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Abstract: The present area of European wetlands is only a fraction of their area before the start of large-scale human colonization of Europe. Many European wetlands have been exploited and/or managed for various purposes. Large wetland areas have been drained and reclaimed mainly for agriculture and establishment of human settlements. These threats to European wetlands persist. The main responses of European wetland to ongoing climate changes will vary according to wetland type and geographical location. Sea level rise will probably be the decisive factor affecting coastal wetlands especially along the Atlantic coast. In the boreal part of Europe, increased temperatures will probably lead to lowered annual evapotranspiration and lowered organic matter accumulation in soil. The role of vast boreal wetlands as carbon sinks may thus be suppressed. In central and western Europe, the risk of floods may support the political will for ecosystem-unfriendly flood defence measures, which may threaten the hydrology of existing wetlands. Southern Europe will probably suffer most from water shortage, which may stregthen the competition for water resources between agriculture, industry and settlements on the one hand and nature conservancy, including wetland conservation, on the other.

Actual State of European Wetlands and Their Possible Future in the Context of Global Climate Change

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Abstract

The present area of European wetlands is only a fraction of their area before the start of large-scale human colonization of Europe. Many European wetlands have been exploited and/or managed for various purposes. Large wetland areas have been drained and reclaimed mainly for agriculture and establishment of human settlements. These threats to European wetlands persist.

The main responses of European wetland to ongoing climate changes will vary according to wetland type and geographical location. Sea level rise will probably be the decisive factor affecting coastal wetlands especially along the Atlantic coast. In the boreal part of Europe, increased temperatures will probably lead to lowered annual evapotranspiration and lowered organic matter accumulation in soil. The role of vast boreal wetlands as carbon sinks may thus be suppressed. In central and western Europe, the risk of floods may support the political will for ecosystem-unfriendly flood defence measures, which may threaten the hydrology of existing wetlands. Southern Europe will probably suffer most from water shortage, which may stregthen the competition for water resources between agriculture, industry and settlements on the one hand and nature conservancy, including wetland conservation, on the other.

Keywords. Wetlands; carbon sequestration; hydrology; biodiversity; climate stabilization; ecosystem services

Introduction

The area of the European continent is about 10^7 km^2 including the European part of Russia. Excluding Russia and marine areas, the area is about 6 710 000 km^2. The climate of Europe is characterized by marked climatic gradients (from the cold climate in the polar regions in the north to the dry and warm Mediterranean climate in the south, and from the oceanic climate in the west to the continental climate in the east). Europe is a densely populated continent (67 inhabitants per km^2) but with a high heterogeneity in the distribution of the human population. More densely inhabited areas are located in central-west Europe, especially in urban areas where 400 inhabitants per km^2 is frequent. In all the other zones of Europe, large areas remain with a rather low human population density (e.g., less than 10 inhabitants per km^2 in parts of Finland, Spain, Greece or Poland). Much of the area of Europe has been settled at least since the beginning of the Middle Ages (i.e., for about 1500 years).

While the formation and original distribution of wetlands is largely due to climatic and edaphic factors of habitats, their further fate in Europe is closely connected with the human settlement and the associated history of landscape changes as well as their intensity. Within these changes, large wetland areas have been lost by their drainage for various purposes. Few wetland complexes have remained untouched or reasonably well preserved, while many of the remaining wetland sites have been fragmented into isolated bio-geographical islands. The total area of European wetlands included in the last wetland inventory is estimated at about 500 000 km^2 (excluding the European part of Russia). This represents about 7 % of the land area considered (Nivet and Frazier 2004, Table 1).

Wetland types

The varied climatic characteristics, together with variation in other site characteristics such as hydrology and bedrock type, account for a great variety of wetland types. Numerous classifications of wetlands have been proposed that can be applied also to European wetlands (Cowardin and Golet 1995; Gopal et al. 1990; Orme 1990; Hejný et al. 1998; Keddy 2000; www.ramsar.org). In addition, there is a European classification of wetlands with the CORINE system developed for the purpose of EU legislation (European Commission 1991). In this paper, we employ a highly simple wetland classification. According to the existence of contact with the sea, we distinguish between coastal wetlands and inland wetlands, both of different types. For the sake of simplicity, river deltas, which are transitional between coastal and inland wetlands, are included among the inland wetlands. Within the category of inland wetlands, we pay most attention to palustrine, lacustrine and riverine wetlands. In addition, we treat separately human-made wetlands. In Europe, freshwater wetlands prevail among the inland wetlands, but salt marshes and both permanent and temporary oligo- to hyperhaline shallow lakes and other standing water bodies must not be neglected. Examples of wetland areas, notable because of their large area or conservation value, are given in Fig. 1. The structure and functioning of biotic components of all wetland types has been thoroughly treated by Gopal and Masing (1990).

Nivet and Frazier (2004) have reported on the most recent inventory of European wetlands. The area of coastal wetlands is currently estimated at about 46 000 km^2, which is about 2 % of the total area of European wetlands including Russia. With their total area of 2486000 km^2, inland wetlands comprise the largest proportion of the total wetland area. The reported area of human-made wetlands is about 20000 km^2 or 1% of the total wetland area. However, there still seems to be large uncertainty in these estimates because not all national inventories are complete, different national inventories use different definitions of wetland types and, last but not least, some of the areas reported as wetlands apparently also comprise former wetlands that have been drained.

Coastal wetlands

Despite its relatively small geographic size, Europe has a very long coastline, approximating 326 000 km (Pruet and Cimino 2000). The European coastline comprises the main marine regions of the northeast Atlantic, part of the Arctic, the Baltic Sea, the North Sea, the Mediterranean Sea and the Black Sea. Much of the European coastline consists of a chain of extensive estuaries, lagoons and intertidal bays interspersed through stretches of rocky shore and sandy beaches. These areas support various wetland types. Airoldi and Beck (2007) distinguish macroalgal beds, seagrass meadows, biogenic reefs, sedimentary habitats (mudflats, sandflats and subtidal soft bottoms) and emergent coastal wetlands including salt marshes.

In addition to climate, the occurrence of various wetland types is determined mainly by local geomorphological features and tidal range. The tidal range is up to several meters along the Atlantic coast (including the North Sea). Along the North Adriatic Sea the tide range can be 1 m, both diurnal and semidiurnal (depending on the Moon phase) while it is much narrower to negligible (usually several centimeters) along the remaining European coasts. In strongly tidal areas of the Atlantic and the North Sea, the seaward zone of gently sloping shores is occupied by soft-sediment habitats emerging at low tide. Muddy habitats usually occur in sheltered areas, such as sea lochs, enclosed bays and estuaries, whereas sandflats and coarser sediments tend to occur in more exposed situations on the open coast. Salt marshes are developed on suitable more elevated sites all along the European coast, often bordering on estuaries. A wide tidal range is responsible for the occurrence of tides even within predominantly freshwater wetlands fringing the estuaries of rivers or streams flowing into the Atlantic or the North Sea. This is the case even for the flat parts of the Mediterranean coast with an average maximum tide of 0.20 m. Long wave based phenomena caused by barometric changes (seiches) can be reflected on the coasts as 2 m wide changes of sea level in a few hours (Ranwell 1972).

Palustrine wetlands (peatlands)

The first group of natural inland wetlands occurs in habitats characterized by the presence of organic soils, waterlogged or saturated with water, with fairly narrow annual water-table fluctuations (peatlands). Definitions vary to some extent among countries, but peat thickness usually needs to be at least 30 cm for a site to be classified as peatland. Mires represent a subset of peatlands. Mires are living ecosystems, where peat is being formed and accumulated. In addition, peatlands also comprise drained sites e.g., in agricultural use, where a peat layer is still present (e.g., Joosten and Clarke 2002). According to Joosten and Clarke, the original peatland area in Europe (excluding Russia) had been about 374 500 km^2, that is around 6% of total land area; more than 50% of the original area has ceased to accumulate peat due to human exploitation, and almost 20% has ceased to exist as peatlands. Lappalainen (1996), on the other hand, has estimated that peatlands cover about 960 000 km^2, that is about 20 % (!), of the land area of Europe.

Peatlands are essentially either ombrotrophic (bogs), fed predominantly with rain water, or minerotrophic (fens), fed additionally with ground water, or surface runoff. Bogs (Fig. 2b) may develop where there is a positive rainwater balance (precipitation > evaporation + runoff). Raised bogs are or were common in North-West Europe (southern Fennoscandia, the British Isles, Germany, the Baltic states, Poland, Russia) as well as in subalpine regions further south. Most of these peatland complexes started as minerotrophic, and developed towards ombrotrophy during millennia. They characteristically involve a minerotrophic lagg and a raised ombrotrophic part. In regions with very high precipitation, such as western Ireland and Scotland, another type of bogs is found: the blanket bogs. In northern

Fennoscandia and Russia, as well as the lowlands in southern Europe, fens predominate. The northern aapa mires (Fig. 2a) are sustained by ample snow-melt waters.

Lacustrine wetlands

The lacustrine wetlands have developed mostly in the littoral zones of both shallow and deep standing waters, mainly natural lakes, and also on gently sloping shores of reservoirs and fishponds (Fig. 3a; Dykyjová and Květ 1978; Jörgensen and Löffler 1990; O´Sullivan and Reynolds 2005). These wetlands thus form ecotones between the respective water bodies and surrounding land (Holland et al. 1991; Gopal et al. 1993; Hillbricht-Ilkowska and Pieczyńska 1993; Lachavanne and Juge 1997) Apart from the climatic conditions and the shore slope, it is the water-level fluctuations and their timing (Hejný 1957, 1971; Hejný et al. 1998) as well as the granulometric composition and chemical composition of the wetland sediments that are decisive for the development of various types of lacustrine wetlands (Mitsch and Gosselink 2000). Usually, these wetlands show a distinct zonation according to the ecophases (*sensu* Hejný et al. 1998) prevailing at each particular site in the littoral ecotone. Hydric phase (or hydrophase), littoral, limosal and terrestrial ecophases can be distinguished, with the water table at more than about 1 m, less than that, more or less at, and below the ground level, respectively. For an illustration and characteristics of various zonations of lacustrine wetlands see, e.g., Hejný et al. (1998). The amplitude of these water-level fluctuations is relatively narrow in oceanic and sub-oceanic regions of Europe with frequent rain- and snowfall. By contrast, prolonged dry periods occur either regularly or frequently in areas of Europe with a Mediterranean or continental climate. Consequently, wide amplitudes of water-level fluctuations characterize their lacustrine wetlands.

According to the shore exposure to wave action one can distinguish either accumulation or erosion littoral habitats. The former ones occur in sheltered situations where ample accumulation of detritus-derived autochthonous sediments rich in organic matter takes place. The latter habitats occur in wind- and wave exposed situations where most of the plant litter and detritus is washed away into the adjacent water body, and the underlying mineral layer consists of sand, gravel or withered bedrock. One can add sedimentary wetland habitats with the deposition of allochthonous sediments at the mouths of running waters entering standing water bodies. The granulometric composition of these sediments depends on the inflow velocity at each site while their chemical composition reflects that of soils in the catchment areas of the inflowing streams or rivers. For more details on the formation of lake sediments see, e.g., Bloesch (2004).

Marginal wetlands, in which the terrestrial ecophase prevails for most of an average year (Květ et al. 2002), often reach to long distances from the shoreline in areas that have not been artificially drained. These marginal lacustrine wetlands are very similar to palustrine wetlands. At eu- to mesotrophic sites, they are fen-like and colonized either by shrubby or forest vegetation dominated by hygrophytic woody plants (typically *Salix* or *Alnus*), or by both natural and human-made wet grassland dominated by hygrophytic grasses and sedges. At oligotrophic sites, the character of the marginal lacustrine wetlands resembles that of transition mires or even bogs and the water stored and flowing out of them is more or less dystrophic.

The functional interaction between the lacustrine wetlands and the adjacent water body or land depends, naturally, on the width of the littoral zone which, in turn is determined by the shore slope. Only rather wide littoral belts, like that of lake Neusiedlersee/Fertö in Austria/Hungary (Löffler 1974; Löffler and Gunatilaka 1994), possess structural and functional features of ecosystems showing a high degree of independence of their adjacent biomes. Narrower littoral wetlands strongly interact with adjacent both land and water. Nevertheless, the predominance of the detritus-bases food web is characteristic of all lacustrine littoral wetlands dependent mainly on the primary production by macrophytes, while the grazing-predatory food chain predominates in the food web in open-water (pelagial) habitats dependent mainly on the primary production by phytoplankton (Straškraba 1963, 1968; Straškraba et al. 1967; Gopal et al. 1993; Hillbricht-Ilkowska and Pieczyńska 1993).

Inland salt marshes and saline lakes (Fig. 2e) occur predominantly in south-western and south-eastern Europe (e.g., in Spain, Hungary, Balkan countries), on sites where summer evaporation is intense and brings about capillary rise of soil water rich in salts (sulphates and/or chlorides) from the subsoil. This provides a unique type of wetland ecosystem for the European ecodiversity, which is more abundant in other continents (Comín and Alonso 1988; Comín and Williams 1993). Even elsewhere in Europe, small inland salt marshes can be found around mineral springs.

Riverine wetlands

Diverse and highly dynamic systems of habitats are associated with riverine wetlands, i.e., those fed with running water – from springs and small streams through preserved segments of floodplains to both freshwater and brackish habitats of large river deltas (e.g., Purseglove 1988; Junk and Welcomme 1990; Prach et al. 1996; Middleton 2002; Haslam 2008). The hydrological régime is decisive for the structure and functioning of the riverine wetlands

(Duever 1990; Mitch and Gosselink 2000). It varies according to the climatic zones and geomorphological features of the respective river and stream headwater as well as remaining catchment areas. At high altitudes it is only on mountain plateaux or gentle slopes that smaller or larger floodplains develop around springs and along slowly flowing and often winding streams. In many of them, smooth transitions can be observed to peat-forming wetland systems, i.e., mires. In steep mountains, on the other hand, there is often hardly any place for the formation of a floodplain of an appreciable size along swiftly running streams or rivers, often squeezed into narrow gorges or ravines.

In the foothills, where the water flow slows down, relatively large floodplains can be formed, which are differentiated into a shifting and meandering river or stream bed, leaving behind partly or fully cut-off backwaters, oxbow lakes and high-water whirlpools, whose natural land-filling can be checked by disturbances during higher floods (Fig. 3b); they revert these habitats to or near to their initial stages. In spite of a common strong water flow regulation by dams and canals, high spring floods occur especially in floodplains of both small and large rivers fed with water from thawing snow in spring. Examples of such rivers are the Danube, Rhine, Rhone or Ebro. High spring floods occur especially in floodplains of both small and large rivers fed with water from thawing snow in spring. Examples of such rivers are the Danube, Rhine or Rhone. For the lower reaches and delta of the Danube, for example, the high-water period can extend into the summer months when the water from thawing snow combines with water from heavy June or July precipitations in the Danube catchment area, especially in the Alps and Carpathians. Such heavy floods occurred, e.g., in June and July 1966 and August 2002.

Generally less dynamic (with notable exceptions such as the floods in England in 1997) is the hydrological régime of rivers and streams with completely or prevailingly lowland catchment areas with little or no snow accumulating during winter. Here, it is the actual precipitation in the catchment areas that controls the water table and flow velocity. As a result, the fluctuations of these hydrological parametres are less regular here than in watercourses fed with water from abundant snow in the mountains. Nevertheless, the resulting diversification of the floodplains of predominantly lowland watercourses is similar to that of the previous type of floodplains. Both sharp boundaries between land and water and smooth ecotones between them are abundant in natural floodplains (Naiman and Décamps 1990). Specific for floodplains of watercourses in the Mediterranean parts of Europe is a regular alternation of relatively high-water periods in the rainy winter and low-water periods in the dry summer (Britton and Criveli 1993).

Highly dynamic and exposed to frequent disturbances is the herbaceous and shrub vegetation fringing river and stream banks (Prach et al. 1996, 2003). Sites exposed to more or less frequent disturbances by the flood water are colonized by a mosaic of temporary stages of a hydrarch succession of wetland herbaceous vegetation, from submerged and floating-leaved hydrophytes (Fig. 2f) through helophytes (e.g., *Phalaris arundinacea* or *Glyceria maxima*) to marsh plants such as sedges (*Carex* spp.) (Fig. 2c). Calmer floodplain sites are occupied, as a rule, by forest vegetation (Penka et al. 1985, 1991). In eutrophic habitats, they are dominated by softwood trees (e.g., *Salix, Populus*) at low elevations above the normal water table, while hardwood trees (e.g., *Fraxinus, Ulmus, Quercus*) dominate at higher elevations (Fig. 2d). In oligotrophic habitats, the dominant softwood trees tend to be *Alnus* and *Salix.* Over large areas of all European floodplains, alluvial forest has been forced to give way to plantations of fast-growing trees (e.g., introduced cultivars of *Populus* or *Eucalyptus*), or to more or less intensely managed alluvial grassland. Local drainage has even enabled crop cultivation at places. Large areas of floodplain forests are still preserved in various European floodplains (e.g., the Rhine in Alsace, the Danube near Vienna, the Morava/March and Dyje/Thaya rivers in southern Moravia, western Slovakia and Lower Austria, the Drava and Sava rivers in Slavonia). Diverse algal vegetation as well as species-rich assemblages of fish and amphibians occur in the still preserved alluvial backwaters, oxbow lakes and pools (e.g., Pechar et al. 1996; Prach et al. 2003).

Unfortunately, only few floodplains or their segments have preserved their natural or semi-natural structure and dynamics, as a result of large-scale straightening and channelization not only of larger rivers, but also of small watercourses all over Europe during the last 200 years (Purseglove 1988; Haslam 2008). As to large European rivers, segments of natural or near-natural floodplains remain, e.g., along the Danube, Rhine, Elbe and Loire and some of their tributaries. On some sites, attempts have been made to restore the natural floodplain dynamics, e.g., on the Rhine in Alsace (France), on the Morava river in the Czech Republic, Austria and Slovakia or on the Elbe river in Germany.

The deltas and estuaries of large European rivers represent highly complex systems of habitats characteristic of floodplains, also with sand bars, tidal mudflats and lagoons with water that shows a gradient of salinity depending on the ratio between the freshwater and saltwater inputs to each particular zone or site of the delta or estuary at particular phases of the hydrological régime of the respective river and of the tidal régime of the respective sea (e.g., Rodewald-Rudescu 1974). This variation of environmental conditions is reflected in a high biodiversity of the deltas and estuaries, unless they have been heavily modified by water engineering. The largest and relatively well preserved European deltas are those of the Danube and Volga, but valuable wetlands are also found in the Rhone and Ebro deltas or in the estuaries of the Rhone, Elbe, Oder or Loire.

Human-made wetlands

Human-made wetlands comprise diverse types of human-made biotopes created for various purposes. For instance, drainage or irrigation ditches may constitute the last remnants of formerly large wetland areas. Paddy rice fields can be found in southern Europe where most of them occupy former natural wetlands. Buffer zones involving natural wetlands, wetlands created for capturing agricultural runoff, and constructed wetlands designed for wastewater treatment, have received increasing attention since the 1980s from both the technological and scientific points of view and presently occur in most European countries (Vymazal et al. 1998). Artificial lakes of all sizes have been created for various reasons in river floodplains. Provided they are in a good ecological state, their littoral zones have the potential to host littoral and submerged vegetation which is very similar to that of natural lakes.

In terms of area, shallow lakes created for fish rearing, or fishponds, probably represent the largest proportion of artificial wetlands in Europe. They have been constructed since the Middle Ages in countries of Central Europe as well as France, Serbia or Ukraine. In the Czech Republic, which does not have large natural lakes, the fishponds represent about 50 % (or 560 km^2) of the country's total wetland area. Although the fishponds were constructed mostly for fish rearing in the course of history (Šusta 1898), they have successively become harmonious parts of the surrounding landscapes and have evolved into ecosystems in many respects similar to natural shallow lakes (Dykyjová and Květ, 1978; Kořínek et al. 1987; Kubů et al. 1994; Pechar et al. 2002). In addition to fish production, they have provided numerous additional ecosystem goods and services such as flood control, water retention, modification of local climate and enhancement of biodiversity (Hejný et al. 2002; Pechar et al. 2002). These benefits were the main reasons for declaring the well preserved fishpond-rich landscape of the Třeboň Basin (Czech Republic) a biosphere reserve by UNESCO (Květ et al. 2002).

Other inland wetlands

Temporary freshwater pools, ponds and marshes (both natural and human-made) are also abundant and represent, together with rice fields, a well recognized type of habitats and ecosystems (European Commission 1991) as they contain a diverse and distinguished flora and fauna. Particularly valuable plant species, often with a short life cycle, occur on emerged bottoms, shores or banks of both standing and running waters. When inundated, their propagules can survive long periods in a dormant state (see, e.g., Hejný 1960, 1969; Hejný et al. 1998, for weeds of rice fields and temporary vegetation of emerged fishpond bottoms and shores in Central Europe). In areas with a continental climate and salt-rich subsoils, the temporary wetlands exhibit a slight to medium salinity (e.g., Löffler 1982 for the so-called "Lacken" in the Seewinkel near Lake Neusiedlersee).

Wetland research and university teaching of wetlands ecology

Wetland research has a relatively long tradition in Europe. It has developed simultaneously at several scientific centres of marine and coastal ecology, limnology, telmatology (i.e., peatland science) and aquatic botany or zoology, especially since the times of the IBP (International Biological Programme, 1965 – 1974, see Westlake et al. 1998). Research centres where wetlands are studied can be found in most European countries. The level of wetland research (both fundamental and applied) is generally high at these centres although the emphasis on various aspects of wetland ecology varies among them. European authors have either written, edited or significantly contributed to several textbooks or handbooks devoted to wetlands ecology and management (e.g., Moore and Bellamy 1974; Gore 1983a,b; Moore 1984; Whigham et al. 1993; Paavilainen and Päivänen 1995; Fustec and Lefeuvre 2000; Vymazal 1995, 1998, 2006; Westlake et al. 1998; Charman 2002; Jeglum and Rydin 2006; Haslam 2007). University education in wetlands ecology is carried out within the curricula of a number of European universities, albeit not always in courses so entitled. Quite often, for example, the courses of limnology deal also with wetlands.

The Society of Wetland Scientists has relatively recently (in 2004) established its European chapter whose annual meetings (since 2006) aim at becoming a representative forum of European wetland scientists. The Wetlands Working Group of INTECOL has a broad base of collborating wetland scientists who gather at International Conferences on Wetlands every four years. Three out of eight of these Conferences held so far took place at the European wetland research centres at Třeboň (Czechoslovakia), Rennes (France) and Utrecht (The Netherlands).

Wetland uses

Wetlands have been exploited and/or traditionally managed for various purposes since the very beginning of the human settlement in the area (Haslam et al. 1998; Löffler 1990 and another ten chapters (17 to 26) in Patten 1990, 1994). Many of the traditional uses such as fishery, harvesting of reed, mowing of wet grasslands, hunting and floodplain forestry have locally been preserved till today. These uses are considered sustainable provided their extent and technology comply with the carrying capacity of the ecosystems (Verhoeven et al. 2006). The same applies to one of the recent wetland uses by modern society, i.e., soft tourism. The intensity of research also has to be adjusted to the ecological sensitivity and resilience of each wetland studied.

Along with the above-mentioned (potentially sustainable) uses, various types of unsustainable wetland uses occur (Williams 1990). They cover peat and sand or gravel extraction and drainage for agricultural or forestry use. Although these uses have occurred all through the history of the human settlement in Europe, both their extent and impact have dramatically increased since the middle of the 19th century.

In many densely inhabited regions, most nutrient-rich waterlogged sites with mineral soil as well as fens were drained for agriculture quite early (alluvial sites along the River Po in Italy are among the earliest documented). All uses involving drainage lead to decreased beta-diversity in the flora and fauna (e.g., Laine et al. 1995; Vasander et al. 1997). Agricultural use also has lead to the loss of the carbon sink function – subsidence and a gradual loss of soil organic matter. Agricultural use involving both drainage and nutrient enrichment affects the sites with organic soils differently from those with mineral soils. While sites with mineral soils rapidly respond by changes in plant diversity, on sites with organic soils the peat or humus mineralisation enhances the CO_2 efflux from the soil. Forestry use is somewhat less aggressive, since a plant cover with new C inputs into the soil is maintained for most of the time. The effect of forestry use on the C sink function varies with wetland type and climate, often leading to C loss from the soil, but in some relatively poor peatland types in Fennoscandia, a C sink may be maintained. Forested drained sites are somewhat easier to restore than the more disturbed agricultural sites.

Extraction of peat is always linked with a lowering of the water table on the respective sites. Thus, apart from direct loss of the peat, another type of loss comes into question, namely that due to mineralization of the peat, which leads to further subsidence of the extracted mire surface. The speed of this subsidence varies from place to place according to both climatic and soil factors. It may also be substantial in peatlands used for intensive agriculture. The most striking examples of peat subsidence originate from regions where peatlands were drained in the late middle ages or soon afterwards (the Netherlands, East Anglia). In the Netherlands, subsidence mainly through oxidisation and compaction occurred at a rate of about 0.3 m per hundred years. Consequently, the main embanked rivers were soon flowing some 1.5 to 2 m above the general level of the peat, a difference that has now increased to 3.5-4 m (Williams 1990).

Both riverine and lacustrine wetlands occurring in Europe tend to be affected by large-scale eutrophication (e.g., Phillips 2005) occurring under the impact of agricultural and forestry management of their catchments, effluents from human settlements, feedlots and industrial plants, and atmospheric deposition, especially of nitrogen compounds. Most European wetland restoration projects aim at mitigating the effects of eutrophication of various types of wetlands – from floodplains and shallow lakes to wet grassland – on their ecosystem structure and functioning. Most successful are such projects that succeed in reducing the input of organic pollutants and mineral nutrients from whole catchments, also including the most important point sources of these substances or intentional fertilization of wetland habitats (Jörgensen and Löffler 1990; Eiseltová 1996; Eiseltová and Biggs 1995; O´Sullivan and Reynolds 2004, 2005; Verhoeven 2006).

Other threats include land filling, building of navigation canals, accelerated water discharge caused by straightening of watercourses, permanent inundation by reservoirs, fragmentation of residual wetland biotopes, pollution. In addition, an unproved assumption that all wetlands are important sources of greenhouse gases (especially CH_4) and therefore speed up the climate change, may be misused as an argument for further drainage of wetlands.

Wetland conservation and restoration

In spite of many destructive uses of wetlands in Europe, important activities exist there, whose aim is to protect or even restore wetlands. At the international level, the European Union has signed international conventions aimed at nature protection, including the Ramsar Convention on the Conservation of Wetlands (www.ramsar.org), the Bonn Convention on Migratory Species (www.cms.int), and the Rio Convention on Biological Diversity (www.biodiv.org/convention/default.shtml). To date, the Ramsar Convention is the primary basis for the conservation of most valuable wetlands in Europe. There are 47 contracting parties to the Ramsar Convention in Europe (of the world's total of 160), which have designated 898 European wetlands of international importance. These 898 sites represent about 50% of the total number of all Ramsar sites worldwide. However, these sites occupy only 14% of the area of all Ramsar sites of the world. This fact reflects the fragmentation of the still existing wetlands but, at the same time, also a fairly strong public awareness of the wetlands values. The Montreux record, listing Ramsar sites exposed to actual or potenrial unfavourable changes in the past, present or future times, comprises 23 European Ramsar sites.

Other administrative and legislative tools have strengthened or enlarged the impact and extent of wetlands conservation and wise use at the EU level and in its particular member countries. At the European level, the Bern Convention (www.coe.int/T/E/Cultural_Co-operation/Environment/Nature_and_biological_diversity/ Nature_protection/) has led the development of policy and action in nature conservation in Europe. It lists protected species and requires its parties to prevent the disappearance of endangered natural habitats including wetlands. Within EU legislation, the Birds Directive (79/409/EEC) and the Habitats Directive (92/43/EEC) have been promoted to rectify or reduce damage to European natural habitats and associated species. The Birds Directive is aimed at the protection of endangered bird species through designation of areas where these species are given special protection. Following the same principle, the Habitats Directive is aimed at the conservation of wild fauna and flora on the European territory on the basis of protection of their natural habitats. Following the criteria set out in the directives, each Member State must draw up a list of sites hosting the wild species of fauna and flora and put in place a special management plan to protect them, combining long-term preservation with economic and social activities, as part of a sustainable development strategy. Special Protection Areas for Birds (SPAs) and Special Areas of Conservation (SACs) are designated according to the Birds Directive and the Habitats Directive, respectively, and approved by the European Union to become part of a European Ecological Network called Natura 2000. By December 2008, 24 831 sites belonging to 27 European State Members covering 859 411 km^2 are included in this Network (http://ec.europa.eu/environment/nature/natura2000), which represents 17% of the whole European territory. Wetlands are particularly important in the Natura 2000 network.

Indirect protection to a variety of habitats also comes from EU Directives that regulate water quality, especially the Water Framework Directive (2000/60/EC). Additional policies concern coastal and marine areas (Airoldi and Beck 2007).

The European Directive (2007/60/EC) on the assessment and management of flood risks has recently been established to reduce adverse consequences associated with floods on the human health, the environment, cultural heritage and economic activity. For this, the European member countries ought to establish flood risk management plans based on flood hazard maps and flood risk maps at the scale of river basin by 22 December 2015. This European law clearly states that a preliminary flood risk should be assessed by December 2013 considering potential impacts of climate change, or to use already existing management plans on the occurrence of floods and the role of floodplains with respect to this risk, both in inland (river and lake floodplains) and coastal areas. While the objective of preventing and buffering damages to human health and economic activity may be encouraged, an adequate integration conservation of floodplains with their role as valuable natural ecosystems is not assured (Comín et al. 2008).

Of the Europe-based organisations taking care of the scientific basis for wetlands management, conservation and restoration, one should mention Wetlands International (www.wetlands.org) as a global science-based non-profit organisation dedicated solely to wetland conservation and sustainabloe development. Wetlands International, whose office is in Ede, The Netherlands, closely cooperates with the Ramsar Secretariat at Gland, Switzerland.

The basic principles of wetland conservation, restoration and creation are described, e.g., by Bobbink et al. (2006). There are numerous examples of successful conservation and restoration measures in European wetlands of all types (e.g., Gilman 1994; Janda and Ševčík 2002; Bragg et al. 2003; Farrel and Doyle 2003; Vasander et al. 2003). They are based largely (though not solely, see Sliva and Pfadenhauer 2003, Gorham and Rochefort 2003) on the successful conservation or restoration of the respective wetland's hydrological regime. New wetlands have spontaneously developed or have been created, e.g., in the littoral zones of artificial lakes (e.g., Rajchard et al. 2008) or in association with reclaimed Dutch polders, such as Wolderwijd en Nuldernauw adjacent to South Flevoland polder (Anonymous 2003). The European Comission promoted the long-term programme Life-Nature which included many cases of wetland restoration during the last 15 years (D.G. Environment-EC 2007). This programme was responsible for the restoration of coastal and inland wetlands all around Europe and elsewhere as it involved also neighbouring countries. It is still operating, active and stimulating the cooperation of managers, stakeholders, scientists and landowners. International and national legislation primarily aimed at improving the quality of surface waters (e.g., the Convention on the Protection of the Rhine (http://www.iksr.org), Convention on the International Commission for the Protection of the Elbe (http://www.ikse-mkol.org), the Danube River Protection Convention (http://www.icpdr.org) as well as various river and floodplain restoration projects have indirectly contributed to the conservation of existing wetlands.

As 60-90% of the European wetland area disappeared during the last century (Mitsch and Gosselink 2000), there is an important deficit of wetlands with respect to earlier times and also to potential wetlands recovery. A practical limit to the official approach to wetland protection in Europe is prioritising between wetland protection and restoration on the one hand and agriculture and tourism exploitation on the other. Frequently immediate economic profit prevails over interests of nature conservation and restoration. The recent incorporation of further countries into the European Union could be an opportunity to integrate these wetland activities into the socio-economic development of these countries.

Climate change

Sea level rise

It is generally accepted that the global climate change will bring about a rise of water level in all seas. IPCC models estimate the global average rise at about 3 to 4 mm per year. The highest sea level rise is expected in the Arctic region, thus affecting also the northern coast of Europe (Scandinavian countries and Russia) (Meehl et al. 2007). The local sea level rise will further be modified by vertical land movement. Taking vertical land movement into account gives slightly larger sea level rise projections relative to the land in the more southern parts of the UK where land is subsiding, and somewhat lower increases in relative sea level for the north. We have, for example, derived projected relative sea level increases for 1990–2095 of approximately 21–68 cm for London and 7–54 cm for Edinburgh (5th to 95th percentile for the medium emissions scenario) (Lowe 2009). The sea level rise will bring about also increased frequency and amplitude of extreme sea level events. This increase is also determined by the atmospheric storm intensity and movement and coastal geometry. Within Europe, increases in extreme sea level events are to be expected along the continental North Sea coast (Christensen et al. 2007), thus affecting costal areas of all countries from Denmark in the North to northern France in the South.

Temperature and precipitation

According to most regional climate change models, the annual mean temperatures are likely to increase more in Europe than is the global mean increment. In addition, spatial and temporal differences in the intensity of the warming will be substantial. In winter, the largest warming is likely to take place in northern Europe. In summer, on the other hand, the maximum temperatures are likely to increase most in southern Europe (Christensen et al. 2007).

Annual precipitation is expected to increase in northern Europe but decrease in most of southern Europe. The seasonal patterns may, however, be more important than average annual sums of precipitation. In northern Europe, the increased annual precipitation will be caused mainly by increased precipitation in the winter months. Nevertheless, water input from increased precipitation will be offset by the effects of higher temperatures: because of higher winter temperatures, the snowy season is likely to be shorter and the snow depth will probably decrease over much of northern Europe. Also, the increased evapotranspiration owing to higher summer temperatures is expected to override the increased summer precipitation. Consequently, summer drought will probably be the most important stressful effect of the changing climate on inland wetlands. Its risk will penetrate further northwards in comparison with the present-day situation. The frequency and intensity of summer droughts is most likely to increase from the north to the south.

Meteorological extremes

More frequent occurrence of extreme meteorological conditions (temperature, precipitation, air humidity) is envisaged and may be more important than the overall trends. Depending on the local and regional climate character, the resulting meteorological events may include strong winds, heavy rains possibly followed by floods, a greater frequency of extremely high temperatures for a given region, and longer periods without precipitation. Also, events of low frequency but intense ones (e.g., droughts) are important phenomena related with climate teleconnections (e.g., Atlantic and tropical air pressure oscillations such as El Niño Southern Oscilation) which regulate the dynamics of many inland wetlands all around Europe (Rodo et al. 1997; Rodo 2003).

Anticipated effects of climate change on wetlands

One of the first assessments of possible climate change effects on wetlands can be found in Boer and de Groot (1990). Our considerations in further text are largely in agreement with their assumptions given on pages 41-46. The authors regard the sea-leve rise, changing air and water temperatures and evaporation to precipiration ratios as the main driving forces affecting wetlands.

Sea level rise

Among various impacts of the ongoing climate change, the sea level rise will probably be the most important factor affecting coastal wetlands (mainly mudflats and salt marshes) because of the strong dependence of these habitats on water-level fluctuations and tidal régimes. It has been suggested that the projected sea-level rise could cause the loss of up to half of the present European coastal wetlands, with some of the largest losses expected to occur around the Mediterranean and Baltic Seas (Airoldi and Beck 2007 and references therein). With a higher sea level, salt water will penetrate deeper into estuaries, converting a part of brackish aquatic and wetland ecosystems into saline ones. At the same time, some freshwater wetlands connected with the sea will become brackish.

In a natural coastal zonation, the sea level rise would just cause a landward shift of all wetland zones. This is, however, unlikely to happen in the densely populated coastal areas of Europe, because most of the suitable upland areas are already used by people for various purposes. The coastal wetlands may therefore be sandwiched and squeezed between the shifting boundary of the shoreline on the seaward side and the fixed boundary given by the current land use on the landward side (Doody 2004). In some areas along the Atlantic coast (mainly in the Netherlands), the advance of sea water will be resisted by building new or strengthening the existing barriers. In these areas, the hydrology of the remaining wetlands would be fully controlled by the associated technical measures and the space left for wetlands will again depend on priorities of land use. Provided the coastal wetlands are given sufficient priority, their anticipated loss can theoretically be minimized or compensated for by political and socio-economic tools such as wise and timely land-use planning and consequent management measures. This will possibly happen in some large protected areas such as the Wadden Zee (Fig. 1), which has received a continued attention by both nature conservationists and national and local administrations (Hofstede 2003). Outside the strictly protected areas, we must fear that the area of coastal wetlands will be forced to shrink.

This will be the case if a defensive short-term strategy (reactive strategy according to the Millenium Ecosystem Assessment, Finlayson et al 2005) is followed in order to avoid sea water intrusion into the coastal zone. Such approach will be sustainable neither economically nor ecologically (with respect to the preservation of a healthy coastal zone). As an alternative, an adaptive long-term strategy (proactive strategy according to Finlayson et al 2005) can be adopted, which should include re-allocation of land uses and re-definition of services provided by the coastal zone ecosystem. A socio-economically acceptable compromise would probably be a mixed strategy, establishing defensive structural measures where important social assets are established and let the coast dynamics orientate future distribution of ecosystems and land uses. In the socio-economic context, one should consider that coastal wetlands provide a high value (Martinez et al. 2007) to the coastal zone inhabitants including protection against storms and other impacts of sea level rise induced by climate change.

Temperature increase

It is commonly accepted that the anticipated increase in temperatures will considerably affect both coastal and inland wetlands. The temperature increase will directly affect biological processes such as photosynthesis, respiration and transpiration. It will affect the biological processes also indirectly through changed physico-chemical properties of ecosystem components, such as changed solubility of various substances in water. In addition, the increased evaporation to precipitation ratio is expected to lead to lowered water levels and/or increased probability of drought.

The described climate development is generally unfriendly to inland wetlands, with increased summer dryness being the key factor. It will translate into wider water level fluctuations (both seasonal and irregular) and a generally greater water shortage in most wetland types over much of Europe. Maintenance or restoration of a hydrological régime ensuring the continued existence of any wetland will gradually become more and more difficult as it will require water supply from larger catchments or infiltration areas.

Boer and de Groot (1990) argue that the temperature rise and increased evaporation to precipitation ration could have a profound impact on inland wetlands because of internal eutrophication, salinization, dessication and invasion of thermophilous species. They conclude that the isolation of individual wetlands can increase because of the fragmentation of biocorridors as a result of water shortage.

Riverine wetlands, including those in estuaries and river deltas, may be reduced in area, especially in the South European and inland East European regions, as a result of decreased water discharge in rivers and streams in the growing season. The same is true for wetlands associated with lakes and other standing waters, where the water shortage will be associated with accelerated land-filling and a consequent establishment of terrestrial species of plants and animals. This development may eventually lead to a shift towards terrestrial ecosystems. The reduced water volume will also result in higher concentrations of dissolved nutrients and suspended solids in both running and standing waters. Additionally, higher temperatures will promote the mineralisation of soil organic matter resulting in an increased availability of nutrients in wetland soils.

In peatlands (both bogs and fens), the anticipated water shortage in summer will lead to lowered water levels and, thus, oxic conditions in a deeper surface layer, but also increasing dryness of the surface peat. Oxic conditions allow for an increased rate of decomposition of the organic matter contained in peat or peaty soil. But if the water level falls deep enough, dryness may impede decomposition in the topmost layers (Lieffers 1988; Laiho et al. 2004). Many anticipated effects will depend on the range of water level fluctuations. If dry and wet years alternate, increasing the instability of water levels, the systems will enter a stage of "constant disturbance", with a limited number of plant species tolerating both extremely wet and extremely dry conditions, forming distinct community compositions (Laitinen et al. 2008). In such cases, C sequestration can cease, and extensive C loss from soil may take place, since the "best" C accumulators disappear from the plant community, and decomposition during dry periods may compensate and even exceed any accumulation during wet years. An essentially similar situation ~~as~~ has been found during dry years in contemporary mire ecosystems (e.g., Schreader et al. 1998; Alm et al. 1999a; Moore et al. 2002). Leaching of dissolved organic carbon and nutrients may be accelerated as in other wetland types. More or less permanently lowered water levels, on the other hand, will lead to a "forest succession" with increasing abundance of shrubs and trees (e.g., Laiho et al. 2003), except for bog sites so poor in nutrients or so cold that increased tree growth is not feasible (Vasander 1982). The succession will continue until a new equilibrium between the vegetation composition and the new water level régime has been achieved, which may take several decades. This development will lead to changes in the runoff patterns, and may eventually lead to decreased leaching of DOC and some elements such as K, whereas the leaching of other elements, such as Ca and Mg, may increase. The drier systems will become more acid. They will lose most of their specialized wetland vegetation, which will be replaced by common forest species. This change will lead to lowered beta-diversity (e.g., Laine et al. 1995; Vasander et al. 1997). In most cases, C loss from soil can still take place, even though its rate may slow down as the decomposition potential of the exposed peat decreases (Jaatinen et al. 2008). On the most productive bog sites with increased tree growth and, consequently, litter inputs, but still relatively poor substrate quality, the C sink function may continue, at least at the higher latitudes. While C losses are likely to increase in the temperate and southern boreal regions, C sequestration may increase in the subarctic regions. The palsa mires (with local permafrost formations), specific for cold regions, may disappear, and permafrost melt in the northernmost Scandinavia and northern Russia, especially, may lead to yet partly unpredictable changes.

As a result of temperature changes and the associated changes in water availability, the latitudinal zonation of different peatland types may change considerably. However, predictions differ according to the presumed driving forces. Crawford (2008) stresses that the greater increase in winter temperatures than in summer ones will lead to an expansion of oceanic climate into northeastern Europe and Siberia. This in turn may support a southward expansion of *Sphagnum*-dominated mires in spite of a northward expansion of boreal forest, as it has been commonly assumed.

Prolonged dry periods which have been observed in southwestern Europe since the second half of the 20th century can change the spatial distribution of wetland habitats, particularly inland wetlands. An example is Lake Gallocanta in Aragon, NE Spain, a playa lake in a closed endorheic basin. It serves as a climatic sensor with its water level mostly fluctuating in accordance with its climate regulated water balance. More frequent and prolonged dry periods have been observed in Lake Gallocanta in accordance with global climate change (Rodo et al. 1997). More frequent and prolonged dry periods will turn this temporary wetland, an area of high biodiversity at the European scale, into a dry salt pan (Comín et al. 1991).

The fire hazard will increase especially in summer-dry Mediterranean wetlands as well as peatlands with a dried-out surface vegetation and peat layer.

Further interacting effects

There is still much controversy on the relative importance of the impacts of climate change *versus* current direct human global changes on wetlands ecology, as for other types of ecosystems and the Earth ecosystem (Vitousek et al. 1986, Fig. 4). However, in contrast with suggested impacts of the climate change, many negative impacts of changes in land use and land cover on European wetlands have already been demonstrated and quantified (Anderson 2008). So, the question may be on the interaction between climate change and other global changes rather than specifically on the impacts of temperature and rainfall changes on wetland functioning. And the response should discuss prioritization of objectives not to continue wetland decline, which took place at high rate in the last century (Mitsch and Gosselink 2000).

Land use and land cover may significantly affect climate at the regional and local scales. Recent modelling studies also show that in some instances these effects can extend beyond the areas where the land cover changes occur, through the teleconnection processes. (Christensen et al. 2007). For the fate of wetlands in a changing climate, various interacting effects may become more decisive than the anticipated increased impact of the atmospheric greenhouse effect. Drainage of wetlands is a more potent driver of local climate change than the changed greenhouse gas balance. The reduced transformation of incoming solar radiation into latent heat of evaporation leads to increased overheating of dried surfaces. This holds not only for "reclaimed" wetland areas, but for drained and urbanised areas in general (Denman et al 2007). An extreme situation is represented by urbanised areas, which create urban heat islands associated with considerable warming (Arnfield 2003). The rapid urbanization of the European landscapes (Antrop 2004) cannot leave the mostly fragmented European wetlands unaffected.

Socioeconomic trends resulting from the public perception of climate change may singificantly interact with the direct impacts of the changing climate. The socio-economic priorities are likely to differ between regions exposed to different main impacts. Preservation of carbon storage is a key issue for the northern part of Europe, where large areas of peatlands occur. Extreme meteorological events and their consequences such as downpour rains followed by floods are likely to be perceived most sensitively in Central and Western Europe. They may promote public requirements for technological (hard) flood control measures resulting in faster water discharge, which would threaten the hydrology of existing wetlands. Continental and south European wetlands will probably suffer most from water shortage. Consequently, competition for water between agriculture and urban land use on the one hand and environmental protection on the other hand may substantially reduce the water supply to wetlands.

Need for a change in the perception of wetland values

In recent years, the scientific community has contributed to the formation of environmental policies by synthesising scientific knowledge in the form of background materials addressed to decision-makers. Apart from scientific knowledge, these documents incorporate the elements of strategic considerations (scenarios, strategies), which facilitate the assessment of the limits and/or alternatives of future development (e.g., Finlayson et al 2005). The principles of nature and ecosystem conservation, which are generally accepted as social priorities, have, under European conditions, the chance to raise funds for maintaining the desirable state or restoration of valuable sites including wetlands. The financial means invested in this way have already brought visible results (http://ec.europa.eu/environment/life).

Although there has been a considerable improvement in the human attitude toward wetlands over the last decades (especially in areas where most wetlands had previously been lost), the fast climate changes and their anticipated impacts call for further change in the human perception of wetlands. The impact of climate change on biodiversity has long been of widespread concern. In addition, however, it is worth considering that there is a feedback relationship between the wetland ecosystems (the same as any living systems) and their environment including climate. This becomes particularly important for large wetland areas such as boreal peatlands and deltas of large rives. This feedback relationship encompasses not only the greenhouse gas balance, which is in the focus of attention today, but also the climate stabilization through the airconditioning effect of evapotranspiration. In addition, the specific features of wetlands, such as their hydrology, predispose them for playing an important role in large landscape complexes, where their impact considerably surpasses their physical boundaries. This statement applies mainly to such hydrological functions of wetlands as water retention on the one hand and flood mitigation on the other.

It must be taken into account that a wetland function can be performed if the ecosystem is well established and that long-term water retention and flood mitigation in floodplains requires a dynamic floodplain (Comín et al. 2009). Otherwise, the negative impacts of artificially created infrastructures for water retention can override the water retention function and eliminate it in the long term.

Biodiversity

Biodiversity support is commonly listed as one of the important wetlands values (Mitsch and Gosselink 2000; Gopal et al. 2000, 2001). Under biodiveristy we do not undestand only a weighted variety of species, but also that of their life forms, habitats and niches occupied by them. In this respect, European wetlands are highly diverse ranging from acidic and nutrient poor bogs on the one hand to highly fertile and productive wetlands in estuaries, salt marshes and freshwater littoral or riparian wetlands on the other. Some wetlands (such as many mires and springs) are island ecosystems *sensu* MacArthur and Wilson (1967). They serve as refuges of rare and relic species and their relative isolation in the landscape may promote microevolution of specialized phenotypes. Ecotonal wetlands (such as littoral reed belts or riparian wetlands) host species both from the adjacent larger-scale ecosystems and species which are confined only to the ecotone itself (Naiman and Décamps 1990). The biodiversity has also a time dimension associated with water level fluctuations (Hejný 1957; Hejný et al. 1998). When the water table sinks to or below the ground surface, specialized plant species often appear, which can survive long-term flooding in the bank of dormant propagules. Examples are the communities of emerged shores or bottoms of lakes, pools and ponds (e.g., Hejný and Husák 1978; Rejmánek and Velasquez 1978; Hroudová 1981; Prach et al. 1987; Šumberová et al. 2005, 2006). Their preservation in European landscapes is enabled by harmonising the water level fluctuations with the species requirements during their life cycle[1].

Rajchard et al. (2008) have suggested that littoral zones of quarries and sandpits formed by surface mining provide oligotrophic habitats for wetland species that are endangered and disappearing from the surrounding eutrophicated wetlands in intensely managed areas. In reality, the fulfilment of this potential depends on other factors such as shore morphology and intensity of recreational use).

[1] This is true, e.g., for *Coleanthus subtilis* (Tratt.) Seidl, a tiny (3 to 11 cm tall) annual grass which occurs on the bare or almost bare soils of emerged lake or pond bottoms and shores after its dormant caryopses have survived a long-term flooding of the biotope. The life cycle of the shoots of this grass lasts only 4 to 6 weeks, and the plants usually flower and fruit in June and July. The reproduction and hence also survival of *C. subtilis* at each particular site of its occurrence is ensured by a periodical drawdown of the water table at that time of year. One plant can produce over one thousand ripe caryopses. In Europe, its geographical range of occurrence is narrow, covering only Central Europe and within it especially the basin of Třeboň and adjacent areas in the Czech Republic and Lower Austria. *C. subtilis* has thus become one of the 434 plant species protected within the EU "Natura 2000" framework (Habitat Directive 92/43, Annex 2). It is also listed in the Red List of threatened plants of the Czech Republic and in the IUCN List of Threatened Plants. The central area of its European occurrence lacks natural lakes, but is rich in artificial fishponds where the water table can be set at a certain level at any time. For securing permanent occurrence of *C. subtilis* within this area, agreements have therefore to be made with the fishpond owners as to the occasional maintenance of a low water table in the respective fishponds at the optimum of this species´ seasonal development. Such an arrangement can result in a certain loss of the fish crop in a fishpond whose water area is temporarily diminished by the drawdown (summer drainage is not any more a regular part of the fishpond management in Central Europe), and provisions have to be made for a financial compensation of this loss. The obtaining of reliable data on the occurrence of *C. subtilis* on a certain territory thus requires a period as long as several years. It is advantageous that the protection of *C. subtilis* at any site brings with itself the protection of the whole rather rare plant community colonizing the emerged pond bottom or shore. For a thorough treatment of the biology and ecology of *C. subtilis* see, e.g., Hejný (1969).

To date, beta diversity of wetlands seems to be broadly accepted by the European population. Since it is reflected by both national and EU legislations, it frequently provides the most powerful argument for protecting a particular wetland site. This argument will become even stronger in the near future as a result of the implementation of the EU Habitat Directive (NATURA 2000). Because of the administrative feasibility of this approach, biodiversity is used in advocating protection of wetland sites whose other values (see the text below) are obvious but are not protected by legislative tools. Such substitute arguments for wetlands conservation are often only unwillingly accepted by the predominantly technocratically oriented decision-makers. Development of complementary legislative and administrative tools, based on the assessment of all wetland functions in the landscape, is a pre-requisite of establishing a more balanced basis for sustainable decision-making concerning wetlands.

Greenhouse gas balance

The greenhouse gas balance is currently in the centre of attention of both the scientific community and the general public because it is considered to be one of the main causes of the global climate change (Janssens et al. 2005). In contrast with terrestrial ecosystems, wetlands emit methane as an important component of their greenhouse gas budget (Segers 1998, LeMer and Roger 2001). The greenhouse gas balance of a wetland is the outcome of the rate of net CO_2 uptake (CO_2 sequestration) on the one hand and the rates of CH_4 and N_2O efflux (greenhouse gas emissions) on the other hand. This outcome, expressed as radiative forcing, may be either positive or negative depending on the rates of the processes involved. The dynamics of greenhouse gas exchange is largely determined by specific site conditions including hydrological conditions, soil type, vegetation, and management and meteorological and climatic conditions. Depending on meteorological conditions, wetlands (the same as other ecosystems) may act as CO_2 sinks in some periods and as sources in others. The emissions of CH_4 and N_2O from wetlands are similarly variable in time.

Compared to other terrestrial ecosystems of Europe, especially forests (Janssens et al. 2005), less information is available on the greenhouse gas balance of wetlands. Among wetland types, *Phragmites*-dominated wetlands are understood relatively well (Brix et al. 2001). Case studies have been published for boreal sedge fens (Aurela et al. 2004; 2007), temperate wet grasslands (Hendriks et al. 2007, Dušek et al. 2009) and constructed wetlands for wastewater treatment (Picek et al. 2007). Special attention has been paid to the determinants of methane dynamics (Kaki et al. 2001; Kankaala et al. 2003; 2004; Rinne et al. 2007).

There is insufficient information as yet needed to provide simple guidelines for management aimed at achieving a positive balance of greenhouse gases in the existing variety of wetland types. Yet, the current knowledge provides a basis for some important generalizations. Of all natural wetland types, peatlands are by far the most important ecosystems affecting the global balance of greenhouse gases. Peatlands globally represent a highly important store of carbon, sink for carbon dioxide and a significant source (from the point of view of its importance for the greenhouse effect) of atmospheric methane. In general, nitrous oxide (N_2O) emissions are small in natural peatlands (Joosten and Clarke 2002). In addition to live peatlands (mires), littoral wetlands with abundant plant cover, such as reed (*Phragmites australis*) dominated marshes in Central and North Europe, can be important sinks for carbon (Brix et al. 2001). Floodplains can play an important role by accumulating organic matter and carbon if floods are maintained and the river-floodplain connectivity lets the plant communities (especially riparian woodlands) develop at an integrated ecohydrological rhythm (Cabezas et al., 2009).

Two types of impact considerably affect the greenhouse gas balance of wetlands: changed hydrology and nutrient enrichment. More frequent summer droughts increase the frequency of situations under which wetlands, especially peatlands, act as sources of CO_2. At the same time, the CH_4 emissions decrease. There is also evidence that peatlands “reclaimed” for agricultural use are releasing significant amounts of nitrous oxide (N_2O) because they have become enriched with mineral nutrients including nitrogen. Long-term nutrient enrichment of wetlands with organic soils can also promote CO_2 efflux. (Zemanová et al. 2008). Eutrophication of permanent wetlands associated with standing waters can promote anaerobic decomposition processes including methane production.

Generally, it is important to consider that wetlands have been both taking up and releasing greenhouse gases continuously since their formation and thus their influence on the atmosphere must be modelled over time. When this is considered, the sequestration of CO_2 in peat outweighs the CH_4 emissions. In terms of greenhouse gas management, the maintenance of large carbon stores in undisturbed peatlands should be a priority, as recently pointed out by Joosten and Clarke (2002).

Climate stabilization

The air-conditioning effect of the water cycle has been recognised as one of positive functions of vegetated areas sufficiently supplied with water such as most wetlands (Mitsch and Gosselink 2000). Its importance has been demonstrated by a negative practical experience: drainage associated with changed land use has been recognised as one cause of climate change at local and regional scales (Christensen 2007, Denman 2007). Yet, no information exists as yer about its importance for asssociated changes in energy fluxes at the global scale.

Evapotranspiration of water from vegetated surfaces not only increases air humidity, but also cools surfaces from which water vapour evaporates by the amount of energy needed for vaporisation (latent heat). The evaporated water vapour then condenses in cool air or on cool surfaces, which thereby receive the energy of the released latent heat. In this manner, the evaporation and condensation processes have a double air-conditioning function - plant stands are cooled in the evapotranspiration process while heated are places where the water vapour precipitates. Evapotranspiration possesses a huge capacity to equalize temperature differences in time and space. This air-conditioning effect is associated with enormous energy fluxes[2]. Kravčík et al. (2008) explain the effect of drainage on temperature extremes and the role of water and wetlands in mitigation of climate change.

As a consequence of the cooling effect of evapotranspiration, the vegetation cover well supplied with water is substantially cooler than adjacent dry surfaces (Pokorný 2001, Ripl 2003, Brom and Pokorný 2009). This is conveniently documented by means of thermo-vision. While the surface temperature of vegetation well supplied with water is close to that of the ambient air, dry surfaces can be warmer by 10-30 ^{o}C (Fig. 5). On a larger scale, the effect of vegetation on temperature distribution can be shown by means of satellite pictures in the thermal IR spectrum (Fig. 6). The use of satellite pictures for evaluation of indicators of landscape ecological functions based on temperature, biomass and humidity is described by Hesslerová and Pokorný (2009).

The predicted more frequent summer droughts are likely to bring about a general decrease of evapotranspiration from most vegetated areas: the heat balance will be shifted towards an increased sensible heat flux (thus to a higher Bowen ratio) more often than now. The value of wetlands, as “oases” in dry landscapes, will therefore increase, provided they

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[2] Let us consider an herbaceous wetland which evaporates 6 l of water per 1 m^2 during a summer day. The solar energy consumed in evapotranspiration of 6 litres is equal to 4.2 kWh (latent heat of 1 litre of water = 0.7 kWh, or 2.45 MJ). This amount of energy represents an average 24-h flux of about 180 $W.m^{-2}$. Expressed per an area of 5 km^2, the latent heat flux equals 900 MW, which is equivalent to the power output of a large electric power station.

remain saturated with water. This is particularly important in river and stream floodplains, where the water supply to wetlands can partly be subject to human control.

Flood mitigation

Flood mitigation has already obtained increasing public and scientific attention in connection with the extreme flood events in recent years. The risk of floods will become even more important in the near future because of the anticipated wider water-level fluctuations. This gives an opportunity to reconsider the hitherto only theoretically appreciated role of wetlands in flood mitigation. This consideration pertains especially to non-degraded peatlands and floodplains. The case study of the extreme summer flood in 2002 (Lhotský 2006, Fig. 7) demonstrates the potential of both natural and man-made wetlands for flood mitigation. In comparison with drained land, wetlands have a considerably greater ability to attenuate peak discharges (Procházka et al 2009, Fig. 8).

Large water-unsaturated peatlands can function as water stores and retain extreme rainfall. The retention capacity of peatland ecosystems is higher during the vegetation season than in winter because of the periodic sinking of the groundwater table due to evapotranspiration (Kolmanová et al. 1999, Fig. 9). If, however, a peatland ecosystem is strongly influenced by drainage and/or opencast peat mining, the surface peat layer is frequently dry and impermeable. Such a peat has a low capability to absorb the rainfall water. In such cases, all surplus water is quickly discharged from the area. Even partial drainage reduces the potential of the peatland area to attenuate discharges following downpour rains. A ditch network accelerates the discharge, affecting the timing and intensity of the peak flow downstream.

Ecosystem services

The concept of ecosystem services may change the perspective, which the wetlands are perceived from. By services we mean different benefits or goods, which wetlands (or any other ecosystems) provide to the human welfare. In the Millenium Ecosystem Assessment (Finlayson et al 2005), ecosystem services are defined as “the benefits people obtain from ecosystems”. Flood and drought mitigation, water purification, carbon sequestration, biodiversity refuge, production of commodities (fish, reed, wood etc.), mitigation of storm effects, coastal erosion, and recreation are examples of services provided by wetlands (Mitsch and Gosselink 2000; Costanza et al. 1989; Jeník et al. 2002). These services can be expressed also in monetary values (Turner et al. 2008), although the introduction of economic tools would rise the level of complexity of these proposed ecological-economic systems. Financial evaluation of the services has at least three following consequences: (1) wetland values are more understandable to technically oriented decision-makers and the general public; (2) The values are additive, i.e., we can approach the overall value of a wetland and compare it with other types of land uses – which should alter the decision-making process in land use towards wetland promotion; (3) The valuation of particular services will help to understand their importance from the perspective of contemporary human welfare.

If the overall value of ecosystem services per unit area is compared among different world biomes, wetlands are quite valuable, especially river estuaries (first position) and floodplains (second position together with seagrasses, Costanza et al.1997). However, examples of complete calculation of wetland values are scarce. Successful practical solutions, based on the acknowledgement and application of this concept to the life of modern human society, remain rather a challenge for the future (Ruhl et al. 2007) than beeing succesfully applied in contemporary decision-making processes. The reason is the difficulty to connect a broad multidisciplinary ecological approach, encompassing the quantification of different processes (fluxes of water, carbon, nutrients, etc) ~~as~~ related to the ecosystem structure (biodiversity, land use), with the economic sphere (marketable and non-marketable services, values and prices, discount rates), which quite often results in an instinctive rejection of ~~the~~ environmental issues.

The case study of the Lužnice river floodplain (Pithart et al., 2008) may serve as an example of calculation of selected ecosystem services in a Central European wetland (Table 2). The results show a relatively small contribution of the production of commodities to the overall ecosystem value. On the other hand, the values of flood mitigation and biodiversity refugium are quite high. This assessment shows that revitalisation of floodplain segments within previously chanellised river beds surrounded by arable land may increase their value tremendously, because the value of ecosystem services depends strongly on regular flooding, and biodiversity depends on connectivity between the river bed and adjacent wetland areas of differnet elevation. Other services of this site, which are not covered by this study, but apparently exist (groundwater dotation, climate regulation, water purification) would even more enhance this point of view.

Conclusions

(1) Current status of wetlands

  (a) There is a strong deficit of wetland number and area compared to wetlands existing in the early 20th century. Consequently, the ecological restoration of wetlands is still a major activity to be performed to obtain the benefits of wetlands functions all around Europe.

(2) Response of wetlands to anticipated climate change

  (a) Sea level rise will be the main factor affecting coastal wetlands. It is also an opportunity for developing adaptive coastal management and recovering disappeared and degraded wetlands.

  (b) Wider water-level fluctuations will occur in most inland wetlands.

  (c) Human impact on wetlands (especially drainage) can strongly interact with or even prevail over the effects of climate change.

(3) Socio-economic aspects

  (a) Preservation of carbon storage should be the priority in the northern part of Europe, where large areas of peatlands occur.

  (b) Extreme meteorological events and their consequences such as downpour rains followed by floods are likely to be perceived most sensitively in Central and Western Europe. They may promote public requirements for technological (hard) flood control measures resulting in faster water discharge from the respective catchments, which would threaten the hydrology of existing wetlands.

  (c) Continental and south European wetlands will probably suffer most from water shortage. Consequently, competition for water between agriculture and urban land use on the one hand and environmental protection on the other hand may substantially reduce the water supply to wetlands.

(4) Need for a changed attitude toward wetlands:

  (a) The anticipated climate change imposes a threat to the current condition of European wetlands in addition to the historically existing and recently strongly increasing human impact.

  (b) At the same time, our facing the climate change encompasses an opportunity for developing adaptive wetland management and recovering disappeared and degraded wetlands.

  (c) Socio-economic appreciation of wetlands will be enhanced if the scientific community is able to develop a broadly accepted system of economic evaluation of wetland ecosystem services such as carbon sequestration, climate stabilisation, or flood mitigation.

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References

Adam P (1990) Saltmarsh Ecology. Cambridge University Press, Cambridge

Airoldi L, Beck MW (2007) Loss, status and trends for coastal marine habitats of Europe. Oceanography and Marine Biology: Oceanogr Marine Biol Annu Rev 45: 345-405

Alm J, Schulman L, Walden J, Nykänen H, Martikainen PJ, Silvola J (1999) Carbon balance of a boreal bog during a year with an exceptionally dry summer. Ecology 80, 161-174

Anderson, J. 2008. Climate change induced water stress and its impact on natural and managed ecosystems. European Parliament-Policy Department Economic and Scientific Policy, IP/A/ENVI/FWC/2006-172/LOTI/C1SC12

Anonymous (2003) Information sheet on Ramsar wetlands. Wolderwijd en Nuldernauw. Available via HTTP: http://www.wetlands.org/reports/ris/3NL042en.pdf. Accessed 3 October 2009

Antrop M (2004) Landscape change and the urbanization process in Europe. Landsc Urban Plan 67: 9-26

Arnfield AJ (2003) Two decades of urbane climate research: a review of turbulence, exchanges of energy and water, and the urban heat island. Int J. Climatol. 23: 1-26.

Aurela M, Laurila T, Tuovinen J-P (2004) The timing of snow melt controls the annual CO_2 balance in a subartic fen. Geophys Res Lett 31: L16119

Aurela M, Ruitta T, Laurila T, Tuovinen J-P, Vesala T, Tuittila ES, Rinne J, Haapanala S, Laine J. 2007. CO_2 exchange of a sedge fen in souther Finland - the impact of a drought period. Tellus 59B: 826-837.

Bloesch J (2004) Sedimentation and lake sediment formation. In: O'Sullivan PE, Reynolds CS (ed), The Lakes Handbook, Vol.1, Limnology and Limnetic Ecology.limnetic ecology. Blackwell Publishing Comp., Malden, Oxford, Carlton, pp 197-229

Bobbink R, Beltman B, Verhoeven JTA, Whigham DF (ed) (2006) Wetlands: functioning, biodiversity conservation, and restoration. Springer, Berlin

Boer MM, de Groot RS (ed) (1990) Landscape-Ecological Impacts of Climatic Change. IOS Pres, Amsterdam/Washington/Tokyo, 429 pp.

Bragg OM, Lindsay R, Risager M, Silvius M, Zingstra H (ed) (2003) Strategy and action plan for mire and peatland conservation in Central Europe, Central European Peatland Project (CEPP). Wetlands International Publ.18. Ede, The Netherlands

Britton RH, Crivelli AJ (1993) Wetlands of southern Europe and North Africa: Mediterranean wetlands. In: Whigham DF, Dykyjová D, Hejný S (ed) Wetlands of the World I, Kluwer Academic Publishers, Dordrecht/Boston/London, pp 129-194

Brix H, Sorrell BK, Lorenzen B (2001) Are *Phragmites*-dominated wetlands a net source or net sink of greenhouse gases? Aquat Bot 69: 313-324

Brom J, Pokorný J (2009) Temperature and humidity characteristics of two willow stands, a peaty meadow and a drained pasture and their impact on landscape functioning. Boreal Env Res 14: 389-403

Cabezas A, Comín FA, Walling DE (2009) Changing pattens of organic carbon and nitrogen accretion on the middle Ebro floodplain (NE Spain). Ecol Eng 35:1547-1558

Christensen JH, Hewitson B, Busuioc A et al (2007) Regional climate change. In: Solomon S, Qin D, Manning M et al (ed) Climate change 2007: The physical science basis. Contribution of Working group I to the Fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, pp 847-940

Comín FA, Alonso M (1988) Spanish salt lakes, their chemistry and biota. Hydrobiol 159: 237-245

Comín FA, Williams WD (1993) Parched continents: our common future? In: Margalef R (ed) Limnology now. A paradigm of planetary problems. Elsevier, pp 473-527

Comín FA, Calvo A, González M, Sorando R, Gallardo B, Cabezas A, Garcaía M, González E (2008) If flooding is the answer, what is the question?. In: Gumiero B, Rinaldi M, Fokkens B (ed) River restoration 2008. CIRF, Venice, pp: 513-518

Comín FA, Julia R, Comín MP (1991). Fluctuations: the key aspect for the ecological interpretation of saline lake ecosystems. Oecologia AquaticaOecol Aquat 10:127-135

Costanza R, Farber S C, Maxwell J (1989). The valuation and management of wetland ecosystems. Ecol Econ 1:335-361

Costanza R, d'Arge R, de Groot R, Farberk S, Grasso M, Hannon B, Limburg H, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253-260

Cowardin LM, Golet FC (1995) US Fish and Wildlife Service 1979 wetland classification: A review. Vegetatio 118: 139-152

Crawford RMM (2008) Plants at the margin. Ecological limits and climate change. Cambridge University Press, Cambridge

Denman KL, Brasseur G, Chidthaisong A et al (2007) Couplings between changes in the climate system and biogeochemistry. In: Solomon S, Qin D, Manning M et al (ed) Climate change 2007: The physical science basis. Contribution of Working group I to the Fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, pp 499-588

Doody JP (2004) Coastal squeeze: a historical perspective. J Coast Conserv 10: 138

Duever MJ (1990) Hydrology. In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 1. Natural and human relationships, SPB Academic Publishing, The Hague, pp 61-89

Dykyjová D, Květ J (ed) (1978) Pond littoral ecosystems. Structure and functioning. Ecological Studies 28, Springer-Verlag, Berlin/Heidelberg/New York

Eiseltová M (ed) (1994) Restoration of lake ecosystems. IWRB Publ 32: 1-182, International Waterfowl and Wetlands Research Bureau, Slimbridge, U.K.

Eiseltová M, Biggs J (1995) Restoration of stream ecosystems, IWRB Publ 37: 1-170. International Waterfowl and Wetlands Research Bureau, Slimbridge, U.K.

European Commission (1999). Interpretation manaul of European Commission habitats. EUR/5, 2nd edition. European Commission, Brussels

Farrel CA, Doyle GJ (2003) Rehabilitation of industrial cutaway Atlantic blanket bog in County Mayo, North-West Ireland. Wetlands Ecol Manage 11: 21–35

Finlayson CM, D’Cruz R, Davidson N et al (2005) Millenium ecosystem assessment. Ecosystems and human well-being: Wetlands and Water. Synthesis. World Resources Institute, Washington, DC

Fustec É, Lefeuvre JC (2000) Fonctions et valeurs des zones humides. Dunod, Paris, 426 pp

Gilman K (1994) Cors Erddreiniog, Anglesey: A case study of wetland conservation (North Wales). In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 2. Case Studies. SPB Academic Publishing, The Hague, pp 439-456

Gopal B, Masing V (1990) Biology and ecology In: In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 1. Natural and human relationships. SPB Academic Publishing, The Hague, pp 91-239

Gopal B, Hillbricht-Ilkowska A, Wetzel RG (ed) (1993) Wetlands and ecotones: Studies on land-water interactions. National Institute of Ecology and International Scientific Publications, New Delhi

Gopal B, Junk WJ, Davis JA (2000) Biodiversity in wetlands: Assessment, function and conservation. Volume 1. Backhuys Publishers, Leiden

Gopal B, Junk WJ, Davis JA. (2001) Biodiversity in wetlands: Assessment, function and conservation. Volume 2. Backhuys Publishers, Leiden

Gopal B, Květ J, Löffler H, Masing V, Patten B (1990) Definition and classification. In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 1. Natural and human relationships. SPB Academic Publishing, The Hague, pp 9-15

Gore AJP (ed) (1983a.) Ecosystems of the world 4A. Mires: swamp, bog, fen, and moor. General studies. Elsevier Scientific Publishing Company, Amsterdam

Gore AJP (ed) (1983b.) Ecosystems of the world 4A. Mires: swamp, bog, fen, and moor. Regional studies. Elsevier Scientific Publishing Company, Amsterdam

Gorham E, Rochefort L (2003) Peatland restoration: A brief assessment with special reference to Sphagnum bogs. Wetlands Ecol and Manage 11: 109–119

Haslam SM (2008) The riverscape and the river. Cambridge University Press, Cambridge

Haslam SM, Klötzli F, Sukopp H, Szczepański A (1998) The management of wetlands. In: Westlake DF, Květ J, Szczepański A (ed) The production ecology of wetlands. Cambridge University Press, Cambridge, pp 405-464

Hejný S (1957) Ein Beitrag zur ökologischen Gliederung der Makrophyten der tschechoslowakischen Niederungsgewässer. Preslia 29: 349-368

Hejný S (1960) Ökologische Charakteristik der Wasser- und Sumpfpflanzen in den slowakischen Tiefebenen (Donau- und Theissgebiet). VydavateĺstvoSAV, Bratislava

Hejný S (1969) *Coleanthus subtilis* (Tratt.) Seidl in der Tschechoslowakei. Folia Geobot Phytotax 4: 345-399

Hejný S (1971) The dynamic characteristics of littoral vegetation with respect to changes of water level. Hidrobiologia Bucuresti 12: 71-85

Hejný S, Husák Š (1978) Higher plant communities. In: Dykyjová D, Květ J (ed.) Pond littoral ecosystems. Structure and functioning. Ecological Studies 28. Springer-Verlag Berlin/Heidelberg/New York, pp 23-64.

Hejný S, Segal S, Raspopov IM (1998) General ecology of wetlands. In: Westlake DF, Květ J, Szczepański A (ed) The production ecology of wetlands. Cambridge University Press, Cambridge, pp 1-77

Hendriks DMD, Van Huissteden J, Dolman AJ, Van der Molen MK 2007. The full greenhouse gas balance of an abandoned peat meadow. Biogeosci 4: 411-424

Hesslerová P, Pokorný J (2009) The synergy of solar radiation, plant biomass and humidity as an indicator of ecological functions of the landscape: a case study from Central Europe. Integrated Env. Assessment and Monitoring, in press.

Hillbricht-Ilkowska A, Pieczyńska E (ed) (1993) Nutrient dynamics and retention in land/water ecotones of lowland, temperate lakes and rivers. Developments in Hydrobiology 82. Kluwer Acad. Publishers, Dordrecht / Boston / London

Hofstede JLA (2003) Integrated management of artificially created salt marshes in the Wadden See of Schleswig-Holstein, Germany. Wetlands Ecol and Manage 11: 183-194

Holland MM, Risser PG, Naiman RJ (eds.) (1991) Ecotones. Chapman and Hall, New York, London

Hroudová Z (1981) Seasonal vegetation dynamics on emerged pond bottom. Sborník Jihočeského muzea v Českých Budějovicích. Přírodní vědy 21: 37-49. [In Czech with Engl. Summ.]

Jaatinen K, Laiho R, Vuorenmaa A, del Castillo U, Minkkinen K, Pennanen T, Penttilä T, Fritze H (2008) Responses of aerobic microbial communities and soil respiration to water level drawdown in a northern boreal fen. Environ Microbiol 10: 339-353

Janda J, Ševčík J (2002) Avifauna of the Třeboň fishponds and new wetlands. In: Květ J, Jeník J, Soukupová L (ed) Freshwater wetlands and their sustainable future. A case study of the Třeboň Basin Biosphere Reserve, Czech Republic. Unesco, Paris, and the Parthenon Publishing Group, Boca Raton, pp 475-480

Janssens IA, Freibauer A, Schlamadinger B, Ceulemans R, Ciais P, Dolman AJ, Heimann M, Nabuurs G-., Smith P, Valentini R, Schulze ED (2005) The carbon budget of terrestrial ecosystems at country-scale – a European case study. Biogeosci 2: 15-26

Jeník J, Hátle M, Hlásek J (2002) Preservation of ecological and socio-economic roles of human-managed wetlands. In: Květ J, Jeník J, Soukupová L (ed) Freshwater wetlands and their sustainable future. A case study of the Třeboň Basin Biosphere Reserve, Czech Republic. Unesco, Paris, and the Parthenon Publishing Group, Boca Raton, pp 481-486

Joosten H, Clarke D (2002) Wise use of mires and peatlands - background and principles including a framework for decision-making. International Mire Conservation Group and International Peat Society.

Jörgensen SE, Löffler H (ed) (1990) Guidelines of lake management, Vol. 3. International Lake Environment Committee, UN Environment Programme, Otsu, Shiga, Japan

Junk WJ, Welcomme RL (1990) Floodplains. In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 1. Natural and human relationships, SPB Academic Publishing, The Hague, pp 491-524

Kaki T, Ojala A, Kankaala P (2001) Diel variation in CH_4 emissions from stands of *Phragmites australis* (Cav.) Trin. Ex Steud. and *Typha latifolia* L. in a boreal lake. Aquat Bot 71: 259–271

Kankaala P, Makela S, Bergstrom I, Huitu E, Kaki T, Ojala A, Rantakari M, Kortelainen P, Arvola L (2003). Midsummer spatial variation in CH_4 efflux from stands of littoral vegetation in a boreal meso-eutrophic lake. Freshw Biol 48: 1617–1629

Kankaala P, Ojala A, Kaki T (2004). Temporal and spatial variation in methane emission from a flooded transgression shore of boreal lake. Biogeochem 68: 297-311

Keddy PA (2000) Wetland ecology. Cambridge University Press, Cambridge

Kolmanová A, Rektoris L, Přibáň K (1999): Retention ability of pine peat bog ecosystem and its response to downpour precipitation. – In: Vymazal J (ed.) Nutrient cycling and retention in natural and constructed wetlands., Backhuys Publ., Leiden, pp 177-182

Kořínek V, Fott J, Fuksa J, Lellák J, Pražáková M (1987) Carp ponds of Central Europe. In: Michael RG (ed) Managed aquatic ecosystems. Ecosystems of the world no. 29, Elsevier, Amsterdam, pp 29-62

Kravčík, M., Pokorný, J., Kohutiar, J., Kováč, M., Tóth, E. (2008) Water for the Recovery of the Climate – A New water Paradigm. Available via HTTP:

http://www.waterparadigm.org/indexen.php?web=./home/homeen.html. Accessed 13 November 2009

Kubů F, Květ J, Hejný S (1994) Fishpond management in Czechoslovakia. In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 2: Case studies. SPB Academic Publishing, The Hague, pp 391-404

Květ J, Jeník J, Soukupová L (ed) (2002) Freshwater wetlands and their sustainable future, A case study of the Třeboň Basin Biosphere Reserve, Czech Republic. Man and the Biosphere Series Vol. 28. UNESCO Paris, and Parthenon Publishing Group. Boca Raton/London/New York/Washington, D.C.

Lachavanne JB, Juge R (ed) (1997) Biodiversity in land-inland water ecotones. Man and the Biosphere Series 18, I-XVIII and 1-308. UNESCO Paris, and Parthenon Publishing Group, Carnforth

Laiho R, Vasander H, Penttilä T, Laine J (2003) Dynamics of plant-mediated organic matter and nutrient cycling following water-level drawdown in boreal peatlands. Global Biogeochemical Cycles 17(2), 1053, doi:10.1029/2002GB002015

Laine J, Vasander H, Laiho R (1995) Long-term effects of water level drawdown on the vegetation of drained pine mires in southern Finland. J Appl Ecol 32: 785-802

Laitinen J, Kukko-Oja K, Huttunen A (2008) Stability of the water regime forms a vegetation gradient in minerotrophic mire expanse vegetation of a boreal aapa mire. Ann Bot Fenn 45: 342-358

Lappalainen E (ed) (1996) Global peat resources. International Peat Society and Geological Survey of Finland

LeMer J, Roger P (2001) Production, oxidation, emission and consumption of methane by soils: a review. Eur J Soil Biol 37: 25–50

Lhotský R (2006) Retenční funkce Třeboňské rybniční soustavy [Retention Function of the Trebon Fishpond System, in Czech]. Vodní hospodářství 56: 410-418

Lieffers VJ (1988) *Sphagnum* and cellulose decomposition in drained and natural areas of an Alberta peatland. Can J Soil Sci 68, pp 755-761

Löffler H (1974) Der Neusiedlersee. Naturgeschichte eines Steppensees. Verlag Fritz Molden, Wien/München/Zürich

Löffler H (1982) Der Seewinkel. Die fast verlorene Landschaft. Verlag Niederösterreichisches Pressehaus, St.Pölten/Wien

Löffler H (1990) Human uses. In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 1. Natural and human relationships. SPB Academic Publishing, The Hague, pp 17-27

Löffler H, Gunatilaka A (1994) The shallow lake and reed (*Phragmites australis*) wetland of Neusiedlersee (Austria). In: Patten BC (ed) Wetlands and shallow continental water bodies. Volume 2. Case studies. SPB Academic Publishing, The Hague, pp 183-202

Lowe AJ, Howard T, Pardaens A, Tinker J, Jenkins G, Jeff Ridley J, Leake J, Holt J, Wakelin S, Wolf J, Horsburgh K, Reeder T, Glenn MG, Bradley S, Stephen DS (2009) Marine & coastal projections. Available via HTTP: http://ukclimateprojections.defra.gov.uk/content/view/825/518. Accessed 3 October 2009

MacArthur R, Wilson EO (1967) The theory of island biogeography. Princeton Univ. Press, Princeton, N.J., USA

Martínez ML, Intralawan A, Vázquez G, Pérez-Maqueo O, Sutton P, Landgrave R 2007 . The coasts of our world: Ecological, economic and social importance. Ecol Econ 63: 254-272

Meehl G, Stocker TF, Collins WD et al (2007) Global climate projections. In: Solomon S, Qin D, Manning M et al (ed) Climate change 2007: The physical science basis. Contribution of Working group I to the Fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, pp 747-846

Middleton B (ed) (2002) Flood pulsing in wetlands. John Wiley & Sons, Inc., New York

Mitsch WJ, Gosselink JG (2000) Wetlands. Third edition. John Wiley & Sons, New York

Moore PD, Bellamy DJ (1974) Peatlands. Elek Science, London

Moore PD (ed) (1984) European mires. Academic. Press, London

Moore TR, Bubier JL, Frolking SE, Lafleur PM, Roulet NT (2002) Plant biomass and production and CO_2 exchange in an ombrotrophic bog. J Ecol 90: 25-36

Naiman RJ, Décamps H (1990) The ecology and management of aquatic-terrestrial Ecotones. Man and the Biosphere Series Vo. 4. UNESCO Paris, and Parthenon Publishing Group, Carnforth

Nivet C, Frazier S (2004) A review of European wetland inventory. Wetlands International and the Dutch Institute for Inland Water Management and Waste Water Treatment (RIZA). Available via HTTP: http://www.wetlands.org/RSIS/WKBASE/pewi/intro.htm. Accessed 3 October 2009

O‘Sullivan PE, Reynolds CS (ed) (2004) The lakes handbook. Vol.1, Limnology and limnetic ecology, Blackwell Publishing Comp., Malden, Oxford, Carlton

O‘Sullivan PE, Reynolds CS (ed) (2005) The lakes handbook, Vol.2, Lake restoration and rehabilitation. Blackwell Publishing Comp., Malden/Oxford/Carlton

Orme AR (1990) Wetland morphology, hydrodynamics and sedimantation. In: Williams M (ed) Wetlands. A threatened landscape. Basil Blackwell, Oxford, pp 42-94

Paavilainen E, Päivänen J (1995) Peatland forestry: Ecology and principles. Ecological Studies 111. Springer, Berlin/Heidelberg/New York

Patten BC (ed) (1990) Wetlands and shallow continental water bodies. Volume 1: Natural and human relationships. SPB Academic Publishing, The Hague

Patten BC (ed) (1994) Wetlands and Shallow Continental Water Bodies. Volume 2: Case studies. SPB Academic Publishing, The Hague

Pechar L, Hrbáček J, Pithart D, Dvořák J (1996) Ecology of pools in the floodplain. In: Prach K, Jeník J, Large ARG (ed) Floodplain ecology and management. SPB Academic Publishing bv, Amsterdam, pp 209-226

Pechar L, Přikryl I, Faina R (2002) Hydrobiological evaluation of Třeboň fishponds since the end of the nineteenth century. In: Květ J, Jeník J, Soukupová L (ed) Freshwater wetlands and their sustainable future. A case study of the Třeboň Basin Biosphere Reserve, Czech Republic. Man and the Biosphere Series Vol. 28. UNESCO Paris and Parthenon Publishing Group. Boca Rayton/London/New York/Washington, D.C., pp 31-61

Penka M (ed) (1985) Floodplain forest ecosystem I. Before management measures. Academia, Praha

Penka M (ed) (1991) Floodplain forest ecosystem II. After management measures. Academia, Praha

Phillips GL (2005) Eutrophication of shallow lakes, In: O‘Sullivan PE, Reynolds CS (ed) The lakes handbook. Vol.2, Lake restoration and rehabiltation. Blackwell Publishing Comp., Malden/Oxford/Carlton, pp 261-278

Picek T, Čížková H, Dušek J (2007) Greenhouse gas emissions from a constructed wetland – plants as important source of carbon. Ecol Eng 27: 153-165

Pithart D, Křováková K, Dušek J, Žaloudík J (2008) Case study: Ecosystem services of a floodplain with a preserved hydrological regime, Czech Republic. In: Saalismaa N et al (ed) The role of environmental management and eco-engineering in disaster risk reduction and climate change adaptation. Pro Act Network, Geneva, pp 34-36

Pokorný J (2001) Dissipation of solar energy in landscape – controlled by management of water and vegetation. Renewable Energy 24: 641-645

Prach K, Jeník J, Large ARG (ed) (1996) Floodplain ecology and management. SPB Academic Publishing bv, Amsterdam

Prach K, Pithart D, Francírková T (ed) (2003) Ekologické funkce hospodaření v říčních nivách [Ecological Functions of Floodplain Management, in Czech]. Institute of Botany, Czech Academy of Sciences, Třeboň

Procházka J, Brom J, Pechar L (2009) The comparison of water and matter flows in three small catchments in the Šumava mts. Soil Water Res 4 (Special Issue 2), in press

Pruett, L. & Cimino, J. 2000. Global Maritime Boundaries Database (GMBD). Fairfax, Virginia, U.S.: Veridian — MRJ Technology Solutions. Available via HTTP: http://earthtrends.wri.org/text/ coastal-marine/variable-61.html. Accessed 3 October 2009.

Purseglove J (1988) Taming the flood. Oxford Univ. Press, Oxford/New York, 307 pp

Rajchard J, Fridrichovský V, Křiváčková O, Navrátilová J (2008) Colonization of waterbirds of artificial lakes after surface mining: a case study. Acta Zool Sin 54: 602-614

Ranwell DS (1972) Ecology of salt marshes and sand dunes. Chapman and Hall, London

Rejmánek M, Velásquez J (1978) Communities of emerged fishpond shores and bottoms. In: Dykyjová D, Květ J (ed) Pond littoral ecosystems. Structure and functioning. Ecological Studies 28. Springer-Verlag, Berlin/Heidelberg/New York, pp. 206-211

Rinne J, Ruita T, Philatie M, Aurela M, Haapanala S, Tuovinen JP, Tuitilla ES 2007. Annual cycle of methane emission from a boreal fen measured by the eddy covariance technique. Tellus 59B: 449-457

Ripl, W. (2003) Water: the bloodstream of the biosphere. Phil T Roy Soc B 358: 1921-1934

Rodewald-Rudescu L (1974) Das Schilfrohr. Die Binnengewässer Vol. 27, Schweizerbart´scher Verlag, Stuttgart

Rodo XE, Baert E, Comin FA (1997) Variations in seasonal rainfall in Southern Europe during the present century: relationships with the North Atlantic Oscillation and the El Niño-Southern Oscillation. Climatol Dyn 13: 275-284

Rodo XE (2003) Interactions between the tropics and the extratropics. In: Rodo XE, Comín FA (ed) Global climate: Current uncertainties and research in the climate system. Springer, Heidelberg, pp 237-274

Ruhl JB, Kraft SE, Lant CL (2007) The law and policy of ecosystem services. Island Press, Washington DC

Rydin H, Jeglum JK (2006) The biology of peatlands. Oxford University Press

Schreader CP, Rouse WR, Griffis TJ, Boudreau LD, Blanken PD (1998) Carbon dioxide fluxes in a northern fen during a hot, dry summer. Glob Biogeochem Cycles 12: 729-740

Segers R (1998) Methane production and methane consumption: a review of processes underlying wetland methane fluxes. Biogeochem 41: 23–51

Seják J, Dejmal I et al (2003) Hodnocení a oceňování biotopů České republiky [Assessment and Valuation of Biotopes of the Czech Republic, In Czech]. Český ekologický ústav, Praha

Sliva J, Pfadenhauer J (1999) Restoration of cut-over raised bogs in southern Germany - a comparison of methods. Appl Veg Sci 2: 137-148

Straškraba M (1963) Share of the littoral vegetation in the productivity of two fishponds in Southern Bohemia. Rozpravy Čs. Akademie Věd, Řada Mat. Přír. Věd 73(13):1-64

Straškraba M (1968) Der Anteil der höheren Pflanzen an der Produktion der stehenden Gewässer. Mitt Intern Ver Theor Angew Limnol 14: 212-230

Straškraba M, Kořínková J, Poštolková M (1967) Contribution to the productivity of the littoral region of ponds and pools. Rozpravy Čs. Akademie Věd, Řada Mat. Přír. Věd 77(11):1-80

Šumberová K, Horáková V, Lososová Z (2005) Vegetation dynamics on exposed pond bottoms in the Českobudějovický basin (Czech Republic). Phytocoenologia 35: 421–448

Šumberová K, Lososová Z, Fabšičová M, Horáková V (2006) Variability of vegetation of exposed pond bottoms in relation to management and environmental factors. Preslia 78: 235–252

Šusta J (1898) Fünf Jahrhunderte der Teichwirtschaft in Wittingau. Herrcke u. Lebeling, Stettin

Turner RK, Georgiou S, Fisher B (2008) Valuing ecosystem services. The case of multi-functional wetlands. Earthscan, London

Vasander H (1982) Plant biomass and production in virgin, drained and fertilized sites in a raised bog in southern Finland. Ann Bot Fenn 19: 103-125

Vasander H, Laiho R, Laine J (1997) Changes in species diversity in peatlands drained for forestry. In: Trettin CC, Jurgensen MF, Grigal DF, Gale MR, Jeglum JK (ed) Northern Forested Wetlands: Ecology and Management. CRC Press, Lewis Publishers, Boca Raton, pp 109-119

Vasander H, Tuittila ES, Lode E, Lundin L, Ilomets M, Sallantaus T, Heikkilä R, Pitkänen ML, Laine J (2003) Status and restoration of peatlands in Northern Europe. Wetlands Ecol Manage 11: 51-63

Verhoeven JTA, Beltman B, Bobbink R, Whigham DF (2006) Wetlands and natural Resource management. Ecological Studies 190. Springer, Berlin/Heidelberg/New York

Vitousek PM 1994. Beyond global warming: Ecology and global change. Ecology 75:1861-1876

Vymazal J (1995) Algae and Element Cycling in Wetlands. Lewis Publishers, Boca Raton

Vymazal J, Brix H, Cooper PF, Green MB, Haberl R (ed) (1998) Constructed wetlands for wastewater treatment in Europe. Backhuys Publishers, Leiden

Vymazal J, Greenway M, Tonderski K, Brix H, Mander Ü (2006) Constructed wetlands for wastewater treatment. In: Verhoeven JTA, Beltman B, Bobbink R, Whigham DF, 2006. Wetlands and natural resource management. Ecological Studies 190, Springer, Berlin, Heidelberg, New York, pp 69-96

Westlake DF, Květ J, Szczepański A (ed) (1998) The production ecology of wetlands. Cambridge University Press, Cambridge

Whigham DF, Dykyjová D, Hejný S (1993) Wetlands of the world I: Inventory, ecology and management. Kluwer Academic Publishers, Dordrecht

Williams M (ed) (1990) Wetlands. A threatened landscape. Basil Blackwell, Oxford, 419 pp

Zemanová K, Čížková H, Edwards K, Šantrůčková H (2008) Soil CO2 efflux in three wet meadow ecosystems with different C and N status. Comm Ecol 9 (Suppl): 49-55

Table 1 Estimated wetland coverage in Europe as identified by the European inventory dataset (according to Nivet and Frazier 2004)

Number of countries:	47
Total land area of study Region (km^2) (excluding marine areas; including Asian part of Russia, and Azerbaijan) (km^2)	23703572
Total area of wetlands identified (km^2)	2667420
% of land area (excluding marine areas) covered by these wetlands	11.3
Land area of Region (km^2), excluding Russia and marine areas:	6707772
Total area of wetlands identified in this study, excluding Russian wetlands (km^2):	491060
% of land area, excluding Russia and marine areas, covered by these wetlands:	7.3

Table 2 Quantification of selected ecosystem services for a natural segment of the Lužnice River Floodplain (South Bohemia, Czech Republic)[a]

Service	**Calculation**	**Value ($USD.ha^{-1}.yr^{-1}$)**
Flood mitigation	Total retention volume 4.7 mil m^3, per ha 10250 m^3	11 788
Biodiversity refuge	Average point value 38, monetary value of one point per ha 8000 USD	15 000
Carbon sequestration	Sequestration of 2029 t of C per ha, i.e. 7.54 t of CO_2, market price of emission limit 20 USD per t.	144
Fish production	Catches: 3,4 t of fish in the area, 7,2 kg per ha, average price 5 USD per kg	37
Hay production	183 ha of cut medows with production of 20q of hay per ha, price 10 USD per q	78
Wood production	Growth rate 5 $m^3.ha^{-1}.year^{-}1$; 33 USD per m^3 and 61 ha of the floodplain forest	21
Total		**27068**

[a] The 470 ha area of this floodplain segment has a well preserved hydrological régime and is fully adapted to periodical flooding including its agricultural and social aspects. The land cover is formed by a mosaic of water bodies, river bed, meadows, wetlands and floodplain forest. Flood mitigation was estimated by calculation of the aboveground retention volume (digital elevation model). For the evaluation, the alternative cost of a cubic meter in a man-made construction was multiplied by the calculated retention volume**.** Carbon sequestration was based on annual measurement of CO_2 fluxes between a local herbaceous wetland ecosystem and atmosphere by the eddy-covariance method. The amount of carbon sequestered per unit area per year was multiplied by the marketable price of emission limits in 2008. In order to assess the ecosystem value as biodiversity refugium, all major biotopes were mapped and their contributions to the total area of the study were calculated. Each biotope was given a value in points according to Seják et. al. (2003). Monetary value of one point was derived from the average cost of a revitalisation project bringing an increase in the point value. Fish production was estimated on the basis of the number of catches obtained from the Czech Angling Union. Local market prices of particular fish species were used for total price calculation. The hay and wood production and prices in 2008 were obtained from local farmers, the area of meadows and forest (without solitary trees and willow carrs) was calculated by GIS methods. All services were calculated for the whole area and related to one hectare of the floodplain.

Figure Captions

Fig. 1 Map of selected wetlands of Europe: 1 – Peatlands of Ireland; 2 – Scottish peatlands – predominantly mires (blanket bogs) ; 3 – Peatlands of the Pennines; 4 – Estuary of the Rhine and Maas; 5 – Mudflats and coastal salt marshes and estuaries of the Waddenzee (Wattenmeer); 6 – Mountain peatlands and lake littoral wetlands of Scandinavia, incl. Finland; 7, 8, 9 – Estonian and Latvian peatlands, lake littoral marshes and coastal marshes; 10 – The Pripyat and Polesie wetlands (mainly peatlands and floodplains) in Belarus and the Ukraine; 11, 12, 13 – Coastal marshes, lagoons and estuaries of the southern Baltic Sea; 14, 15 – Floodplains of the lower Oder and Elbe rivers; 16, 17 – Peatlands of the Giant Mts. (Krkonoše) and Ore Mts. (Erzgebirge); 18 – Peatlands of the Bohemian and Bavarian Forest Mts.; 19 – Fishponds, peatlands and floodplains of southern Bohemia (basins of České Budějovice and Třeboň); 20 – Peatlands of the Black Forest (Schwarzwald) Mts.; 21 – Littoral wetlands of Lake Constance and Bavarian subalpine lakes; 22 – Floodplains of the Morava and Dyje (Thaya) rivers and of the Danube in the Czech Republic, Austria, Slovakia and northern Hungary; 23 – Lake Neusiedler See / Fertö littoral wetlands, saline lakes (Lacken) of the Seewinkel and fens of the Hanság in Austria and Hungary; 24, 25 – Floodplains of the middle Danube and of the Tisza, Drava and Sava rivers; 26 – Floodplain of the lower Danube and the Danube delta with adjacent coastal lagoons; 27 – Estuary of the Neretva river; 28 – Estuary of the Dniper river; 29 – Delta of the Po river and adjacent coastal lagoons and salt marshes; 30 – La Camargue – delta of the Rhone river; 31 – Delta of the Ebro river; 32 – Lake Gallocanta and adjacent inland saline lakes and salt marshes; 33 – Doňana – floodplain and estuary of the lower Quadalquivir river; 34, 35 – Estuaries of the Garonne and Loire rivers; 36 – Coastal wetlands and mudflats of the Brittany and Normandy coasts. Note: Wherever lakes are shown in the map, littoral wetlands also occur on most gently sloping lake shores.

The map has been compiled on the basis of the following sources: Gore (1983), Mitsch and Gosselink (2000), Airoldi and Beck (2007), map of European Ramsar sites (http://www.wetlands.org/reports/rammap/mapper.cfm#; 11.4.2010) and map of wetland concentration in Europe (http://www.eea.europa.eu/data-and-maps/figures/wetland-concentration-in-europe-2000; 11.04.2010).

Fig. 2 Examples of European wetland habitats. (a) Mesotrophic fen (example of aapa mires) at Lompolojänkä in Finnish Lapland. Photo T. Penttilä. (b) Pristine ombrotrophic bog forest in southern Finland. Photo R. Laiho. (c) Mesotrophic fen in the floodplain of the Lužnice river, Czech Republic. Photo J. Ševčík. (d) Floodplain forest at the confluence of the Dyje and Morava rivers, Czech Republic. Photo Czech Ramsar Committee. (e) Example of an inland saline shallow lake with its littoral wetlands: Lake Gallocanta, Spain. Photo J. Dušek. (f) Example of a Mediterranean estuary: Hutovo Blato in the estuary of the Neretva river, Croatia. Photo T. Kušík.

Fig. 3 Aerial views of a riverine and a lacustrine wetland in Central Europe. (a) a bay of a human made shallow lake: Velký Tisý Fishpond, Czech Republic. Photo J. Ševčík. (b) a well preserved floodplain: the upper Lužnice River, Czech Republic. Photo P. Znachor.Both sites belong to wetlands of southern Bohemia, Czech Republic (area no. 19 in Fig. 1).

Fig. 4 Network of the major relationships between the global processes causing global changes and global climate change in the industrial era. Modified from Vitousek (1994)

Fig. 5 Temperature distribution in the town of Třeboň and surrounding Wet Meadows (South Bohemia, Czech Republic) visualised by an infra-red camera. Photo J. Brom

Fig. 6 Comparison of temperature distribution in two differently treated areas of cultural landscape: the Most Basin and the Třeboň Basin, Czech Republic. a, b – Most Basin, coal mining area. c, d – Třeboň Basin, landscape with large areas of artificial lakes (fishponds). a, c – pictures in visible light spectrum. b, d – false colour spectrum indicating temperature differences, with the highest temperatures being yellow, orange, and red and lowest temperatures being blue and green (see the relative temperature scale, e). 1 – the Krušné hory mountains, 2 – Most city, 3 – Sokolov city, 4 – open-cast mines and unvegetated heaps, 5 – Rožmberk lake, 6 – Třeboň town. Large hot spots correspond mostly to mines and heap areas while cold areas correspond to the Krušné hory mountains in Fig. 6b and to lakes and forests in Fig. 6d. Landsat TM data (Most Basin: scene 192-025, acquisition date 1 July 1995; Třeboň Basin: scene 191-026, acquisition date 10.7.1995)

[a]Multispectral spaceborne data acquired by Landsat Thematic Mapper and Enhanced Thematic Mapper+ sensors were used to compare two types of cultural landscapes in the Czech Republic. The Most Basin has been largely influenced through the open mining of the tertiary lignite. Large areas have been drained and turned either into

deep and vast mines or spoil heaps. The Třeboň Basin Biosphere Reserve represents a contrasting example of a landscape which, although historically largely influenced by humans, has maintained high nature qualities. It is a rural landscape characterised by a frequent occurrence of human-made lakes (fishponds) which have been constructed since 15^{th} century and cover ca. 12% of the total area. Apart from the fishponds there are also other wetlands in the area such as floodplain segments and mires. The Třeboň Basin, generally well saturated with water, shows a substantially lower temperature variance with temperature extremes completely missing.

Fig. 7 Discharges of three main tributaries to the Orlík Reservoir in South Bohemia, Czech Republic, during floods in August 2006. Dotted line – the Vltava at České Budějovice, dashed line – the Otava at Písek, solid line - the Lužnice at Bechyně. Modified from Lhotský (2006)[a]

[a]The extremely high precipitation values of August 2002 resulted in the largest flood ever reported in Central Europe. The water discharges were estimated as representing the 100-year to 500-year maxima in different catchments. The flood dynamics differed between the catchments in response to their hydrology. A notable example is provided by the Třeboň fishpond system, situated in the catchment of the upper Lužnice River. The Lužnice River has a well preserved natural floodplain connected with an elaborate system of artificial lakes (fishponds) of about 60 km^2 in area. The Třeboň fishpond system retains 50 - 70 million m^3 water during floods. In the extreme floods of summer 2002, the fispond system retained 110-114 million m^3 of water. As a result, flood culmination was delayed by 68 hours and the flood peak was substantially damped.

Fig. 8 Average daily discharges from three small sub-mountain catchments (the Šumava mountains, Czech Republic). Solid grey line – drained catchment of the Mlýnský brook, covered with pastures and mown meadows, solid black line – unmanaged catchment of the Horský brook covered with forest and wetlands, dotted grey line – forested catchment of the Bukový brook. Modified from Procházka et al. (2009)[a]

[a]During douwnpour rains in August 2002, precipitation reached over 100 mm in a few days. Several times as high flow rates from a drained pasture (900 $l.s^{-1}$) as from forested and wetland catchments (300 $l.s^{-1}$) were recorded. This comparison illustrates the greater retention capacity of the wetland and the forest in comparison with the drained catchment, which is then translated in the attenuation of discharge peaks.

Fig. 9 – Precipitation and water discharge as related to the groundwater level in the Red Bog (Červené Blato), the Třeboň Basin, Czech Republic. Open columns – precipitation, full squares – discharge from the bog, open circles – groundwater table in the bog-pine forest, full triangles – groundwater table in the regenerating part of the bog. Modified from Kolmanová et al. (1999).

Fig. 1
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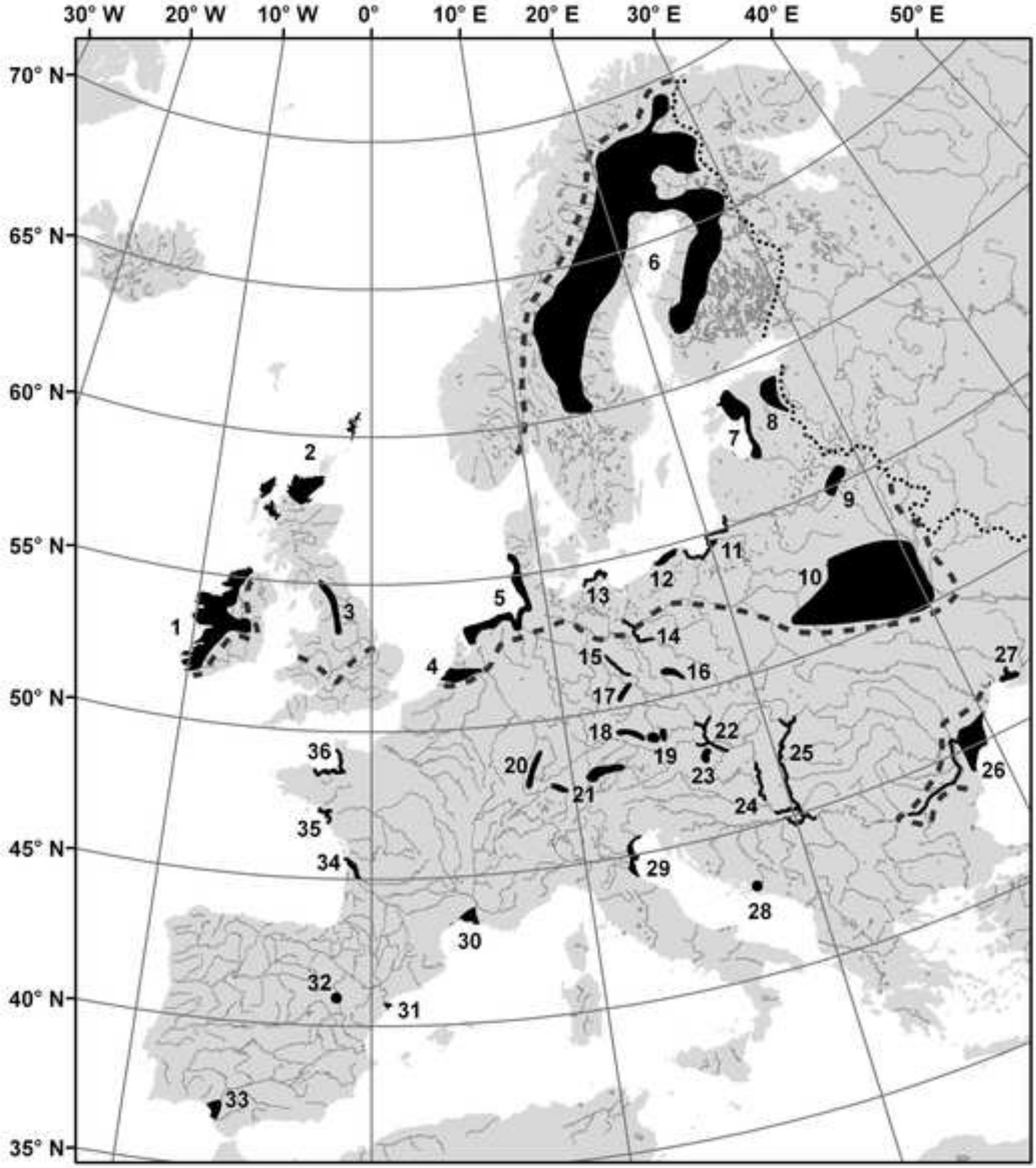

Fig. 2

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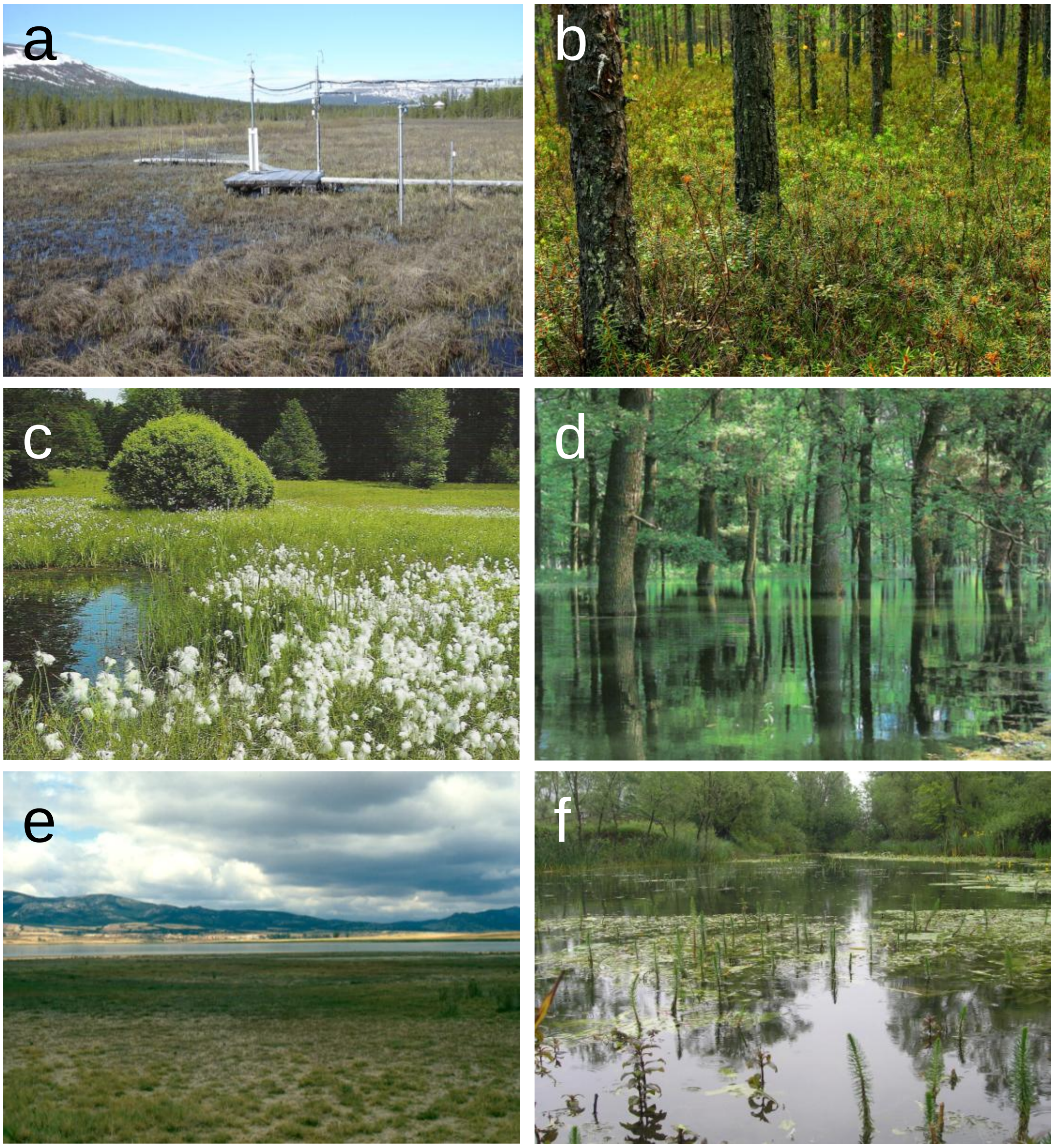

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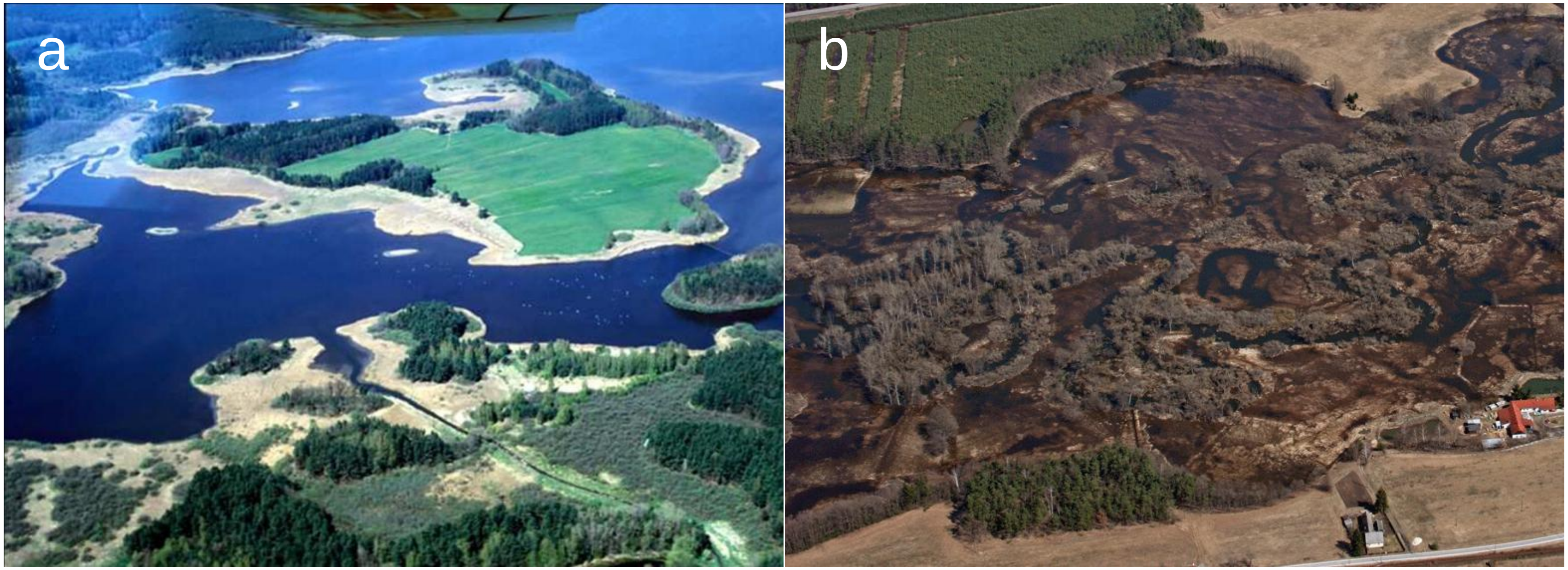

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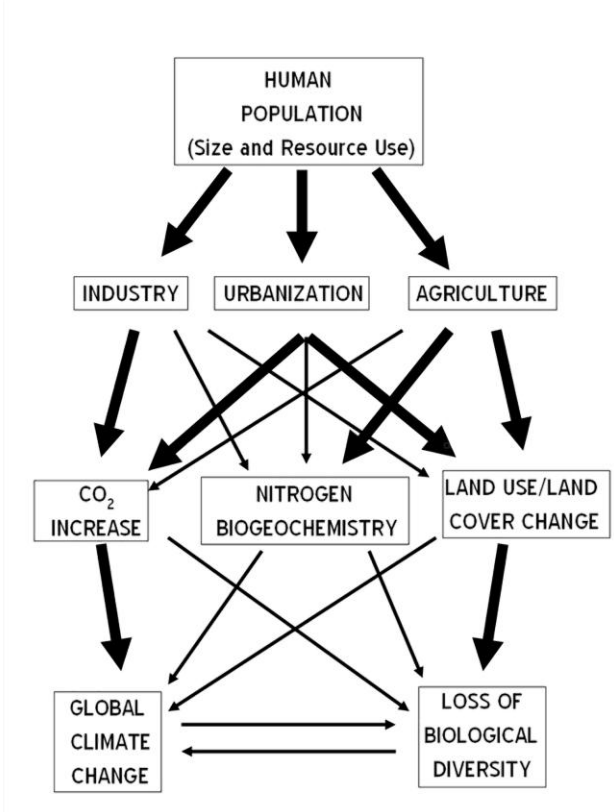

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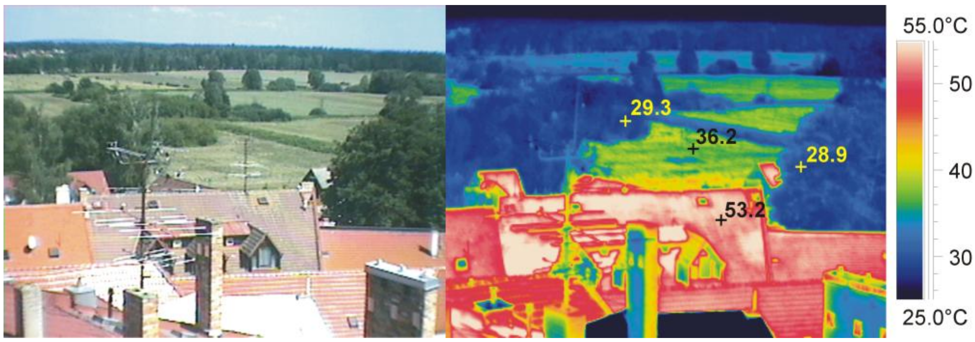

Fig. 6 (colour)

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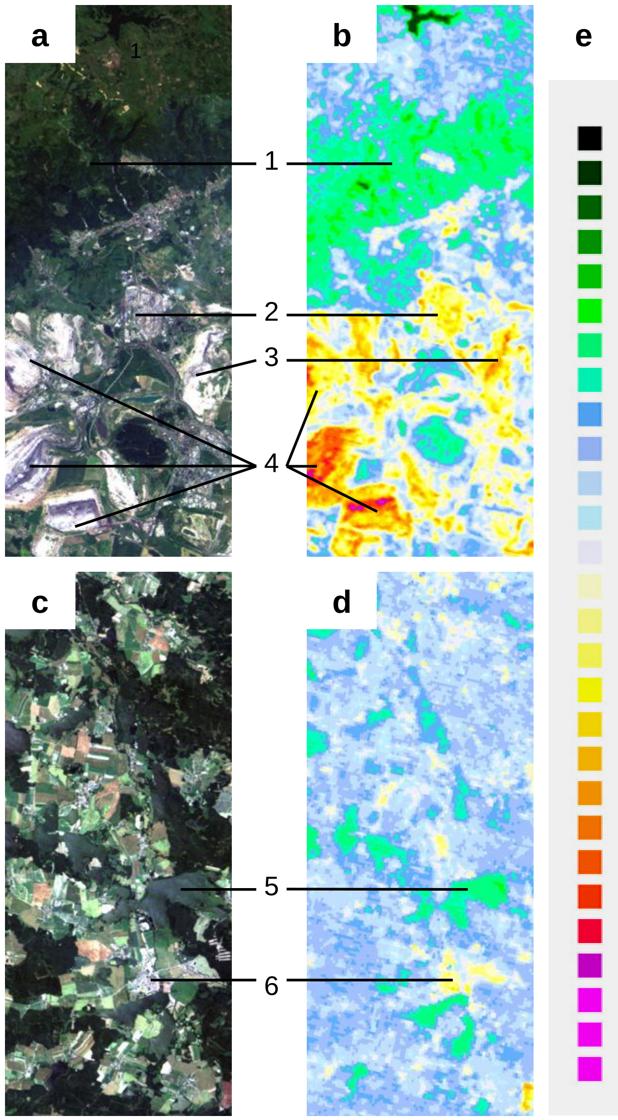

Fig. 7

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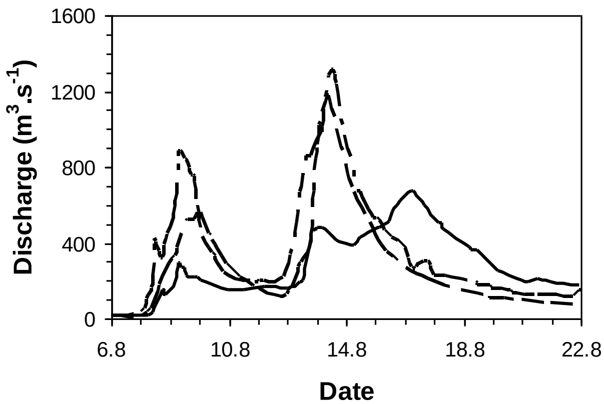

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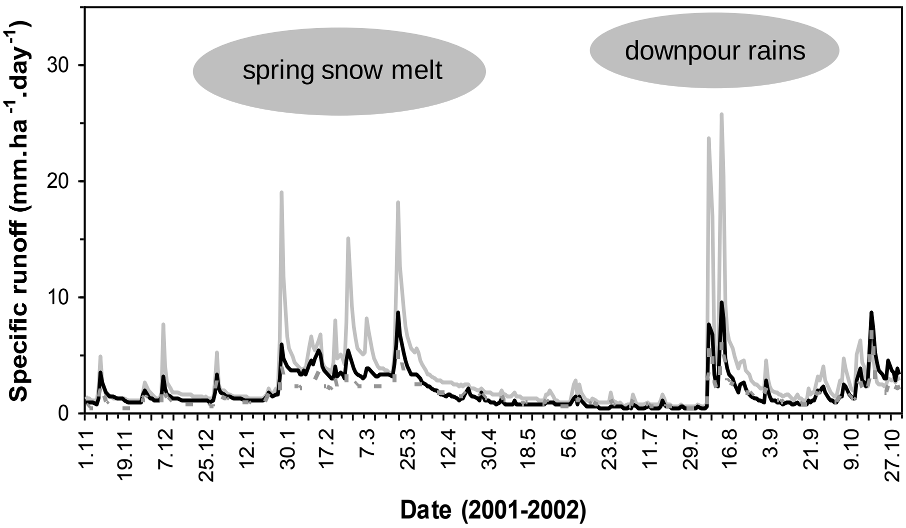

Fig. 9
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