1. Introduction

The sorptive interaction of DOM in soil is a highly reactive and dynamic process, influencing the carbon cycle and the fate of contaminants and other organic and inorganic species in the agro-ecosystem. The major control of DOM stabilization in soil is conferred by the soil’s mineral constituent (Kleber et al., 2005; Rumpel and Kögel-Knabner, 2011; Sanaullah et al., 2011), whereas soil organic matter (SOM) generally tends to restrict DOM-soil interactions (Kaiser et al., 1996). Several major DOM-sorption mechanisms may operate concurrently in soil, among them ligand-exchange reactions with Fe- and Al-oxide surfaces, anion exchange with positively charged sites on soil minerals (limited at alkaline pH), cation-bridging between both negatively charged soil minerals and DOM functional groups, and van der Waals interactions. In addition to the DOM characteristics and mineral matrix composition, the solution chemistry also controls DOM sorption capabilities by affecting the electrostatic forces existing between sorbates and sorbents (Jardine et al., 1989; Rashad et al., 2010; Shen, 1999). Ligand-exchange complexation of DOM at metal-oxide surfaces has been frequently implicated as a dominant mechanism for DOM sorption in soils (Kahle et al., 2004; Kaiser and Zech, 2000a, 2000b; Kaiser et al., 1996, 1997; Nambu and Yonebayashi, 2000). Clay minerals have also been shown to contribute significantly to DOM sorption (Mikutta et al., 2009), but with a higher tendency toward the relatively less energetic van der Waals interactions (Jardine et al., 1989; Polubesova et al., 2008) or cation- and water-bridging mechanisms (Baham and Sposito, 1994; Chorover and Amistadi, 2001).

Most of the reported data on DOM sorption to soil involves soils with a significant amount of SOM as sorbents, thus reflecting the mixed effects of organic matter and the mineral constituent on DOM sorption (Kaiser et al., 1996; Kothawala et al., 2009; Lilienfein et al., 2004; Ling et al., 2006). Moreover, the evaluation of DOM sorption in such systems necessitates the consideration of the amount of native carbon released as DOM during the interaction. The incorporation of a parameter representing the released DOM in the sorption isotherm’s model (Kaiser et al., 1996; Lilienfein et al., 2004; Nambu and Yonebayashi, 2000) has its toll for the accuracy of determined sorption parameters. Consequently, the vast majority of studies dealing with DOM sorption focus on its interaction with pure minerals (Chorover and Amistadi, 2001; Gu et al., 1994; Kaiser and Guggenberger, 2007) or with isolated soil clay fractions (Navon et al., 2011), sometimes stripped of their native organic matter (Kahle et al., 2003, 2004). In either case, the natural organization of the mineral matrix within the soil is utterly overlooked (Kang and Xing, 2008; Ohno et al., 2007; Rennert and Mansfeldt, 2003).

In light of the vast knowledge that has been gained regarding sorptive interactions of DOM with major mineral species of soils, and of the fundamental need to fill in the knowledge gaps in natural soil systems, the goal of the current study was to elucidate sorption and desorption interactions of DOM in soil systems and to link them to soil mineral matrix characteristics. Because of the scarcity of studies made on DOM-sorption in alkaline soil systems (Kahle et al., 2003; Rashad et al., 2010; Shen, 1999) and the wide occurrence of such systems in major agricultural regions (e.g., Mediterranean) we chose to focus the current study on a set of alkaline soils also exhibiting...
distinct mineral-phase compositions as well as negligible levels of native SOM. Our major hypothesis was that sorption and its degree of irreversibility would be governed by the mineral makeup (oxides and clays) of the soils. An original feature of the experimental set up was the performance of successive cycles of sorption–desorption procedures. In this study we also tested the hypothesis that sorption of DOM onto mineral surfaces may modify surface properties and reactivity (Murphy et al., 1990; Tipping and Cooke, 1982; Vermeer and Kooij, 1998) and reduce the mineral surface area available for interaction. This should lead to a negative relationship between the extent of sorption irreversibility and a soil’s capacity for a successive sorption phase.

2. Materials and methods

2.1. Soils

Soil cores (90–120 cm; B horizons) were collected from three sites in Israel located along the Mediterranean coastline in areas which are traditionally used for intensive agriculture. The soils (Fluent, Rhoodoxeral and Loess) were collected from Akko (N 33° 12’ E 35° 738’), Basra (N 32° 43’ E 35° 549’), and Nir-Oz (N 31° 45’ E 35° 041’), respectively. The soil samples were air-dried and sieved (<2 mm).

Fundamental soil chemical and physical analyses were performed according to Sparks (1996) and Dane and Topp (2002), respectively. Particle size distribution was determined by the hydrometer method. Soil pH was determined in an equilibrated soil aqueous extract (1:10 ratio, in double-distilled water). Total organic carbon (TOC) content was determined by the method of US Environmental Protection Agency (1985) and the resultant texture was determined by the hydrometer method. Particle size distribution was determined by the hydrometer method. Soil pH was determined in an equilibrated soil aqueous extract (1:10 ratio, in double-distilled water).

2.2. DOM

DOM was extracted from mature composted biosolids (sewage sludge and tree clippings, 1:1 v/v) obtained from a commercial compost factory (Dilila, Israel). DOM was extracted by mixing a homogeneous compost sample with distilled water (1:1 ratio) and shaking (200 rpm) for 12 h. The DOM solution was isolated by centrifugation (10,000 g for 20 min) and subsequent pressure filtration through a 0.45-μm membrane filter. The DOM solution exhibited pH, electrical conductivity (EC), and TOC values (measured with a VCPH model TOC analyzer, Shimadzu Scientific Instruments, Kyoto, Japan) of 9.5, 2.3 dS m⁻¹, and 300 mg L⁻¹, respectively. The isolated DOM was subsequently freeze-dried.

2.3. Sorption–desorption experiments

Sorption–desorption processes were investigated in batch. Aliquots (20 mL) of DOM solution were mixed with 2-g portions of soil in 40-mL centrifuge tubes. Seven DOM concentrations were implemented (from -3 to -75 mg C L⁻¹), corresponding to the wide range of environmental soil conditions. Experiments were run in duplicate. The DOM solutions were prepared by dissolving freeze-dried DOM material in a background solution containing 3.7 mM CaCl₂ and 1.5 mM NaN₃. The solutions were passed through a 0.45-μm membrane filter prior to their introduction to the soils. An equilibrium time of 4 d was determined in preliminary kinetic experiments. The possibilities of DOM–vessel interaction and/or microbial degradation of DOM were excluded by verifying that the TOC of a DOM solution run without added soil did not change during a similar equilibration period. In addition, an organic carbon mass balance evaluation was performed for an equilibrated DOM-soil sorption system. The mass balance (considering solution as well as solid phase measurements) accounted for >90% of the organic carbon in the system.

Tubes containing soil and a DOM-free background solution served as controls for the estimation of the amount of native SOM released during the sorption interaction. These amounts were considered in the calculation of the amounts of DOM sorbed onto soil as follows: sorbed DOM = [initial solution DOM – (measured equilibrium solution DOM – native SOM released as DOM)]. Although these amounts were limited to begin with (2–5 mg C L⁻¹) an effort was made to further reduce them by shaking the soil portions to be used in the sorption experiments with distilled water for 2 h (200 rpm) prior to the DOM application. The tubes were centrifuged (10,000 g for 10 min) and the rinsing solutions decanted, except for 5 mL. The subsequent addition of DOM solutions of predetermined concentration generated the actual initial DOM concentrations which were calculated and considered thereafter in the calculation of sorbed DOM. The tubes were vortexed to re-suspend the settled soil material and the 4-d equilibration period started.

At the end of the sorption-equilibration period, the tubes (including the controls) were centrifuged and a 10-mL aliquot of the supernatant was drawn, filtered (0.45 μm), and analyzed for TOC content. A fresh 10-mL aliquot of DOM-free background solution was then added and a 4-d desorption phase began. In the calculation of the amount of DOM desorbed from the soil, the amount released in the corresponding control experiment (<1 mg C L⁻¹) was deducted. Four similar sequential desorption steps were performed, after which a successive sorption phase was initiated using DOM solutions similar in concentration to those employed in the original sorption phase. Finally, a successive desorption phase was initiated in a manner similar to the first desorption phase and included three desorption steps. Complementary sorption–desorption experiments were conducted with DBC-treated samples (i.e., metal oxide-free soils). Following a similar metal-extraction procedure as described above (Section 2.1), the treated soil samples were sequentially rinsed with distilled water until their EC and pH values approached those of the untreated samples. Then, DOM was applied at a single, intermediate concentration level (~40 mg C L⁻¹) and sorption–desorption was evaluated. However, this time desorption was induced by exchanging the entire equilibrium solution volume with a background solution and performing only a single desorption step. Comparing sorption–desorption between DBC-
treated and non-treated samples allowed the assessment of how central metal oxides are in the sorption and desorption of DOM.

2.4. Sorption–desorption data analysis

Concentrations of DOM sorbed onto soil following sorption, desorption, successive sorption, and successive desorption phases were plotted against their corresponding solution equilibrium concentrations in sorption–desorption isotherms. Due to the associated non-linearity of the isotherms, Freundlich and Langmuir models were employed for the estimation of sorption parameters. Sorption capacity was conveyed with the Langmuir $Q_{\text{max}}$ parameter while sorption affinity was expressed as a distribution coefficient ($K_d$) extracted from the Freundlich equation for two equilibrium DOM concentrations (low and high). The fitting of these non-linear sorption models to the sorption isotherm data was performed using a spreadsheet capable of performing non-linear least squares regression (Bolster and Hornberger, 2007). Sorption–desorption hysteresis levels were calculated (for each DOM concentration level independently) as the percent of DOM amount which had been sorbed and not desorbed in the course of desorption or successive desorption phases.

3. Results and discussion

3.1. Soil mineral matrix characteristics

The three soils investigated varied distinctively with respect to particular traits that could potentially affect DOM sorption. Soil texture varied from clay in the Akko soil to sandy loam in the Basra and Nir-Oz soils (Table 1). Accordingly, the SSA values in the two latter soils were significantly smaller. The XRD analysis of the three soils revealed varying proportions of major mineral groups: in Akko, clay minerals dominated whereas in the two sandy loam soils clay minerals made up only a secondary constituent; quartz practically dominated the mineral matrix of the Basra soil (~90% according to Ravikovich (1981)), was a major constituent in the Nir-Oz soil (though less dominant than in Basra), and was an important constituent in the Akko soil although only secondary to clay minerals. Traces of hematite were positively detected by the XRD analysis only in the Akko and Basra soils. Complementary DCB extraction and ICPAES quantification of metal ions confirmed that the sum of Fe, Al, and Mn oxide minerals was in the order: Akko > Basra > Nir-Oz (3:4:2:1:1:0).

3.2. Sorption of DOM

DOM sorption isotherms (Fig. 1) were constructed in a cumulative manner, where the sorbed amount in the successive sorption step considered the non-desorbed DOM amount remaining in the soil (Fig. 1B) and in a non-cumulative manner, where the sorbed amount in the successive sorption step disregarded the non-desorbed amount (i.e., as if no sorbed DOM was present; Fig. 1A). The successive sorption behavior greatly resembled the original sorption pattern in each of the three soils with no major differences in sorption parameters between the first and second sorption phases (Table 2). This suggests that the binding interactions in the two stages (first and successive sorption) are similar, thus the sorption parameters calculated for the first sorption cycle were compared thereafter with literature data.

The sorption isotherms obtained for the three studied soils illustrate that, while sorption capacity for DOM may possibly be enhanced by a fine soil texture, a certainly more critical factor is the content of metal-oxide minerals, as previously affirmed (Jardine et al., 1989; Kaiser et al., 1996). Particular support for that assertion was given by the greater sorption to the Basra soil than to the Nir-Oz soil despite the smaller SSA in the former soil. The clay- as well as metal-oxide-rich Akko soil (Table 1) sorbed significantly larger amounts of DOM than the other two soils at any given equilibrium concentration.
saturation. Kothawala et al. (2009) reported \( Q_{\text{max}} \) values ranging from 500 to 567 mg C kg\(^{-1}\) for 20 soil samples from B and C horizons which exhibited TOC and DCB-extractable Fe levels (\( \text{Fed} \)) that closely matched those of the currently investigated soils. Ling et al. (2006) reported a \( Q_{\text{max}} \) range of 268–700 mg C kg\(^{-1}\) for 10 soil samples covering a broad range of TOC contents. Shen (1999) presented a \( Q_{\text{max}} \) range of 65 to 1300 mg C kg\(^{-1}\) for 6 soils which greatly varied with respect to both TOC and clay contents, whereas Jardine et al. (1989) extended the \( Q_{\text{max}} \) range to 1344–2392 mg C kg\(^{-1}\) for a set of subsoils characterized by high \( \text{Fed} \) levels (16–21 g kg\(^{-1}\)). In another study conducted by Jardine et al. (1992), this time on DOM sorption to a sandy aquifer material which was low in TOC as well as in metal oxides, a considerably lower \( Q_{\text{max}} \) value was reported (245 mg C kg\(^{-1}\)).

Kothawala et al. (2009) concluded that the amount of Fe and Al associated with mineral oxides is the strongest predictor of \( Q_{\text{max}} \) whereas the influence of soil texture is largely irrelevant. On the contrary, Shen (1999) found a compelling positive correlation between \( Q_{\text{max}} \) and soil clay content, and Ling et al. (2006) observed a similar trend, albeit much less inclusive. The decisive work of Jardine et al. (1989) emphasized that both metal oxide and clay minerals could potentially be important vectors in the sorption behavior of DOM in soils. On the one hand, \( \text{Fed} \) was found to account for 50 to 70% of the DOM sorbed depending on soil type; on the other, pure kaolinite was shown to be equivalent to pure gibbsite with respect to its sorption capacity for DOM. Likewise, kaolinite’s sorption capacity equaled that of whole soil samples rich in metal oxides. Quite the reverse of these results, significantly smaller sorption capacities as well as affinities were recorded by Kaiser and Zech (2000a) for DOM sorption onto kaolinite and illite relative to its sorption onto Al(OH)\(_3\) and goethite. Validating the discrepancy between the two latter groups’ results is the fact that the \( Q_{\text{max}} \) levels recorded for kaolinite in the two studies were not very different.

3.3. Desorption of DOM

DOM desorption isotherms revealed strong sorption–desorption hysteresis with the three soils (Fig. 2A): up to 83% of sorbed DOC was retained in the soil after four sequential desorption steps. Many reports implied that DOM–mineral interactions are often irreversible (Gu et al., 1994, 1995; Joo et al., 2008; Kahle et al., 2003; Kaiser and Zech, 1999; Mikutta et al., 2009). Hysteresis levels (determined as in the present study) as high as 79 to 86% were calculated for wetland pond DOM sorbed onto hematite following five sequential desorption steps (Gu et al., 1994). Even higher hysteresis levels were obtained for the sorption of isolated hydrophilic and hydrophobic fractions of the same DOM source (Gu et al., 1995). The sorption of forest floor DOM on a variety of metal-oxide as well as clay minerals was shown to be highly irreversible (97–99% hysteresis; Kaiser and Zech, 2000a). Sorption–desorption of peat humic acid with pure quartz and with Fe-oxide- and Al-oxide-coated quartz sorbents resulted in hysteresis levels in the ranges of 30–40%, 93–99%, and 92–97%, respectively (Joo et al., 2008).

In contrast with the multitude of studies examining desorption of DOM from model mineral materials, the potential for DOM sorption irreversibility in soil systems has not received much attention. Recently, Navon et al. (2011) reported virtually complete hysteresis in the sorption–desorption interaction of sewage-sludge compost DOM with a vertisol (68% clay) and with its isolated clay fraction. Kahle et al. (2003) reported hysteresis values ranging from 69 to 86% (average 82%) and from 79 to 89% (average 86%) for the interaction of forest...
The sorption–desorption hysteresis levels recorded in the current study were comparable to those reported by Kahle et al. (2003) and were mostly in the range of 60–80%. The Akko and Nir-Oz soils exhibited somewhat higher hysteresis levels than the Basra soil across the concentration range, while below an initial DOM concentration of about 40 mg C L$^{-1}$, the Akko soil showed more hysteretic behavior than the Nir-Oz soil. The trend in the level of sorption–desorption hysteresis corresponds to the trend observed for SSA (Akko > Nir-Oz > Basra). Likewise, Joo et al. (2008) observed lower hysteresis levels in the sorption of humic acid to pure quartz relative to sorption onto metal-oxide-coated quartz. The role of quartz in promoting the release of sorbed DOM may be direct, in terms of its relatively low sorption affinity (e.g., formation of van der Waals bonds), or indirect, involving its low effective surface area.

In addition to the recognized importance of metal-oxide content in the sorption of DOM to soils, this trait was further shown to affect the desorption process. Comparing the few studies in which both the level of DOM sorption–desorption hysteresis in soil (or a soil clay fraction) and soil Fe$_{ox}$ levels were provided, a general positive relationship between the degree of hysteresis and the content of Fe-oxides was found (Fig. 3A). In addition to the merely correlative efforts made to associate sorption and desorption with soil mineral traits, some direct experimental evidence (including from the current investigation) exists for the role of Fe-oxides in controlling the sorption and desorption of DOM. Assimilating four studies in which sorption of DOM to soils or soil clay fractions was comparatively performed prior to, and subsequent to the neutralization of effective sorption sites on metal oxides (Fig. 3B), a clear positive correlation was found between the initial soil Fe$_{ox}$ content and the extent of reduction in sorption due to the neutralizing action. Beyond restricting sorption, the neutralization of metal-oxide sorption sites appears to generally facilitate the desorption process relative to non-treated samples (Fig. 3B).

3.4. A successive cycle of DOM sorption–desorption

The hypothesis assuming a negative relationship between the extent of sorption irreversibility and the capacity of a soil for a successive sorption phase was refuted by the sorption and successive sorption data obtained in the current study. Despite the extensive sorption–desorption hysteresis levels and the resulting presence of organic matter residues in the soil, the successive sorption behavior greatly resembled the original sorption pattern in each of the three soils (see Fig. 1A for non-cumulative data) with no dramatic differences in sorption parameters between the first and second sorption phases, as discussed earlier. This implies that the irreversible binding of DOM did not reduce the mineral surface area of the soils enough to bring about a modification of overall surface properties with respect to sorption reactivity. The calculated sorption capacities ($Q_{max}$) of the soils based on the cumulative isotherms (Fig. 1B) were 8281, 1553 and 1016 mg DOC kg$^{-1}$ for the Akko, Basra, and Nir-Oz soils, respectively. These results suggest that the ultimate sorption capacity of soils may not be correctly calculated from a single sorption cycle (commonly employed methodology). Instead, we would recommend the future use of experimental setups that involve multistage sorption phases (as currently initiated) to assess sorption processes under conditions that better reflect those in the field.

In further investigating the possible effects of the presence of native SOM on DOM sorption, results from the regular sorption experimental setup were compared with those from additional experiments using the control soil samples as sorbents: five sequential steps of aqueous purging (controls) did not bring about any difference in the sorption to Akko or Nir-Oz soils, as compared with sorption to non-purged samples (regular experimental setup). The Basra soil alone responded to this experimental manipulation, with a 28% larger amount of sorbed DOM in the control treatment. Basra soil is unique with respect to its mineral matrix in a few aspects which may have led to the observed effect. First, soil surface coverage by organic matter is significantly higher in Basra than in the other soils, as evidenced by the ~2.8 times higher TOC/SSA ratio in the Basra soil. Second, the proportion of potential metal-oxide sorption sites on the soil surface is highest in the Basra soil, with 3.3 and 4.6 times higher $2\{\text{Fe}_o, \text{Al}_{o}, \text{Mn}_{o}\}$/SSA ratios if compared with the Akko and Nir-Oz soils, respectively (Table 1). Consequently, the washing process in the Basra control treatment might have removed some organic coating, which although constituting only a small proportion of the SOM (4%), may have interfered with DOM sorption to more effective (e.g., metal-oxide) sorption sites which were presumably less accessible in the non-purged samples (Kaiser et al., 1996). The more extensive organic coating of mineral particles in the Basra soil was further expressed by 1.6 times higher amount of native SOM cumulatively released (as DOM) from the Basra soil relative to the other soils during the sequential aqueous purging process (data not presented).

As mentioned, the Basra soil, like the other two soils, exhibited similar sorption patterns in the original and the successive sorption phases despite the presence of organic matter residues in soil in the second sorption phase. Apparently in contrast, the enhanced sorption to the purged control Basra soil may suggest that even a diminutive coverage of mineral surface area by organic coatings may impede the sorption of DOM. The nature of organic association with the mineral matrix may thus be an important factor behind its potential to hinder sorption. The native SOM removed during the purging process of the control samples was doubtlessly more intimately bound to effective sorption sites in comparison with the externally applied DOM in the sorption experiments.

![Fig. 3](image-url)
Desorption following the second sorption phase yielded isotherms which were similar to those generated in the first desorption phase (Fig. 2B). Considering the finding that the irreversible binding of DOM did not modify soil surface reactivity with respect to a further sorption process, it should not have been expected that successive desorption would diverge from the first desorption pattern.

4. Conclusions

The DOM sorption data obtained for a set of alkaline, organic-poor soils supported earlier accounts of a strong positive effect of metal oxide content on sorption affinity and capacity. In contrast, the role of soil texture in DOM sorption could not be established.

4.1. The effect of metal oxides and clay content on DOM sorption to soil

In an effort to resolve the apparent conflict discussed in Section 3.2 concerning the relative roles of metal oxides and clay content in the sorption of DOM to soil, a comprehensive dataset on DOM sorption was constructed from 13 studies, all dealing with sorption to whole soil samples. Two sorption parameters were comparatively assessed and plotted against Fe\textsubscript{d} and clay contents (Fig. 4): Langmuir $Q_{\text{max}}$, representing sorption capacity and the sorbed DOM concentration corresponding to an initially applied concentration of 700 mg C kg\textsuperscript{-1} (Ci\textsubscript{700}), equivalent to the maximal DOM load applied in the current study (70 mg C L\textsuperscript{-1}). Considering that the sorption levels measured in the current study were in the low range of the constructed dataset, conducting the comparison at this level of applied DOM largely restricted the comparison to the linear portion of the isotherms and thus enabled a reasonable evaluation of sorption affinities. In cases where $Q_{\text{max}}$ values were not reported, yet the sorption behavior approached saturation or at least diverged from linearity so as to allow a good fit to the Langmuir model, a non-linear least squares regression was performed to calculate them (Bolster and Hornberger, 2007).

Positive correlations were observed between soil Fe\textsubscript{d} content and DOM-sorption capacity as well as sorption degree at Ci\textsubscript{700} (Fig. 4). A Fe\textsubscript{d}-normalized $Q_{\text{max}}$ term for soils was generalized by averaging all data presented in Fig. 4 (top, left). The obtained value (141,888 mg C kg Fe\textsuperscript{-1}) was found to be 3.3 times higher than the averaged equivalent determined for pure Fe-oxide minerals based on an ample dataset (Chorover and Amistadi, 2001; Gu et al., 1994, 1996; Hur and Schlautman, 2003; Kaiser et al., 1997, 2005; Kang and Xing, 2008; Meier et al., 1999; Sharma et al., 2010; Tipping, 1981; Wang et al., 1997; Zhou et al., 2001). Despite the wide range of sorption capacities measured (stretching over two and three orders of magnitude for soils and pure minerals, respectively), the large difference between the Fe-normalized $Q_{\text{max}}$ terms for soils and pure minerals likely suggests an additional contribution to sorption by other soil mineral constituents such as oxides other than Fe (e.g., Al and Mn) or phyllosilicate minerals. While the DOM-sorption capacity of phyllosilicates such as kaolinite and illite (Jardine et al., 1989; Kaiser and Zech, 2000a) is typically 5% of the average capacity determined for metal oxides (weight-normalized), the proportion of phyllosilicate minerals in the soil matrix may be substantial. While some weak positive correlations can be noted in Fig. 4 (right) between sorption capacity and sorption degree at Ci\textsubscript{700} to soil clay content, they are rather vague. The comprehensive positive correlation between DOM sorption and soil Fe-oxide content supports the common proposition that the characteristically high-energy ligand-exchange complexion of DOM with single-coordinated OH groups on metal-oxide surfaces (Afifi et al., 1995; Kaiser and Guggenberger, 2000; Kleber et al., 2005) is a dominant mechanism for DOM sorption in soils.

![Fig. 4. Parameters of DOM sorption vs. soil-extractable Fe (left) and soil clay content (right) from a set of studies: sorption capacity (top) and sorbed DOM assessed for initial concentration of 700 mg C kg soil\textsuperscript{-1} (bottom). Data were collected from: current study (●); Jardine et al., 1989, 1992 (*); Kable et al., 2003 (Δ); Kaiser and Zech, 1997, 2000a (□); Kothawala et al., 2009 (+); Ling et al., 2006 (-); Nambu and Yonebayashi, 2000 (◊); Navon et al., 2011 (○); Shen, 1999 (—); Ussiri and Johnson, 2004 (▲); Vandenbruwane et al., 2007 (×).](image-url)
4.2. The effect of soil alkalinity on DOM sorption

The alkalinity of the investigated soils did not seem to limit their DOM sorption capabilities relative to less alkaline soils having comparable Fe-oxide contents (Fig. 4, left), as might have been expected based on studies demonstrating a sorption-limiting effect of alkaline soil solution conditions in mineral as well as in soil systems (Grünéwald et al., 2008; jardine et al., 1989; Shen, 1999). While elevated internal repulsion between negatively charged DOM functional groups and negatively charged soil minerals under alkaline conditions may be expected to limit DOM sorptive interactions, cation-bridging between the two compartments can effectively negate such a limiting effect (Grünéwald et al., 2008). Insights as to DOM sorptive interactions in alkaline soil systems are imperative facing the scarcity of information regarding the stabilizing effects of soil minerals on dissolved organic compounds in calcareous soils (Rumpel and Kögél-Knabner, 2011).

4.3. Environmental implications

The fate of DOM in soil is crucial not only from the perspective of soil carbon stabilization but also for the evaluation of contaminants’ reactivity in the soil system. The binding possibilities of anthropogenic compounds adsorbing to the soil system with treated wastewater irrigation or with the application of sewage sludge may diverge between direct interactions at mineral soil surfaces, complexion with soil-bound DOM moieties, and the binding to DOM free in the soil solution. In a previously study (Oren and Chefetz, 2012), we demonstrated that DOM is chemically fractioned during contact (i.e., sorption and desorption phases) with mineral soils. Hence, the magnitude of sorptive interaction of DOM in the soil mineral matrix and the extent of sorption–desorption hysteresis might be important factors controlling contaminants’ chemical associations and ultimate environmental fate.

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