



Redox chemistry of nickel in soils and sediments: A review



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HIGHLIGHTS

- The impact of redox potential (E_H) on mobilization of Ni in soils was reviewed.
- Mobilization of Ni increased under reducing conditions in various soils.
- Oxic conditions can lead to an increase mobilization of Ni in other soils.
- Mobilization of Ni in soils is indirectly affected by E_H -dependent pH changes.
- Dynamics of Ni were controlled mainly by the chemistry of Fe, Mn, Mg, and DOC.

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ABSTRACT

Knowledge on the redox geochemistry of Ni is behind in comparison to other heavy metals. Hence, this article reviews the direct and indirect impact of redox potential (E_H) on mobilization and release dynamics of Ni in soils and sediments across the world. Nickel can show a different behavior in response to E_H . Mobilization of Ni increased at low E_H in various soils; however, oxic conditions can lead to an increased mobilization of Ni in other soils. Those differences occur because the mobilization of Ni is often indirectly affected by E_H , e.g. through E_H -dependent pH changes, co-precipitation with iron (Fe) and manganese (Mn) (hydr)oxides, complexation with soil organic carbon, similar position of Ni and magnesium (Mg) in the soil solid phase, and/or precipitation as sulphides. Dissolved concentrations of Ni showed a similar pattern like Fe and increased at low E_H in many soils, which might be explained by the reductive dissolution of Fe (hydr)oxides and the release of the co-precipitated/sorbed Ni. Few other studies indicated that Ni might be associated with Mn oxides rather than with Fe oxides. Additionally, the formation of soluble complexes with dissolved organic carbon may contribute to a mobilization of Ni at low E_H . Nickel and Mg are similarly affected by redox changes especially in serpentine soils. This review summarizes the recent knowledge about the redox chemistry of Ni and contributes thus to a better understanding of the potential mobilization, hazard, and eco-toxicity of Ni in frequently flooded soils and sediments as agricultural ecosystems.

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1. Introduction

Nickel (Ni) is an essential nutrient for animals and a beneficial element for plants (Kabata-Pendias, 2011). However, as with other trace elements (TEs), elevated Ni concentrations in soils can have negative impact on plants, microorganisms, and animals which increase the risk of Ni in soils (Thakali et al., 2006; Antoniadis et al.,

2017a,b). During the last decades, Ni has become a serious concern as its concentration has reached up to 26,000 mg kg⁻¹ in polluted soils and 0.2 mg L⁻¹ in polluted surface waters; which is 20–30 times higher than found in unpolluted areas (Ma and Hooda, 2010). High contents of Ni in soils as a potentially toxic element (PTE) can cause toxicity and adverse impacts on soil functions as well as considerable environmental problems regarding the mobility and thus the soil-plant transfer and consequent the transfer of Ni into the food chains. Therefore, soil pollution by Ni could soon be a global environmental concern and a significant issue for environmental protection (Kabata Pendias, 2011; Yusuf et al., 2011).

Nickel in soils can be derived from both, parent materials and

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anthropogenic deposits (Cheng et al., 2009, 2011; Starr et al., 2003). Redox conditions play an outstanding role in controlling the behavior of PTEs including Ni in soil environments. Nickel can occur in a number of oxidation states: 1, 2, 3 and 4, but only Ni²⁺ is stable over the wide range of pH and redox conditions found in the soil environment (Ma and Hooda, 2010). Research on Ni redox geochemistry has, however, lagged behind that for other transition elements. Recent studies (e.g., Antić-Mladenović et al., 2017; Rinklebe et al., 2016a) have been dedicated to the relation of Ni to soil redox potential (E_H). Those studies showed a different behavior of Ni in soils as affected by the soil type, changes in E_H/pH, and the factors governing the chemistry of iron (Fe), manganese (Mn), magnesium (Mg), dissolved organic carbon (DOC), and sulfate (SO₄²⁻).

A review about the geochemical behavior of Ni in soils is important to contribute to a better understanding and prediction of the potential mobilization, hazard and eco-toxicity of Ni. Specifically, detailed knowledge about the behavior of Ni in periodically flooded soils is required to understand the underlying mobilization processes and enable a more accurate prediction of Ni release into soil solution, groundwater, and surface water in response to the changes on E_H conditions.

Up to date, only few reviews about Ni are available. For instance, Yusuf et al. (2011) provided a comprehensive review about the uptake of Ni and its essentiality and toxicity in plants. Iyaka (2011) published a short review about the role of Ni as a nutritional TE for animals, microorganisms, and plants. However, to our best knowledge, the redox chemistry of Ni in soils is not reviewed up to date. Therefore, the aim of this review is to address the geochemistry of Ni and the factors governing the release dynamics of Ni in different soils under changing reducing-oxidizing conditions.

2. Total concentrations of Ni in soils and sediments

The concentrations of Ni in soils highly depend on the contents in the soil parent material (Starr et al., 2003; Ma and Hooda, 2010). The mean concentrations of Ni (mg kg⁻¹) in various types of rocks are 2000 in ultramafic igneous, 140 in basaltic igneous, 68 in shales and clays, 50 in black shales, 20 in limestone, 8 in granitic igneous, and 2 in sandstone (Ma and Hooda, 2010). However, the content of Ni in surface soils reflects the additional impact of both soil-forming processes and anthropogenic activities. Soils and sediments throughout the world contain Ni in an enormous range (Table 1). The total Ni contents in uncontaminated soils range between 13 and 40 mg kg⁻¹ (Ma and Hooda, 2010; Kabata-Pendias, 2011). The content of total Ni in many arable soils seldom exceeds 50 mg kg⁻¹, but in ultramafic bedrock, such as serpentine or peridotite, naturally enriched with Ni, it can reach more than 10,000 mg kg⁻¹ (Cheng et al., 2009; Antić-Mladenović et al., 2011; Ho et al., 2013; Hseu et al., 2017). The highest contamination by Ni, up to 26,000 mg kg⁻¹, was reported for topsoils near the Ni–Cu smelter at Sudbury, Canada (Cox and Hutchinson, 1981). Elevated concentrations of Ni in soils have negative impact on plants and microorganisms (Thakali et al., 2006; Yusuf et al., 2011; Ličina et al., 2013).

The total concentrations of Ni in different sediments varied widely from values lower than the detection limits to 5200 mg kg⁻¹ (Table 1). The highest Ni concentrations (500–5200 mg kg⁻¹) in sediments were reported by Noël et al. (2015) in mangrove sediments in New Caledonia (Table 1). The highest Ni concentrations measured by Noël et al. (2015) are similar to that in other mangrove sediments from New Caledonia (Marchand et al., 2012), but are higher than the background level reported for mangrove sediments in other parts of the world (Lewis et al., 2011). Enrichments of Ni in

Table 1

Total concentrations of Ni in different soils and sediments originate from different countries.

Values [mg kg ⁻¹]	Country	References
Soils		
450	China	Quiiping et al., 1984
21–132	Germany	Frohne et al., 2014
3240	Italy	Bini et al., 1988
6.2–136.9	Norway	Almas et al., 1995
1243	Albania	Shtiza et al., 2005
20–100	Siberia	Niechayeva, 2002
28–34	Russia	Protasova and Kopayeva, 1985
9–17	Lithuania	Kadunas et al., 1999
60	Finland	Koljonen, 1992
10.6–21.0	Argentina	Roca et al., 2008
2.3–186.0	Spain	Bech et al., 2008
18.0–78.8	Germany	Rinklebe and Shaheen, 2014
80.9	Germany	Shaheen et al., 2015b
119	Canada	Shaheen and Rinklebe, 2015
–26,000		Cox and Hutchinson, 1981
		Temple and Bisessar, 1979
		Frank et al., 1976
74.3	Egypt	Shaheen et al., 2015a
236.4	Greece	Shaheen et al., 2015a
29.0	Germany	Shaheen et al., 2015a
12.7–79.6	Egypt	Shaheen et al., 2015a
5.2–105.4	Germany	Shaheen et al., 2015a
63.0–65.0	Egypt	Shaheen et al., 2014a
72–112	Germany	Rennert and Rinklebe, 2010
550	Serbia	Antić-Mladenović et al., 2011
143–160	Serbia	Antić-Mladenović et al., 2017
70	Serbia	Rinklebe et al., 2016a
71	Germany	Shaheen et al., 2014b
770	Australia	Severne, 1974
1700–5000	New Zealand	Lyon et al., 1970
3563–7375	Rhodesia	Nielsen et al., 1977
2–1150	Great Britain	Grove and Ellis, 1980
304–9288	Russia	Barcan and Kovnatsky, 1998
31–101	Holland	Hemkes et al., 1980
64.5–277.2	Iran	Abbaslou et al., 2014
32.8	Iran	Khanlari and Jalali, 2008
1314–4048	Taiwan	Cheng et al., 2011
391.2	Taiwan	Cheng et al., 2009
–5765		
3400	Taiwan	Hsiao et al., 2007
98.0	Germany	Hindersmann and Mansfeldt, 2014
100–2000	Maryland, United States	Rabenhorst et al., 1982
100–4500	Oregon, United States	Burt et al., 2001; Alexander et al., 2007
100–3900	California, United States	Lee et al., 2004; Oze et al., 2008
200–1700	British Columbia, Canada	Bulmer and Lavkulich, 1994
3200–7200	Santa Elena, Costa Rica	Reeves et al., 2007
2300–6000	Niquela'ndia, Brazil	Garnier et al., 2006
300–1100	Aosta Valley, Italy	D'Amico et al., 2008
100–1600	Trás-os-Montes, Portugal	Diez Lázaro et al., 2006
1300–4100	Szkлары Massif, Poland	Kierczak et al., 2007, 2008
200–3600	France	Massoura et al., 2006
800–1200	Harsin and Khoy, Iran	Ghaderian et al., 2007
7000–8500	New Caledonia	Becquer et al., 2006
6567	Wasgamuwa, Sri Lanka	Herath et al., 2015
Sediments		
32.5–56.1	Chittagong, Bangladesh	Wang et al., 2016
15.6–83.1	Southern Poland	Szarek-Gwiazda et al., 2011
20.8–64.5	Arvand River, Persian Gulf, Iran	Sarasiab et al., 2014
37–163	Korotoa (Bangladesh)	Islam et al., 2015
200	Buriganga River (Bangladesh)	Ahmad et al., 2010
28	Padma River (Bangladesh)	Datta and Subramanian, 1998
33	Jamuna River (Bangladesh)	Datta and Subramanian, 1998
26	Bangshi River (Bangladesh)	Rahman et al., 2014
15.7	Gomti River (India)	Singh et al., 2005
101–129	Gediz River (Turkey)	Akcay et al., 2003
1.21	Okumeshi River (Nigeria)	Raphael et al., 2011
56–244	Buriganga River (Bangladesh)	Mohiuddin et al., 2011
16–43	Juru River, Penang, Malaysia	Idriss and Ahmad, 2013

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