



## Review

## Does control of soil erosion inhibit aquatic eutrophication?

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## ABSTRACT

Much of the phosphorus (P) from erosive soils is transported to water bodies together with eroded soil. Studies clarifying the impact of soil erosion on eutrophication have sought largely to quantify the reserves of P in soil particles that can be desorbed in different types of receiving waters. Aquatic microbiology has revealed that the cycling of P is coupled to the availability of common electron acceptors, Fe oxides and SO<sub>4</sub>, through anaerobic mineralization in sediments. Eroded soil is also rich in Fe oxides, and their effect on the coupled cycling of C, Fe, S, and P has been neglected in eutrophication research. Soil erosion, and its control, should therefore be studied by considering not only the processes occurring in the water phase but also those taking place after the soil particles have settled to the bottom. We propose that in SO<sub>4</sub>-rich systems, Fe oxides transported by eroded soil may promote Fe cycling, inhibit microbial SO<sub>4</sub> reduction and maintain the ability of sediment to retain P. We discuss the mechanisms through which eroded soil may affect benthic mineralization processes and the manner in which soil erosion may contribute to or counteract eutrophication.

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## 1. Introduction

Soil erosion and aquatic eutrophication are two global environmental problems that affect human well-being in many ways (Diaz and Rosenberg, 2008; Lal, 1998; Nixon, 1995; Verstraeten et al., 2006). They are considered to be linked, since primary production in waters is assumed to be enhanced by the phosphorus (P) carried away by eroded soil from P-amended agricultural land (Carpenter, 2005; Dorich et al., 1980; House et al., 1998; Pote et al., 1996; Rekolainen et al., 2006; Sharpley et al., 1991). The theoretical justification for this assumption is based on soil science. According to the specific ligand exchange theory (Hingston et al., 1967), P bound by Fe and Al oxides is in dynamic 'equilibrium' with P in solution. While in contact with dilute rain or snow melt-water on the surface of a field, during transport in ditches and streams, and finally in P-deficient bodies of water, soil particles gradually lose P by its desorption of some of the oxide-bound reserves (Carritt and Goodgal, 1954; Froelich, 1988; Hartikainen et al., 2010; Yli-Halla et al., 1995).

Soil erosion can be caused both by natural processes and by human activities such as forestry and construction. Agriculture carries both the greatest risk for soil erosion and the greatest potential for reducing P loads related to erosion. To reduce eutrophication,

water protection measures have been introduced with a view to preventing soil detachment and transport from cultivated fields. Farmers worldwide are advised to favor reduced tillage, contour cropping, and cultivation of cover crops, to establish buffer strips and riparian zones, and to construct settling ponds and wetlands, partly at least with a view to reducing P losses (Boardman et al., 2009; Carpenter, 2005; Ekholm et al., 2007; Schoumans et al., 2011).

Aquatic microbiology has shown that the cycling of P is closely linked to the availability of common electron acceptors, Fe oxides and SO<sub>4</sub>, through the anaerobic mineralization reactions taking place in sediments (Burgin et al., 2011). Further research is needed on aquatic P cycling regarding not only the processes occurring in the water phase (desorption) but also those taking place after the soil particles have settled on the sediment surface. Bottom sediments play a crucial role in aquatic nutrient cycling, as, among other things, they 'buffer' the systems against changes in nutrient loading by immobilizing P (Boström et al., 1988). With an increase in eutrophication, the buffering ability tends to weaken, thus contributing to a regime shift or even to a 'point of no-return' in the entire ecosystem (Genkai-Kato, 2007; Scheffer et al., 2001). Accelerated benthic release of P is part of the vicious circle in which the system feeds itself and exacerbates the symptoms of eutrophication in many water bodies, e.g., the Baltic Sea (Conley et al., 2002; Pitkänen et al., 2001; Vahtera et al., 2007).

Even if O<sub>2</sub> is present at the sediment–water interface, only the top few millimeters of aquatic sediments are aerobic (Jørgensen

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and Revsbech, 1985). This layer is important for the oxidation of organic matter and the end products of anaerobic mineralization, such as the reduced species of Fe and S. However, sediment is largely anaerobic, and P cycling is primarily governed not by ligand exchange reactions but by the bulk mineralization rate due to the partial or complete reduction of Fe oxides (Anschutz et al., 1998; Canfield et al., 2005; Lovley, 1991; Severmann et al., 2010; Thamdrup et al., 1994) and to the mineralization of organic P. The pathway by which Fe oxides are reduced crucially affects P processing. Microbial reduction of Fe oxides does not necessarily lead to massive benthic P fluxes, assuming re-oxidation of Fe, whereas benthic P release is radically accelerated with the onset of SO<sub>4</sub> reduction and subsequent chemical Fe oxide reduction (Roden and Edmonds, 1997). However, SO<sub>4</sub> reduction is inhibited by the presence of Fe oxides (Lovley, 1991). On the basis of the recent literature on sediment microbial processes and element cycling, we propose that soil transported from terrestrial to aquatic systems may maintain 'favorable' (Fe reduction) and suppress 'adverse' (SO<sub>4</sub> reduction) microbial sediment processes. While losses of desorbable P are admittedly reduced with soil erosion, fluxes of Fe oxides are also lowered, which may thus increase the release of P settled on sediments. The overall effect on primary production depends on which is more important: a decrease in desorbable P with the flux of eroded soil or an increase in benthic P release due to the concomitant lower flux of Fe oxides.

In this article, we review the transport of Fe and P from terrestrial to aquatic systems and examine the role of benthic mineralization in P cycling in an effort to describe the potential role of eroded soil in eutrophication. Our point of departure is the processing of P in the bottom sediments of SO<sub>4</sub>-rich systems. Although we use eutrophication as an example, eroded soil may also have an unprecedented impact on other environmental problems, with subsequent economic and social implications.

## 2. From fields to aquatic sediments: soil as a carrier of Fe and P

The detachment of soil, total Fe, and total P from terrestrial systems and their transport to an aquatic environment is a relatively well-known process. The average concentration of Fe in surface soil is 3.5% (Chesworth, 1991), being a function of soil characteristics. In the post-glacial soils of Finland, the Fe concentration ranges from 1.6% to 7.1% of soil dry matter: the finer the soil texture the higher the concentration (Table 1). In the older and more weathered soils in the Guadalquivir Valley (SW Spain), the

concentration is on average only 0.9%. The Fe concentration in riverine suspended solids reflects that in the parent soil, as possibly modified by the selective erosion of, for instance, fine particles (Saavedra and Delgado, 2005, Table 1). According to Canfield et al. (2005), about half of the Fe in riverine particulates consists of largely inert Fe in silicates, the remaining Fe being in possibly reducible forms, such as in oxides.

Oxides, hydroxides and oxide–hydroxides of Fe(III) (referred to here as Fe oxides) are ubiquitous in the environment, with concentrations ranging from one to several hundred g kg<sup>−1</sup> in aerobic soils (Cornell and Schwertmann, 1996). Natural Fe oxides exhibit variable mineralogy, crystallinity, grain size, and impurity content (Carlson, 1982; Schwertmann, 1991). This heterogeneity affects their availability to Fe-reducing microbes and also their chemical reactivity (Hyacinthe et al., 2006). In older soils in a warm and dry climate, Fe oxides tend to be present in crystalline forms but, under boreal conditions (young soils, cold and wet climate), poorly crystalline forms dominate. It is the poorly crystalline forms (also called short-range ordered oxides, sesquioxides, and amorphous oxides) that have a particularly high capacity to adsorb P at about neutral pH (Borggaard et al., 1990) and that are microbially most readily available (Roden and Zachara, 1996). Organic C (i.e., aliphatic and aromatic carboxyl and hydroxyl functional groups) can also bind Fe oxides (Pullin and Cabaniss, 2003) and act as a carrier of Fe from terrestrial to aquatic systems.

Whereas both Fe and P are naturally present in soils, application of chemical fertilizers to fields has increased the reserves of soil P. Globally, the accumulation of P in agricultural soils is about 11.5 Tg year<sup>−1</sup>, and has more than doubled during the past 40 years (Bennett et al., 2001). The build-up of P in soil is made possible by the high ability of Fe and Al oxides to capture P by specific ligand exchange reactions (Borggaard et al., 2004; Hingston et al., 1967; Parfitt et al., 1975). The erosion process transports P-rich surface soil to water bodies, in which some of the oxide-bound P can be released due to changes in, say, the ionic composition of the surrounding solution (e.g., Martin et al., 2004).

The global river export of suspended solids to the ocean is 13–19 Pg year<sup>−1</sup> (Beusen et al., 2005; Syvitski et al., 2005). Most of the solids settle out, becoming bottom sediments in estuarine and coastal regions (Gibbs, 1981). In turn, global river fluxes of total and sulfide-reactive Fe (e.g., Fe oxides) are about 960 Tg and 270–420 Tg year<sup>−1</sup>, respectively, mainly in particulate form and also largely trapped by estuaries (Poulton and Raiswell, 2002; Raiswell, 2006). Finally, rivers carry about 18–30 Tg year<sup>−1</sup> of P to the ocean (Carpenter, 2005; Compton et al., 2000). More than 90% of this flux consists of particulate P, of which 25–45% has been estimated to be 'reactive' (Ruttenberg, 2003).

Particulate P losses from arable land are spatially highly variable and affected by tillage depth and intensity, plant cover, soil texture and quality, slope, and hydrology (e.g., McDowell and McGregor, 1984; Puustinen et al., 2005, 2007). Particulate P tends to be the dominant P form in runoff from fine-textured soils without a permanent plant cover (Ulén and Kalisky, 2005). In the Nordic countries, particulate P can contribute up to 93% of the total P in agricultural runoff (Ulén et al., 2007). To be utilized by algae, P has to be released from particles into dissolved orthophosphate, the only P form ubiquitously taken up by algae (Cembella et al., 1984; Ekholm et al., 2005). Algal-assays showed that in eroding river banks only <1–13% of P was available (Ellis and Stanford, 1988) and that in eroding lake banks no P was available (Young et al., 1985). By contrast, the mean availability of particulate P in agricultural rivers ranged from 5% (Ekholm, 1994) to 41% (Persson, 1990), the majority of the results falling in the range of 20–30% (DePinto et al., 1981; Dorich et al., 1980; Källqvist and Berge, 1990; Williams et al., 1980). Thus, particulate P from cultivated soils appears to be

**Table 1**  
Concentrations of Fe and P in soil and riverine suspended solids.

Sample type	Area	Total Fe	Total P
		% dry matter	
Soil	Global mean	3.5 <sup>a</sup>	0.11 <sup>b</sup>
	Finland (young soils)	1.6–7.1 <sup>c</sup>	0.12 <sup>d</sup>
	Spain (old soils)	0.9 <sup>e</sup>	0.046 <sup>e</sup>
Riverine suspended solids	Global mean	4.8 <sup>f</sup>	0.12 <sup>f</sup>
	USA	2.7–7.3 <sup>g</sup>	
	Finland	6.1 <sup>h</sup>	0.19 <sup>i</sup>
	Spain	3.3 <sup>e</sup>	0.17 <sup>e</sup>

<sup>a</sup> Chesworth (1991).

<sup>b</sup> Compton et al. (2000).

<sup>c</sup> HF–HClO<sub>4</sub> extraction (Sippola, 1974).

<sup>d</sup> Yli-Halla et al. (2005).

<sup>e</sup> Saavedra and Delgado (2005).

<sup>f</sup> Martin and Meybeck (1979).

<sup>g</sup> Canfield (1997).

<sup>h</sup> K<sub>2</sub>S<sub>2</sub>O<sub>8</sub> digestion (Mäkiäho, 2007).

<sup>i</sup> Data base of the Finnish Environment Institute.

more readily available than particulate P from pristine areas, although synthesis is problematic due to the considerable methodological diversity among the bioavailability tests (Rekolainen et al., 1997).

If 20–30% of particulate P becomes available in receiving waters, losses of bioavailable particulate P may be at a similar level to, or even exceed, those of the readily available dissolved reactive P from cultivated fields (Uusitalo, 2004). For example, the mean losses of particulate P in the surface runoff from a plowed clayey field in Finland were  $3.7 \text{ kg ha}^{-1} \text{ year}^{-1}$  (Puustinen et al., 2005). Algal-assays and chemical extractions with anion exchange resin suggested that 16% of the particulate P was available under aerobic conditions (Uusitalo and Ekholm, 2003), which translates to a mean loss of  $0.59 \text{ kg ha}^{-1} \text{ year}^{-1}$  of algal-available P. This value is similar to that for the losses of dissolved reactive P from the field, i.e.,  $0.58 \text{ kg ha}^{-1} \text{ year}^{-1}$  (Puustinen et al., 2005). On the basis of such calculations, current erosion control measures seem feasible, unless the losses of dissolved reactive P are not increased by practices such as reduced tillage (McDowell and McGregor, 1984; Muukkonen et al., 2009; Ulén and Kalisky, 2005). The situation may, however, be reversed if we also take into account the reactions taking place after the soil particles have settled on the bottom.

### 3. Sediments make all the difference

Phosphorus cycling being a crucial limiting element in aquatic systems, the existing literature on P cycling understandably concentrates on the link between soil erosion and P transport, or P bioavailability. However, after the soil particles have found their way to bottom of a water body, they will eventually face an anoxic environment. These conditions have a drastic impact on Fe cycling and thus also crucially affect the fate of P. Our novel hypothesis links erosion to the pathway by which organic C is mineralized in sediments, i.e., by providing Fe oxides, eroded soil may shift the sediment microbial processes favorable for P binding.

In sediments, Fe oxides are reduced by two mechanisms: (1) via microbial dissimilatory Fe reduction, in which microbes use the oxides as terminal electron acceptors in respiration, and (2) chemically by sulfides ( $\text{H}_2\text{S}$ ,  $\text{HS}^-$ ) formed in microbial  $\text{SO}_4$  reduction (Canfield et al., 2005; Lovley, 1991). In both mechanisms, organic C provides the primary source of energy to reduce the oxides. When the flux of organic C to the sediment is low and the sediment is rich in Fe oxides, Fe reduction is likely to dominate in surface sediments. However, when the availability of organic C increases due to eutrophication, Fe reduction gives way to  $\text{SO}_4$  reduction. Such a change in sediment microbiological processes may contribute to a regime shift in the entire ecosystem: the ability of sediments to retain P being drastically reduced by the onset of  $\text{SO}_4$  reduction (Hyacinthe and Van Cappellen, 2004; Lehtoranta et al., 2009; Roden and Edmonds, 1997; Smolders and Roelofs, 1993; Sugawara et al., 1957). As previously stated, the availability of Fe oxides inhibits  $\text{SO}_4$  reduction. Quantitatively, terrestrial sources are the most important carriers of Fe oxides to marine waters (Raiswell, 2006). Yet, the role of soil erosion in providing Fe oxides for benthic mineralization, and thereby potentially lowering benthic P fluxes, has not been investigated.

The finding that greater primary production and high concentrations of  $\text{SO}_4$  increase P mobilization is not a new one (see Hasler and Einsele, 1948), but the mechanism has only recently been elucidated. Microbial and chemical reductions of Fe have crucially different consequences for Fe cycling. When Fe oxides are directly reduced by microbial dissimilatory reactions, the highly soluble Fe(II) produced is able to diffuse upward in the pore water, and will ultimately be oxidized by, say,  $\text{O}_2$  or  $\text{NO}_3^-$  (van der Zee et al., 2005). As a result, a newly precipitated layer of Fe oxides will be formed,

this layer being able to capture P also diffusing upward in the sediment. In this way, an Fe atom can be re-used in microbial mineralization reactions up to hundreds of times, forming “a ferrous wheel” (Burgin et al., 2011), provided that the bottom fauna mixes the top sediments (Canfield et al., 2005). Thanks to the “ferrous wheel”, Fe reduction may form a significant part of bulk anaerobic respiration, and thereby lower the labile C pool for  $\text{SO}_4$  reduction and  $\text{CH}_4$  formation.

Microbes can only partly reduce crystalline Fe oxides (Roden and Urrutia, 2002), whereas sulfides eventually reduce all Fe oxides (Canfield, 1989) and, what is more important, form insoluble Fe sulfides (Thamdrup et al., 1994). As a consequence, Fe will be buried in a solid form in sediment, which prevents the upward diffusion of Fe and the formation of the Fe oxide layer on the sediment surface. Now, P can be freely transported into the water column, and little Fe-bound P will be buried. Such a situation prevails in the eutrophic sub-basins of the Baltic Sea, which exhibit high benthic release of P (Lehtoranta, 2003; Pitkänen et al., 2001; Vahtera et al., 2007).

### 4. Outline of the possible effects of eroded soil on eutrophication

The following section summarizes the proposed effects of eroded soil on eutrophication in an estuary (or any recipient rich in  $\text{SO}_4$ ). If the estuary receives a high riverine input of Fe oxides and a modest input of dissolved P, it may show only a moderate level of eutrophication (Fig. 1A). The settling flux has a low C/Fe ratio and the sediments are mainly in an Fe reduction state. As a result, the estuary may respond toward load reduction measures, because the P concentrations are not maintained by strong P release from sediments due to the coupled cycling of P and Fe. Benthic fauna is exuberant and keeps the “ferrous wheel” turning.

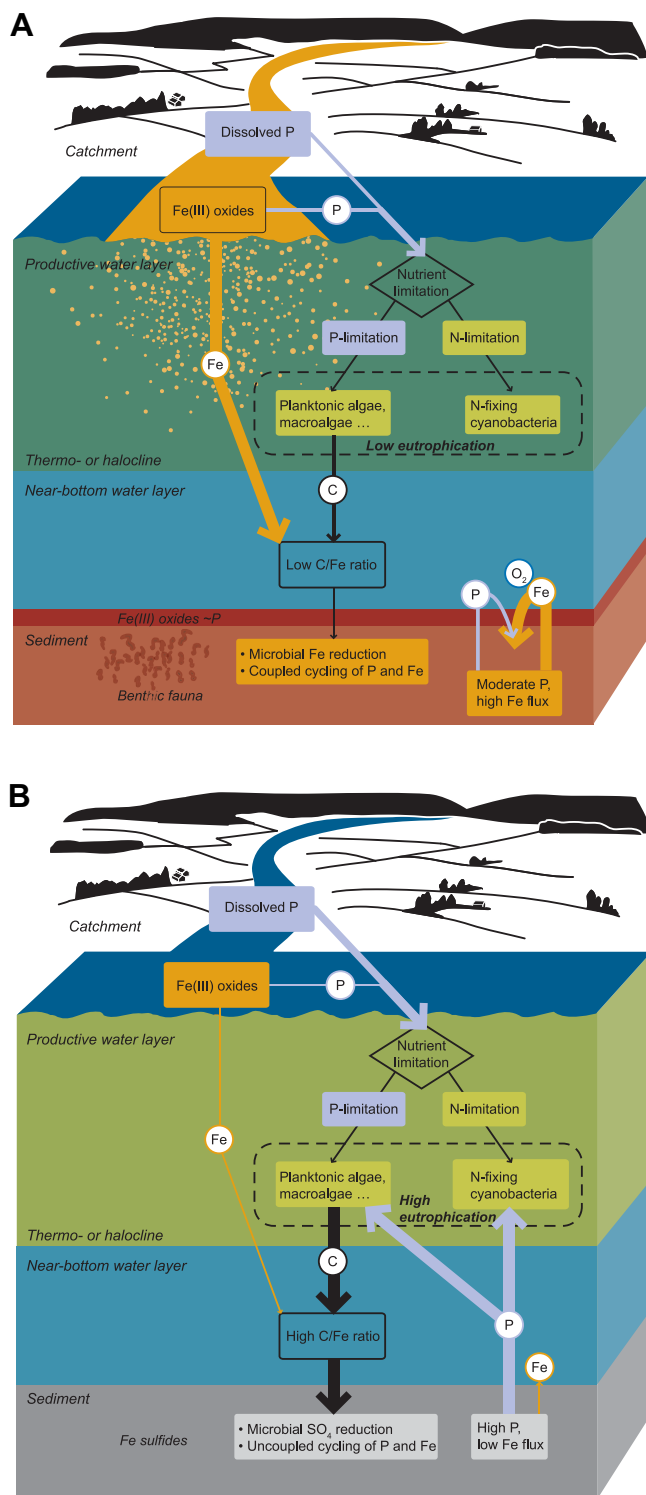
Let us assume then that erosion control measures are implemented in the catchment of the estuary. While the input of Fe oxides is lowered with soil flux, that of dissolved P is increased, a common outcome of applying reduced tillage to arable fields (McDowell and McGregor, 1984; Muukkonen et al., 2007; Puustinen et al., 2005; Ulen et al., 2010). The readily bioavailable P triggers algal production and increases the flux of autochthonous organic C to the sediment surface. As C inputs increase,  $\text{SO}_4$  reduction becomes more important (Howarth et al., 2011). Its importance is also promoted by the lowered flux of Fe oxides. When  $\text{SO}_4$  reduction rates increase, more of the sediment Fe is transformed to non-sorptive Fe sulfides (Roden and Edmonds, 1997), leading to massive release of Fe-bound P stored in sediments (Fig. 1B). The estuary is now highly eutrophic. It may show extensive cyanobacterial blooms, and the sediment is largely devoid of benthic fauna. This poor state can be difficult to reverse, as it requires a shift in sediment microbial processes in addition to successful load reduction.

If P increases primary production and the settling flux of organic C, the effect of this flux on sediment processes depends on the hydromorphology and chemistry of the receiving body of water. For example, in deep systems in which organic C is largely mineralized already in the water phase, the effect of benthic processes in nutrient cycling may be negligible. Bathymetry is also important in that it affects sediment accumulation areas. Finally, any barriers to  $\text{O}_2$  transport to near-bottom waters, such as the thermocline and halocline, affect the sediment state, for instance, by affecting mineralization and preventing the re-oxidization of Fe(II) (Reed et al., 2011).

Obviously,  $\text{SO}_4$  abundance is a crucial characteristic of the systems. In  $\text{SO}_4$ -poor waters (e.g., boreal lakes),  $\text{SO}_4$  reduction may play only a minor role. In  $\text{SO}_4$ -poor P-limited systems, one would

expect desorbable P in soil particles, in addition to the P already in a dissolved form, to govern primary production. Defining a  $\text{SO}_4$ -poor system is, however, problematic, as sulfide layers can even be found in the sediments of boreal freshwater lakes (Salonen et al., 1993).

If eroded soil improves the ability of sediments to retain P by promoting Fe reduction and subsequent coupled Fe and P cycling,



**Fig. 1.** Proposed effect of eroded soil on eutrophication in an estuary or any other  $\text{SO}_4$ -rich system. (A) An estuary receiving a high amount of Fe oxides, (B) An estuary receiving a low amount of Fe oxides.

the net effect of soil erosion on eutrophication depends on the balance between (1) labile soil P that can support algal primary production and (2) the lowering of benthic P release caused by Fe oxides. Evidently, the balance is site-specific. We can, however, approximate it by determining the rate of Fe that is available for Fe-reducing microbes and algal-available P in the incoming fluxes. Alternatively, we can estimate the “ultimately” available P by chemical extractions as done by Uusitalo and Turtola (2003) and Uusitalo et al. (2003). They found that dithionite extracted 34–58% of particulate P in field runoff and interpreted dithionite-extractable P as approximating the P that could be released in sediments after reduction of all Fe oxides. In this respect, dithionite extraction would mimic the conditions prevailing when  $\text{SO}_4$  reduction is the dominant respiration pathway in sediments. However, if surface sediments are in a microbial Fe reduction state (perhaps as a result of soil erosion), the above interpretation may not be valid, overestimating the amount of ultimately available P.

## 5. Precautions and research needs

As already discussed, Fe oxides play a dual role in the eutrophication process, as not only do they sequester P but they also provide electron acceptors in mineralization. We conclude that research on P should be widened to cover the role of electron acceptors (Fe,  $\text{SO}_4$ ) and labile C on the biogeochemical processing of P. Specific research topics here include the determination of critical  $\text{SO}_4$  concentrations in receiving waters, and the behavior of different Fe forms transported to sediments. In addition, we should study how different human actions, such as agricultural practices, affect the transport of Fe. To predict the role of Fe oxides in the overall P cycling in different types of receiving waters, we should be able to quantify four Fe forms in soils, riverine suspended solids, and bottom sediments:

- (1) Fe responsible for binding and release of P.
- (2) Fe available for microbial Fe reduction.
- (3) Fe reacting with sulfides formed in  $\text{SO}_4$  reduction.
- (4) Residual Fe inert toward both microbial Fe reduction and sulfides.

Poorly crystalline Fe oxides are thought to have the highest ability to bind P, although the potential role of solid forms of Fe(II), such as vivianite and hydroxides, in P cycling needs to be clarified (Bruun Hansen and Fiil Poulsen, 1999; Patrick and Khalid, 1974). Further, the poorly crystalline Fe oxides are generally more readily available for Fe reducers than are crystalline oxides. Yet, under optimal laboratory conditions, *Shewanella putrefaciens*, a facultative anaerobe, has been reported to reduce goethite, hematite, magnetite, and even structural Fe in clay minerals (see Glasauer et al., 2003). Under more stringent natural conditions, Fe reduction has been closely correlated only with the presence of poorly crystalline Fe oxides (see Glasauer et al., 2003). Phosphorus bound to Fe oxides may lower the susceptibility to microbial reduction (Borch et al., 2007; Glasauer et al., 2003).

Insight into the effect of eroded soil could be gained by systematic laboratory studies. The net ecosystem effect could be examined by looking at chemical and biological monitoring data from catchment–recipient pairs differing in, say, land-use, soil, water chemistry, and basin morphometry, and representing an extensive geographic area. We began to evolve the theory outlined here from our observations on two contrasting Finnish estuaries in the Baltic Sea. The estuary of the Paimionjoki, which receives a high amount of eroded soil from its intensively cultivated catchment, showed high concentrations of dissolved Fe (the end product of microbial Fe reduction) and low concentrations of dissolved P in the



sediment pore water. The opposite, indicating the presence of SO<sub>4</sub> reduction, was found in the sediments of the Kymijoki estuary, which is less affected by land-derived solids (see Lehtoranta et al., 2005; Lukkari et al., 2008). The latter estuary was more eutrophic, although the level of land-based P flux was similar in both estuaries.

Studies should also be conducted to establish the proportion of organic C that is mineralized via Fe reduction and SO<sub>4</sub> reduction under different conditions. The potential role of eroded soil itself is also interesting: as clay minerals may be able to preserve organic C, they may lower mineralization activity and the risk of SO<sub>4</sub> reduction in surface sediments (Arnarson and Keil, 2001; Ding and Henrichs, 2002; Hedges, 1992; Mayer, 1994). In this respect, we need to know whether the clay transported from fields to waters still has the capacity to 'fix' C. Finally, eroded soil may increase the settling of algae (Avnimelech et al., 1982) and, by rendering water turbid, reduce light penetration and primary production (Grobbelaar, 1985).

## 6. Concluding remarks

Erosion control measures are applied all around the world. In many cases the motivation is to protect the soil itself, whereas in others it is to reduce the off-site impact of erosion (Boardman and Poesen, 2006). The off-site impact covers esthetic issues, higher costs of water treatment and detrimental effects on fish and other aquatic biota (Bilotta and Brazier, 2008; Sutela and Vehanen, 2009). There appears to be no information on the extent to which voluntary, subsidized or compulsory erosion control measures are performed specifically to reduce eutrophication. Yet, judging by the large number of research articles on the P in eroded soil (e.g., DePinto et al., 1981; Persson, 1990; Sharpley et al., 1991; Uusitalo et al., 2000; Williams et al., 1980), and advisory texts addressing the issue (e.g., Schoumans et al., 2011; SERA-17, 2011), abating eutrophication appears to be a major impetus for erosion control measures.

Does controlling soil erosion inhibit aquatic eutrophication? We conclude that the link between erosion and aquatic eutrophication is more complex than previously thought and thus needs to be examined from a wider perspective than merely accounting for the loading and bioavailability of soil-bound P. Our proposition that soil erosion may play a role in aquatic eutrophication is an attempt to combine the relatively isolated research fields of soil chemistry and biogeochemical processes driven by microbes in aquatic sediments. It is analogous to lake restoration with Fe addition. Here, too, the other role of Fe, i.e., the effect on mineralization pathways and thereby on coupled C, S, and P cycling, has received little attention.

The proposition appears to be theoretically sound, but further studies are naturally needed to corroborate or falsify the hypothesized mechanisms through which eroded soil may counteract eutrophication by affecting cycles of Fe, S, and C in recipient sediments. It would seem that the quality and quantity of eroded soil as well as the characteristics of the receiving water body dictate the outcome of erosion control measures on the trophic status of the system. Given the large variation in terrestrial systems (e.g., landscapes, soils, agricultural practices) and aquatic systems (e.g., chemical composition, morphometry), site-specific factors should be taken into account when estimating the extent of the conditions under which soil erosion may contribute to or counteract eutrophication. Note also that other benefits of erosion control may outweigh the costs, even in cases when eutrophication targets are not met.

Climate change may further increase erosion, depending on trends in precipitation and temperature. Like the growing population and changing diet preferences, climate change, too, may

create pressure to clear new agricultural land. Not only agriculture, but also construction and forestry practices increase fluxes of soil to water. In such cases, however, the eroded soil may contain less labile P, and so should have a less marked eutrophying effect than soil transported from P-amended fields. By contrast, other human actions, such as dam and reservoir construction, or increasing the sealed surface area, reduce the flux of suspended solids (Beusen et al., 2005; Syvitski et al., 2005).

By affecting anaerobic microbial processes, eroded soil may have an impact on other environmental problems besides eutrophication. Erosion as such may increase both the emission and sequestration of C (Quinton et al., 2010; Van Oost et al., 2007), but the role of eroded matter on benthic mineralization and on greenhouse gas fluxes from sediments appears not to have been addressed. The mobilization of some toxic substances, e.g., as (McArthur et al., 2004), is also governed by the reductive dissolution of Fe oxides, and here, too, eroded soil may have a role.

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