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Changes in soil salinity in the habitats of five halophytes after 20 years

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ABSTRACT Long-term soil salinity data are required to properly manage saline habitats. This study presents easy procedures for comparing soil salinity profiles, which were used to assess four saline wetlands in NE Spain. Soil salinity measurements at sites where different halophytes were dominant were recorded throughout the year in 1979-1980 and 1999-2000, with additional measurements in 1985. Electrical conductivity of saturation extracts (ECe) and the concentrations of major ions were measured in four hundred soil samples. To confirm the consistency of the quality of the lab measurements, we used saturation percentage (SP) statistics and the relationships between ECe and both the cationic and the anionic concentrations in the extracts ($R^2 > 97\%$), and the relationship with the EC of the extracts at a soil-to-water ratio of 1:5 by weight (EC 1:5). The ECe of the soil samples ranged from 4 to 114 dS m^{-1}, and the averages in the four wetlands (AN, PC, AL, and AG) were 38.6, 10.1, 22.3, and 68.5 dS m^{-1}, respectively. The median of the sodium adsorption ratio [SAR, (meq $L^{-1})^{0.5}$] in the samples was 69, 126, 64, and 28, respectively. At AN, the maximum average ECe in the upper 100 cm of the soil was 35.0 dS m^{-1} for *Artemisia herba-alba* and 75.2 dS m^{-1} for *Suaeda vera*; at PC, the maximum value was 19.3 dS m^{-1} for *Puccinellia festuciformis*; and, at AL, it was 30.3 dS m^{-1} for *Salicornia ramosissima* and 44.6 dS m^{-1} for *S. vera*. At AG, the maximum average ECe was 69.2 for *S. vera*, and 85.2 dS m^{-1} for *Arthrocnemum macrostachyum*. The ECe and SAR remained stable between the two sampling periods in three of the wetlands but, in AL, which was converted into a paddy, ECe was reduced from ~ 40 dS m^{-1} to ~ 2 dS m^{-1} and SAR was reduced from > 77 to ~2. The presence of saline non-sodic and saline-sodic soils at nearby wetlands reflects the pedodiversity of the area.

Keywords: *Artemisia herba-alba, Arthrocnemum macrostachyum, Puccinellia festuciformis, Salicornia ramosissima, Suaeda vera*.

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1. Introduction

Saline soils often occur in closed depressions and other poorly drained areas in arid environments, which can lead to the development of saline wetlands. Historically, peasants disdained those areas because they had little or no agricultural value; however, today, biodiversity conservation and other environmental issues have increased the concern for the protection of wetlands (Ramsar Convention Secretariat, 2010), even though the degradation and destruction of saline wetlands are common. In the Central Ebro Valley in northeastern Spain, there are many athalassohaline wetlands, where degradation and destruction had been a long-standing problem (Domínguez et al., 2013a). The composition and distribution of halophytic vegetation in these sensitive areas have been documented by botanists (Braun-Blanquet and Bolós, 1958; Conesa et al., 2011), but much less is known about the soils of these saline wetlands. In the field, soil salinity is often appraised based on the presence of halophytes; however, this approach is problematic because the responses of plants to salinity are complex (see reviews by Flowers and Colmer, 2008; Chávez and González, 2009) due to the interactions between stress factors and the many physiological mechanisms that provide salt tolerance (Barrett-Lennard, 2003; Pessarakli, 2010; Nazar et al., 2011), and seasonal changes in soil salt concentrations. Typically, the importance of those factors has been evaluated in controlled experiments, but quantitative studies of soil salinity in the natural habitats of halophytes are less common.

The redistribution of the salts in geological materials and shallow saline water tables contributes to the formation of the saline soils in wetlands of arid regions (e.g., Kezao and Bowler, 1986; McEwan et al., 2006). Artificial inflows can disturb natural hydrological regimes, which often leads to the replacement of halophytes and affiliated animals and microbes by a “less halophilous” vegetation and affiliated species, which has occurred in some of the wetlands in the Ebro Basin, Spain (Guerrero et al., 1991; Camacho and Wit, 2003). Maintaining the salinity and other natural conditions of soils, including intermittent waterlogging, is essential for the persistence of saline habitats; however, there are limited data on the changes in soil salinity over time (see review by Herrero and Pérez-Coveta, 2005).

Typically salt tolerance by plants is a concept used in agriculture, and is studied using experiments in dishes, pots, or in other controlled environments; however, in natural habitats, where soil salinity can vary seasonally amid a complex set of factors, salt tolerance can be an imprecise concept, particularly if soil salinity measurements are based on samples collected on a single date. Soil salinity often dictates whether one plant species or another can become established and persist. Rural dwellers and botanists alike associate the presence of specific halophytes with the degree of soil salinity, but there are limited quantitative data on the soil salinity of the habitats (examples: Vasek and Lund, 1980; Tòth et al., 1995; Blank et al., 1998; Piernik, 2005; Barrett, 2006) and the germination strategies (Gul et al., 2013) of halophytes. The relationship between soil salinity and the occurrence of inland halophytes is still at the 'rule of thumb' stage (Bennett et al., 2009).

Most of the studies on the soil salinity of wetlands have been conducted on or near marine coastlines. Some examples in Spain are Murillo et al. (1979), De la Rosa et al. (1980), Bescansa and Roquero (1990), Ortiz et al. (1995), Gómez et al. (2005), and Álvarez et al. (2006). Studies on inland saline wetlands in Spain are less common, e.g., Porta et al. (1980), Gumuzzio et al. (1981), Badía and Alcañiz (1994), and Domínguez et al. (2013b). Those types of studies provide valuable information about soil salinity, but few have examined the temporal variations in the salinity of wetland soils.

The wide variation in the procedures used to study soil salinity has made it difficult to compare soil salinity data between studies and for assessing long-term (e.g., decades) changes. The repeatability of soil sampling and verification of the consistency of analytical procedures are essential, especially if long-term sampling involves different technicians and labs. Straightforward procedures for presenting salinity data will benefit the various disciplines that study the environment. Comparable and standardized data would facilitate the long-term monitoring of saline wetlands. The approach in our research is naturalistic or observational, i.e., without controlled experiments, but it is similar to the so-called long-term soil experiments (LTSEs). According to Richter et al. (2007), 'LTSEs that are simple and efficient in design may be more likely to survive compared with complex experiments that require much labor and expense to maintain.' In that way, the study presented here is an extension of previous studies by Herrero (1982, 2008) on athalassohaline wetlands in northeastern Spain.

The objectives of this study were (i) to demonstrate a procedure for describing soil salinity that facilitates comparisons between sites and over time, (ii) to compare the soil salinity of four saline wetlands in NE Spain and the salinity where the dominant plants live, and (iii) to compare the salinity of these soils after 20 years.

2. Materials and methods

2.1. Study area

The four saline wetlands studied were in the Central Ebro Basin on the north side of the Ebro River, Spain (Figure 1). The wetlands (from north to south) are Paúl de Anzano (AN), Plan de Callén (PC), Paúl de Almuniente (AL), and Saladar de Agustín (AG). Table 1 shows their coordinates and surface area size. The elevation declines from north (500 m a.s.l.) to south (328 m a.s.l.). The underlying material is saliferous Miocene horizontal strata of lacustrine origin. Alternating sandstone and lutites predominate in AN, PC, and AL, and gyprock is predominant in AG.

Figure 1

Table 1

The climate is Mediterranean and, at the study sites, aridity was lowest in the north (AN) and highest in the south (AG). Mean annual temperature ranged from 13.7 ºC to 14.4 ºC and annual rainfall ranged from 550 mm to 388 mm. A water deficit occurred from February to November and, based on the records from the weather stations closest to each wetland (< 13 km way), mean annual deficits were 536 mm (AN), 600 mm (PC and AL), and 653 mm (AG). The soils of the wetlands were characterized by the persistence of shallow groundwater (up to 41 cm) and saturated or wet soils, depending on rainfall (AN and AG), or rainfall and irrigation (PC and AL).

Based on the criteria of the Soil Survey Staff (2010), the soils were Typic Aquisalid (AN, AL), Oxyaquic Xerofluvent (PC), and Gypsic Aquisalid (AG). Land use differed among the four wetlands. AN has been grazed by sheep since antiquity. The PC and AL depressions were leveled and converted to plots for irrigation in the 1950s; the PC plot which had impervious bare soil, remained uncultivated during the years of our study; the AL plot was uncultivated in 1979 but, since it was re-leveled and reshaped in 1981 it has been used as a paddy. AG has never been cultivated and is rarely grazed by sheep.

Artemisia herba-alba Asso, *Arthrocnemum macrostachyum* (Moric.) Moris in Moris & Delponte, *Puccinellia festuciformis* (Host) Parl., *Salicornia ramosissima* Woods, and *Suaeda vera* Forssk. ex J.F. Gmel were the dominant plants at the sampling sites. For inventories of the vegetation see Herrero (1982) and for a description of the halophilous vegetation at AG see Conesa et al. (2011).

2.2. Sampling

The types of vegetation present dictated the selection of sampling sites within each wetland. Overall, 404 soil samples were collected from 52 auger holes and four pits (mean sampling depth = 105 cm) at 12 sampling sites at different dates (Table 1). The objective for each augering was to reach a depth of 2 m, and the augerings that were repeated at each site were within 2 m of the remains of the initial pit or auger hole.

At AN (mean = 151 cm), PC (mean = 170 cm), and AL (mean = 119 cm), all but two of the holes were deeper than 1 m and, at AG, the maximum depth was 88 cm (mean = 63 cm). The variation in the depths was due to impenetrable compacted dry layers or to supersaturated materials, which often had their clay minerals in a dispersed state at PC and AL, and which were not retained by the auger. All of the augerings at AG encountered impenetrable material within 85 cm of the surface. The depths of the auger samplings reflected those characteristics of the soil profiles (Table 1). At several sites, we sampled phreatic, irrigation, or drainage waters (Table 2).

Table 2

2.3. Soil analyses

The 404 soil samples were air-dried before they were passed through a 2-mm sieving mill. All of the samples contained little or no coarse fragments. All of the 4415 analytical determinations were performed using the fine earth. We prepared the saturated paste (United States Salinity Laboratory Staff, 1954) and the saturation percentage (SP), i.e., the gravimetric water content of the paste (Rhoades et al., 1999, page xiv), was recorded. The first author prepared 243 pastes from the samples collected in 1979, 1980, and 1985, and another person prepared 161 pastes from the samples collected in 1999 and 2000. The equipment and procedures were similar in all determinations; furthermore, in 1999 and 2000, the technician adjusted the criteria for the end-point of the pastes.

ECe was measured in the extracts of the saturated paste. Major ions were titrated in the extracts, except potassium, which was assumed to be minimal in non-fertilized, highly saline soils and was confirmed by the saturation extracts of the 63 soil samples in which K^+ was measured in Spanish saline soils (Ayers et al., 1960; Bech et al., 1988; Badía and Alcañiz, 1994; Álvarez-Rogel et al., 2001; Ortiz et al., 2009; Domínguez et al., 2013b). In those samples, the highest potassium ratio (PR) to the sum of the other major ions ($PR = 100 \times K^+ / (Na^+ + Ca^{2+} + Mg^{2+})$) was 10.9%, only six of the samples had a PR > 5% and the mean was 2.36%. In our analyses of 359 saturation extracts from the saline soils of Monegros, the mean PR was 2.17%, the highest was 5.96%, and only 7 samples had a PR > 4%. The sulfate concentrations in the samples from 1979 were estimated as the difference between major ions (Bower and Wilcox, 1965; Skarie et al., 1987).

The EC of the extracts at a soil-to-water ratio of 1:5 by weight (EC 1:5) was measured in 250 soil samples. The pH was measured in 230 soil samples at a soil-to-water ratio of 1:2.5 (pH1:2.5); 166 samples of the saturation extracts (pHe) and 74 samples of the saturated paste (pHp). The calcium carbonate equivalent was measured in 155 samples using a Bernard calcimeter and gypsum was titrated in 56 samples using thermogravimetry (Artieda et al., 2006). The official Spanish methods for soil and water analyses (MAPA, 1994) were used as a reference.

To the extent possible, the data were evaluated using resistant statistics of the exploratory data analysis (Tukey, 1977; Chambers et al., 1983). Data displayed in boxplots are presented with confidence intervals about the median (Hettmansperger and Sheather, 1986). The normality of the data distributions was evaluated using the Kolmogorov-Smirnov Test (Steel and Torrie, 1980). Regression lines were calculated using the Least Squares Method.

2.4. Validation of the analytical results

The reliability and comparability of the data obtained by the labs over a 20-year period were assessed using the untransformed results of the analyses of the soil samples. Some studies of soil salinity studies overlook or do not report saturation percentage (SP). SP is an almost invariant property of soils that is primarily influenced by the distribution of particle sizes, and it was used to verify the overall correspondence of the repeated samplings and the stability of the end-point criteria for the preparation of the saturated paste and the water extraction procedure.

The ECe-EC 1:5 relationships were used to verify the consistency of the EC data. Furthermore, the validity of those two measures of salinity was assessed in combination with SP based on the relationships between soil extracts at various dilutions (Abrisqueta et al., 1962; Carpena et al., 1968). Shaw (1994) expressed those relationships with a power q ($1 \geq q \geq 0$) of the quotient of the dilutions in both extracts. That author incorporated the water content of the air-dried sample used for the preparation of extracts, which provided the basis for the equation described by Sumner et al. (1998, page 7):

$$ECe = EC1{:}5\ [(500 + 6ADMC)/SP]^q$$

where

ECe and EC1:5 are the electrical conductivities of saturated paste and 1:5 extracts, respectively, in dS m^{-1};

ADMC is the air dry moisture content;

SP is the saturation percentage;

q is a coefficient, $1 \geq q \geq 0$

We use that equation and ignored the moisture content of the air-dried samples because of the presence of gypsum and or more soluble salts that contraindicate the heating of the samples (Herrero et al., 2009).

The electroneutrality of the extracts was quantified in the samples in which Ca^{2+}, Mg^{2+}, Na^+, CO_3^{2-}, HCO_3^-, Cl^-, and SO_4^{2-} concentrations were measured. To that end, we calculated the percent deviation (PD) between the anions and cations, above, as PD = 100 × (anions - cations) / [(anions + cations) / 2]. We excluded from the checking those samples where sulfate was estimated by difference.

2. 5. Composition of the soil solutions

To compare the soils of the four wetlands over time, we used the samples from the 52 auger holes only (Table 1) because each of the sampling dates and sites was equally represented. For each auger hole, we used a single synthetic value of ECe and the ionic content of the saturation extracts. Synthetic values were calculated as the mean of ECe or ionic contents in each soil sample weighted by its depth interval (Herrero and Pérez-Coveta, 2005; http://digital.csic.es/handle/10261/60892) up to 100 cm or up to the maximum depth in the augerings that were shallower than 100 cm. The single synthetic values of ionic concentrations in the soil solutions are displayed in the Schoeller-Berkaloff diagrams, and the clustering of similarly shaped lines in a diagram indicated a wetland soil signature.

For the same samples, the median sodium adsorption ratio (SAR) was calculated to produce single synthetic values of SAR. Weighted means for the non-additive magnitudes SP, pH, or SAR were not computed. The average values of ECe and SAR expressed by the single synthetic values of each auger hole were used in interpreting soil salinity in relation to the dominant plant species.

3. Results and discussion

3.1. The consistency of the soil analyses

3.1.1. Saturation percentage

The saturation percentage (SP) in the soil samples ranged from 20% to 82%. The saturation percentages deserve close examination because SP influences all of the subsequent analyses of the saturated paste extracts. Differences between years might have arisen from differences in the intensity of grinding, the shape of the container in which the paste was prepared, the quantity of the paste, the rate at which water or earth was added, or the technician's judgement about the saturation end-point. With the exception of the potential subjectivity of the technicians, few published studies address those factors; however, they might be important for long-term soil monitoring and research projects.

Assessing possible operator bias using direct, pairwise comparisons between sampling periods at specific sampling sites and depths seems unwise given the lateral textural variations even over the short distances between the initial and subsequent auger holes. Therefore, the SP distributions obtained by the two operators were compared using only the data from the sampling sites to which both operators contributed. Furthermore, the comparison was restricted to samples that were from the same site, and were collected from similar depths.

Thus, the comparison was based on 150 soil samples from 1979 and 1980 (SP mean = 45.4%; standard deviation = 12.4%; median = 46.5%; range = 20-74%), and 113 samples from 1999 and 2000 (SP mean = 41.1%; standard deviation = 13.5%; median =41.0%; range = 22-82%). The approximately 5% difference in the measures of central tendency between the two sampling periods is acceptable given the degree of experimental error and the intrinsic lateral variability of the soils. In addition, the similar range of both groups suggests that there was no bias associated with the two operators.

3.1.2. ECe and EC 1:5

The validity of the ECe measurements was tested based on the relationships between ECe and EC1:5 in the two sampling periods. The EC1:5 was measured in 89 samples collected in 1979, 1980, or 1985, and in 161 samples collected in 1999 or 2000. The linear regressions for the two data sets and for the 250 samples, collectively, were very similar (Table 3). The standard error was high because the dispersion of EC1:5 among the samples that had ECe > 15 dS m^{-1} was high, which is a common phenomenon that can be attributed to (i) soluble minerals in concentrations that allow the saturation of the concentrated, but not the diluted extract, (ii) the solubility of the various salts that are affected by common ions, and (iii) the formation of neutral ionic pairs. EC1:5 is a very easy procedure that is not influenced by operator subjectivity; therefore, the stability of the relationship between ECe and EC1:5 provides a sound basis for concluding that the ECe determinations in the different sampling periods were unbiased.

Table 3

The two last equations in Table 3 are for the regression of ECe on EC1:5 transformed using the quotient of the extract's dilution first, and then by this quotient raised to the exponent 0.6 (following Shaw, 1994). After testing the values of q from 0 to 1 in 0.1 increments, the 0.6 was chosen because this value of the exponent produced the strongest correlation with ECe (R^2 = 95.5%). The consistent and similar relationships between ECe, SP, and EC1:5 in the analyses performed by the two lab operators (Table 3) reinforce the overall robustness of the analyses and the absence of bias in the lab results from the two sampling periods.

Table 3

3.1.3. Electroneutrality

The median PD of the major ions in the soil samples from 1979, 1980, and 1985 (-5.93, n = 88) indicated an acceptable excess of cations vs. anions, but the median PD in the samples collected in 1999 and 2000 (21.03, n = 159) indicated appreciably more anions than cations. Furthermore, the range of PD among the samples collected earlier (-19.29 to 9.64) was much smaller than it was among the samples collected in the later period (-24.43 to 9.64), which had more outliers than did the earlier period. The deviations from electroneutrality are relevant for the samples from the later sampling period (Figure 2), which had higher concentrations of anions than cations, and exhibited greater dispersion than did the samples from the first period. Only the PD data from the first sampling period followed the Gaussian distribution (based on a Kolmogorov-Smirnov test), which suggests that the causes of the deviations differed between the two sampling periods. In all of the samples, sulfate was the last ion titrated. In the first sampling period, fresh extracts were analyzed in small batches; however, in the second sampling period, technical difficulties in the lab forced us to store the extracts for weeks before the sulfate titration, which affected the extracts and resulted in titrations that yielded distorted concentrations of sulfate. The sulfate estimated by difference (*sulf. diff.*) in the second sampling period (1999-2000) was regressed on the sulfate titrated after storage (*sulf. t.a.s.*), which was *sulf. diff.* ≈ 0.67 × *sulf. t.a.s* (R^2 = 95.2%). That result confirmed the high quality of the ion measurements.

Figure 2

3.1.4. The amounts of the major ions and the electrical conductivity

In the soil samples, cation concentrations ranged from 6.3 to 2479.4 meq L^{-1} and anion concentrations ranged from 6.7 to 3272.6 meq L^{-1}. The horn-shaped silhouette in the scatter diagrams of cations and anions vs. ECe (Figure 3) reflects the positive correlation between the ion/ECe ratio and ion concentrations, which indicates the presence of neutral ionic pairs. Differences in the distribution of lab errors associated with the dilutions used for the high concentrations, which are required by the titration procedure, might have caused the dispersion in the high concentrations.

Figure 3

The quotient of 10 (Bower and Wilcox, 1965, page 939) was applicable to the low concentrations in the scatter diagrams, only. The increase in the ion/ECe ratios, the dispersion, and the maximum concentrations were higher in the samples collected in 1999-2000 than they were in the samples from 1979-1985 because of the delay in the sulfate titrations of the extracts from the 1999-2000 samples. Nevertheless, the analytical results were consistent, even when the two sampling periods were assessed separately (Figure 3). The regressions of the decimal logarithms of the cation and anion concentrations against the logarithms of ECe (Marion and Babcock, 1976) were calculated for all of the samples, collectively, and for the two sampling periods, individually (Table 4). The coefficients of determination of the six regressions ranged from 97.6% to 98.5% and, in all cases, the intercept and slope differed significantly from 0. These coincidences provided additional evidence of the consistency of the lab analyses.

Table 4

The correlation coefficients of the regressions between ECe and the most abundant ions (Table 5) were high and similar to those found in soil samples from agricultural fields near PC and AL (Herrero and Pérez-Coveta, 2005). Furthermore, the differences in these coefficients between the two sampling periods seem acceptable given that the samples from each sampling period were analyzed in different labs. Sulfate was the only ion that had a correlation coefficient with ECe that was smaller in the second period than it was in the first period, which was caused by the storage of the extracts before the sulfate titration. As in many other inland saline environments, in the saline wetlands in our study, chloride was abundant and was the ion that had the strongest correlation with electrical conductivity (r = 0.951 and 0.987 in the first and second sampling periods, respectively). The strength of those correlations and others (Table 5) suggests that the EC measurements can be surrogates for ionic concentrations, although the veracity of the surrogation should be verified and defined for specific soils by calculating the specific regression equations from a subset of soil samples.

Table 5

3.2. Comparisons between the soil solutions from each wetland

The measurements of **pH** at various soil-to-water ratios (pH1:2.5, pHe, pHp) used different samples; yet, the results were consistent (Figure 4), and the pHp values were between pH1:2.5 and pHe values in the four wetlands. The median pH1:2.5 (10.13), pHp (9.42), and pHe (9.70) in PC were much higher than they were in the other three wetlands, where median pH1:2.5 ranged from 8.59 to 9.26, median pHp ranged from 8.40 to 8.99, and median pHe ranged from 7.79 to 8.16. The range of pH1:2.5 in the soils of PC and AL overlapped, slightly, all of the pHp values in AL soils were within the range of values in AN and PC soils, and the median pHp in the AN and AL samples was the same (8.99). In addition, most of the pHp values in the AG soils were within the range of values in the AN soils, but the median pHp in the AG soils (8.40) was lower than the value in the AN soils (8.99). The pHe of the soil samples from PC (lowest = 9.27) did not overlap the values in the other three wetlands (highest = 8.74, which was an outlier from AL). The substantial differences between the pH1:2.5, pHp, and pHe from PC and the values in the other wetlands were consistent with the presence of CO_3^{2-}, which occurred in 30 of 32 soil samples from PC (mean = 12.5 meq L^{-1}), but was negligible in the samples from AN and AL, and, in AG, only occurred in three surficial samples from a single site AG5 (< 2 meq L^{-1}).

Figure 4

In 108 samples (26.7% of the 404 samples analyzed), the **saturation percentage** (**SP**) was ≤ 33. All but 11 of those samples were from AG and PC. All but four of the 19 samples (4.7% of the total) that had a SP > 69 came from AG; the remainder was from a single pit (AL0). A distinct bimodal distribution in SP was apparent in the AG wetland, only (Figure 5). The mode at ~ 25 SP was associated with the high gypsum contents (Figure 6), and the mode at ~ 65 SP was attributed to shallow horizons that had organic matter content > 1.5%.

Figure 5

Figure 6

Among the four wetlands, the distributions of SP were most similar in the soils samples of AN and AL (Figure 7). The range of SP in the samples from AG was much broader than it was in the other wetlands; however, the bimodality in the AG soils (Figure 5) reduces the applicability of the median in the comparison with the median SP in the other wetlands. That bimodality is related to the contrast in the composition of the soil between dark, OM-rich, shallow horizons and the white or pinkish, deeper, gypsum-rich horizons, which is reflected in the correlation between SP and percent gypsum content (R^2 = 81.4%, n = 55 samples). The correlation between SP and the sum of gypsum and calcium carbonate equivalent (R^2 = 77.9%), reflects the importance of gypsum content on SP.

Figure 7

The boxplots of ECe provide an easy means of comparing the **soil salinity** at the four wetlands (Figure 7) and show that the median and mean ECe values of the wetlands were well differentiated. The soils in PC had the lowest average ECe, and all but one of the samples were above the standard threshold for saline soils (ECe > 4 dS m^{-1}). Furthermore, the range of ECe in the soil samples from PC (24.3 dS m^{-1}) was much smaller than the range of values in the other three wetlands. The soil samples at AG had the highest average ECe (range = 102.3 dS m^{-1}), and only a single outlier sample from AN had an ECe that surpassed the maximum ECe at AG. The range of ECe values was broadest in the soils at AN (110.8 dS m^{-1}). The distributions of ECe values were most similar in the soils at AN and AL (range = 99.2 dS m^{-1}), although the average values at AL were lower than they were at AN. The average ECe provides an indication of the soil salinity at the four wetlands. AG had the highest mean ECe (68.5 dS m^{-1}) and the means were much lower at AN (38.6 dS m^{-1}), AL (22.6 dS m^{-1}), and PC (10.1 dS m^{-1}). The more or less marked bimodality in the distributions of ECe in the soil samples from the four wetlands (Herrero, 2008, Figure 22) is an important feature. The sampling strategy, which was based on the differences in vegetation, contributed to the bimodality, but an assessment of the samples based on sampling period indicated temporal changes in ECe (see section 3.5).

The relationships between ECe and EC1:5 were distinctly different at each of the wetlands (Figure 8). An almost linear relationship was apparent at AN, PC, and AL. The correlations between ECe and EC1:5 were much more linear when the analyses were based on the transformation of EC1:5 by the dilution ratio (Table 6). For each wetland, the regression equations of ECe on EC1:5 were calculated in three ways: (i) without transformation of EC1:5 by the dilution ratio, i.e, q = 0; (ii) transformed by the ratio, i.e., q = 1; and (iii) transformed using the values of q that yielded the best linear adjustment after examining the values of q from 0 to 1 in 0.1 increments. Those transformations improved R^2 and reduced the standard errors although the effects were small in the soil samples from AG, which probably was due to the predominance of gypsum in many of the soil samples and the occurrence of neutral ionic pairs in some of the extracts from AG. The equations that have q = 0 (i.e., do not include SP) can be used (Table 6) to estimate ECe based on EC1:5, except for AG, for which acceptable R^2 values were achieved only when SP was included in the equation. An evaluation of the parameters in those types of equations within the context of the objectives of a study can help in deciding whether a specific equation can be useful in the assessment of ECe in a particular soil.

Figure 8

Table 6

SAR is a measure of **sodicity**, which is an important aspect of salt-affected soils. The four wetlands in this study differed in the SAR of their soils (Figure 9). The median SAR was 69 at AN, 126 at PC, 64 at AL, and 28 at AG, which were much higher than the conventional threshold (SAR = 13) for sodic soils. The value of SAR for predicting hydraulic properties diminishes for soils that have low clay content such as the gypseous soils at AG, where the mean calcium carbonate equivalent plus gypsum was 69% by weight (n = 53 samples). Compared to the other three wetlands, the SAR of the soil samples from AG was low and within a narrow range because of (i) Ca^{2+} saturation caused by the abundance of gypsum, and (ii) the high Mg^{2+} concentration (mean = 734.0 meq L^{-1} vs. 48.0 meq L^{-1} at AN, 0.4 meq L^{-1} at PC, and 20.0 meq L^{-1} at AL). The high SAR of the soil at PC (Figure 9) is consistent with the low or negligible Mg^{2+} concentrations and the imperviousness of the soil, where the SAR in the upper 20 cm (130) was associated with persistent ponding, even after light rain.

Figure 9

At PC, sodicity was particularly high and ECe low; conversely, at AG sodicity was particularly low and ECe high (Figure 10). At AN and AL, the values were intermediate. Most of the soil samples were saline-sodic (ECe > 4 dS m^{-1} and SAR > 13). Only three of the 128 soil samples from AN, one of the 32 samples from PC, and none of the 132 samples from AG had an ECe < 4 dS m^{-1}. Most of the 38 samples that had an ECe < 4 dS m^{-1} were among the

111 samples collected from AL in 1999 and 2000. Nine of the samples from AN, none of the samples from PC, and three of the samples from AG had a SAR < 13. Forty-two of the samples from AL had a SAR < 13, and 39 of these were collected in 1999 and 2000, which reflected the substantial reduction in SAR between 1979-80 and 1999-2000. In addition, the absence of non-saline sodic soil samples is consistent with what is known about the soils of the Central Ebro Valley (Nogués, 2002; Herrero and Pérez-Coveta, 2005; Nogués et al., 2006; Playán et al., 2008; Domínguez et al., 2013b).

Figure 10

Almost all of the samples from AN, AL, and PC had Na^+ / Cl^- ratios > 1 but, at AG, the ratios were < 1 and, with few exceptions, very close to 1 (Figure 11). Despite the SO_4^{2-} saturation caused by the abundance of gypsum, the low ratio in AG appears to have been due to the high Mg^{2+} concentrations (Figure 12).

Figure 11

Figure 12

3.3. ECe and SAR in soil samples from different years

The scatterplots of ECe and SAR at AN and PC suggest a moderate reduction in ECe and a slight decrease in the SAR between the two sampling periods. The high SAR at PC was influenced by the composition of the geological materials, the absence of gypsum in the soil, and the ionic composition of the irrigation water (Table 2). The past basin and border irrigation with water of low salinity led to high sodicity and pH in the PC soils. PC and AL were irrigated with water from the same canal, but the presence of gypsum in the soil at AL prevented the dispersion of clays, which permitted leaching without sodication, which is reflected by the intermediate ECe and SAR values in 1985and the non-irrigated plot in 1979 (Figure 10).

Temporal stability in soil salinity was a noticeable feature at AG. This salinity is related to the salinity of the phreatic (Table 2), which had about twice the electrical conductivity (EC) of seawater. The Table shows slight difference in the EC of the phreatic at AG5 and AG9, both sites with dead *A. macrostachyum,* against the EC at AG8, a site with *S. vera*. Our study provides a baseline for environmental control and management in this saline wetland that is threatened by the projected irrigation of the conterminous lands.

3.4. Ionic signature of soils

The soils at AN, PC, and AG each had one ionic signature and there were two at AL (Figure 12). Given the strong correlation between ionic concentrations and ECe (Figure 3), those ionic signatures are reflected in the clusters at the scatter diagram of SAR vs. ECe (Figure 13).

At AN, the difference between *S. vera* (AN3) and *A. herba-alba* (AN4) within the ionic signature (Figure 12), which was one order of magnitude for some ions, was consistent with the ECe at AN3 (57.8 dS m^{-1}) and AN4 (19.4 dS m^{-1}). The range of ionic concentrations at AN3 was broader than it was at AN4, which reflected the differences in the vertical movement of salts between the sampling dates because of the combined effects of flooding, variations in water table level, and evaporation. The effects were smaller in AN4 because this site was on the upper level of a 40-65 cm escarpment, where fluctuations in the water table had less of an influence. Among the four wetlands, AN3 had the highest Na^+ concentrations and the Cl^- and Mg^{2+} concentrations were exceeded only by the soils at AG.

At PC, the low concentrations of Ca^{2+} and Mg^{2+} and the high concentrations of CO_3^{2-} plus HCO_3^- produced a very distinct signature, and the composition differed little between sampling periods and sites (Figure 12). That signature was consistent with the soil sodicity (Figure 9) and high pH (Figure 4) at PC, where the SAR was highest. At PC, the differences in ECe and SAR between bare soils and vegetated sites did not appear to have influenced the development of clumps of *P. festuciformis* given their changing and erratic location along 15 years of observation.

The soils of the AL wetland had two ionic signatures: one represented the first sampling period (1979 and 1980), and the other represented the second period (1999 and 2000), after the saline wetland was converted into a paddy. After the conversion, the ionic concentrations of Na^+ and Cl^- were reduced by up to three orders of magnitude, but the reductions in the concentrations of SO_4^{2-}, and Mg^{2+} were less pronounced. Following annual applications of high quantities of fresh irrigation water, the soil at AL underwent desalination with no sodication because the concentrations of Ca^{2+} and Mg^{2+} remained considerable relative to the concentration of Na^{+}. Gypsum crystals within the soil (Herrero, 1982; 2008) and the Gypsic Haploxerepts at the bottoms of nearby valleys (Nogués et al., 2006) were sources of Ca^{2+}, which prevented sodication of the soils.

At AL, in the first sampling period, ionic concentrations were higher at AL1 (*S. vera*) than they were at AL2 (*S. ramosissima*), but that difference had disappeared after 20 years, which reflected the homogenizing effect of fresh water irrigation despite the different salinities of the phreatic water at AL1 and AL2 in 1999 (Table 2). The changes of the soil after 20 years were reflected in: (i) the reductions in ECe from 43.5 dS m^{-1} in AL1 and 27.8 dS m^{-1} in AL2 to ECe < 2 dS m^{-1} in both of these paddy sites, and SAR from 79 to 2, and (ii) the intermediate salinity and sodicity (ECe = 15.2 dS m^{-1} and SAR = 36) recorded in 1985 at ALx, which was between the AL1 and the AL2 sites.

The ionic signature at AG (Figure 12) reflected the ionic homogeneity and temporal stability of the soil salinity at the sampling sites. AG was the only one of the four wetlands where the individual concentrations of both Mg^{2+} and SO_4^{2-} were equal to or greater than the concentrations of the other ions. Furthermore, the concentrations of almost all of the ions were higher at AG than they were at the other wetlands (except CO_3^{2-} plus HCO_3^{-}). The ranking by increasing ionic concentration was AG8 (*S. vera*), AG7 (mixture of *A. macrostachyum* and *S. vera*), AG6 (*A. macrostachyum*), and AG5 (bare soil), which paralleled the average ECe at these sites: 54.9 dS m^{-1}, 62.8 dS m^{-1}, 71.6 dS m^{-1}, and 78.4 dS m^{-1}, respectively. A distinguishing feature at AG was the abundance of gypsum (mean = 63%), which meant that Ca^{2+} was always saturating the soil solutions. Coupled with the abundance of Mg^{2+}, that factor explained the moderate and very similar SAR values (~ 28 for the four sites) despite the high concentrations of Na^{+} in the AG wetland.

3.5. Plants as indicators of soil salinity

The scatter diagram of single synthetic values of SAR and ECe reflects the differences between the soils where different plant species occurred (Figure 13). In the figure, the dashed lines indicate the traditional agricultural thresholds for the phases of soil salinity as defined by Soil Survey Division Staff (1993) and modified by Nogués et al. (2006). Most of the soils were strongly (ECe ≥ 8 dS m^{-1}) or very strongly (ECe ≥ 16 dS m^{-1}) saline, and all surpassed the threshold (SAR = 13) for sodic soils. The only exception was the wetland that had been converted into a paddy, which involved 20 years of irrigation by applying a running water table. Based on the mean ECe of the soils in which they occurred, the ranking of the plants in ascending order was as follows: *P. festuciformis, A. herba-alba*, *S. ramosissima*, S. *vera*, and *A. macrostachyum*. Based on median SAR, the ascending ranking was *A. macrostachyum*, *A. herba-alba*, *S. ramosissima, S. vera,* and *P. festuciformis* (Figure 13).

Puccinellia festuciformis at PC was unique because it occurred in highly sodic soils (SAR range = 62 - 250), and was associated with lower ECe than were the other plant species, which is consistent with some species of *Puccinellia* referred to as 'alkali grass', and with the high frequency of pH > 9 in the soil samples (Figure 4). In the soil samples from PC, Mg^{2+} concentrations were < 2 meq L^{-1} and 13 of 32 samples were below the threshold for analytical detection. Furthermore, Ca^{2+} concentrations were low (mean sum of both ions = 1.5 meq L^{-1}) which led to much higher and a wider range of values of SAR (Figure 13) at PC than were at the other wetlands, despite the relatively low Na^{+} concentrations in the soil at PC (Figure 12). The soil at PC, which had a mean ECe = 16.0 dS m^{-1} and a median SAR = 130 in samples from the upper 20 cm, was prone to ponding after every rain, and the water became warm and hypoxic in summer. Those results are consistent with the ecological conditions associated with other species of *Puccinellia* (Tarasoff et al., 2007; Bennet et al., 2009; Jenkins et al., 2010).

A. herba-alba and *S. ramosissima* occurred in relatively mildly saline soil. The ECe and SAR of the soils in which *S. ramosissima* occurred were very similar (ECe ~ 27 dS m^{-1} and SAR ~ 78) for the three sampling dates in 1979-1980 (Table 1). For *A. herba-alba*, the averages were ECe = 19.4 dS m^{-1} and SAR = 34 based on the six sampling dates, and the ranges of values of were greater than they were for *S. ramosissima*. The highest ECe for *A. herba-alba* was higher than the highest ECe of *S. ramosissima* (Figure 13).

S. vera, which was found in three of the wetlands (AN, AL, and AG), occurred in soils that had a higher ECe that did the soils in which the plant species mentioned in the above paragraph occurred. The *S. vera* sites had an average ECe = 56.3 dS m^{-1} and SAR = 42. At AN3, the average ECe = 58.8 dS m^{-1} and SAR = 100, which were somewhat higher than those at AL1 (ECe = 43.5 dS m^{-1} and SAR = 90). At AG, the soils of *S. vera* had an average ECe =

58.5 dS m^{-1} and SAR = 27. The highest values for *S. vera* soils (ECe = 75.2 dS m^{-1} and SAR = 117) occurred at AN3 (Figure 13).

Among the five halophytes, *A. macrostachyum* occurred in soils that had the highest ECe (mean = 71.6 dS m^{-1}, maximum = 85.2 dS m^{-1}) (Figure 13), which was similar to the values in the area of bare soil (mean ECe = 78.4 dS m^{-1}, maximum = 88.6 dS m^{-1}). In AG, ECe and SAR did not discriminate (Figure 13) the habitats of *S. vera*, *A. macrostachyum*, from bare soil. The results are consistent with the perception in the field of flooding as key factor for the occurrence of *S. vera* and *A. macrostachyum*, with the latter best adapted to long periods of flooding.

Based on the maximum single synthetic soil salinity (ECe dS m^{-1}) of the soils in which they occurred, the ranking of the plants in ascending order was as follows: *P. festuciformis* (19.3), *S. ramosissima* (30.3) *A. herba-alba* (35.0), *S. vera* (75.2), and *A. macrostachyum* (85.2) (Figure 13). SAR is associated with the hydric properties of the soil and its relationship to the presence of specific plant species is indirect, but the high and unsurpassed values at PC were consistent with the high pH and imperviousness of the soil. At the wetlands in this study, only *P. festuciformis* appeared to have the capacity to live in those conditions. In AN, AL, and AG, the presence of gypsum in the soil, which is a source of sulfur, likely enhanced the resistance of plants to salinity (Nazar et al., 2011).

The results of our study can serve as a guide for making plant presence a relative or semi-quantitative indicator of the average ECe and SAR of the soils in saline wetlands; however, the data on the average soil salinity associated with individual halophytes in natural environments cannot be used to assess salt tolerance in agricultural environments. The agricultural perspective of salinity tolerance relies on thresholds or production functions of salinity, which are defined for each plant species or cultivar based on controlled experiments. If our soil salinity data are used in descriptive plant ecology, they have to be evaluated along with many other factors such as the survival of propagules, which is influenced by surficial soil features (Domínguez et al., 2013b), plant scalding caused by ponding on impervious soils (Herrero, 2008), or facilitation and interference phenomena (Callaway and Walker, 1997; Raevel et al., 2012).

4. Conclusions

The similarities in the values of saturation percentage of the soils in four wetlands in NE Spain obtained by two labs 20 year apart substantiate the validity of comparing the soil pastes and ionic compositions from the two sampling periods. Furthermore, the relationships between pH, electrical conductivity, and ionic concentrations were consistent among the sampling periods. Collectively, those results reduce the concern for a bias in the data that might have arisen because of the protracted time between soil sampling periods and that the analyses were performed at two labs 20 years apart. We recommend that the assessments of soil salinity using ECe report the saturation percentage. The technical difficulties associated with the analyses of sulfates in the first sampling period underscore the robustness of the methods used to assess soil salinity.

The ECe and ionic composition of the four saline wetlands were characterized using samples collected in different seasons and years over a 20-years period. That approach provides much more reliable and representative information than do many of the standard reports based on single samplings. The ionic composition of the soil solutions differed among the wetlands, which was reflected in their ionic signatures. The averages and the distributions of the values of ECe and SAR in the soils of the four saline wetlands and their relationships were distinct. Collectively, those features underscore the pedodiversity of the saline wetlands in the region.

The ionic composition of the soils at the three wetlands when there were no changes in land use was similar between the two sampling periods. One saline-sodic wetland was subjected to puddling and annual inundation with fresh irrigation water for rice cultivation, which led to the development of a non-saline, non-sodic soil. The results of this study provide a baseline for long-term studies that are designed to detect changes in soils induced by natural or anthropic factors.

Jointly, ECe and SAR indicate the contrasts in the soil salinity of the habitats of the dominant plants in the wetlands examined in this study. The five halophytes can be estimators of average soil salinity. Similar studies in other saline habitats might verify the suitability of those plants as quantitative indicators of soil salinity in other regions.

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Table 1. Saline wetlands studied in NE Spain. Pits are marked with an asterisk in the last column.

Sampling site Dominant plant	**Sampling date**	**Depth reached (cm)**	**Number of soil samples taken**
AN3 *Suaeda vera*	1979-04-18	110	10
	1979-07-10	140	*5
	1979-08-28	140	14
	1980-02-03	120	12
	1999-03-30	220	11
	1999-08-19	210	9
	2000-02-13	174	8
AN4 *Artemisia herba-alba*	1979-04-18	150	15
	1979-08-28	65	7
	1980-02-03	120	12
	1999-03-30	220	11
	1999-08-19	100	5
	2000-02-13	184	9
PC1 Bare soil	1985-04-03	148	7
	1999-03-30	200	9
PC2 *Puccinellia* sp.	1985-04-03	160	8
	1999-03-30	180	8
AL0 *Suaeda vera* with bryophytes	1979-03-04	85	*5
AL1 *Suaeda vera*	1979-04-13	110	9
	1979-07-10	145	*6
	1979-08-24	100	9
	1980-02-09	105	10
	1999-02-21	130	10
	1999-08-01	158	5
	2000-02-12	130	4
AL2 *Salicornia ramosissima*	1979-04-17	120	10
	1979-08-24	80	8
	1980-02-09	105	10
	1999-02-21	130	10
	1999-08-01	140	5
	2000-02-12	130	5
ALx (located between AL1 and AL2) Rice fallow.	1985-03-12	105	5

AG5 Bare soil with stumps of *Arthrocnemum macrostachyum*	1979-04-19	75	7
	1979-08-20	80	8
	1980-03-09	70	6
	1999-02-28	88	8
	1999-07-22	44	5
	2000-02-12	80	4
AG6 *Arthrocnemum macrostachyum*	1979-04-19	65	7
	1979-08-20	65	7
	1980-03-09	70	6
	1999-02-28	70	5
	1999-07-22	56	6
	2000-02-12	56	3
AG7 *Arthrocnemum macrostachyum* and *Suaeda vera*	1979-04-21	50	5
	1979-08-20	55	6
	1980-03-09	55	6
	1999-07-22	41	4
	2000-02-12	38	2
AG8 *Suaeda vera*	1979-05-12	65	7
	1979-08-20	64	7
	1979-09-16	60	*2
	1980-03-09	65	7
	1999-02-28	60	5
	1999-07-22	65	6
	2000-02-12	74	4
AG9 *Arthrocnemum macrostachyum*, dead	1979-07-29	80	*0
			404

Table 2. Analyses of water; electrical conductivity (EC) in dS m^{-1}, ions in meq L^{-1}.

Date	Origin of the water	pH	EC	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	SAR	CO_3^{2-}	HCO_3^{-}	SO_4^{2-}	Cl^{-}
						PC						
05/06/85	PC2 augering		8.49	3.0	2.0	79.2		50.1	8.4	10.1	13.6	61.7
05/06/85	irrigation canal	8.19	0.33	1.5	0.5	0.9		0.9	0.3	2.0	0.4	0.4
18/06/85	PC2 augering	8.11	8.12	1.0	1.0	90.1		90.1	ip	18.5	23.8	53.7
04/10/85	PC2 augering		8.23			86.0						
04/10/85	drainage ditch		1.78			15.6						
						AL						
21/02/99	AL1 augering		1.36	1.5	7.3	3.9	-	1.9	ip	5.0	2.9	3.3
21/02/99	AL2 augering		0.95	1.9	3.8	1.3	-	0.8	ip	2.8	1.2	2.4
01/08/99	AL1 augering	7.61	4.87				0.7		ip	8.4		
01/08/99	AL2 augering	7.95	1.93				0.4		ip	13.0		
						AG						
16/06/79	AG 8pit	7.58	98.35			891.3						
29/07/79	AG9 augering	7.52	102.20			965.2						
28/02/99	AG5 augering		105.70						2.4	2.9		
12/02/00	AG5 augering	7.05	105.10						ip	5.6		1266.8

Table 3. Regressions of ECe on EC1:5 (both in dS m^{-1}) for several sampling periods at four saline wetlands in NE Spain, and for two transformations of EC1:5 using the saturation percentage (SP). The Table also shows the number of samples (n), the coefficient of determination (R^2), and the standard error (S).

Years	n	*a*	*b*	R^2	S dS m^{-1}
		ECe = *a* + *b* × EC1:5			
1979, 80, and 85	89	1.28	7.24	80.7	10.63
1999 and 2000	161	-0.24	7.06	83.3	12.34
All	250	0.30	7.13	82.7	11.70
		ECe = *a* + *b* × EC1:5 × (500 × SP^{-1})			
All	250	‡ 4.46	0.47	92.4	7.80
		ECe = *a* + *b* × EC1:5 × $(500 \times SP^{-1})^{0.6}$			
All	250	0.46	1.53	95.5	5.98

‡ differs significantly from 0 ($p < 0.005$).

Table 4. Equations of the form log [ion] = *a* + *b* × log ECe for the ionic concentrations in the soils at four saline wetlands in NE Spain, with the number (n) of soil samples, and the coefficient of determination (R^2).

	Sampling years	n	*a*	*b*	R^2
Cations	All	395	0.906	1.16	98.3
	1979, 1980 and 1985	234	0.747	1.26	98.1
	1999 and 2000	161	0.941	1.13	98.3
Anions	All	247	1.018	1.11	97.6
	1979, 1980 and 1985	89	0.740	1.25	97.8
	1999 and 2000	158	1.042	1.13	98.5

Table 5. Correlation coefficients between ECe (dS m^{-1}) and the most abundant ions (meq L^{-1}) in saturation extracts (p = 0.000) of the soils at four saline wetlands in NE Spain.

Extracts from	Mg^{2+}	Na^{+}	SO_4^{2-}	Cl^{-}
1979, 1980, and 1985	0.753	0.880	0.929	0.951
1999, and 2000	0.868	0.968	0.898	0.987

Table 6. Regression equations of ECe on [EC1:5 × $(500/SP)^q$], with the form CEe = *a* + *b* × CE1:5 × $(500/SP)^q$, based on soil samples collected at the four saline wetlands; n is the number of samples, R^2 is the coefficient of determination, and S is the standard error.

Wetland	**q**	***a***	***b***	**R^2** %	**S** dS m^{-1}
AN n = 77	0	‡-7.12	7.70	86.9	6.59
	1	-4.69	0.67	87.8	6.35
	0.5	‡-7.74	2.38	91.6	5.28
PC n = 32	0	0.14	8.13	93.9	1.62
	1	‡-3.84	0.71	94.8	1.49
	0.7	‡-2.50	1.49	95.8	1.34
AL n = 69	0	0.04	7.44	96.8	3.09
	1	-0.38	0.74	98.8	1.88
	0.9	-0.36	0.94	98.9	1.84
AG n = 72	0	‡35.27	3.58	13.0	18.60
	1	‡17.29	0.37	84.4	7.87
	0.8	‡ 9.11	0.75	86.1	7.44

‡ differs significantly from 0 (p < 0.005).

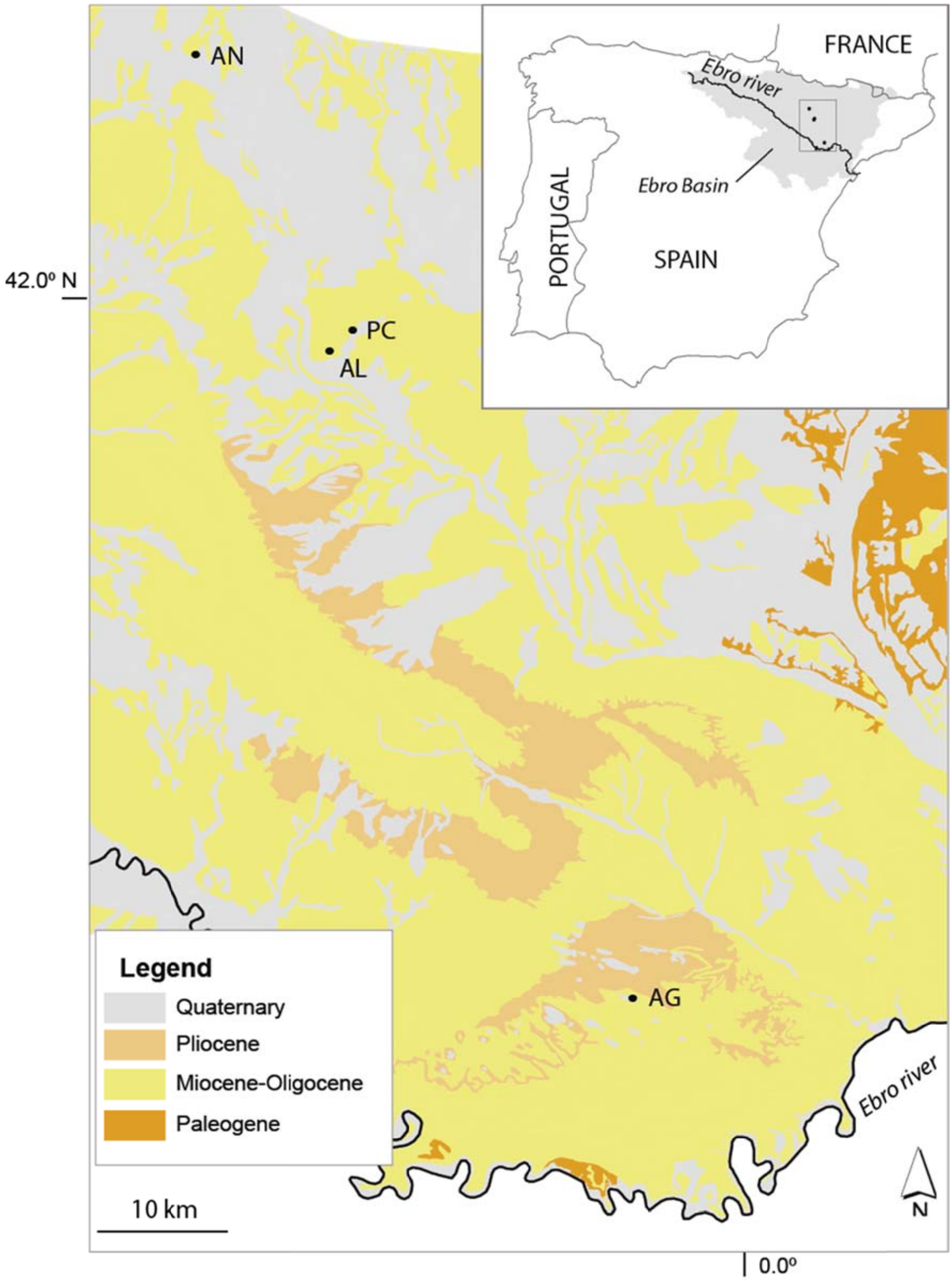


Figure 1. Simplified geological map (modified from www.chebro.es) with the location of the four saline wetlands AN, AL, PC, and AG studied.

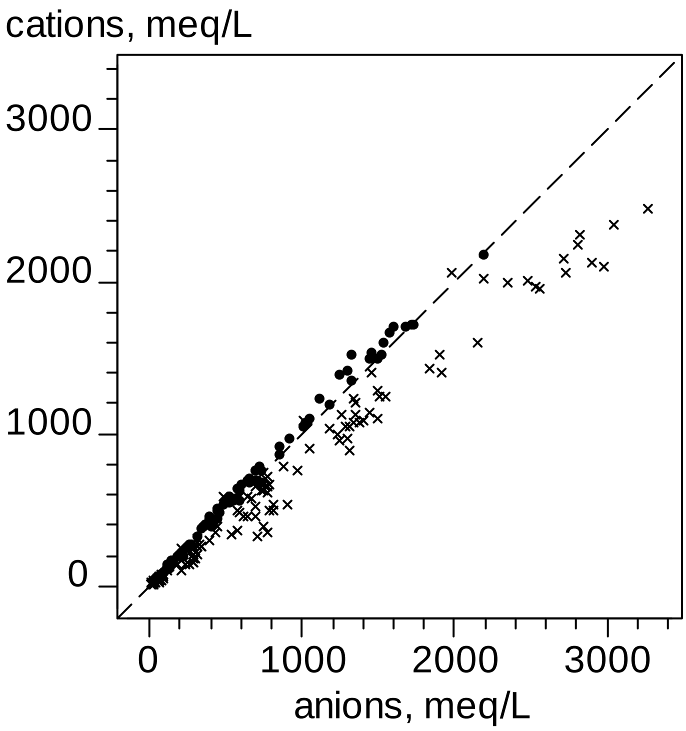
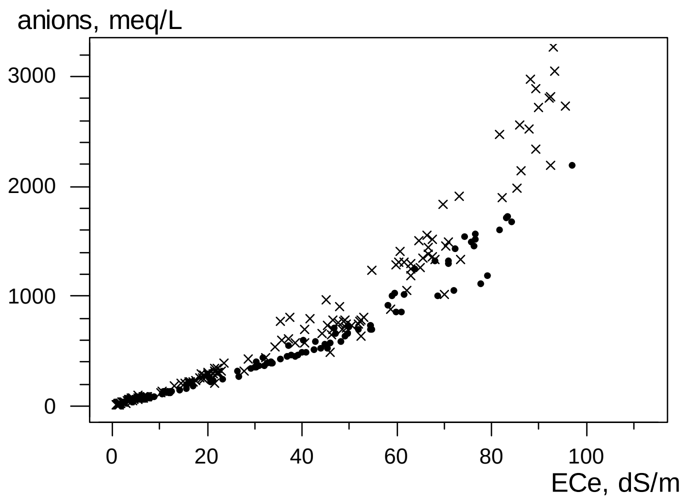


Figure 2. Scatter diagram of the sums of the concentrations of the major ions at the soil of four saline wetlands in NE Spain determined in 88 saturation extracts from 1979, 1980, and 1985 (dots); and in 159 extracts from 1999 and 2000 (crosses).



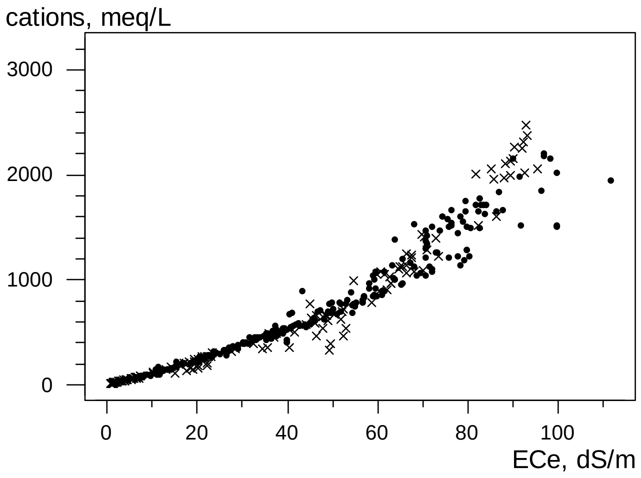


Figure 3. Scatterplots of the cations and anions concentrations against ECe in the saturation extracts of the soil samples from 1979, 1980 and 1985 (dots) and from 1999 and 2000 (crosses) at four saline wetlands in NE Spain.

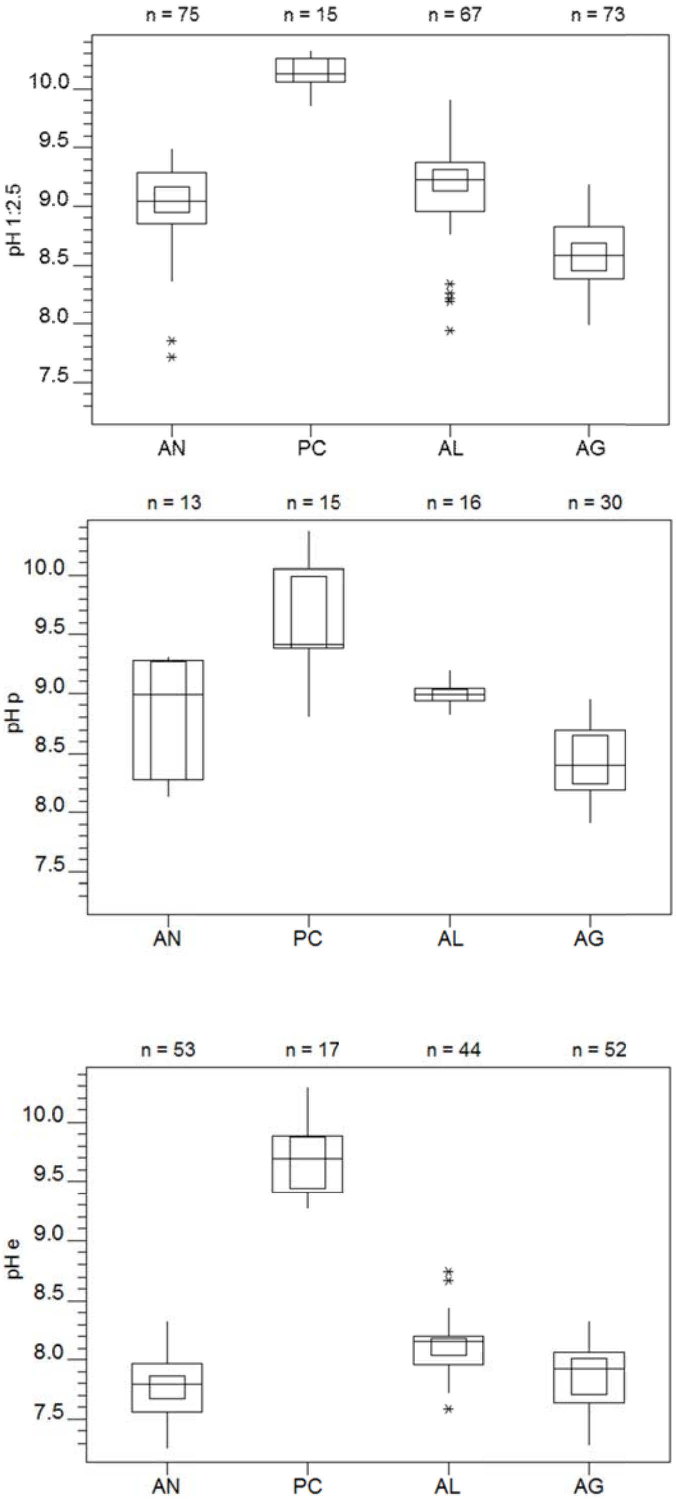


Figure 4. Boxplots of pH in 1:2.5 extract, in the saturated paste, and in the saturation extract from soil samples collected at four saline wetlands in NE Spain.

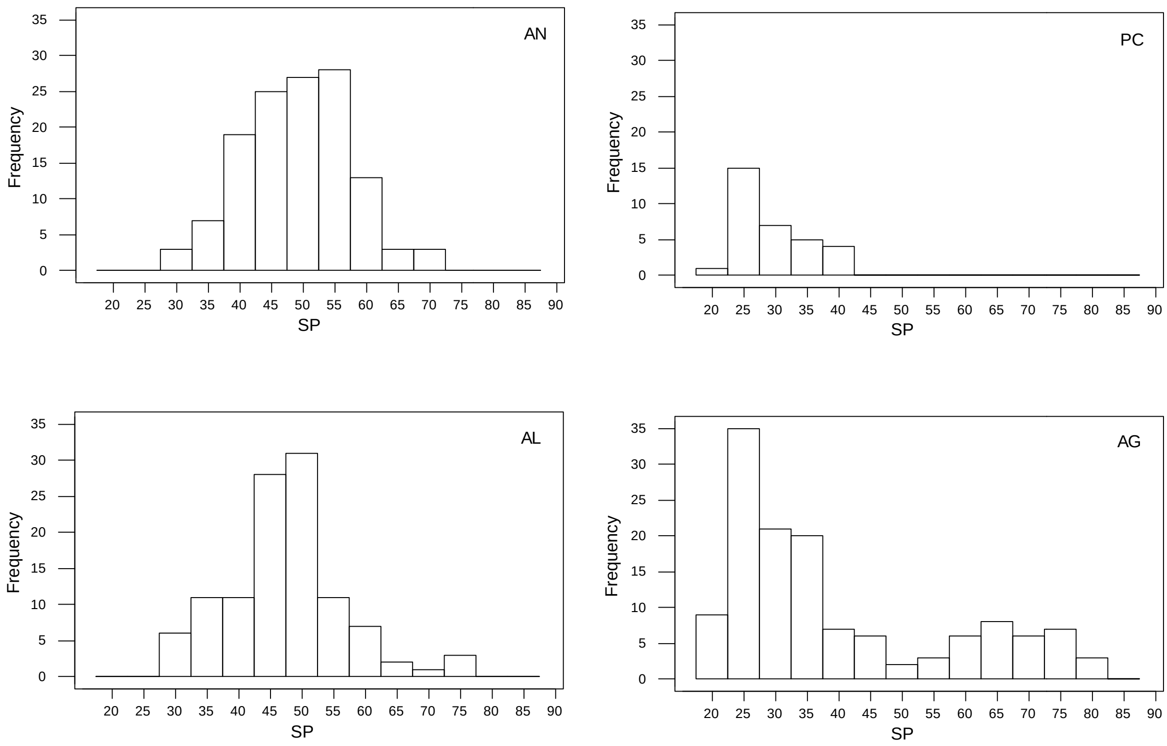


Figure 5. Saturation percentage (SP) of the soil samples from four saline wetlands in NE Spain.

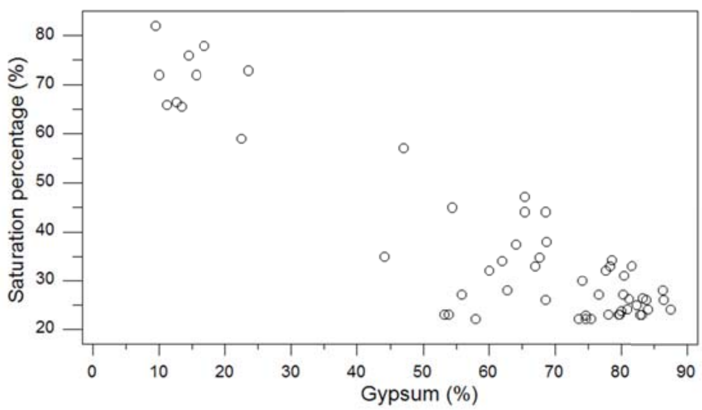


Figure 6. Scatter diagram of the saturation percentage (SP) and gypsum content in 55 soil samples from a saline wetland (AG) in NE Spain.

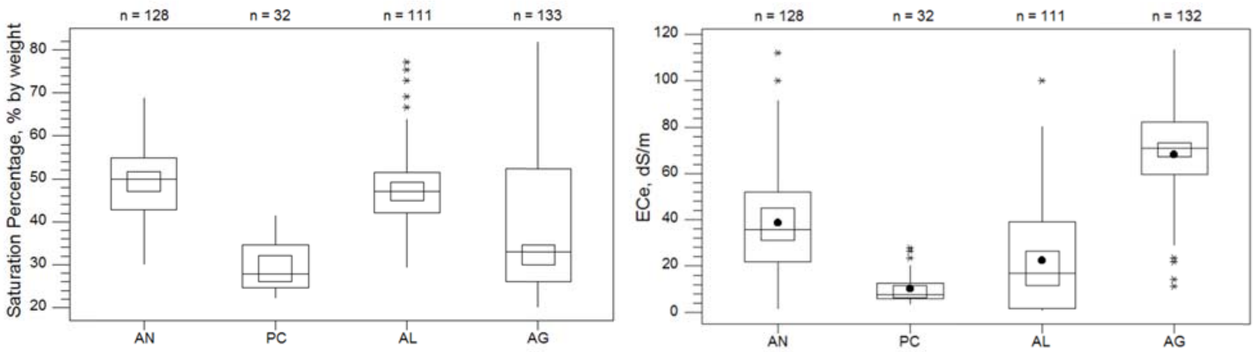


Figure 7. Saturation percentage and ECe of soil samples from four saline wetlands in NE Spain (n = number of samples).

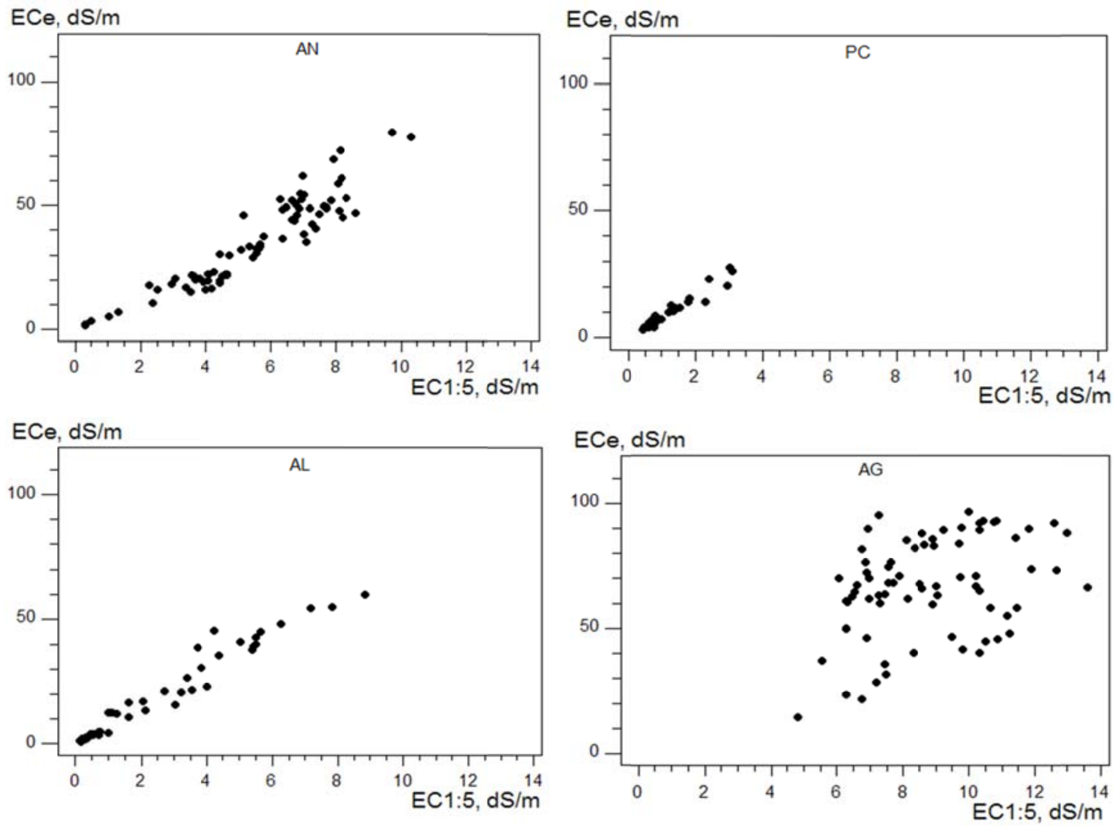


Figure 8. Scatter plot of ECe versus EC1:5 in the soil samples from four saline wetlands in NE Spain.

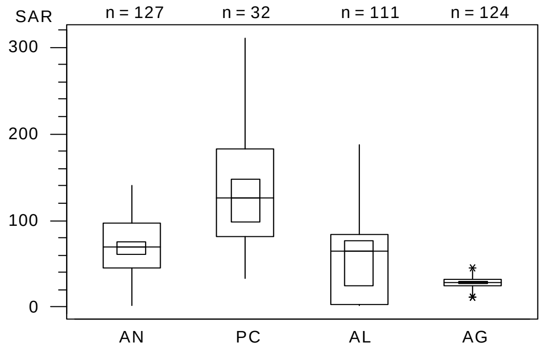


Figure 9. Sodium adsorption ratio (SAR) in the saturation extracts of soil samples from four saline wetlands in NE Spain.

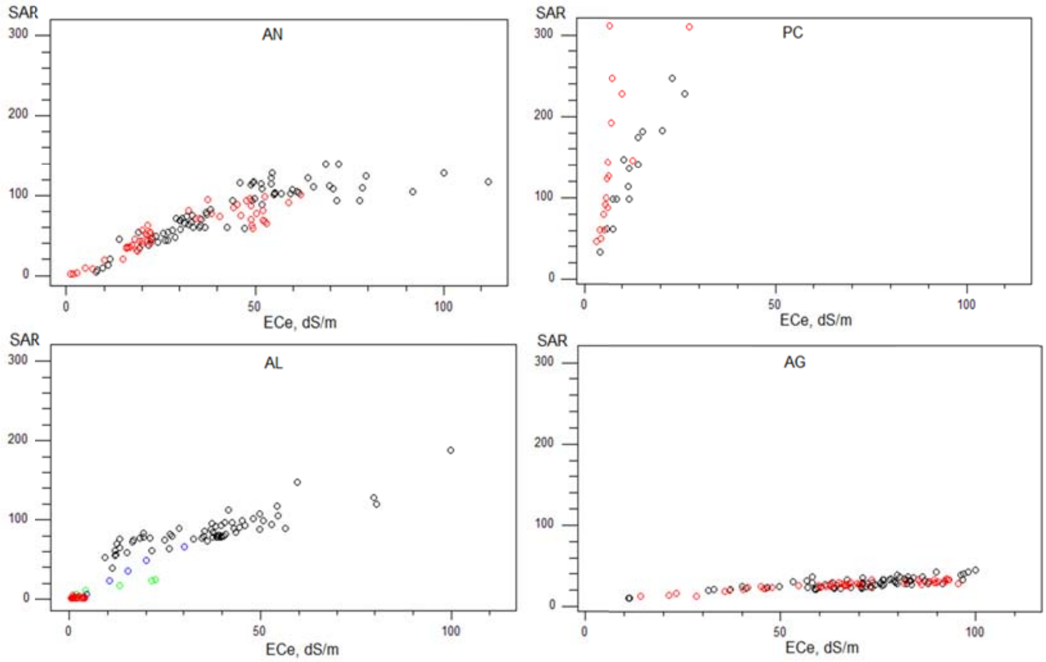


Figure 10. Scatterplots of SAR versus ECe in the soil samples from four saline wetlands in NE Spain. Black circles are for soil samples from 1979 and 1980; red circles from 1999 and 2000; blue are for year 1985; and green are for 1979 in a non-irrigated field with a dense cover of *Suaeda vera* and bryophytes (AL0).

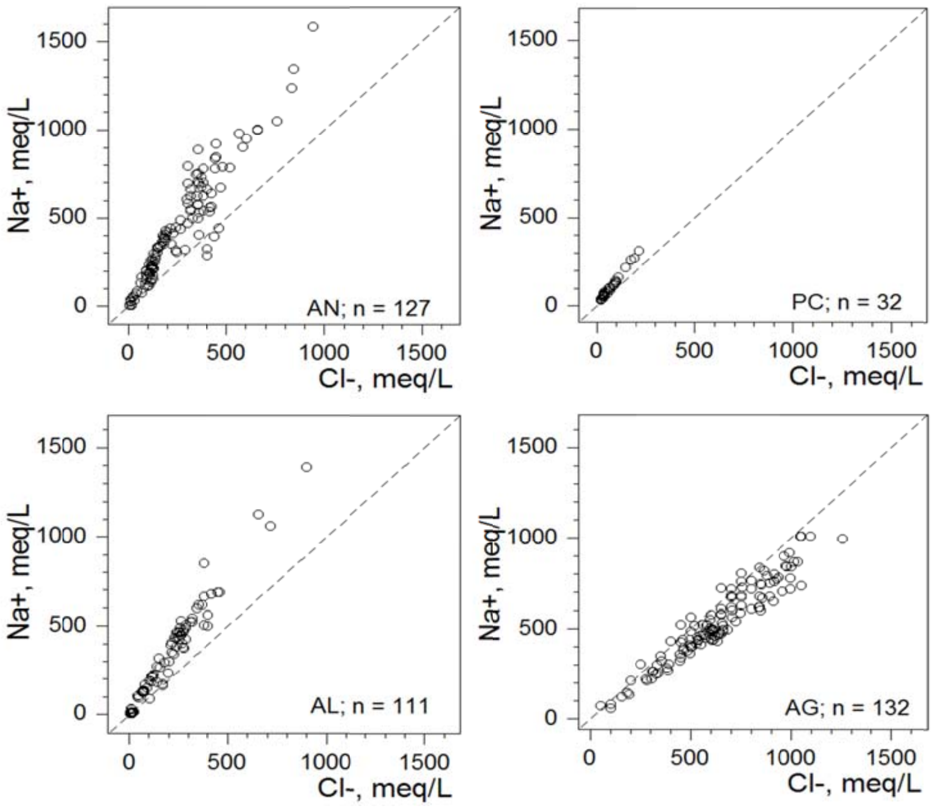


Figure 11. Scatterplots of the concentrations of Na^+ and Cl^- in the saturated extracts of soil samples from four saline wetlands in NE Spain. Dashed line is $Na^+ = Cl^-$.

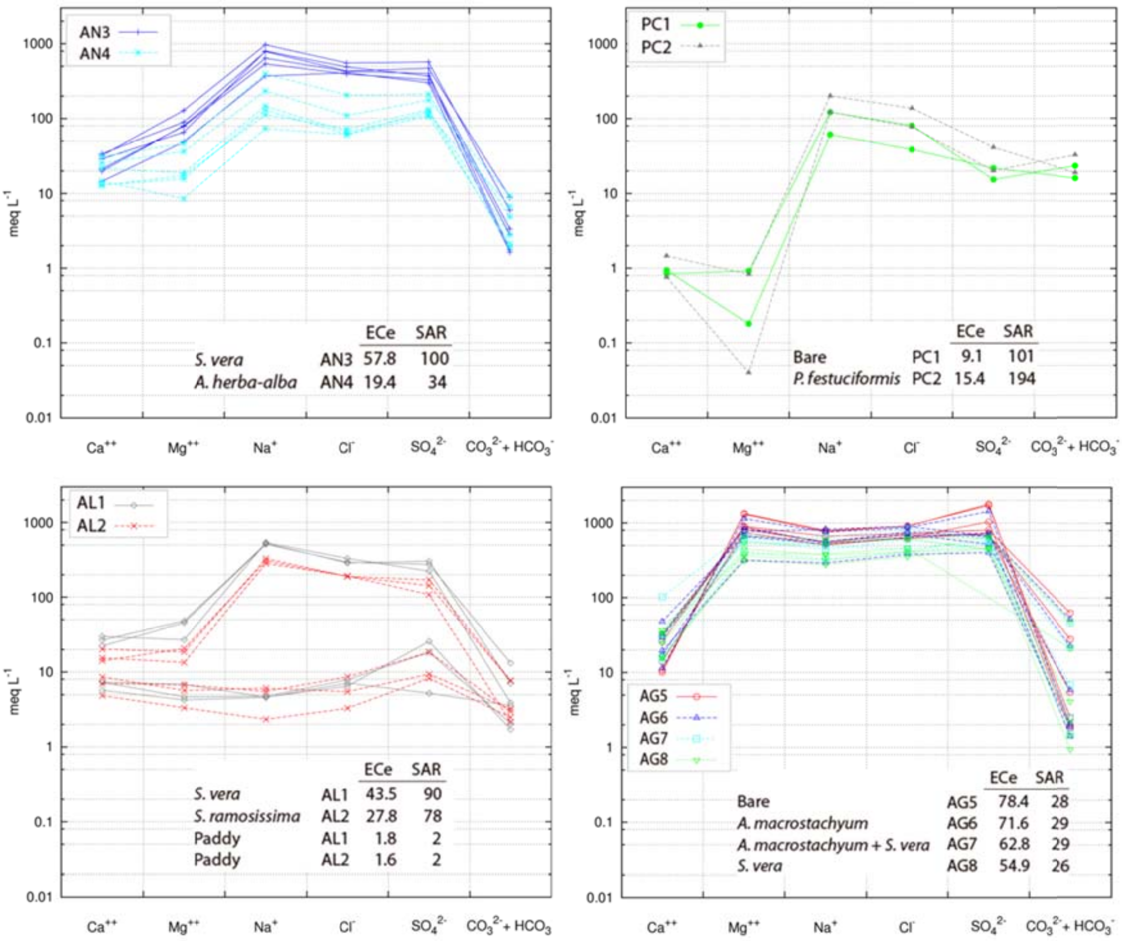


Figure 12. Schoeller-Berkaloff diagrams of the saturated extracts of soil samples from four saline wetlands in NE Spain. Ionic composition was weighted for each site to 100 cm, or to the maximum depth if 100 cm was not reached. ECe and SAR are displayed by habitat. The inserts display the average ECe and SAR of the single synthetic values for each habitat.

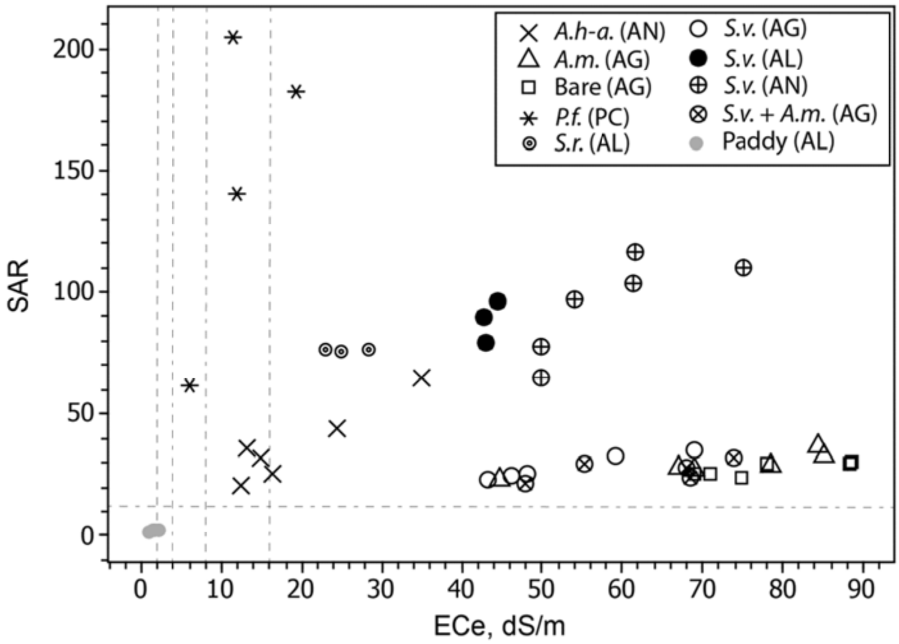


Figure 13. Scatterplot of the median SAR versus mean ECe at the augerings computed until 100 cm depth or to the maximum depth if the augering was < 100 cm. Dashed lines mark thresholds usual in agricultural research.