

# Mitigation of Diffuse Phosphorus Pollution during Rewetting of Fen Peat Soils: A Trans-European Case Study

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**Abstract** Intensive cultivation of fen peat soils (Eutric Histosols) for agricultural purposes, started in Europe about 250 years ago, resulting in decreased soil fertility, increased oxidation of peat and corresponding CO<sub>2</sub>-emissions to the atmosphere, nutrient transfer to aquatic ecosystems and losses in the total area of the former native wetlands. To prevent these negative environmental effects set-aside programs and rewetting measures were promoted in recent years. Literature results and practical experiences showed that large scale rewetting of intensively used agricultural Histosols may result in the mobi-

lisation of phosphorus (P), its transport to adjacent surface waters and an accelerated eutrophication risk. The paper summarises results from an international European Community sponsored research project and demonstrates how results obtained at different scales and from different scientific disciplines were compiled to derive a strategy to carry out rewetting measures. A decision support system (DSS) for a hydrologically sensitive area in the Droemling catchment in north-eastern Germany was developed and is presented as a tool to regulate rewetting in order to control P release. It is demonstrated that additional

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laboratory experiments to identify essential processes of P release during rewetting and the site-specific management of the water table, the involvement of specific knowledge and experience of the stakeholders are necessary to develop an applicable DSS. The presented DSS is practically used to prevent freshwater resources from diffuse P pollution.

**Keywords** Agricultural watershed · Land use change · Assessment · Measurement · Phosphorus · Histosol

## 1 Introduction

Histosols (named according to incomplete decomposition of primary organic matter) cover an area of 209,805 km<sup>2</sup> in the European Communities (EC) which accounts for about 5.25% of the total EC area. The largest areas of Eutric Histosols in European countries are in Poland (12,110 km<sup>2</sup>), Germany (10,500 km<sup>2</sup>), Belgium and The Netherlands (2,590 km<sup>2</sup>), Ireland (2,275 km<sup>2</sup>) and the UK (1,280 km<sup>2</sup>) (Commission of the EC 1985). Eutric Histosols (synonym “fens”), recharged by groundwater, are the dominant type of peat soils in the Pleistocene landscape of north-eastern (NE) Germany as well as in large parts of Poland and the Baltic countries (Zak et al. 2004). Global experience shows that intensive agricultural use of Histosols results in decreasing soil fertility, oxidation of peat and corresponding CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O emissions to the atmosphere, nutrient transfer to aquatic ecosystems and losses in the total area of these native wetlands (Broll et al. 2002). In Europe the cultivation of these fens, lasting for about 250 years, had also resulted in obvious peat degradation and oxidation, and overall loss of ecological value and functions. One of the geocological functions is the filtration of freshwater as the fens generally form buffer zones between the glacial mineral soils and adjacent surface freshwater resources. In this respect the biogeochemistry of phosphorus (P) is of particular importance as P is the limiting nutrient for algae blooming in aquatic environments (Gibson 1997; Sharpley and Rekolainen 1997; Edwards and Chambers 2002). Set-aside programs and rewetting measures were promoted in several countries to prevent the overall negative environmental effects of fen peat cultivation and restore some of the previously degraded fens. However, possible

effects of setting-aside, rewetting and restoration measures on the biogeochemistry of P in fens and adjacent freshwater has yet not been an important field of research at the European level, despite of its great importance (Leinweber et al. 2002).

There are some indications in the literature (Martin et al. 1997; Robinson et al. 1998; Renger et al. 2002; Litaor et al. 2003; Meissner et al. 2003), that the restoration of Histosols and wetlands for environmental protection may result in significant P mobilisation and risk of accelerated eutrophication. This can be explained by the following hypothetic sequence of processes: (1) the mineralization of organically bound P during drying, aeration and oxidation of peat plus input of mineral and organic P fertiliser in periods of intensive agricultural led to P enrichment in the soil; (2) much of this P was retained by adsorption to Al, Fe, and Mn oxides and hydroxides under neutral to slightly acidic conditions; (3) rewetting of these soils has resulted in lowered redox potentials and the dissolution of the Fe and Mn oxides and hydroxides, and consequently, in the release of previously adsorbed P into soil solution water. Despite some supporting hints in the literature (e.g. Ausborn et al. 1997; Ivanoff et al. 1998; Kalbitz et al. 1999; Zak et al. 2004; Gelbrecht et al. 2005) it was unknown to what extent the above processes occur in different fenlands, and scientifically based rewetting strategies are missing.

The European research project PROWATER (Program for the prevention of diffuse pollution from degraded and rewetted peat soils), running from May 2000 until December 2003, aimed at a better understanding of the P biogeochemistry in degraded and rewetted fenlands and the development of guidelines for rewetting to minimise the risk of P transfer to waterways (Meissner and Leinweber 2004). The objective of the present paper is to demonstrate how results obtained with different methodologies at different scales and various experimental sites were compiled to derive risk assessment and decision support tools. After brief method descriptions we will summarise results from selected publications of the consortium, present and discuss unpublished data, and finally show how the findings were integrated in a decision support system (DSS) for rewetting management. The DSS was developed specifically for the Droemling Nature Reserve (Droemling) fen area in NE-Germany, hence the majority of examples and field data refers to the Droemling.

## 2 Experimental Sites

A field research program on P mobilisation was carried out at seven representative fen areas in Europe and Israel (Table 1; Sweden not shown). Each experimental site was equipped with measuring techniques and devices for periodic soil analyses and soil solution sampling. Samples from representative soil profiles were used for in-depth characterisation of the P status of the soils, and for P sorption batch- and P mobilisation microcosm-experiments. The study areas represent the climatic conditions of the geographical areas in western maritime (Great Britain), Baltic/northern lowland (Sweden, Germany, and Poland), subalpine (Slovenia) and Mediterranean zones (Israel). They all have a common history of wetlands that were drained and used for agriculture and were all rewetted in the last decade or are targeted for rewetting. Detailed geological, hydrological and other site-specific information was given by Meissner and Leinweber (2004).

## 3 Methods, Results and Discussion

### 3.1 Characterisation of Soil P Status, P Adsorption and P Mobilisation

Methods for the characterisation of soil P status applied in PROWATER can be grouped into (1) total P ( $P_t$ ) determinations, conventional agronomic and environmental soil P tests, and (2) analyses of P species by sequential fractionations and  $^{31}\text{P}$  Nuclear Magnetic Resonance ( $^{31}\text{P}$  NMR) spectroscopy. Table 2 shows important characteristics of topsoils in the experimental areas which were related to the P status. The pH values ranged from medium acidic (Droemling: 5.2) to slightly alkaline (Hula Valley: 7.3). The total organic

C ( $C_t$ ) contents indicated a wide range of peat decomposition and mineral matter input. The lowest values were explained by mineral matter input combined with strong peat decomposition prior to a shift in management to natural succession in the Droemling (site SU), and the extreme peat decomposition due to tillage for arable use, and the Mediterranean climatic conditions in the Hula Valley.

The  $P_t$  contents were larger than in a wide range of agriculturally used mineral soils from NE-Germany (Leinweber et al. 1997) by up to a factor of 3. This was explained by P enrichments with non-decomposed organic matter in the native Histosols and by additional P inputs due to formerly intensive agricultural use. The plant available P (Soil P tests) was evaluated by the common method used in each of the study sites. Ammonium lactate (AL) was used in Sweden and Poland, double lactate (DL) was used in Germany, and Olsen extraction was used in the UK and Israel. Procedures used for these extractions were described in detail elsewhere (Otabbong et al. 2004). Soil P tests in topsoils and their evaluation according to the national classification schemes mostly revealed low (l) to medium (m) values. High values (h) were only derived for the most intensively used soils in the Droemling (fens strongly degraded to mineral soils, not shown) and in the Hula Valley (Table 2). Since this classification depended on the extractant and the evaluation of the P concentration determined, both differing among the national research groups and experimental sites, we compared the various national soil P tests. For a set of 43 samples from all experimental sites, the extraction efficiency increased in the order  $P\text{-DL} < P\text{-AL} < P\text{-Olsen}$ . The large extraction efficiency of 0.5 M  $\text{NaHCO}_3$  at pH 8.5 (P-Olsen) was interpreted by its ability to extract organically bound P which neither AL nor DL were able to access (Otabbong et al. 2004).

**Table 1** Geographical position and basic climatic characteristics (annual means) of the PROWATER study areas, arranged according to latitude

Study area	Country	Latitude	Longitude	Temperature (°C)	Precipitation (mm year <sup>-1</sup> )
Trebel Valley	Germany	54°05'N	12°45'E	7	600
Droemling Nature Reserve	Germany	52°50'N	11°10'E	8	600
Biebrza Valley	Poland	52°N	21°E	8	580
Somerset Levels and Moors	Great Britain	51°10'N	3°W	11	700
Ljubljana Marshes	Slovenia	45°58'N	14°28'E	10	1400
Hula Valley	Israel	33°N	35°30'E	19	500

**Table 2** Basic chemical characteristics and P contents of representative topsoil samples from the PROWATER study areas

Study area	pH	$C_t$ [mg kg <sup>-1</sup> ]	$P_t$	$Fe_{ox}$ [mmol kg <sup>-1</sup> ]	$Al_{ox}$	$P_{ox}$	DPS [%]	Soil P tests [g kg <sup>-1</sup> ]	$S_{max}$ mg kg <sup>-1</sup>	$k$ L mg <sup>-1</sup>
Trebel Valley	5.6	39.9	1,024	299	28	18	11	12/l	3,523	0.45
Droemling GL	5.3	21.8	776	146	47	11	11	21/l	n.d.	n.d.
Nature SU	5.7	12.8	802	184	17	11	11	72/m	n.d.	n.d.
Reserve AL	5.2	39.3	949	416	35	18	8	14/l	n.d.	n.d.
Biebrza Valley	6.0	41.2	1,755	282	113	32	21	10/l	853	0.34
Somerset Levels and Moors	5.4	27.7	1,171	279	166	16	9	29/m	2,032	1.22
Ljubljana Marshes	6.8	16.4	1,815	244	170	29	14	24/l	667	9.87
Hula Valley	7.3	13.7	1,239	530	31	n.a.	n.a.	35/h	1,699	0.77

Abbreviations: *ox* = oxalate extractable Al and Fe, *GL* = extensive grassland, *SU* = natural succession, *AL* = alder forest (Histosols), Soil P test according to national recommendation schemes, *l* = low, *m* = medium, *h* = high;

*n.a.* = not available,  $S_{max}/k$  = P sorption maximum/constant as an expression of the relative bonding energy (parameters calculated from the Langmuir equation)

The contents of poorly crystalline Al- and Fe-oxides, extracted with acid oxalate ( $Al_{ox}$  and  $Fe_{ox}$ , respectively) (Schwertmann 1964), were larger than corresponding values in sandy loams and loams by factor of up to 10 (Lookman et al. 1996; Leinweber et al. 1999). However, because the oxalate extractable P ( $P_{ox}$ ) values were comparable with those for the above mineral soils (Lookman et al. 1996; Leinweber et al. 1997), the resulting degree of P saturation (DPS, calculated as  $P_{ox}/0.5(Al_{ox}+Fe_{ox}) \times 100$ ) estimates were generally low (8–21%). These data indicated a minor risk of P leaching losses if the threshold of 25% for mineral soils was assumed (Breeuwsma and Silva 1992; Schoumans and Groenendijk 2000). An extensive study of DPS distribution in the Hula Valley demonstrated a strong correlation between the sorption maxima ( $S_{max}$ ) derived from evaluations of adsorption isotherms by the Langmuir equation and the DPS, indicating that DPS could be applied both to mineral and organic soils at the catchment scale (Litaor et al. 2003). In that study, the DPS values were much lower in Histosols ( $6.5 \pm 5$ ) than in redoximorphic Gley soils from marl ( $28 \pm 18$ ) and Vertisols ( $78 \pm 28$ ). Spatial analysis with a sequential Gaussian simulation technique further indicated that the Histosols had a much lower risk (<10%) of exceeding the commonly used threshold of 25% DPS than redoximorphic Gley soils from marl and Vertisols (>60% probability) (Litaor et al. 2003). However, it is doubtful if the thresholds developed for assessments of P leaching risks in sandy acid soils could be applied for the problem of fen peat rewetting. Therefore, complementary and more problem-specific analyses were carried out.

The P adsorption characteristics of the topsoils were evaluated in batch experiments according to Barrow and Shaw (1975). The experimental data were fitted to three different adsorption models: Freundlich (Barrow 1978), Langmuir (Langmuir 1918) and Tempkin (Hayward and Trapnell 1964; Berkheiser et al. 1980). The equations were used to determine the following indicators of P retention and release capacity: P sorption maximum ( $S_{max}$ ), P binding energy ( $k$ ), and equilibrium P concentrations at zero sorption point ( $EPC_0$ ). Regardless of model used, values for  $S_{max}$  were largest in the degraded and humified topsoils. For example, Langmuir-derived  $S_{max}$  values ranged from 667 mg kg<sup>-1</sup> for Ljubljana-Marsh to 3,523 mg kg<sup>-1</sup> for the Trebel Valley (Table 2). Smaller values were measured in the subsoils; 217 mg kg<sup>-1</sup> for the Somerset Levels and Moors to 861 mg kg<sup>-1</sup> for the Trebel Valley (Meissner and Leinweber 2004). Correlation analysis revealed reasonably strong relationships ( $P < 0.05$ ) between the Langmuir sorption parameters ( $S_{max}$ ,  $k$ ) and the  $Fe_{ox}$  content of the seven non-calcareous soils (Trebel, Droemling, Biebrza, Somerset and Ljubljana). The dominant role of poorly crystalline Fe influencing the P sorption characteristics of non-calcareous soils is consistent with the findings of many previous workers, but whose focus has been on mineral soils (e.g. Breeuwsma and Silva 1992; Lookman et al. 1996; Leinweber et al. 1997; Schoumans and Groenendijk 2000). The current study indicates that, in spite of high  $C_t$  contents (128–399 g kg<sup>-1</sup>, Table 2), Fe chemistry can play a key role in determining the mobility of soil P in Histosols.

The association of P with poorly crystalline forms of Fe has implications for the mobility of soil P in rewetted soils, both in the short- and long terms. For example, during the early stages of flooding of most non-calcareous soils, the concentration of  $\text{Fe}^{2+}$  increases in the soil solution through the reductive dissolution of oxides and hydroxides of  $\text{Fe}^{3+}$  (e.g.,  $\text{Fe}(\text{OH})_3$ ):  $[\text{Fe}(\text{OH})_3 + 3\text{H}^+ + \text{e}^- \rightarrow \text{Fe}^{2+} + 3\text{H}_2\text{O}]$  and can be related to Eh and pH. As a consequence, P that is associated with  $\text{Fe}^{3+}$  through sorption and co-precipitation is also released to the soil solution. There may also be a subsequent release of occluded P from the more amorphous (less crystalline) forms of Fe oxides (De Datta et al. 1989). It is important to note that the expected increase in the pH of a non-calcareous soil during the reduction of its  $\text{Fe}(\text{OH})_3$  would favour P desorption from excess  $\text{Fe}^{3+}$  oxides, as well as layer silicates and Al oxides (Ponnamperuma 1972); this would further increase the solution P concentration during the early stages of flooding, particularly in a soil that had a recent P fertiliser history (Willett 1979). The reductive dissolution of Fe–P compounds, such as  $\text{FePO}_4 \cdot 8\text{H}_2\text{O}$  to more soluble  $\text{Fe}_3(\text{PO}_4)_2 \cdot 3\text{H}_2\text{O}$  or  $\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}$  (vivianite) has also been proposed as a mechanism for increases in solution P concentrations in recently flooded rice soils (De Datta et al. 1989). The importance of these reactions in rewetted Histosols has not yet been researched.

The Hula soil, on the other hand, is characterised by its slightly calcareous nature (pH 7.3), and its inclusion in the correlation analysis between the P sorption parameters and  $\text{Fe}_{\text{ox}}$  removes any statistical significance in the relationships. Up to 50% of the P in the Hula soil (0–20 cm depth) is in the  $\text{H}_2\text{SO}_4$ -extractable form according to the modified Hedley P fractionation scheme; whereas approximately half of this proportion is associated with Ca in the other 7 soils (Meissner and Leinweber 2004). It is possible therefore, that precipitation with Ca and/or adsorption onto  $\text{CaCO}_3$  is the major P retention mechanism in this soil, in spite of its relatively high content ( $530 \text{ mmol kg}^{-1}$ ) of  $\text{Fe}_{\text{ox}}$  (Table 2). Moreover, geochemical calculations, based on published mineral solubility data, indicate that Ca–P minerals dominate P in this soil (Shenker et al. 2005).

Since the P adsorption isotherms do not allow direct conclusions on P species and characteristics of the corresponding binding sites, we sequentially extracted the P compounds according to Hedley et al. (1982).

This fractionation quantifies the following operationally defined P fractions: (1) water soluble P; (2, and 3) loosely bound P (extracted by anion exchange resin and by 0.5 M  $\text{NaHCO}_3$ , respectively); (4) adsorbed P, mainly on Al and Fe oxides and hydroxides (extracted by 0.1 M  $\text{NaOH}$ ); (5) Ca- and Mg-bound P (extracted by 1 M  $\text{H}_2\text{SO}_4$  or  $\text{HCl}$ ); and (6) P resistant to the previous extracts. The extracted P was further separated into organic and inorganic components ( $P_o$  and  $P_i$ , respectively). Horizon samples from entire soil profiles, samples from anaerobic incubations studies and topsoil horizons from field sites after rewetting periods were subjected to the Hedley fractionation. In the degraded and humified topsoil peat from the Trebel Valley the order of fractions was  $\text{NaOH}-P_o > \text{NaOH}-P_i > \text{NaHCO}_3-P_o > \text{Resin}-P > \text{NaHCO}_3-P_i > \text{H}_2\text{SO}_4-P$  (Schlichting et al. 2002). This indicated large P proportions in organically-bound and labile forms, which may become mobile following rewetting.  $^{31}\text{P}$  NMR spectra confirmed this as ortho- $P_i$  accounted for 32–50% in the upper parts of soil profiles and was the most abundant P form in topsoils at all sites except Ljubljana (Meissner and Leinweber 2004). Furthermore, the proportions of ortho-P-diester, regarded as more labile than ortho-P-monoesters (Newman and Tate 1980; Hinedi et al. 1988; Bedrock et al. 1995; Makarov et al. 1996) were significantly correlated with the concentrations of labile and moderately labile organic P extracted by  $\text{NaHCO}_3$  and  $\text{NaOH}$  in the Hedley-fractionation ( $r=0.650^{**}$  and  $0.724^{**}$ ) (not shown).

For an evaluation of the potential P mobilisation due to rewetting, two sets of microcosm experiments were carried out, using disturbed peat samples from two contrasting pedoclimatic environments, peat from a cold-humid fen area of the Trebel Valley, Germany, and a peat from a semiarid wetland of the Hula Valley, Israel.

Topsoil and subsoil peat samples (5–20 and 20–40 cm profile depth, respectively) from the Trebel Valley, were placed in slitted 15-cm-PVC columns with 5.1 cm inner diameter. These columns were wetted by placing into 1,000-mL-beakers which were filled with distilled water. Experiments were run at 6 and 20°C to determine temperature effects on the biochemical transformations. Five replicates were used for each soil and temperature treatment. Redox potential (Eh) was recorded by platinum electrodes vertically inserted into the peat, and the Eh values

were read manually every day for a period of 405 days. Solution aliquots of 10 mL were taken at six dates over the incubation period using a pipette for colorimetric determination of soluble reactive P (SRP). Soil solution removed was substituted by the same volume of distilled water. Plant available P was extracted from soil before and after the incubation by the DL-method (Riehm 1948). Furthermore, P fractions were sequentially extracted according to Hedley et al. (1982) as explained above.

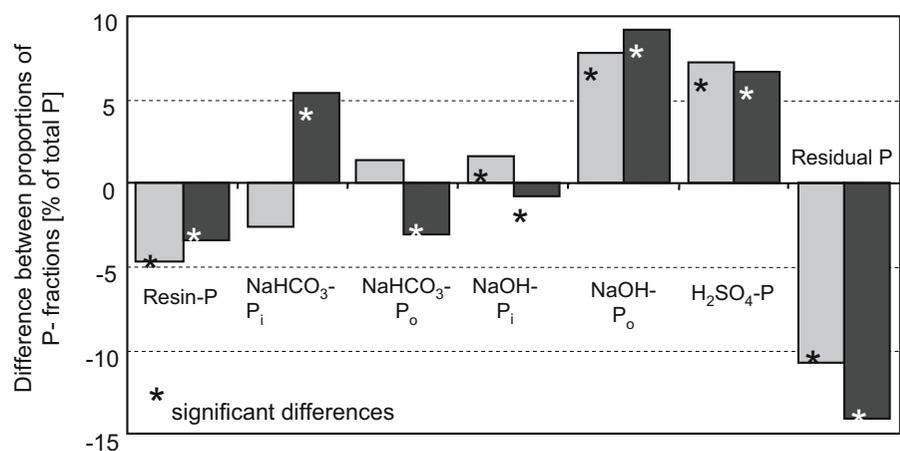
The redox potentials dropped from about +100 mV at the beginning to about -100 mV ( $\pm 50$  mV) during 90 days and remained relatively constant at this low level until the end of incubation (405 days; not shown). Temperature had a strong effect on the SRP concentration in the soil solutions. Reductive incubation at 6°C led to continuous increase in SRP concentration from  $\approx 0$  mg L<sup>-1</sup> to 0.23 mg L<sup>-1</sup> (topsoil) and 0.06 mg L<sup>-1</sup> (subsoil), respectively. At 20°C the SRP concentration increased from  $\approx 0$  mg L<sup>-1</sup> to 0.51 mg L<sup>-1</sup> (topsoil) and 0.16 mg L<sup>-1</sup> (subsoil) within the first 200 days. Then the SRP concentrations remained at a high level until the end of the experiment after 405 days (0.34 mg L<sup>-1</sup> and 0.12 mg L<sup>-1</sup>, in top- and subsoil, respectively).

Sequential P fractionation indicated generally large proportions of organically-bound and labile P in fen peat (Schlichting et al. 2002). Therefore, it was interesting to test if these fractions were predominant sources of SRP mobilised. We sequentially extracted samples taken before and after incubation in the laboratory, and after 18 month of rewetting in the field (Trebel Valley). Surprisingly, the distribution of

P among sequentially extractable fractions was on average more strongly altered during incubation at 6°C than at 20°C. Incubation at 6°C was a better reflection of the temperature in the northern and central European experimental areas, and accordingly, changes in P fractions obtained at 6°C were more relevant for the prediction of P transformations in the Droemling area. Figure 1 shows that reductive incubation experiment and the field study agreed in increased proportions of NaOH-P<sub>o</sub> and H<sub>2</sub>SO<sub>4</sub>-P at the expense of residual P amount rather than resin P after 405 days (laboratory) and 548 days (field). Generally, there was a clear trend towards mobilisation of P into more easily extractable fractions. The reduction of the most soluble P fraction (resin-P) is explained by the dilution of P-concentration in soil solution through the addition of distilled water at the beginning of the experiment and the successive equilibration between weekly adsorbed P on the solid peat surface and SRP. In summary, these results showed that the anaerobic incubation experiment reflected rewetting-induced changes under field conditions, and that the degraded and rewetted peat resulted in equilibrium SRP concentrations in the range of 0.1–0.5 mg L<sup>-1</sup>, which unequivocally pointed to the risk of accelerated eutrophication.

Bottom-sealed 128-cm PVC columns with internal diameter of 103 mm were used for soil incubation in controlled conditions of temperature (25°C) and water depth. Degraded peat soil from the Hula Valley, Israel (pH 7.5; 34% organic matter; 14% CaCO<sub>3</sub>), was thoroughly mixed with 0.1% (w/w) ground wheat straw, serving as an electron donor, and packed in the

**Fig. 1** Changes in the proportions of sequentially extracted P fractions in a soil from the Trebel Valley (20–40 cm depth) due to an anaerobic incubation in the laboratory at 6°C for 405 days (light grey), and a rewetting in the field for 548 days (black). Proportions of P fractions in % of total P after minus before exposure to anaerobic conditions. Positive differences indicate an increase, and negative differences indicate a decrease due to rewetting



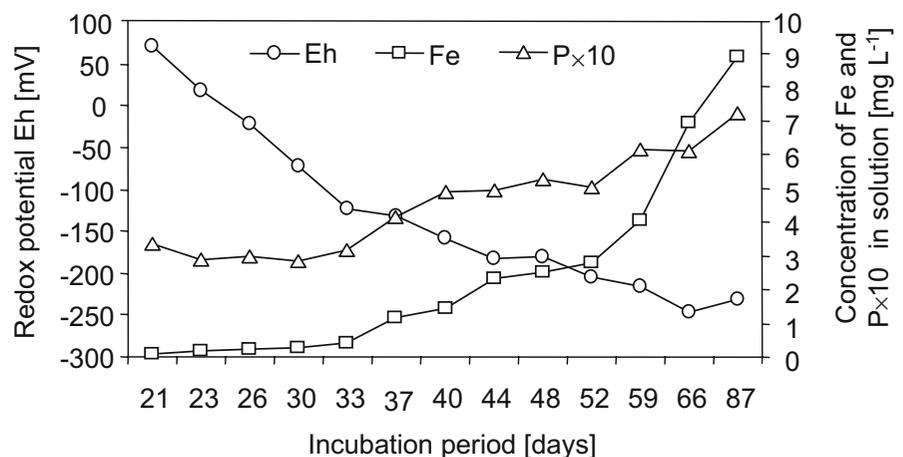
column to form a bulk density similar to that found in the field ( $1.05 \text{ kg dry weight L}^{-1}$ ). Water level was maintained at depth of 65 cm below the soil surface. Continuous measurements of redox and pH at depths of 30, 70, and 100 cm below soil surface were performed as described by Shenker et al. (2005). Soil solution samples were collected periodically from the saturated zone. Soils were sampled for analyses before packing in the biogeochemical microcosms and after 200 days of incubation.

The initial redox potential at 70 cm depth (5 cm below water table,  $488 \text{ mV}$ ,  $\text{pe}+\text{pH}=15.8$ ) had fallen within three days after rewetting to  $265 \text{ mV}$ ,  $\text{pe}+\text{pH}=12$ , then it remained quite stable until day 15 (data not shown). From day 17 the Eh declined within another 2 weeks to  $\text{Eh}=-73 \text{ mV}$  and continued to decline gradually to  $\text{Eh}<-200 \text{ mV}$  at which it remained to the end of the experiment (Fig. 2). During this period P concentration had increased more than two-fold, from  $0.33 \text{ mg L}^{-1}$  in the first month of the incubation to  $0.72 \text{ mg L}^{-1}$  at the end of the experiment. This increase had started concurrently with the fall of the redox potential to Eh values lower than about  $-100 \text{ mV}$  and with the initiation of the increase of Fe concentration, which increased from below  $0.3 \text{ mg L}^{-1}$  at the first month of the incubation to about  $9 \text{ mg L}^{-1}$  at the end of the experiment. Using the ferrozine method (Stookey 1970) it was shown that practically all the dissolved Fe was in the form of  $\text{Fe}^{2+}$ , hence it is concluded that the increase in its concentration during the incubation was by reductive dissolution. The Mn increase from  $0.2$  to  $>2 \text{ mg L}^{-1}$  was also related to the same mechanism, while Al and

Ca, both major elements in the soil and of importance in P chemistry in soils, did not dissolve during the incubation period. The increase of P concentration was thus related to the dissolution of Fe, and to a lesser extent of Mn minerals. Since the molar ratio of the released P to the released (Fe+Mn) was much smaller than one ( $<0.1$ ) it was concluded that the release of P during the reduction period was not related to dissolution of any Fe–P or Mn–P mineral, but rather it resulted from release of occluded or surface adsorbed P in Fe hydroxides that were dissolved.

The chemical soil analyses done in the labs can be summarised as follows: agronomic P tests indicated mostly low to medium P contents in soil and the DPS indicated a low risk of leaching losses. However, these tests were not specific for peat soils and possible effects of rewetting and more specific investigations of P adsorption and species revealed contradicting results. The decreased P adsorption on less degraded subsoils and in reduced soils as derived from adsorption isotherms (Brand-Klibanski et al. 2007) may indicate an increased P mobilisation as adsorption sites are removed under reductive conditions. The significance of this possible reaction was confirmed by the predominance of potentially labile, Al- and Fe-bound ( $\text{NaOH-}$ ) and labile ( $\text{NaHCO}_3\text{-}$ ) P fractions in topsoils and the correlations of these sequentially extracted fractions with proportions of labile P-diester from  $^{31}\text{P}$  NMR. Evidence for the dissolution reactions under reductive conditions was provided by anaerobic incubation and microcosms experiments (Shenker et al. 2005). Despite different peat characteristics and exper-

**Fig. 2** Changes of redox (Eh) and concentrations of Fe and P in the soil solution of degraded peat soil from the Hula Valley during 87-day reductive incubation in biogeochemical microcosms. In order to fit into scale P concentrations (ranging from  $0.33$  to  $0.72 \text{ mg L}^{-1}$ ) are multiplied by 10



imental conditions, samples both from the Baltic and the Mediterranean zone released much P into soil solution reaching concentrations of 0.2–0.7 mg L<sup>-1</sup> following Eh falls to below -100 to -200 mV. The mechanism of P release as a consequence of the reductive dissolution of Fe-oxides (Brümmer, 1974) was unequivocally supported by direct determinations of Al-, Ca-, Fe- and Mn- concentrations and molar ratios during the incubation. However, the significance of these processes had to be studied under field conditions.

### 3.2 Groundwater Levels, Redox Potentials and P Concentrations in Soil Solution, Ground and Surface Water

All experimental sites in Europe (Table 1) were equipped with semi-automated field research stations (UGT Ltd., Müncheberg, Germany). Matrix water potential was measured with septum tensiometers at depths of 0.25, 0.60 and 0.90 m. At each of these depths soil solution was collected in a three-week interval by suction ceramic cups using a vacuum pump to generate approximately 50 kPa suction. At the same depths, soil moisture content was determined continuously by frequency or time domain reflectometry (FDR, TDR) soil moisture probes (Delta-T-Devices Ltd., Burwell, Cambridge, England) and electronic thermometers were installed. Redox probes consisting of Platinum (Pt) electrodes (Schmidt 1998), were inserted of 0.1 m-intervals down to a depth of 1 m. The redox potentials were recorded automatically four times a day against an Ag/AgCl reference electrode (Schott Ltd., Mainz, Germany) and transformed to the standard hydrogen system (Eh) according to Kölling (2000). The 10 cm-increments were chosen to obtain reliable redox profiles down to 1 m depth, considering that the redox can vary at the small scale (Fiedler 1997; Stolt et al. 2000; Mansfeldt 2003). All probes were connected to a data logger for continuous registration and data storage. Groundwater levels at each site were measured manually at a frequency of 3 weeks in shallow (1.5-m depth, 5-cm internal diameter) dip wells.

Table 3 compiles data sets for moisture regimes, redox potentials and resulting P concentrations in soil solution, groundwater and adjacent surface waters, which were hydrologically connected to the rewetted fens. The groundwater levels had fluctuated between

12 cm above and 190 cm below surface, resulting in means between 12 (Droemling, GL) and 144 cm (Hula Valley, un-degraded peat site) below surface. The large variations in groundwater levels between ±9 and ±31 cm indicate temporarily surface-near groundwater respectively or inundation at some sites (Somerset, Trebel, Droemling; the degraded peat site at the Hula Valley) and a lower extent of rewetting at other sites (Biebrza, Ljubljana, and the un-degraded peat site at the Hula Valley). Comparison of groundwater fluctuations with profile redox data shows that the largest redox fluctuations occurred in most cases in the layer around which the water table had fluctuated. The minimum values of redox potentials were far below -100 mV at all sites. Generally the minimum Eh values decreased with the profile depth. However, there were some exceptions with higher Eh values at deeper layers. This might be explained by a combined effect of moisture saturation and rhizodeposit turnover on the redox potential. For instance, intensive rhizodeposit decomposition and O<sub>2</sub> consumption in the rooted zone at 30 cm and groundwater level around 90 cm could result in an Eh maximum at 60 cm. Fast penetration of O<sub>2</sub>-rich water to the subsoil may also explain a higher redox in deeper layers, as was observed at the site Droemling, SU, which had a sandy subsoil horizon below the peat horizon.

The mean concentrations of SRP in soil profiles ranged from 0.05 to about 0.8 mg L<sup>-1</sup>. At the Trebel Valley site, the SRP concentrations were higher in topsoils and lower in subsoils. At sites Somerset, Ljubljana, SU in Droemling, and the degraded peat site in the Hula Valley, the SRP concentrations were higher at deeper layers. Sites of the second group were characterised by lower and more intensively fluctuating groundwater levels. The mean concentrations of SRP in the groundwater often were around or above 0.1 mg L<sup>-1</sup> except for sites Droemling, GL, and Biebrza with distinctly lower concentrations. The surface water had similar (Trebel, Droemling, GL, Biebrza) or higher SRP concentrations (Droemling, SU, AL) than the groundwater indicating additional P inputs. Only at site Somerset the SRP concentrations were lower in adjacent surface water than in groundwater and in the soil solution of the saturated zone. This is explained by the frequent and managed recharging of the field ditch water with relatively low-P water from larger drainage channels in the

**Table 3** Groundwater levels, redox potentials, and SRP concentrations in soil solution, groundwater and adjacent surface water at the experimental sites during sampling periods June 2000–December 2002; the presented data are means  $\pm$  standard deviations of all measurements

Site	Groundwater level	Depth	Redox potential	SRP			
				Soil profile	Groundwater	Surface water	
	cm		mV	mg L <sup>-1</sup>			
Trebel Valley		30	-155 to +167	0.299 $\pm$ 0.373			
	16 $\pm$ 9	60	-193 to +12	0.023 $\pm$ 0.029	0.092 $\pm$ 0.168	0.061 $\pm$ 0.034	
		90	-228 to +12	0.022 $\pm$ 0.027			
Droemling Nature Reserve	GL	30	-204 to -137	0.009 $\pm$ 0.010			
		12 $\pm$ 24	60	-219 to -106	0.006 $\pm$ 0.006	0.033 $\pm$ 0.027	0.047 $\pm$ 0.045
			90	-231 to -140	0.005 $\pm$ 0.005		
	SU	30	-207 to +46	0.009 $\pm$ 0.008			
		19 $\pm$ 17	60	-118 to +75	0.015 $\pm$ 0.015	0.097 $\pm$ 0.121	0.405 $\pm$ 0.573
			90	-130 to +216	0.052 $\pm$ 0.023		
	AL	30	-247 to +816	0.017 $\pm$ 0.015			
		28 $\pm$ 20	60	-205 to +812	0.016 $\pm$ 0.011	0.092 $\pm$ 0.065	0.193 $\pm$ 0.252
			90	-154 to +817	0.042 $\pm$ 0.030		
Biebrza Valley	54 $\pm$ 16	30	-156 to +703	0.038 $\pm$ 0.081			
		60	-511 to +459	0.033 $\pm$ 0.055	0.046 $\pm$ 0.050	0.054 $\pm$ 0.061	
		90	-383 to +403	0.026 $\pm$ 0.027			
Somerset Levels and Moors	23 $\pm$ 20	30	-202 to +5	0.013 $\pm$ 0.018			
		60	-153 to +55	0.121 $\pm$ 0.059	0.244 $\pm$ 0.201	0.079 $\pm$ 0.073	
		90	-184 to +11	0.159 $\pm$ 0.079			
Ljubljana Marshes	28 $\pm$ 12	30	-259 to +215	0.028 $\pm$ 0.009			
		60	-314 to +358	0.024 $\pm$ 0.007	0.119 $\pm$ 0.035	n.a.	
		90	-248 to +258	0.047 $\pm$ 0.015			
The Hula Valley	DP	40 $\pm$ 18	40	-204 to +34	0.309 $\pm$ 0.074	n.a.	n.a.
			110	-287 to -140	0.737 $\pm$ 0.124	n.a.	n.a.
	UDP	144 $\pm$ 31	150	-150 to +275	0.801 $\pm$ 0.122	n.a.	n.a.
			190	-215 to +16	0.345 $\pm$ 0.158	n.a.	n.a.

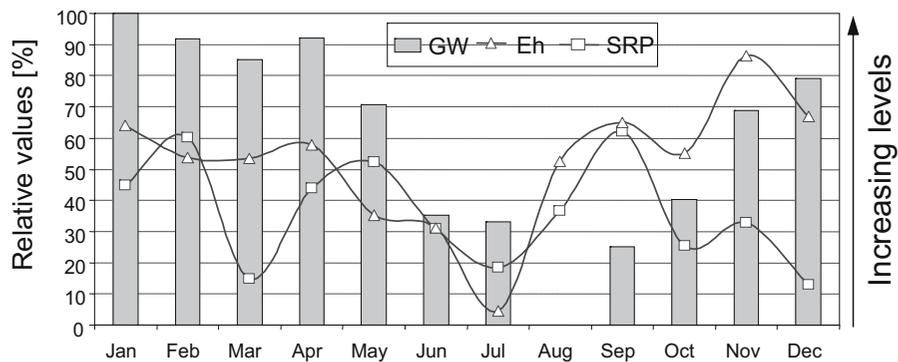
Abbreviations: *GL* = extensive grassland; *SU* = natural succession; *AL* = alder forest (Histosols); *DP* = degraded peat; *UDP* = un-degraded peat; *n.a.* = not available

summer (to meet the evapotranspiration demands) and with relatively low-P agricultural drainage waters during the autumn and winter months.

In general there were only a few sites at which the critical value for freshwater eutrophication of  $>0.1$  mg SRP L<sup>-1</sup> (LAWA 1998) was seldom exceeded in soil solution, groundwater or surface water. These were Droemling-GL, and Biebrza, the latter being not really rewetted as the low groundwater level at 54 $\pm$ 16 cm indicated. At all other sites, the low redox potentials and resulting partially high SRP concentrations in the range of 0.1 to 0.4 mg L<sup>-1</sup> confirmed the findings of the microcosm experiments. Thus the causal chain “rewetting–redox potential fall  $< -100$  to  $-200$  mV–increasing concentrations of SRP above critical values for freshwater eutrophication ( $> 0.1$  mg L<sup>-1</sup>)” was proved to be valid, and provided the basis for the

development of decision support tools for planning and control of rewetting measures.

Figure 3 shows the seasonal fluctuation and gradients of groundwater level, SRP and redox potential. Sites were selected according to a minimum of 12 months of non-disrupted data. The data from the Somerset Levels and Moors were not included because artificial drainage and pumping governed water level, redox and SRP at that site. At the other sites, these parameters varied over seasons. Groundwater level was highest in January and then decreased until August. Recharge of groundwater started in September when the net water balance became positive. In the first four months of the year groundwater level and redox potential were relatively stable showing gradual decreases and a slight increase in April. Despite falling groundwater level the redox



**Fig. 3** Medians of groundwater table (GW), concentrations of soluble reactive P (SRP, 30 cm depth) and redox potentials (Eh, 30 cm depth) at the experimental sites Trebel Valley, Droemling

Nature Reserve and Biebrza Valley. From site-specific ranges, data were normalised so that maximum values are represented by 100 and minimum values by 0%

potential also decreased from April to July. This can be explained by plant growth and the resulting  $O_2$  consumption by the root systems. The drop of SRP in March is explained by initial plant uptake and, possibly, by lateral sub-surface water flow which may have diluted the SRP. The following increase of SRP until May was forced by the decrease in redox potential. Subsequent plant uptake of P may be the reason for minimum SRP concentrations in July. The redox potential and the SRP reached a second maximum in September, which is explained by  $O_2$  diffusion into soil due to low groundwater, reduced plant growth and rhizosphere bioactivity, and beginning SRP input from leaching out of the decomposing vegetation. After a temporary minimum in October, redox potential and SRP peaked again in November, which can be explained by the input of  $O_2$ -rich water from precipitation and the chemical decomposition of dead plant residues both interrupted in December during frost period. Relationships between groundwater and redox confirmed de Mars and Wassen (1999) and Mansfeldt (2003). Furthermore, the present data also support Havens (1997), who reported direct influences of vegetation on redox state of the rhizosphere in seasonally flooded forest soils.

#### 4 Development of a Decision Support System (DSS)

##### 4.1 Natural, Agricultural and Socio-Economic Background for the Droemling

The DSS for the rewetting management of the Droemling area and the maintenance of water quality in the

Ohre River integrated evidence from laboratory experiments, microcosm and field studies on mechanisms and amount of P release (see Sections 3.1. and 3.2.) and scenario calculations of water quality (see Section 4.2.) as well as the perception of local stakeholders (see Section 4.3). The Droemling is the largest fen area in the Federal State of Saxony-Anhalt, Germany. The Saalian glaciation pre-formed the landscape, and the glacial forms were filled with water after ice melting 10,000–8,000 years ago. Loamy, clayey and organic sediments were deposited in a shallow lake and formed a fen of about 320 km<sup>2</sup> size (Fig. 4). Settlements generally started in outer areas of the fen. The mean annual rainfall is about 600 mm and the estimated evapotranspiration ranges from 520 mm in the surrounding elevated areas to at least 660 mm in the central parts. Thus, the fen sites of the central Droemling have an average negative water balance. The Droemling is an important agricultural, recreational, and drinking water supply area. Up to the early 1990s the Droemling was intensively farmed and intensive livestock farming (dairy cattle and beef) with highly fertilised meadows and grassland was characteristic of its central part. The degraded fen sites and the surroundings were intensively tilled and used for arable crops (rye, barley, corn, oil-seed rape) with high fertiliser input. Land use changes were initiated with the establishment of a nature park in 1990 and a nature reserve area (280 km<sup>2</sup>). Intensive cultivated pasture was converted into extensive (unimproved) pasture. Land use changes were accompanied by raising groundwater tables according to the demands of breeding water birds and peat conservation. In selected parts of the central Droemling intensively used

grassland was converted into natural succession including the rewetting of the remaining peat. Furthermore, the Droemling is one of the most important water protection areas of Saxony-Anhalt; about 50% of the amount of water used for artificial groundwater recharge in the “Colbitz” water plant originates from the Droemling (via Ohre River to the pump station Satuelle and in open channel to the groundwater recharge facilities, Fig. 4). Groundwater of this region is naturally cleaned for potable water supply of the Magdeburg urban area with its 600,000 inhabitants. The following stakeholder conflict overrides all other possible conflicts in this region: (1) protection of the raw water quality and assuring the quantity of raw water that is needed for a sustainable supply of drinking water to the region according to the EU Water Framework Directive (European Communities 2000) and (2) rewetting of the Droemling area for ecosystem renewal and nature protection.

#### 4.2 Water Quality in the Ohre River

The water quality of the Ohre River was monitored at gauging station Satuelle because it reflected effects of land use change in the Droemling rewetting area and

the impact on the groundwater recharge area of “Colbitz” water plant (Fig. 4). The concentrations of SRP (Fig. 5), TP, ammonium N ( $\text{NH}_4\text{-N}$ ), sulfate ( $\text{SO}_4$ ) and total organic carbon (TOC) (data not shown) increased after rewetting. The TOC concentrations in most cases met the requirements of German water quality class 3 (“increased polluted”) according to LAWA (1998). A consistent set of P concentrations was available for the period 1996–2005 but not for earlier years because the Magdeburg Water Authority had changed the method of P analysis at Satuelle gauging in 1995. The time series analysis of water quality data was based on 10th and 90th percentiles of discharge to exclude extreme discharge situations (outliers). In this way about 80% of the available data were included. Figure 5 shows a significant trend of increasing SRP concentrations in the Ohre River at Satuelle gauging point between 1996 and 2005. Simultaneously, there was also a tendency towards slightly increased concentrations of TP (data not shown). Therefore, it could be concluded that the water quality of the Ohre River was affected by land-use changes and rewetting of about 3.5% of the Droemling area. However, despite temporal fluctuations, the SRP concentrations were

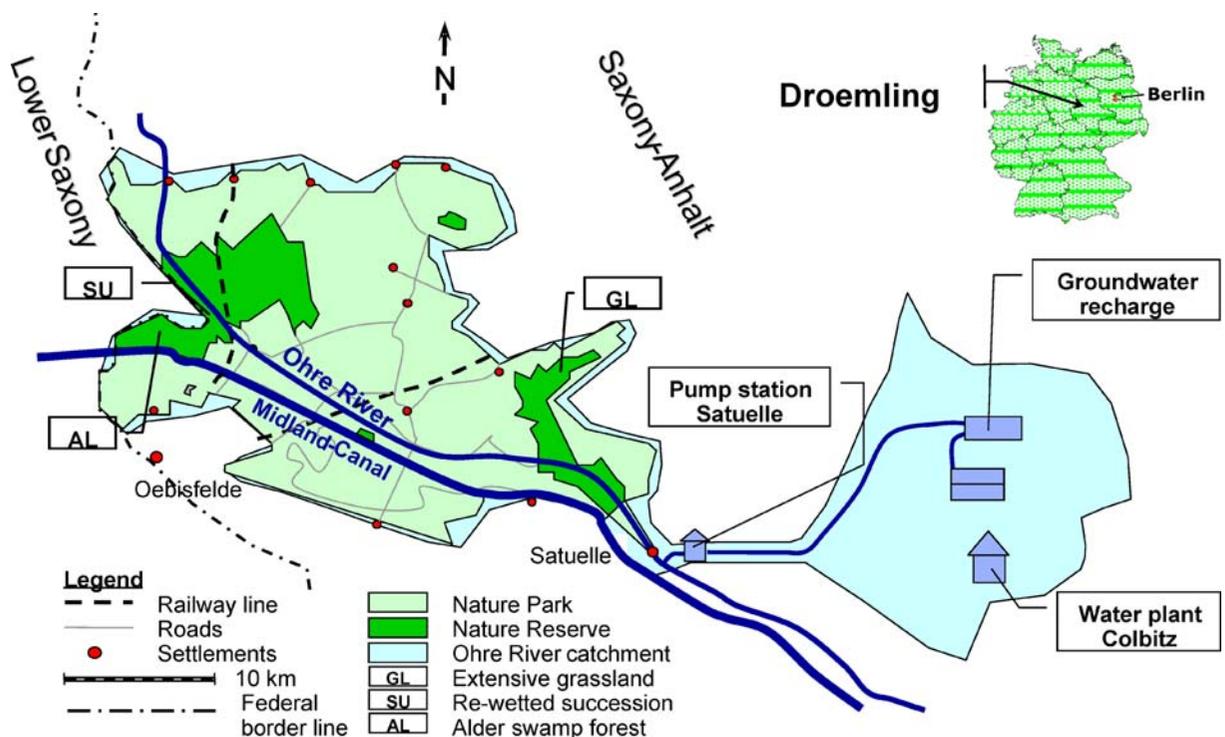


Fig. 4 Sketch of Droemling Nature Reserve and groundwater recharge area of the “Colbitz” water plant, Saxony Anhalt, Germany

always below the limit of German water quality class 2 (“moderately polluted”:  $\leq 0.1 \text{ mg L}^{-1}$ ) according to LAWA (1998). The aim of the DSS application to rewetting management is to maintain this water quality. A trend calculation for the period 1996 to 2005 was used to predict the range of SRP concentration in the Ohre River for a 6-year-period assuming that the actual trend will continue. Presuming that the proportion of extensively used and rewetted area will remain at the present 3.5%, an average SRP concentration of about  $0.04 \text{ mg L}^{-1}$  can be expected in the Ohre River by the year 2011. This corresponds to an increase of around 33% over the SRP concentration determined in 2005. For subsequent scenario calculations we assumed that the slope of the trend lines correspond to larger proportions (10 and 30%) of rewetted area.

**Scenario I:** About 10% of the entire Droemling area will be rewetted in the next 6 years. This could result in an average SRP concentration of about  $0.06 \text{ mg L}^{-1}$  in 2011 equivalent to doubling as compared to the data for 2005.

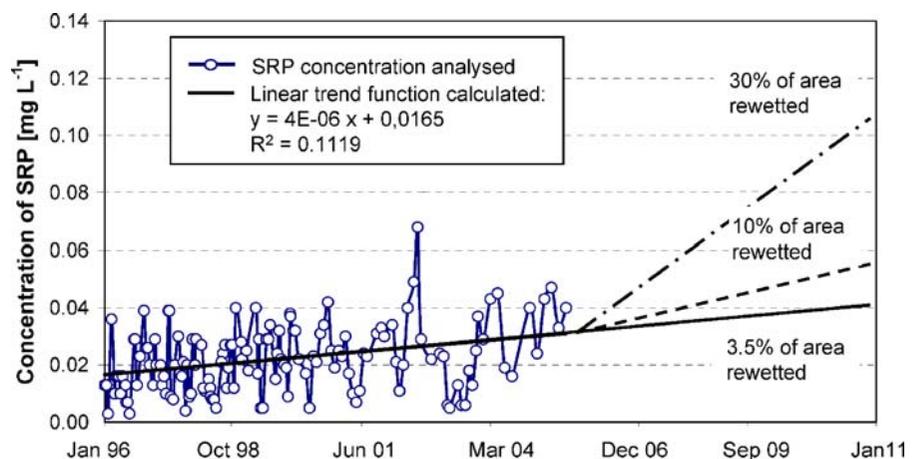
**Scenario II:** About 30% of the entire Droemling area will be rewetted in the next 6 years. In this case an average SRP concentration of about  $0.11 \text{ mg L}^{-1}$  can be predicted for the year 2011, which is equivalent to an increase of about 360% compared to 2005. This scenario is relevant since we already measured an average SRP concentration of  $0.40 \text{ mg L}^{-1}$  in surface water adjacent to the experimental site SU in 2002.

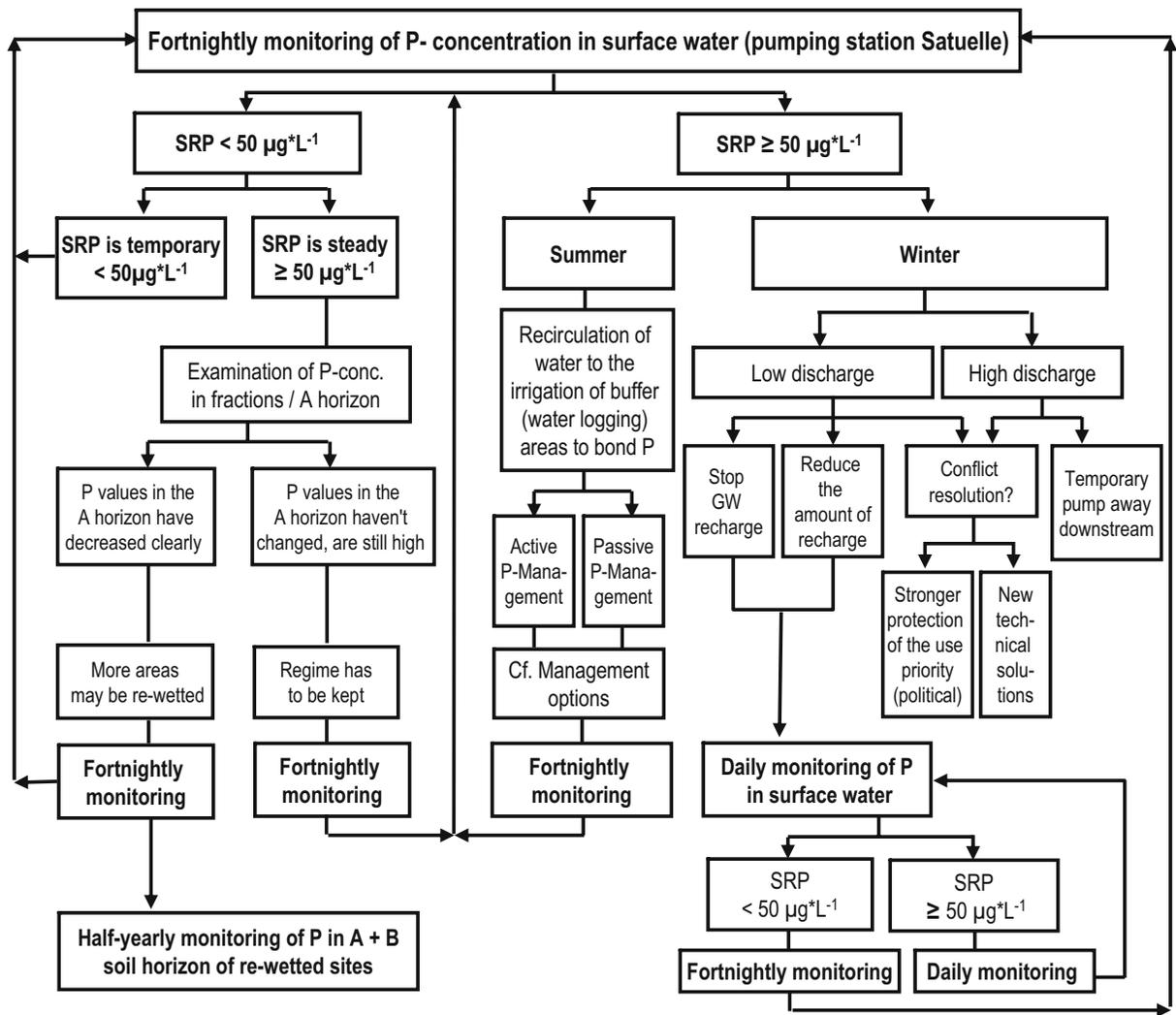
The SRP concentration of  $0.06 \text{ mg L}^{-1}$  for Scenario I was larger by a factor 3 than the trigger value for an accelerated eutrophication in lakes and streams ( $0.02 \text{ mg SRP L}^{-1}$ ; OECD 1982). The calculated SRP concentration for Scenario II ( $0.11 \text{ mg L}^{-1}$ ) is extremely high. These scenarios address the stakeholder conflict between the large-scale rewetting desired for wetland ecosystem renewal, wildlife and biodiversity protection on the one hand, and protection of the raw water quality and assuring the quantity of raw water required for a sustainable supply of drinking water to the region as pointed out by EU Water Framework Directive (European Communities, 2000). Our approach to solve this conflict was the development of a simple DSS which aims at supporting the decision making for a rewetting strategy of the Droemling fenlands and controlling the rewetting effects on water quality and drinking water supply.

#### 4.3 Data and Knowledge Integration into the DSS Structure

At first, the DSS aims to assist maintaining the present water quality situation in the Droemling area by testing alternative management options. Figure 6 shows the structure of the DSS. Its main structure consists of a decision tree with different practical options to manage the rewetting. Since the DSS was based on the P trend analyses of the Ohre River with the extrapolation of two scenarios for increasing the rewetted area (Fig. 5), a fortnightly monitoring of SRP concentration in the Ohre River at gauge Satuelle is the paramount tool in the DSS. The chemical

**Fig. 5** Soluble reactive P (SRP) concentrations in the Ohre River, Droemling Nature Reserve, Saxony-Anhalt, Germany, in the time period 1996 to 2005 and hypothesised projection of P concentrations increase during additional five years under recent rewetting scenario and increased proportions of the rewetted area





**Fig. 6** Schematic presentation of the Decision Support System (DSS) for the planning and control of rewetting effects in the Droemling Nature Reserve, Saxony Anhalt, Germany

analyses in the laboratory and field measurements proved the probability of exceeding the SRP value of  $0.1 \text{ mg L}^{-1}$  indicating German water quality class 2 (“moderately polluted”) for freshwater. For safety reasons and to maintain the present water quality in the Ohre River, half of this value, namely  $0.05 \text{ mg SRP L}^{-1}$ , was set as a simple and practical trigger value to initiate management activities. This limit agrees with recommendations of the EC Cost Action 832 “Quantifying the Agricultural Contribution to Eutrophication” (Flynn et al. 2002; Jarvie and Withers, 2002). Furthermore, this trigger value was accepted by regional stakeholders, especially the “Colbitz” water plant since problematic increases in TOC concentration of the raw water for groundwater

recharge can be avoided if the SRP in the Ohre River remains below  $0.05 \text{ mg L}^{-1}$ . If this situation is constant for a duration of 2–3 years and the whole fenland system is relatively stable without undesired P losses and SRP concentrations increase, then an extension of the rewetting could be tested. Based on the findings of the chemical soil analyses (Fig. 1, Table 2), in-depth examination of the P status of relevant topsoil horizons should prove that the P pools are already in equilibrium with low redox potentials and pose no further risk of undesired P mobilisation. In this case the DSS involves a strict monitoring program of P in the soils (SRP concentration in the rewetted soils twice a year) and in the surface water as well.

If the SRP is above  $0.05 \text{ mg L}^{-1}$  (Fig. 6, right), management must be directed towards maintaining the water quality in the Ohre River or even reducing the SRP. In this case the obvious seasonality in all relevant parameters (Fig. 3) strongly calls for distinguishing between summer and winter situations. Effects of unacceptable high SRP concentrations during summer could be buffered by recirculating the water through P binding areas. There are two alternatives suggested—active or passive P management. Possibilities for the active management are mowing in June, followed by summer pasture and mowing afterwards and/or additional weed removal from ditches and composting the material outside and/or biological measures by fishing white fish in autumn etc.. Options for passive P management are back pumping of P enriched water on the rewetted areas in summer and/or return to extensive land management systems (rotation pasture, hay meadows) and/or avoidance of groundwater table fluctuations etc..

High SRP concentration in winter require considering the amount of discharge. Generally, the seasonality of evapotranspiration and water supply can result in temporary water shortage in the Droemling region. Low discharge immediately requires the reduction or even the cessation of groundwater recharge by the “Colbitz” water plant. Additionally, in this situation SRP in the surface water should be monitored daily. High discharge is mostly connected with flood protection of the area. This means a political decision is necessary, e.g. to use the normally scarce water budget in the area for further rewetting activities (loss of farmland) or transport the flood water as soon as possible out of the area. The immediate transport of water protects the property of the stakeholder but does not solve the problem sustainably, because the P enriched water flows from the Ohre tributary into the Elbe River.

The DSS was transferred into a simple graphical software and delivered to stakeholders and regional authorities in 2004. It was successfully applied in conflict mitigation and provided a long-term perspective for the rewetting process in this region. At present rewetting is limited to the reservation area of the nature park whereas all other areas are used for extensive agriculture. Only a relatively small proportion of the fen peat can be preserved by this strategy. A more complete renewal of the fen peat would demand flooding of a greater area. This seems to be

possible only if the whole catchment of the Ohre River is included in the management regime. Nowadays first steps were initiated towards a cross-boarder reconciliation of water management activities between the Federal States of Saxony Anhalt and Lower Saxony (Fig. 4). Among others, a common water balance for the entire Droemling area is to be provided for the application of the EU Water Framework Directive (European Communities 2000). Moreover, various landscape functions such as sustainable supply of drinking water, extension of the rewetting area or maintenance of agricultural production, all related to commercial activities, have to be ranked by stakeholders.

## 5 Conclusions

The chemical characterisation of the soil P status by using a multi-methodological approach pointed to the enrichment of Fe-bound, potentially mobilisable P forms in the humified fen peat topsoil of agriculturally used, and thus, degraded Histosols. Since the P sorption characteristics were significantly correlated to the contents of poorly crystalline Fe-oxides, redox-driven changes in the Fe status were likely to pose a risk for undesired P mobilisation.

Anaerobic incubation experiments in biogeochemical microcosms enabled quantification of the relationship between fall in redox potential and resulting P release. Irrespective of the peat origin from a cool temperate or mediterranean fen, Eh values of  $-100$  to  $-200 \text{ mV}$  resulted in P concentrations between  $0.2$  and  $0.7 \text{ mg L}^{-1}$  in the corresponding soil solution. Because these values were several times above limits e.g. for “moderately polluted freshwater” (LAWA 1998) the small-scale laboratory-based analyses provided unequivocal evidence for P mobilisation from degraded fen peat under anaerobic conditions.

The upscaling of the investigations to soil profiles and rewetted fen peat fields confirmed the causal chain “rewetting–redox potential fall  $<-100$  to  $-200 \text{ mV}$ –increasing concentrations of SRP above critical values.” Different from incubation experiments, the relationships between redox and P solution were also affected by water level or soil moisture and by the vegetation. Vegetation probably contributed to low redox potential by rhizodeposition and to P concen-

trations by uptake or leaching from decaying biomass. Therefore, the pronounced seasonal and annual variations in groundwater level, redox and corresponding SRP concentrations strongly call for continuous monitoring programs to control large scale rewetting.

It was shown that relevant soil chemical evidence and key values for limits of SRP concentrations could be integrated into a DSS, which also involved the stakeholders view and political and economic issues at the catchment scale. Adaptation and application of the DSS in other countries or catchments may help to set up monitoring programs with a minimum sufficient set of parameters, to reduce the costs for rewetting programs, and to avoid undesired SRP losses and freshwater eutrophication during the rewetting of fen peat.

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