	0.22	0.22	0.94	2.25	0.78	0.19	9.09	4.26	4.36	18.54	44.46	15.51	3.78	0.56	0.55
	3-Feb-16	II		5.05	0.44	0.18	0.12	0.64	2.49	0.84	0.34	8.80	3.59	2.28	12.70	49.36	16.60	6.68	0.51	
	3-Feb-16	III		5.04	0.55	0.18	0.16	0.68	2.11	1.12	0.24	10.93	3.65	3.10	13.40	41.95	22.23	4.74	0.59	
3	3-Feb-16	F1-1-1,60-90 /I	H3	5.02	0.51	0.13	0.10	0.54	1.73	1.25	0.76	10.06	2.53	2.01	10.76	34.56	24.93	15.15	0.52	0.60
	3-Feb-16	II		5.05	0.64	0.15	0.14	0.50	1.78	1.36	0.50	12.58	2.93	2.71	9.89	35.20	26.85	9.83	0.61	
	3-Feb-16	III		5.07	0.70	0.20	0.13	0.50	1.68	1.43	0.44	13.78	4.03	2.47	9.85	33.08	28.14	8.66	0.67	
4	3-Feb-16	F1-1-1,90-100 /I	H4	5.05	0.21	0.11	0.13	0.97	1.78	1.16	0.69	4.12	2.24	2.60	19.16	35.26	22.94	13.69	0.34	0.38
	3-Feb-16	II		5.06	0.31	0.18	0.10	1.23	1.97	0.94	0.33	6.03	3.52	2.04	24.37	39.00	18.52	6.52	0.44	
	3-Feb-16	III		5.05	0.22	0.19	0.15	0.71	2.33	1.09	0.37	4.28	3.67	2.87	14.03	46.15	21.64	7.37	0.36	
5	3-Feb-16	F1-1-1,100-130 /I	H5	5.05	0.35	0.22	0.12	0.37	2.60	1.07	0.34	6.83	4.43	2.28	7.24	51.39	21.16	6.67	0.44	0.55
	3-Feb-16	II		5.07	0.72	0.20	0.18	0.40	2.38	0.86	0.33	14.29	3.97	3.53	7.79	47.00	16.97	6.45	0.70	
	3-Feb-16	III		5.07	0.46	0.16	0.10	0.35	2.85	0.79	0.36	9.06	3.06	1.95	6.90	56.30	15.60	7.12	0.50	
6	4-Feb-16	F1-1-2,0-10/I	H1	5.09	0.15	0.12	0.16	1.28	1.99	1.09	0.31	2.91	2.26	3.20	25.12	39.02	21.45	6.03	0.32	0.39
	4-Feb-16	II		5.01	0.19	0.21	0.18	0.97	2.31	0.95	0.21	3.75	4.15	3.55	19.31	46.12	18.93	4.19	0.37	
	4-Feb-16	III		5.07	0.37	0.16	0.15	0.95	2.28	0.83	0.32	7.36	3.22	2.92	18.80	45.04	16.41	6.25	0.47	
7	4-Feb-16	F1-1-2,20-40/I	H2	5.05	0.58	0.16	0.15	0.57	1.90	1.27	0.42	11.54	3.15	3.03	11.36	37.60	25.05	8.27	0.59	0.54
	4-Feb-16	II		5.06	0.38	0.17	0.14	0.58	2.63	0.79	0.37	7.59	3.32	2.67	11.44	52.04	15.61	7.33	0.47	
	4-Feb-16	III		5.06	0.55	0.17	0.13	0.59	2.18	1.12	0.32	10.96	3.34	2.51	11.59	43.11	22.10	6.39	0.58	
8	4-Feb-16	F1-1-2,40-60/I	H3	5.06	0.17	0.13	0.17	1.20	1.79	1.13	0.47	3.38	2.53	3.38	23.68	35.40	22.28	9.35	0.34	0.34

	4-Feb-16	II		5.06	0.13	0.10	0.18	1.30	1.81	1.12	0.42	2.57	2.04	3.48	25.78	35.73	22.16	8.25	0.31	
	4-Feb-16	III		5.04	0.22	0.19	0.15	1.06	1.69	1.38	0.35	4.32	3.83	2.97	21.05	33.54	27.28	7.00	0.38	
9	4-Feb-16	F1-1-2,60-130/I	H4	5.03	0.49	0.20	0.10	0.64	1.77	1.35	0.49	9.68	3.93	1.95	12.70	35.23	26.76	9.76	0.53	0.42
	4-Feb-16	II		5.03	0.32	0.08	0.09	0.60	2.11	1.51	0.31	6.35	1.67	1.69	11.98	42.03	30.04	6.25	0.39	
	4-Feb-16	III		5.04	0.24	0.13	0.10	0.54	1.87	1.31	0.85	4.67	2.62	1.93	10.68	37.18	26.07	16.86	0.33	
10	4-Feb-16	F1-1-2,130-150/I	H5	5.04	0.39	0.14	0.10	0.49	1.78	1.57	0.57	7.70	2.76	2.04	9.80	35.32	31.07	11.31	0.44	0.44
	4-Feb-16	II		5.05	0.48	0.27	0.10	0.51	1.62	1.50	0.57	9.47	5.33	2.06	10.12	32.01	29.77	11.25	0.54	
	4-Feb-16	III		5.05	0.27	0.15	0.10	0.45	1.61	1.78	0.71	5.34	2.91	1.96	8.80	31.84	35.20	13.95	0.35	
11	6-Mar-17	F1-2-1, 0 cm/l	H0	5.03	0.07	0.10	0.14	0.64	1.44	1.68	0.96	1.45	1.97	2.73	12.66	28.68	33.35	19.16	0.22	0.28
	6-Mar-17	II		5.04	0.09	0.14	0.17	0.57	1.27	1.58	1.22	1.86	2.84	3.33	11.36	25.16	31.26	24.19	0.24	
	6-Mar-17	III		5.03	0.26	0.22	0.15	0.64	1.26	1.51	1.00	5.09	4.35	3.06	12.70	25.02	29.95	19.82	0.38	
12	6-Mar-17	F1-2-1,0-15 cm/I	H1	5.03	0.16	0.17	0.16	0.76	1.52	1.54	0.73	3.16	3.34	3.08	15.00	30.28	30.58	14.56	0.31	0.28
	6-Mar-17	II		5.04	0.14	0.12	0.11	0.74	1.97	0.98	0.98	2.68	2.46	2.12	14.75	39.14	19.43	19.43	0.28	
	6-Mar-17	III		5.04	0.12	0.13	0.09	0.58	1.76	1.33	1.03	2.42	2.60	1.80	11.56	34.89	26.40	20.33	0.25	
13	6-Mar-17	F1-2-1, 15-40 cm/I	H2	5.03	0.23	0.17	0.10	0.53	1.64	1.63	0.74	4.59	3.30	2.07	10.48	32.59	32.33	14.64	0.34	0.35
	6-Mar-17	II		5.03	0.05	0.11	0.08	0.67	1.61	1.39	1.12	0.93	2.17	1.61	13.28	32.00	27.68	22.34	0.20	
	6-Mar-17	III		5.03	0.43	0.26	0.17	0.60	1.83	1.32	0.41	8.55	5.19	3.42	12.00	36.42	26.18	8.24	0.52	
14	6-Mar-17	F1-2-1, 40-80 cm/I	H3	5.04	0.54	0.25	0.22	0.65	1.64	1.19	0.56	10.62	4.94	4.37	12.98	32.49	23.58	11.03	0.59	0.49
	6-Mar-17	II		5.04	0.15	0.19	0.11	0.56	1.92	1.16	0.96	2.93	3.73	2.20	11.01	38.09	22.92	19.12	0.29	
	6-Mar-17	III		5.05	0.51	0.25	0.22	0.59	1.49	1.41	0.58	10.09	4.94	4.44	11.73	29.47	27.85	11.48	0.57	
15	6-Mar-17	F1-2-1, >80 cm/I	H4	5.03	0.07	0.17	0.24	0.98	1.22	1.14	1.21	1.45	3.44	4.77	19.40	24.25	22.56	24.13	0.27	0.34
	6-Mar-17	II		5.04	0.15	0.27	0.27	0.98	1.27	1.12	0.97	2.90	5.42	5.26	19.48	25.28	22.32	19.32	0.35	
	6-Mar-17	III		5.02	0.27	0.22	0.26	0.82	1.37	1.03	1.07	5.28	4.28	5.16	16.39	27.24	20.45	21.21	0.41	
16	6-Mar-17	F1-2-2, 0 cm/I	H0	5.05	0.31	0.28	0.22	0.81	1.43	1.09	0.91	6.17	5.50	4.31	16.03	28.37	21.57	18.05	0.45	0.51
	6-Mar-17	II		5.03	0.66	0.24	0.18	0.85	1.62	0.97	0.51	13.12	4.75	3.62	16.88	32.26	19.28	10.10	0.68	
	6-Mar-17	III		5.04	0.23	0.21	0.23	1.03	1.67	0.77	0.90	4.55	4.23	4.65	20.45	33.23	15.31	17.75	0.40	
17	6-Mar-17	F1-2-2; 0-23 cm/I	H1	5.03	0.27	0.19	0.12	0.98	1.50	0.89	1.09	5.29	3.72	2.29	19.48	29.87	17.73	21.64	0.39	0.48
	6-Mar-17	II		5.04	0.37	0.27	0.25	0.87	1.58	0.99	0.70	7.39	5.42	4.99	17.26	31.34	19.64	13.96	0.50	
	6-Mar-17	III		5.04	0.41	0.32	0.21	0.88	1.74	0.80	0.68	8.18	6.27	4.13	17.45	34.62	15.84	13.50	0.54	
18	6-Mar-17	F1-2-2; 23-55 cm/I	H2	5.05	0.29	0.21	0.22	1.23	1.75	0.76	0.58	5.72	4.18	4.43	24.41	34.67	15.13	11.46	0.45	0.47
	6-Mar-17	II		5.03	0.39	0.19	0.23	1.22	1.74	0.94	0.33	7.71	3.68	4.65	24.27	34.56	18.57	6.56	0.51	
	6-Mar-17	III		5.04	0.31	0.20	0.23	1.16	1.74	0.85	0.56	6.09	3.97	4.52	23.06	34.42	16.89	11.06	0.45	
19	6-Mar-17	F1-2-2; 55-72 cm/I	H3	5.04	0.87	0.22	0.15	0.97	1.59	0.85	0.39	17.22	4.39	3.06	19.24	31.51	16.92	7.66	0.82	0.77

	6-Mar-17	II		5.05	0.76	0.22	0.15	1.03	1.62	0.82	0.45	15.05	4.30	3.01	20.38	32.12	16.18	8.97	0.75	
	6-Mar-17	III		5.05	0.72	0.25	0.16	1.03	1.65	0.91	0.33	14.27	4.90	3.19	20.38	32.75	18.00	6.52	0.73	
20	6-Mar-17	F1-2-2; > 72/I	H4	5.04	0.17	0.10	0.09	2.47	1.75	0.17	0.30	3.45	1.90	1.80	48.91	34.66	3.29	5.99	0.39	0.36
	6-Mar-17	II		5.05	0.11	0.11	0.09	2.16	2.03	0.25	0.31	2.18	2.18	1.68	42.71	40.16	5.01	6.08	0.34	
	6-Mar-17	III		5.04	0.10	0.13	0.12	2.20	1.97	0.22	0.30	1.98	2.62	2.36	43.64	39.00	4.38	6.01	0.34	
21	7-Mar-17	F2-1-1; 0 cm/I	H0	5.03	0.49	0.31	0.24	0.62	1.74	1.07	0.56	9.74	6.14	4.71	12.29	34.58	21.36	11.18	0.58	0.69
	7-Mar-17	II		5.03	0.74	0.28	0.22	0.55	1.54	1.10	0.61	14.61	5.54	4.27	10.95	30.64	21.92	12.06	0.73	
	7-Mar-17	III		5.05	0.76	0.30	0.21	0.59	1.50	1.08	0.62	14.96	5.90	4.24	11.63	29.64	21.30	12.34	0.75	
22	7-Mar-17	F2-1-1; 0-20 cm/I	H1	5.04	1.11	0.29	0.26	0.61	1.29	0.86	0.63	22.03	5.65	5.23	12.07	25.48	17.13	12.41	0.99	1.10
	7-Mar-17	II		5.03	0.85	0.27	0.28	0.75	1.55	0.87	0.47	16.92	5.41	5.57	14.81	30.76	17.30	9.24	0.83	
	7-Mar-17	III		5.03	1.88	0.20	0.21	0.54	1.07	0.76	0.37	37.30	4.02	4.16	10.66	21.34	15.08	7.44	1.48	
23	7-Mar-17	F2-1-1; 20-60 cm/I	H2	5.02	0.60	0.23	0.16	0.53	1.86	1.23	0.41	12.01	4.50	3.19	10.46	37.05	24.56	8.23	0.62	0.64
	7-Mar-17	II		5.05	0.48	0.22	0.19	0.42	1.97	1.20	0.57	9.55	4.30	3.72	8.32	38.92	23.83	11.37	0.53	
	7-Mar-17	III		5.04	0.84	0.20	0.17	0.50	1.74	1.11	0.49	16.66	3.95	3.30	9.85	34.61	21.98	9.65	0.77	
24	7-Mar-17	F2-1-1; 60-90 cm/I	H3	5.05	0.68	0.18	0.12	0.46	2.08	1.12	0.43	13.42	3.52	2.30	9.05	41.13	22.15	8.43	0.65	0.49
	7-Mar-17	II		5.03	0.16	0.13	0.10	0.59	1.96	1.34	0.76	3.24	2.64	1.89	11.63	38.91	26.55	15.14	0.29	
	7-Mar-17	III		5.04	0.50	0.16	0.13	0.49	1.82	1.40	0.55	9.96	3.11	2.52	9.64	36.12	27.75	10.89	0.53	
25	7-Mar-17	F2-1-1; > 90/I	H4	5.03	0.44	0.15	0.11	0.53	2.13	1.21	0.46	8.67	2.90	2.23	10.58	42.33	24.09	9.19	0.48	0.40
	7-Mar-17	II		5.06	0.26	0.18	0.13	0.62	1.86	1.37	0.63	5.10	3.64	2.55	12.34	36.83	27.14	12.38	0.37	
	7-Mar-17	III		5.04	0.22	0.17	0.13	0.60	2.05	1.32	0.57	4.32	3.35	2.64	11.82	40.54	26.13	11.20	0.35	
26	7-Mar-17	F2-1-2; 0 cm/I	H0	5.03	1.29	0.37	0.32	0.52	0.87	0.57	1.10	25.62	7.39	6.30	10.25	17.27	11.23	21.94	1.13	1.24
	7-Mar-17	II		5.03	1.37	0.45	0.36	0.56	0.75	0.54	1.00	27.27	9.01	7.06	11.22	14.88	10.74	19.81	1.22	
	7-Mar-17	III		5.05	1.59	0.51	0.38	0.52	0.61	0.52	0.92	31.52	10.07	7.43	10.36	12.15	10.32	18.15	1.38	
27	7-Mar-17	F2-1-2; 0-25 cm/I	H1	5.04	0.69	0.36	0.39	0.87	1.32	0.69	0.72	13.63	7.14	7.82	17.32	26.17	13.61	14.31	0.76	0.83
	7-Mar-17	II		5.02	1.11	0.29	0.38	0.88	1.31	0.70	0.36	22.02	5.83	7.57	17.50	25.99	13.92	7.17	1.03	
	7-Mar-17	III		5.04	0.61	0.34	0.41	1.10	1.22	0.71	0.65	12.11	6.65	8.16	21.77	24.28	14.07	12.96	0.71	
28	7-Mar-17	F2-1-2; 25-60 cm/I	H2	5.03	1.06	0.26	0.18	0.46	1.30	1.10	0.68	21.08	5.23	3.60	9.12	25.77	21.76	13.45	0.93	1.16
	7-Mar-17	II		5.03	1.32	0.31	0.21	0.51	1.06	0.67	0.95	26.26	6.10	4.23	10.18	21.05	13.26	18.91	1.12	
	7-Mar-17	III		5.02	1.76	0.29	0.21	0.42	0.87	0.63	0.85	34.96	5.77	4.16	8.26	17.40	12.60	16.84	1.41	
29	24-Jun-17	F2-1-1 (Bio)/I	H0	5.07	0.21	0.22	0.19	1.02	1.69	1.06	0.68	4.10	4.26	3.81	20.06	33.42	20.89	13.45	0.38	0.40
	24-Jun-17	II		5.07	0.24	0.19	0.25	1.01	1.69	1.02	0.67	4.70	3.69	5.01	19.85	33.36	20.10	13.22	0.40	
	24-Jun-17	III		5.05	0.26	0.23	0.26	1.33	1.42	0.99	0.54	5.21	4.60	5.19	26.26	28.03	19.61	10.69	0.44	
30	24-Jun-17	F2-1-2 (Bio)/I	H0	5.06	0.51	0.39	0.27	0.89	1.10	0.86	1.04	10.14	7.73	5.29	17.48	21.72	16.93	20.52	0.62	0.68

	24-Jun-17	II		5.07	0.79	0.42	0.26	0.79	1.35	0.53	0.93	15.60	8.23	5.21	15.51	26.71	10.48	18.37	0.82	
	24-Jun-17	III		5.06	0.54	0.34	0.22	0.75	1.31	0.75	1.16	10.55	6.73	4.40	14.74	25.84	14.76	22.81	0.61	
31	24-Jun-17	F1-2-2 (Bio)/I	H0	5.04	0.50	0.31	0.29	1.90	1.22	0.50	0.32	9.77	6.14	5.70	37.46	24.13	9.92	6.37	0.65	0.72
	24-Jun-17	II		5.04	0.55	0.23	0.32	1.84	1.03	0.56	0.50	10.87	4.56	6.35	36.36	20.38	11.05	9.90	0.67	
	24-Jun-17	III		5.05	0.75	0.34	0.33	1.95	0.90	0.43	0.36	14.74	6.79	6.47	38.39	17.70	8.50	7.08	0.84	
32	24-Jun-17	F1-1-2 (Bio)/I	H0	5.03	1.44	0.32	0.25	0.28	1.11	0.55	1.09	28.31	6.27	4.85	5.50	21.98	10.85	21.40	1.19	1.12
	24-Jun-17	II		5.04	1.12	0.30	0.21	0.32	1.23	0.79	1.07	22.04	5.86	4.18	6.37	24.19	15.51	21.19	0.97	
	24-Jun-17	III		5.04	1.51	0.25	0.24	0.25	0.91	0.66	1.22	29.71	4.93	4.66	4.99	17.93	13.10	24.07	1.21	
33	24-Jun-17	F1-1-1 (Bio)/I	H0	5.05	0.40	0.31	0.28	0.99	1.78	0.90	0.40	7.81	6.16	5.50	19.55	35.02	17.70	7.89	0.54	0.58
	24-Jun-17	II		5.02	0.46	0.23	0.24	0.92	1.80	1.12	0.26	9.00	4.54	4.75	18.13	35.49	22.00	5.21	0.55	
	24-Jun-17	III		5.05	0.60	0.24	0.29	0.95	1.51	0.84	0.62	11.88	4.71	5.62	18.80	29.69	16.61	12.21	0.65	
34	24-Jun-17	F3-1-1 (Bio)/I	H0	5.03	4.47	0.04	0.05	0.13	0.10	0.02	0.23	88.12	0.69	1.07	2.49	1.95	0.34	4.64	3.12	3.12
	24-Jun-17	II		5.05	4.37	0.08	0.07	0.15	0.13	0.03	0.22	86.29	1.54	1.46	2.86	2.54	0.49	4.38	3.07	
	24-Jun-17	III		5.06	4.55	0.06	0.07	0.15	0.13	0.02	0.09	89.76	1.10	1.34	3.04	2.60	0.34	1.68	3.18	
35	24-Jun-17	F3-1-2 (Bio)/I	H0	5.03	1.88	0.23	0.62	2.12	0.54	0.12	0.48	37.11	4.54	12.17	41.72	10.73	2.27	-9.37	1.62	1.19
	24-Jun-17	II		5.05	0.84	0.26	0.73	2.11	0.58	0.24	0.30	16.53	5.07	14.30	41.61	11.44	4.70	5.90	0.93	
	24-Jun-17	III		5.04	0.98	0.22	0.73	1.99	0.61	0.19	0.31	19.37	4.40	14.48	39.18	12.03	3.83	6.10	1.01	
36	24-Jun-17	F1-2-1 (Bio)/I	H0	5.06	0.95	0.20	0.23	0.87	1.50	0.75	0.56	18.66	3.95	4.46	17.22	29.53	14.80	11.13	0.86	0.70
	24-Jun-17	II		5.07	0.49	0.24	0.26	0.83	1.75	0.77	0.72	9.63	4.81	5.09	16.45	34.60	15.27	14.13	0.57	
	24-Jun-17	III		5.04	0.65	0.24	0.24	0.64	1.46	1.03	0.78	12.78	4.75	4.79	12.61	28.84	20.22	15.39	0.66	
37	24-Jun-17	F1-1-3 (Bio)/I	H0	5.05	2.00	0.34	0.22	0.29	0.66	0.54	0.99	39.48	6.73	4.38	5.74	13.10	10.69	19.45	1.57	1.53
	24-Jun-17	II		5.06	1.65	0.39	0.22	0.22	0.69	0.76	1.13	32.47	7.63	4.40	4.38	13.65	15.05	22.25	1.34	
	24-Jun-17	III		5.06	2.19	0.30	0.23	0.30	0.69	0.68	0.68	43.18	5.82	4.60	5.86	13.51	13.45	13.43	1.69	

ANNEX 3. Data of Relative Microbial Respiratory Activities (Data of Chapter 5)

No	FIELDS/ SUBSTRATS	F1-1-1			F1-1-2			F1-1-3			F1-2-1			F1-2-2			F2-1-1			F2-1-2			F3-1-2			F3-1-1		
		I	II	III	III	I	II	I	III	I	II	III	I	II	III	I	II	III	I	II	III	I	II	III	I	II	III
1	Mannose	6.62	4.27	7.45	2.34	1.29	5.57	4.53	1.18	6.08	5.07	4.96	7.06	3.53	5.28	5.13	3.95	4.02	2.78	5.52	0.95	6.02	4.31	1.10
2	Mannitol	2.15	2.17	3.68	4.91	0.47	1.08	4.05	4.98	4.54	1.99	2.16	3.78	2.45	4.90	1.38	3.24	6.27	1.05	1.20	0.12	1.95	1.33	3.02
3	Tréhalose	3.07	2.57	2.39	2.03	4.66	3.83	5.05	4.29	2.72	6.83	3.60	5.10	5.25	3.10	4.33	3.47	0.00	5.47	1.29	2.96	3.53	6.97	3.17
4	Arabinose	2.56	6.20	10.21	5.67	6.77	3.96	5.41	8.46	2.68	4.80	0.82	4.44	3.03	2.83	5.60	6.47	4.73	5.96	0.39	7.01	3.41	3.74	7.35
5	Xylose	5.21	9.49	0.00	6.45	1.67	8.73	2.71	3.22	6.89	4.77	7.96	6.95	9.99	5.03	3.07	5.32	1.57	3.03	4.56	2.75	6.26	6.23	2.72
6	Saccharose	6.80	11.20	5.40	9.35	4.34	2.99	6.70	8.23	10.41	6.99	6.92	10.60	9.21	8.72	7.49	8.16	12.82	2.80	5.54	5.69	5.40	7.77	7.25
7	Galactose	3.80	1.11	4.68	3.07	4.36	4.93	3.52	1.69	0.54	5.02	3.53	4.08	4.06	2.72	4.45	2.58	4.69	6.10	3.28	2.30	4.80	3.15	2.59
8	Myo-INOSITOL	2.10	0.00	4.64	3.21	6.75	2.34	3.12	5.23	2.07	3.13	1.64	4.49	1.87	3.81	6.65	2.97	3.28	3.06	0.38	4.29	3.55	2.88	4.06
9	D(-) Sorbitol	7.01	2.45	0.00	4.32	0.00	6.84	3.12	0.60	5.27	5.65	3.70	5.19	3.00	5.09	0.00	4.06	2.16	3.97	3.08	0.62	5.00	3.91	2.96
10	L(+) Rhamnose	1.54	2.41	0.70	2.64	0.80	0.07	1.84	2.41	3.33	1.41	1.38	3.40	1.38	2.95	1.29	2.85	9.21	0.54	3.26	1.61	1.31	1.30	3.69
11	L(-) Arabitol	1.36	1.53	2.15	0.00	3.08	3.08	1.76	0.56	0.51	2.24	0.29	0.00	1.60	1.53	3.01	0.93	7.59	3.50	0.32	5.36	1.42	1.95	8.03
12	meso-Erythrol	0.31	1.44	4.48	2.99	7.25	3.49	3.78	4.99	2.01	1.74	0.26	0.62	1.72	1.95	6.26	1.65	1.76	4.03	4.12	6.41	4.24	0.80	2.85
13	D(+) Glucose	10.71	9.71	4.48	7.89	1.61	11.40	8.81	3.87	9.72	8.17	13.92	8.32	7.13	8.28	1.22	8.27	1.39	5.17	3.60	3.03	5.29	6.49	4.00
14	Ac citrique	1.14	3.72	1.54	4.36	0.00	0.25	4.84	4.62	5.84	2.81	5.29	4.42	2.70	3.27	4.17	2.60	6.72	4.57	6.44	2.84	2.12	4.86	3.97
15	Ac maléique	6.53	4.36	6.23	0.27	6.51	4.22	2.33	1.67	1.56	2.86	0.69	0.32	4.62	3.19	4.41	3.54	0.27	3.99	1.98	4.12	3.62	2.47	3.68
16	Ac malique	3.69	4.06	11.58	6.46	10.80	4.02	8.27	9.88	7.44	6.39	10.16	5.34	5.77	8.15	7.16	7.41	8.53	4.92	4.45	10.76	5.09	5.97	5.51
17	Na-Gluconate	5.24	6.93	0.00	3.56	0.00	7.88	2.96	0.00	2.48	2.87	2.84	3.57	4.95	3.98	4.86	4.56	1.06	5.34	2.77	0.66	6.03	2.78	1.31
18	N-acétylGlucosamine	0.00	0.83	0.12	3.59	0.00	0.00	1.19	2.90	4.40	0.83	4.06	1.19	0.90	2.30	0.39	1.89	3.05	3.49	4.97	0.97	1.55	2.73	3.06
19	L-glutamiic acid sodium	5.85	7.25	5.28	4.03	7.29	5.09	5.79	4.55	1.73	5.29	7.77	4.93	5.99	4.89	5.62	4.47	0.00	2.43	5.05	3.78	5.20	4.77	3.18
20	L-Asparagine	5.15	3.60	8.19	5.33	7.78	0.10	4.58	8.78	6.15	5.31	1.49	4.18	5.41	4.53	7.22	3.84	6.27	1.11	1.31	4.87	5.74	2.69	5.37
21	D,L-Valine	5.62	1.95	0.00	1.06	0.00	5.02	0.05	0.12	2.50	1.36	3.11	0.23	2.07	0.04	2.65	1.63	0.00	2.38	5.05	1.10	0.83	3.48	1.40
22	L-Méthionine	1.75	1.72	1.14	3.87	0.00	0.55	0.98	1.39	3.51	1.16	2.65	2.40	1.13	1.59	1.06	2.06	0.85	3.51	8.49	1.52	0.87	0.73	3.44
23	L-Glutamine	3.17	2.87	2.45	2.36	4.92	2.06	3.01	1.27	1.89	2.66	5.14	3.03	4.27	2.75	3.34	2.68	1.57	4.76	8.85	1.14	2.98	6.78	2.47
24	L-Alanine	0.55	1.15	5.12	3.40	7.40	3.36	4.49	6.61	0.74	3.12	1.12	2.28	3.51	2.86	3.10	2.14	2.25	4.55	2.44	2.47	3.25	0.56	2.36
25	L-Sérine	2.69	2.80	1.31	3.93	0.00	4.04	2.23	1.90	2.06	2.61	3.83	1.42	2.34	3.74	1.06	5.77	0.00	4.04	5.42	5.76	1.40	3.05	3.28
26	L-Histidine	0.77	2.17	0.45	2.45	2.63	0.59	1.49	3.11	1.99	1.51	0.65	2.47	0.33	0.62	0.84	1.72	3.29	2.84	2.47	3.92	1.55	0.16	3.41
27	L-Proline	1.74	2.04	2.19	0.48	4.56	1.41	1.64	1.75	0.00	1.96	0.06	0.00	0.64	0.95	1.08	1.14	3.94	2.45	1.30	10.99	2.30	6.15	0.00
28	L-Leucine	2.89	0.00	4.11	0.00	5.08	3.11	1.74	1.71	0.96	1.45	0.00	0.18	1.14	0.93	3.17	0.63	2.72	2.15	2.47	2.00	5.27	2.00	4.74

ANNEX 4. Data of Soil Analysis (Data of Chapter 5)

Celesta-lab
Acteur de votre environnement

Field F1-1-1

Numéro Labo 1829-042

CARACTÉRISATION PHYSICO-CHIMIQUE

Fraction Granulomètrique en % de terre	
Argiles	19,1
Limons	54,4
Sables	17,5

Texture
limon argilo-sableux (LAS)

Analyse physique		
Matières organiques	(%)	2,3
Carbone	(g/kg)	13,3
Azote	(g/kg)	1,2
C/N		11,5
pH eau		8,4
pH KCl		7,7
Calcaire total	(g/kg)	68
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	8,66

Analyse chimique				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,160		1,000	1,500			■		
Anhydride Phosphorique	(g/kg)	0,074		0,140	0,200		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,422	10,3	0,300	0,400				■	
Oxyde de Magnésium-MgO	(g/kg)	0,436	25,0	0,120	0,170					■
Oxyde de Calcium-CaO	(g/kg)	12,00	100	1,703	1,947					■
Oxyde de Sodium-NaO	(g/kg)	0,138		0,001	0,100					■
Rapport K2O/MgO		1,00	Souhaitable:	2,40						

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F1-1-2

Numéro Labo 1829-041

Fraction Granulomètrique en % de terre

Argiles	17,5
Limons	52,6
Sables	20,4

Texture

limon sablo-argileux (Lsa)

Analyse physique

Matières organiques	(%)	3,3
Carbone	(g/kg)	18,9
Azote	(g/kg)	1,7
C/N		11,4
pH eau		8,2
pH KCl		7,6
Calcaire total	(g/kg)	62
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	9,92

Analyse chimique

				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,660		1,000	1,500				■	
Anhydride Phosphorique	(g/kg)	0,127		0,140	0,200		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,344	*7,4*	0,350	0,450		■			
Oxyde de Magnésium-MgO	(g/kg)	0,526	*26,4*	0,130	0,180					■
Oxyde de Calcium-CaO	(g/kg)	11,97	*100*	1,951	2,230					■
Oxyde de Sodium-NaO	(g/kg)	0,046		0,001	0,100			■		

Rapport K2O/MgO 0,70 Souhaitable: 2,60

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F1-1-2

Numéro Labo 1829-044

Fraction Granulomètrique en % de terre

Argiles	20,4
Limons	65,5
Sables	5,2

Texture

limon argileux (La)

Analyse physique

Matières organiques	(%)	1,7
Carbone	(g/kg)	10,0
Azote	(g/kg)	1,2
C/N		8,2
pH eau		8,3
pH KCl		7,9
Calcaire total	(g/kg)	72
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	9,26

Analyse chimique

				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,220		1,000	1,500			■		
Anhydride Phosphorique	(g/kg)	0,101		0,140	0,200		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,461	*10,6*	0,300	0,400				■	
Oxyde de Magnésium-MgO	(g/kg)	0,604	*32,5*	0,130	0,180					■
Oxyde de Calcium-CaO	(g/kg)	12,34	*100*	1,821	2,082					■
Oxyde de Sodium-NaO	(g/kg)	0,233		0,001	0,100					■

Rapport K2O/MgO 0,80 Souhaitable: 2,30

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F1-2-1

Numéro Labo 1829-038

Fraction Granulomètrique en % de terre	
Argiles	18,3
Limons	54,5
Sables	18,1

Texture
limon sablo-argileux (Lsa)

Analyse physique		
Matières organiques	(%)	2,5
Carbone	(g/kg)	14,8
Azote	(g/kg)	1,5
C/N		9,8
pH eau		8,2
pH KCl		7,7
Calcaire total	(g/kg)	65
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	9,53

Analyse chimique				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,500		1,000	1,500			■		
Anhydride Phosphorique	(g/kg)	0,302		0,140	0,200					■
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,441	9,8	0,300	0,400				■	
Oxyde de Magnésium-MgO	(g/kg)	0,529	27,6	0,130	0,180					■
Oxyde de Calcium-CaO	(g/kg)	11,42	100	1,875	2,142					■
Oxyde de Sodium-NaO	(g/kg)	0,069		0,001	0,100			■		
Rapport K2O/MgO		0,80	Souhaitable:	2,30						

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F1-2-2

Numéro Labo 1829-036

Fraction Granulomètrique en % de terre	
Argiles	15,0
Limons	36,3
Sables	36,8

Texture
sable argilo-limoneux (Sal)

Analyse physique		
Matières organiques	(%)	2,9
Carbone	(g/kg)	16,9
Azote	(g/kg)	1,6
C/N		10,5
pH eau		8,2
pH KCl		7,9
Calcaire total	(g/kg)	91
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	8,84

Analyse chimique				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,610		1,000	1,500				■	
Anhydride Phosphorique	(g/kg)	0,269		0,120	0,180					■
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,321	*7,7*	0,070	0,100					■
Oxyde de Magnésium-MgO	(g/kg)	0,381	*21,4*	0,120	0,170					■
Oxyde de Calcium-CaO	(g/kg)	11,12	*100*	1,739	1,987					■
Oxyde de Sodium-NaO	(g/kg)	0,042		0,001	0,100			■		

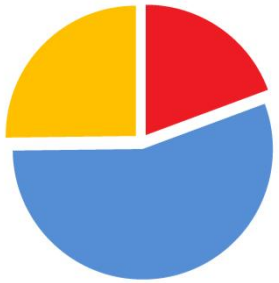
Rapport K2O/MgO 0,80 Souhaitable: 0,60

Numéro Labo 1829-040

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F2-1-1

Fraction Granulomètrique en % de terre	
Argiles	17,7
Limons	50,6
Sables	23,0



Texture
limon sablo-argileux (Lsa)

Analyse physique		
Matières organiques	(%)	2,3
Carbone	(g/kg)	13,7
Azote	(g/kg)	1,3
C/N		10,4
pH eau		8,3
pH KCl		7,8
Calcaire total	(g/kg)	64
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	9,05

Analyse chimique				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,310		1,000	1,500			■		
Anhydride Phosphorique	(g/kg)	0,126		0,140	0,200		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,290	6,8	0,350	0,450		■			
Oxyde de Magnésium-MgO	(g/kg)	0,405	22,3	0,130	0,180					■
Oxyde de Calcium-CaO	(g/kg)	10,92	100	1,780	2,034					■
Oxyde de Sodium-NaO	(g/kg)	0,034		0,001	0,100			■		
Rapport K2O/MgO		0,70	Souhaitable:	2,60						

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F2-1-2

Numéro Labo 1829-039

Fraction Granulomètrique en % de terre

Argiles		20,8
Limons		63,5
Sables		6,5

Texture

limon argileux (La)

Analyse physique

Matières organiques	(%)	1,8
Carbone	(g/kg)	10,6
Azote	(g/kg)	1,3
C/N		8,3
pH eau		8,6
pH KCl		8,2
Calcaire total	(g/kg)	74
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	9,61

Analyse chimique

				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	1,280		1,000	1,500			■		
Anhydride Phosphorique	(g/kg)	0,041		0,140	0,200	■				
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,130	2,9	0,300	0,400	■				
Oxyde de Magnésium-MgO	(g/kg)	0,830	43,0	0,130	0,180					■
Oxyde de Calcium-CaO	(g/kg)	11,89	100	1,890	2,160					■
Oxyde de Sodium-NaO	(g/kg)	0,560		0,001	0,100					■

Rapport K2O/MgO 0,20 Souhaitable: 2,30

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F3-1-1

Numéro Labo 1829-037

Fraction Granulomètrique en % de terre

Argiles		22,4
Limons		56,7
Sables		9,7

Texture

limon argilo-sableux (LAS)

Analyse physique

Matières organiques	(%)	4,1
Carbone	(g/kg)	23,8
Azote	(g/kg)	2,3
C/N		10,5
pH eau		8,5
pH KCl		8,2
Calcaire total	(g/kg)	72
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	12,08

Analyse chimique

				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	2,270		1,000	1,500					■
Anhydride Phosphorique	(g/kg)	0,097		0,140	0,200		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,601	10,6	0,300	0,400					■
Oxyde de Magnésium-MgO	(g/kg)	1,844	75,9	0,150	0,210					■
Oxyde de Calcium-CaO	(g/kg)	9,13	100	2,376	2,716					■
Oxyde de Sodium-NaO	(g/kg)	6,224		0,001	0,100					■

Rapport K2O/MgO 0,30 Souhaitable: 2,00

CARACTÉRISATION PHYSICO-CHIMIQUE

Field F3-1-1

Numéro Labo 1829-043

Fraction Granulomètrique en % de terre

Argiles	8,5
Limons	18,4
Sables	70,2

Texture
sableux (S)

Analyse physique

Matières organiques	(%)	1,7
Carbone	(g/kg)	10,1
Azote	(g/kg)	1,0
C/N		10,4
pH eau		9,0
pH KCl		8,1
Calcaire total	(g/kg)	12
Calcaire actif	(g/kg)	ND
CEC	(Cmol+/kg)	5,67

Analyse chimique

				Optimum		Très faible	Faible	Satisfaisant	Elevé	Très élevé
Azote total (Kjeldhal)	(g/kg)	0,970		1,000	1,500		■			
Anhydride Phosphorique	(g/kg)	0,049		0,060	0,160		■			
Bases échangeables		(g/kg)	(%CEC)							
Oxyde de Potassium-K2O	(g/kg)	0,391	*14,6*	0,070	0,100					■
Oxyde de Magnésium-MgO	(g/kg)	0,805	*70,6*	0,100	0,140					■
Oxyde de Calcium-CaO	(g/kg)	3,76	*100*	1,115	1,275					■
Oxyde de Sodium-NaO	(g/kg)	1,690		0,001	0,100					■

Rapport K2O/MgO 0,50 Souhaitable: 0,70