



Review

Tradeoffs and synergies in wetland multifunctionality: A scaling issue



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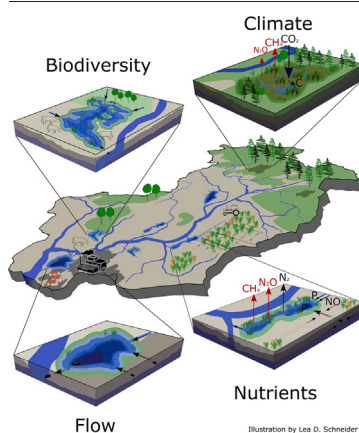
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HIGHLIGHTS

- Wetlands are currently viewed by society as nature-based solutions to a range of societal problems
- To identify incompatibilities between objectives for wetland re-establishment, we reviewed ecosystem and cultural objectives
- Multifunctionality is inconceivable at the level of individual wetlands due to tradeoffs between objectives
- Multifunctionality can instead be achieved at the landscape level, where objectives are optimized in different wetlands

GRAPHICAL ABSTRACT



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ABSTRACT

Wetland area in agricultural landscapes has been heavily reduced to gain land for crop production, but in recent years there is increased societal recognition of the negative consequences from wetland loss on nutrient retention, biodiversity and a range of other benefits to humans. The current trend is therefore to re-establish wetlands, often with an aim to achieve the simultaneous delivery of multiple ecosystem services, i.e., multifunctionality. Here we review the literature on key objectives used to motivate wetland re-establishment in temperate agricultural landscapes (provision of flow regulation, nutrient retention, climate mitigation, biodiversity conservation and cultural ecosystem services), and

Abbreviations: WRT, water retention time (= water volume/throughflow); Aw:Ac, wetland area to catchment area; HL, hydraulic load (= inflowing water volume/wetland area); HE, hydraulic efficiency; GHG, greenhouse gases.

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Biodiversity conservation
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their relationships to environmental properties, in order to identify potential for tradeoffs and synergies concerning the development of multifunctional wetlands. Through this process, we find that there is a need for a change in scale from a focus on single wetlands to wetlandscapes (multiple neighboring wetlands including their catchments and surrounding landscape features) if multiple societal and environmental goals are to be achieved. Finally, we discuss the key factors to be considered when planning for re-establishment of wetlands that can support achievement of a wide range of objectives at the landscape scale.

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1. Introduction

The need for land to increase food and fiber production has dramatically changed agricultural and forest landscapes across the globe, with strong reductions particularly in the areal extent of wetlands (Davidson, 2014; Dixon et al., 2016; Davidson et al., 2018). Historically, wetlands provided important resources, but the need to feed a growing human population resulted in the removal of large wetland areas (Strandin Pers, 2012). This change was also motivated by the perception of wetlands as low productive habitats of no use to humans, wastelands and home of infectious mosquitoes, and therefore of no loss if removed. As a consequence, less than 10 % of the original wetland area remains today in heavily populated areas such as Western Europe (EPCN, 2007).

Even though land use conflicts in relation to wetlands continue to pose great challenges in many parts of the world (Kingsford et al., 2016), the tide has started turning. Current societal attitudes towards wetlands are more positive due to the recognition that wetlands provide valuable regulating, provisioning and cultural services to society (Davidson et al., 2019; Cheng et al., 2020). There are accordingly increased efforts to restore and create wetlands as nature-based solutions to help address problems such as eutrophication, climate change, biodiversity loss, floods and droughts (Zedler, 2003; Natuhara, 2013; Griscom et al., 2017; Thorslund et al., 2017; Jaramillo et al., 2019; Cheng et al., 2020; Bradfer-Lawrence et al., 2021). Thus, the international community has responded through conventions that involve wetland preservation and re-establishment (e.g., CBD/Aichi, IUCN, WFD, Ramsar), and 38 % of all countries currently have wetland protection included in their environmental policy (Peimer et al., 2017). In the EU, the new Common Agricultural Policy (for 2023-7) urges member states to consider wetland protection and functionality in their national plans, and some countries such as Sweden and Denmark have initiated subsidy programs for landowners interested in increasing the area or quality of wetlands (Andersson, 2012; Graversgaard et al., 2021). However, achievements on the ground to date are still far short of objectives (Naturvårdsverket, 2019).

This growing awareness of wetland benefits has supported a shift towards thinking about multifunctionality of individual wetlands and the simultaneous delivery of multiple ecosystem services (Zedler, 2003; Acreman et al., 2011; Blackwell and Pilgrim, 2011; Maclean et al., 2011; Natuhara, 2013). A key challenge, however, is that structural and spatial

wetland properties for delivering specific services may not be best suited for delivering other services (Rouquette et al., 2011; Xu et al., 2018). For example, efficient nutrient retention in a wetland may speed up vegetation succession and lead to changed species abundance and reduced biodiversity (Doherty et al., 2014; Jessop et al., 2015). On the other hand, some ecosystem services may be more easily co-produced, e.g., the capacity for denitrification may positively relate to water storage potential (Jessop et al., 2015). Understanding these synergies and tradeoffs between processes is key to improving guidelines for wetland construction and management.

Another key challenge concerns the different objectives among societal actors for re-establishing wetlands, where landowner interests often deviate from governmental objectives, and include a focus on recreational benefits (e.g., aesthetics, birdwatching, hunting, skating; Söderqvist, 2003; Hansson et al., 2012). Similarly, administrative stakeholders may formulate targets concerning water flow regulation and biodiversity improvements, but landowners may hesitate to construct wetlands that increase flood risk on other productive parts of their land such as crop fields. Landowners therefore construct ponds with steep edges and small flood zones that provide low qualities for biodiversity conservation. Moreover, fear that wetlands support mosquito reproduction may shape the attitudes of landowners and neighbors to planned wetlands (Hanford et al., 2019). Whereas landowner interests are often connected to the size of financial subsidies (Graversgaard et al., 2021), a top-down implementation of wetlands by authorities may be hindered by stakeholder opposition (Gann et al., 2019) and outcomes may depend on what types of behavior are incentivized (e.g., Drescher et al., 2019). Landowner interest is pivotal because long-term interest and management is needed to preserve the functioning of restored or created wetlands.

Some research has considered wetland multifunctionality (e.g., Hansson et al., 2005; Doherty et al., 2014; Jessop et al., 2015; Boughton et al., 2019), including the Wetland Evaluation Technique (WET) which was developed four decades ago to assess multiple, often conflicting functions of individual wetlands (Adamus et al., 1987). Similarly, decision support systems such as systematic conservation planning (Cimon-Morin et al., 2021) or social multi-criteria evaluations (Doyle, 2020) are specifically designed to evaluate tradeoffs during planning of conservation actions based on the demand and supply of different benefits to humans. However, we argue that the inherent tradeoffs and synergies

that limit multifunctionality related to both the ecological and the social dimensions of wetlands have not been fully explored. In the present study, we therefore go beyond the existing studies by, first, reviewing the evidence behind the wetland properties that lead to particular functions, and second, analyzing the relationships between these properties to identify potential tradeoffs and synergies. We also note that it has been debatable if focusing on individual wetlands to improve their multifunctionality is the most rational way forward for reaching multifunctionality at the landscape scale or whether wetland planning and management should increasingly consider developing and maintaining wetlandscapes, i.e., wetlands including their catchments and surrounding landscape features (Zedler et al., 2012). At this larger scale, which is relevant for delivery of ecosystem services to society, a more fruitful strategy may be to aim for heterogeneous arrays of multiple, but not necessarily individually multifunctional, wetlands.

Three main reasons may explain the paucity of empirical studies on multifunctionality. First, the spatial scale needed to estimate tradeoffs is broad (e.g., Jessop et al., 2015), requiring major research efforts involving collaborations with stakeholders at multiple levels (e.g., Boughton et al., 2019). Second, wetland functionality is the result of complex, intertwined social-ecological processes (Dawson et al., 2021), making it difficult to disentangle tradeoffs or synergies in the presence of multiple environmental factors that vary among sites and with different effects on target variables (Bennett et al., 2009). Testing tradeoffs between wetland processes therefore either requires experimental manipulations of wetland properties, or detailed monitoring of a relatively large number of comparable objects. Third, evaluating tradeoffs necessitates a cross-disciplinary approach that is challenging to accomplish at a sufficiently detailed level to quantitatively evaluate process tradeoffs (Johnston et al., 2013). In the literature, the main approaches for examining tradeoffs and synergies of ecosystem services involve mapping co-occurrence patterns by clustering different services across habitat types (e.g., Gomez-Creutzberg et al., 2021) or the identification of negative/positive associations across multiple services in the same habitat type varying in environmental conditions (e.g., Jessop et al., 2015).

To address these open questions and research gaps involving multifunctionality, we organized a workshop involving the coauthors of this paper, whose collective expertise covers both ecosystem (hydrological, ecological, biogeochemical) and societal aspects of wetlands. Using this workshop as a starting point, we reviewed the literature on four overarching objectives associated with wetland re-establishment – (i) water flow and nutrient regulation, (ii) climate change mitigation, (iii) biodiversity conservation, and (iv) cultural ecosystem services. Following the Ramsar convention, we define wetlands as including shallow lakes and small ponds but not shallow coastal waters. For each objective, we identified wetland properties that contribute to functions and ecosystem services whose delivery supports these objectives. We further assessed relationships between structural and spatial properties and ecosystem services with the aim to identify potential tradeoffs and synergies in service delivery provided by wetlands and to derive possible mechanistic connections as basis for identifying limitations or opportunities for multifunctionality across scales, where a great challenge is that different fields use different terms for the same wetland properties. Finally, based on our analysis we discuss approaches for empirically evaluating tradeoffs and synergies and how to implement this information in planning and decision making at the wetlandscape level.

2. Wetland properties in support of key objectives

2.1. Methodology

We used a three-step process, supported by a literature review, to identify wetland properties connected to different objectives of wetland re-establishment. First, during the workshop, participants identified a preliminary list of properties of importance for four main groups of objectives (cf. Maltby, 2018), at local and wetlandscape scales, linked to: (i) water flow and nutrient/pollutant load regulation, (ii) climate change mitigation,

(iii) biodiversity conservation and (iv) cultural ecosystem services (Table 1). Within each group, specific objectives were identified, e.g. emission reduction for climate mitigation objectives or support of specific species groups for biodiversity conservation objectives. Structural properties of individual wetlands and whole wetlandscapes were grouped in four main properties: biophysical properties (e.g., area or depth), water properties (e.g., residence time or nutrient concentration), biological properties (e.g., vegetation or fish presence), and landscape properties (e.g., catchment size or visitor accessibility). Participants then completed a matrix linking properties with identified objectives, as ‘+’ (positive impact), ‘-’ (negative impact) or ‘+/-’ (variable) (Table 1). In the second step, the same experts cross-checked each other's assessments and added supporting references. Finally, Table 1 was distributed to colleagues within each field for deliberation to identify missing relationships and additional references.

Below, we first describe the main connections between environmental properties and the four groups of objectives associated with wetland re-establishment, examining tradeoffs and synergies within groups of objectives. Thereafter, we use Table 1 to identify potential tradeoffs and synergies between groups and finally review empirical studies that examined such tradeoffs and synergies.

2.2. Regulation of water flows and waterborne nutrient/pollutant loads

Hydrological processes including retention, runoff and evaporation are logical starting points for the wetland analysis because they control water availability in the landscape and govern the transport and retention of waterborne nutrients and pollutants. Key objectives for wetland re-establishment therefore include their capacity to buffer high flow events by attenuating flood peaks, to safe-guard water availability during extended droughts by storing water in the landscape and to increase water quality by reducing nutrient or pollutant loads downstream (Brody et al., 2007) (Table 1). The flow modulating capacity of individual wetlands is directly related to wetland size, or the volume of its surrounding topographic depressions, which determines how much excess water can be stored during flood events. To which extent this capacity will be utilized at each flood event additionally depends on the current water level and the magnitude of runoff from the surrounding landscape elements (Aceman and Holden, 2013; Fossey et al., 2016; Åhlén et al., 2022). Wetland prevalence and storage capacity also impact long-term runoff generation over the wetlandscape, which can act in different directions (increasing or decreasing runoff) and evapotranspiration depending on ambient conditions, including hydro-climatic conditions (e.g., Quin et al., 2015; Fossey and Rousseau, 2016; Åhlén et al., 2021).

Nutrient retention varies widely over time, and between wetlands and regions depending on retention target (Land et al., 2016), hydro-climatic conditions, and the characteristics of individual wetlands (Richardson, 1985; Braskerud et al., 2005) and whole wetlandscapes (Quin et al., 2015; Thorslund et al., 2017) (Table 1). For individual wetlands, retention of nutrients and other pollutants can be estimated in three ways depending on purpose: total mass retention, area-specific retention and retention efficiency (% of load). The total mass and area-specific retention generally increase whereas retention efficiency decreases with the nutrient load (Table 1, Kynkäänniemi et al., 2013; Weisner et al., 2016; Audet et al., 2020; Mendes, 2021). The nutrient mass removed is therefore higher in wetlands with high nutrient loads, close to the nutrient source, as in landscapes with intensive agricultural production, but also depends on soil type (Kyllmar et al., 2014).

Water residence time (WRT = water volume/throughflow) plays an important role in nutrient retention, and is typically estimated as hydraulic load (HL = inflowing water volume/wetland area), which is inversely related to WRT, or the wetland size to catchment area ratio (Aw:Ac), which is proportional to WRT. In general, a higher HL (or lower Aw:Ac) tends to lower retention efficiency but increase the area-specific retention (Audet et al., 2020; Mendes, 2021), below a possible threshold above which higher HL will reduce both retention efficiency and area-specific nutrient retention

Table 1

Identified relationships between environmental properties of wetlands and targets of wetland construction and restoration showing the main trends from the literature (references in Appendix A). The signs indicate positive (+), negative (−) or mixed (+/−) effects but variations beyond these relationships also occur. Included relationships should be interpreted as mainly direct, but may be indirect when an intermediate property is not included, and mixed effects include both nonlinear relations and where the direction of the effect depends on other wetland properties. Some environmental properties are typically correlated, and the sign then indicates the unique effect of each property (see footnotes for details). For instance, a wetland with a larger surface area is by necessity shallower at a given volume but some targets respond more to the area and others respond more to the depth. For objectives Hg methylation, CH₄ emissions and N₂O emissions, signs are reversed to show reductions to fit the pattern that ‘+’ is something positive. The strength of support is weak if indicated in brackets.

		Physical properties						Water properties		
		Volume	Area ^a	Depth ^a	Hydraulic efficiency ^b	Shore complexity ^b	Open water	Flood zone ^c	Residence time	Nutrient concentration ^d
Flow regulation	Flood control	+			(+)			+		
	Water storage	+						+		
	Area-specific N retention (g per wetland area)	−	−		+	(−)	−	+	+	+
	Relative N retention (% removed)	+	+		+	(−)	−	+	+	(−)
	Area-specific P retention (g per wetland area)	(+)/−	(+)/−		+	(−)	−	+	+ ⁱ	+
	Relative P retention (% removed)	+		(+)	+	(−)	−	+	+	(−)
	Hg retention	+	+	−	+		−		+	+/−
	Prevent Hg methylation ^j		−	+	−		+	−	+/−	+/−
	Climate mitigation ^k	Reducing CH ₄ emissions		+	+			−	−	−
Reducing N ₂ O emissions				+/−			+	−		−
C sequestration			−	−					+	+
Biodiversity conservation ^l	Emergent vegetation			−	(+)	+		+		+
	Submerged macrophytes		+	−	(−)		+			+/−
	Amphibians		+/−	−	(−)	+	(−)	+		+/−
	Water birds		+	+/−	(+)	+/−	+/−	+		+/−
	Shorebirds		+	−		+	(+)	+		+
	Aquatic macroinvertebrates		+	−		+				+/−
	Spiders/carabids		+					+		(+)
	Pollinators (bees)									+/−
	Mosquitoes		+	−			+	−	+	+
Cultural ecosystem services	Recreation and ecotourism					+	+			−
	Aesthetics					+	+			
	Inspiration									
	Education									
	Sense of place					+				
	Cultural heritage									
	Spiritual and religious									

(Braskerud et al., 2005; Land et al., 2016). Retention efficiency also increases with hydraulic efficiency (HE, Persson et al., 1999), which describes how well bulk flow is distributed within the wetland volume and depends on wetland shape and distance between in- and outlet. HE is lower if water flows mainly through a central channel and higher when complex flow patterns force more water to flow through vegetated wetland parts where denitrification is higher (Kadlec, 2008). Another important shape feature is bottom topography, where deep parts increase particularly P retention (Geranmayeh et al., 2018). Finally, wetlands dominated by event-driven discharge typically show lower retention efficiency than wetlands

with a controlled hydraulic loading rate because high flood events such as spring floods coincide with low WRT, high nutrient transports and low biological activity (Land et al., 2016).

Total nutrient retention depends not only on properties of single wetlands, but also on the distribution of wetland area within catchments. At the wetlandscape level, Aw represents the total wetland area while Ac is the combined multi-wetland catchment area encompassing the same wetlands. Nutrient retention at this scale may be high even for low Aw:Ac ratios if a major part of total Aw is located close to the outlet of the catchment (Thorslund et al., 2017), or is efficiently distributed over the catchment

Water properties	Biological properties					Wetlandscape properties			
	Water fluctuations ^c	Emergent vegetation	Submerged vegetation	Shore grazing/Hay making ^e	Fish presence	Naturalness ^f	Catchment size ^g	Total wetland area ^g	Nutrient load ^d
+/-	+					+/-	+		
+/-	+	+				+	+	+ /(-)	
	+					-		-	
	+	+				+		+ /(-)	
	+					+/-			
-	+								
-	-	-							
+/-	+/-	+	-	+/-				-	
+/-	(+)	+	(-)	(+)				+/-	
+	+		-	+				+	
+	NA		-						
+/-		NA						+	
+/-	(+)	+	+	-				+	
+/-	+/-	+/-	+	+/-				+	(+)
+	(-)	(-)	+/-					+	
(+)	+	+		-				+	
+/-	(+)		+					+	
	+/-		+/-	+				(+)	
+	+	(+)	-	-					
				+					+
+	+/-		+	+	+				-
				+					+
					+				

^a At a given volume, area and depth are by necessity negatively correlated. For flow variables, an important variable is hydraulic load (= inflowing water volume/wetland area) which suggest an opposite relationship to wetland area and catchment size. Depth is also correlated with vegetation, and the effect of depth here is for a comparison when vegetation is the same.

^b Both hydraulic efficiency and shore complexity describe aspects of heterogeneity. Hydraulic efficiency includes properties such as shape and presence of islands that makes water flow over larger or smaller parts of the wetlands, whereas shore complexity accounts for undulated shorelines or variation in shore steepness that differentially affects many organisms.

^c Flood zone describes the presence of shallow areas around the wetland where water could spread during high water tables whereas water fluctuations describe the frequency and amplitude of actual floods but also the occurrence of temporary drought conditions.

^d Nutrient availability describes the actual concentration of nutrients in the water whereas nutrient load describes the amount of nutrients being leaked from the catchment.

^e Shore management involves actions keeping shore vegetation short.

^f Naturalness is a perceived property.

^g Total wetland area and catchment size concerns the effect of these properties on processes in the individual wetlands. It relates to the amount of inflowing water, affecting the hydraulic loading, for flow regulation objectives. However, total wetland area has another meaning for other objectives, and for biodiversity conservation it concerns connectivity to other similar habitats.

^h Accessibility includes proximity to urban areas, parking lots, boardwalks and other features increasing visitor pleasure.

ⁱ Comparing equally large inflows.

^j Focussing on MeHg producing wetland systems, ignoring the set of wetlands that also act as sinks.

^k Greenhouse gas emissions concern per unit wetland area.

^l Main targets include total abundance and/or total species richness.

area so that a large fraction of total runoff, and thereby also the nutrient/pollutant load, goes through several wetlands in the catchment (Åhlén et al., 2020). Nutrient retention may also be low at a higher Aw:Ac ratio if Aw is distributed such that most of the total water discharge does not flow through the wetlands in the wetlandscape (Quin et al., 2015). Finally, the temporal variation of retention at the wetlandscape scale depends on source type (currently active sources at the surface, or long-lived legacy sources remaining from earlier inputs). Active sources at the surface yield relatively stable average load and variable concentrations, whereas legacy sources yield more stable average concentrations (Chen et al., 2021; Destouni et al., 2021).

Wetlands can be hotspots for pollutants such as mercury (Hg), microplastics and pesticides which can contribute to a poor environmental status in rivers and lakes (St Louis et al., 1996; Vymazal and Březinová, 2015; Qian et al., 2021). Hg pollution is a concern primarily in aquatic systems, where methylmercury (MeHg) accumulates up the food chain and reaches toxic levels, and the Hg burden to these systems increases with the relative wetland coverage in the catchment (Chasar et al., 2009; Glover et al., 2010). Hg problems may increase with wetland restoration by turning sites into “hot-spots” for bacterial methylation of inorganic divalent Hg (St Louis et al., 1996; Tjerngren et al., 2012a; Tjerngren et al.,

2012b), but wetlands can also trap total Hg and limit inorganic Hg availability for methylation in downstream systems (Kronberg et al., 2012; Tjerngren et al., 2012b). However, Tjerngren et al. (2012b) found that the risk of MeHg production is mostly greater in restored wetlands than the potential benefit for total Hg retention. Studies on MeHg production in wetlands suggest that net methylation is highest at intermediate nutrient concentrations (Tjerngren et al., 2012a; Tjerngren et al., 2012b), but also varies with sulphate deposition (Åkerblom et al., 2013) and the microbial community (Schaefer et al., 2020; Xu et al., 2021). Redox changes (caused by water fluctuations) may also promote Hg methylation while photochemical demethylation may remove MeHg (Table 1).

In summary, wetlands play a vital role in water, nutrient and pollutant fluxes. The wetland capacity to modulate (extreme) flows depends on its size, or more specifically the storage volume of its surrounding depression. The extent to which a wetland may modulate flows generated by specific precipitation events depends on (i) the extent to which event flows actually pass the wetland, which is governed by its catchment size and the presence of consolidation drainage (McCauley et al., 2015), and (ii) the extent to which the wetland's volume is already filled, as reflected by its pre-flood water level. The total nutrient retention is similarly higher in voluminous wetlands and increases with the mean water residence time. Whereas retention at the individual wetland level can be predicted based on these properties, retention at the wetlandscape level is more affected by the position of wetlands across the catchment where it is important that major nutrient flows are channeled through one or more wetlands.

2.3. Climate mitigation and feedback processes

Wetlands, including shallow lakes and ponds, play globally important roles in carbon cycling and greenhouse gas (GHG) emission by acting as sources and sinks of methane (CH₄), carbon dioxide (CO₂), and nitrous oxide (N₂O), even though estimates are uncertain and vary with wetland definition (Raymond et al., 2013; Saunio et al., 2020; Tian et al., 2020; Rosentreter et al., 2021). However, wetlands also have the capacity to sequester and bury large volumes of C in soils and sediments, which is particularly apparent for *Sphagnum*-rich peatlands (Mitra et al., 2005; Downing et al., 2008). Hence, targets for wetland re-establishment may be to reduce GHG emissions and increase C sequestration (Table 1).

Wetlands are on average net CO₂ sinks because waterlogging limits decomposition of organic matter and store very large amounts of carbon, but wetlands may turn into CO₂ sources due to drainage and disturbance (Bridgman et al., 2006; Hugelius et al., 2020; Evans et al., 2021). Although occupying little space, small permanently inundated wetlands and ponds can be important for C sequestration at the landscape scale by holding disproportionately large stocks of C in soils and sediments (Gilbert et al., 2021). Conventional thinking is that eutrophication increases aquatic primary production, thus enhancing the C sink (Pacheco et al., 2014), but some studies of small agricultural waterbodies show the opposite; that elevated nutrient inputs lead to higher CO₂ emissions (Ollivier et al., 2019; Peacock et al., 2021). Plant species can also affect the C sink, and particularly emergent vegetation such as reeds and bulrushes may be key to enhancing C accumulation in constructed wetlands and ponds (Moore and Hunt, 2012).

CH₄ production is an anaerobic process, while oxidation is an aerobic process. CH₄ emissions are therefore lower in sites where shallow water depth allows oxygenation of the full water column or where deeper anoxic zones are overlain by oxygenated water tables or soil layers where CH₄ produced at depth can be oxidized before reaching the atmosphere (Evans et al., 2021). Emissions are also generally larger from eutrophic ecosystems and at higher temperatures because these conditions favor for CH₄-producing microbes (Gedney et al., 2004; Johansson et al., 2004; Beaulieu et al., 2019), and varies with the wetland vegetation type. Some plant species enhance emissions by transporting CH₄ from anoxic soils and sediments to the atmosphere through their aerenchymatous tissue (Dacey and Klug, 1979; Sebacher et al., 1985) or by releasing labile substrates that fuel CH₄ production (Ström et al., 2012). On the other hand,

free-floating plant species can limit emissions by reducing gas exchange and trapping CH₄-rich gas bubbles before reaching the water surface (Kosten et al., 2016). These effects may be counteracted by herbivorous animals grazing water plants (Winton and Richardson, 2017), and even fish may affect emissions by consuming zooplankton that otherwise prey upon methanotrophic microbes (Devlin et al., 2015).

N₂O emissions are similarly controlled by the depth to water table in non-inundated wetlands, with larger emissions from drier sites due to increased N mineralization, although this effect may be modulated by high nutrient levels with minimal effects in ombrotrophic wetlands (Martikainen et al., 1993). Elevated nutrient levels, specifically N, result in higher N₂O emissions, but emissions may also increase at low organic C:N (Søvik et al., 2006; Peacock et al., 2017; Webb et al., 2021). However, emissions are not always large even from N-rich wetlands, and some wetlands even act as N₂O sinks (Webb et al., 2019; Mander et al., 2021).

In summary, GHG emissions and C storage in wetlands and small waterbodies are mostly controlled by the same drivers: depth to water table, nutrient status, food web structure, vegetation and wetland type (mineral wetlands versus peatlands). However, drivers do not always act in the same direction in an overall C budget but an optimal water level for GHG reduction is suggested to be 10–30 cm below the surface (Evans et al., 2021). In some wetland types, emergent vegetation may enhance CH₄ emissions but also increase C sequestration. Although comparisons of the three GHGs is not straightforward due to their differing atmospheric lifetimes (Forster et al., 2021), calculations suggest that the wetland CO₂ sink “wins out” against the CH₄ source on longer time scales, with a net reducing effect on climatic warming (Günther et al., 2020). Regardless of the timescale and metric used for evaluation, wetland management should aim to enhance CO₂ uptake, minimize N₂O emissions, and accept that CH₄ release is an inevitable function of all wetlands.

2.4. Biodiversity conservation

Biodiversity conservation objectives set by authorities and landowners during wetland re-establishment are often vague and limited to broad indicator species groups such as birds or amphibians (e.g. Everard, 2008). Here, we include those groups but add less well-studied target groups where data are accessible (Table 1). These added groups include taxa whose habitat is open water (aquatic macroinvertebrates and submerged plants) and taxa thriving in wetlands without open water (spiders, carabids and pollinators). We do not separate total abundance and species number, because these typically correlate and we mostly lack information on the processes determining biodiversity in and around wetlands beyond the consensus that heterogeneity is good. Moreover, studies on wetland biodiversity in relation to environmental variables seldom separate species depending on their wetland affinity. Additionally, whereas estimated relationships are typically linear, true habitat requirements are often hump-shaped relative to environmental variables (indicated by +/– in Table 1).

That wetlands are important for biodiversity almost goes without saying, given that many species need water bodies directly or indirectly for their development. Re-establishing wetlands in homogeneous agricultural landscapes always leads to establishment of species that are unable to survive otherwise, and even small ponds that are created for reasons other than biodiversity support may harbor considerable diversity of aquatic macroinvertebrates, birds and amphibians (Knutson et al., 2004; Sanchez-Zapata et al., 2005; Hsu et al., 2011; Strand and Weisner, 2013; Johansson et al., 2019a). It is equally established that larger habitats generally increase species number, as is also shown for most wetland taxa (Pearman, 1993; Hansson et al., 2005; Le Gall et al., 2018; Kačergytė et al., 2021a), because larger areas support larger population sizes and are also typically more heterogeneous (Datry et al., 2014). Beside size, wetlands also differ in quality to different species depending on nutrient conditions, topography, soil properties, management, etc. This environmental variation is important because it increases dissimilarity in species composition between wetlands where mixtures of wetland types result in higher diversity at the landscape level (Scheffer et al., 2006; Thiere et al., 2009;

Žmihorski et al., 2016). Hence, it may be advantageous if the total wetland area is split up into multiple entities (see the SLOSS-debate, Fahrig, 2020) because multiple small wetlands may contain more habitat types than a single large wetland and thus increase both β and γ -diversity (Kačergytė et al., 2021a).

Although most taxa benefit directly from larger habitat sizes, other processes correlated with wetland size may complicate patterns of biodiversity-wetland size relationships. For instance, small and isolated ponds are more likely fish free because fish are less able to colonize these waters, which may favor predator-sensitive groups such as amphibians, several macroinvertebrates and some birds (e.g., Scheffer et al., 2006; Semlitsch et al., 2015; Kačergytė et al., 2021b). Because aquatic taxa such as dragonflies and many midges spend adult stages above the water, fish predation may even affect abundance of terrestrial organisms that feed on or are consumed by these taxa (Nakano et al., 1999; Baxter et al., 2005; Knight et al., 2005), and draining ponds during winter to remove fish may thereby benefit arthropod diversity (Lemmens et al., 2013). Finally, small and shallow wetlands are warmer on average, which may enhance development rates and abundances of amphibians and macroinvertebrates (Greig et al., 2012; Rannap et al., 2020), acting in an opposite direction to the positive species-area relationship.

Emergent vegetation (reed *Phragmites*, cattails *Typha*, etc.) commonly develop in undisturbed wetlands with shallow edges which may benefit animal taxa by providing food, nesting habitat, and by serving as refuges against predation from fish and other predators when occurring at intermediate coverage (Cazzanelli et al., 2008; Ma et al., 2010; Shulse et al., 2012). Because emergent plants generally have strong dispersal capacity and colonize even isolated wetlands, local conditions such as shallow water, high nutrient concentrations and shoreline grazing may be more important than the number of wetlands in the landscape for their abundance (Vretare et al., 2001; Santamaría, 2002; Weisner and Strand, 2002). Submerged plants are more negatively affected by high nutrient concentrations because algal blooms in nutrient-rich ponds increase turbidity, cause periphytic overgrowth and reduce light availability (Weisner et al., 1997). More nutrients in the water typically also benefit higher order consumers through the increased primary productivity both in the water and through emergent insect biomass on nearby shores (Greig et al., 2012), but the algal soup at extreme nutrient loads seems less attractive to birds and other animals (Hansson et al., 1998; Strand and Weisner, 2001). In wetlands with high P-loads, blooms of toxic cyanobacteria may affect populations of insects, such as bees, as well as livestock (May and McBarron, 1973; Delaney and Wilkins, 1995; Stewart et al., 2008).

Although high plant biomass provides food for herbivores, overgrown wetland shores with high biomass are typically dominated by emergent vegetation that may support a lower plant and animal diversity than more open shorelines (e.g., Ma et al., 2010; Lehikoinen et al., 2017; Weilhoefer et al., 2017) even though older areas of emergent vegetation can have a diverse and specialized fauna (Andersen et al., 2021). Therefore, riparian grazing or other management actions that reduce plant biomass, and increase heterogeneity, within the wetland may provide better habitats with more flowering plants for pollinators and higher food availability for predatory arthropods and birds (Žmihorski et al., 2016; Lehikoinen et al., 2017; Lewis-Phillips et al., 2020; Walton et al., 2021). Accordingly, intermediate grazing intensities or mowing in wetlands support a high diversity of both arthropods and birds (Cattin et al., 2003; Smart et al., 2006; Cole et al., 2015; Žmihorski et al., 2016), whereas too heavy grazing is generally negative for both plants and insects because vegetation is cut very short and because cattle entering the water at high densities increase nutrient loads and turbidity (Knutson et al., 2004; Declerck et al., 2006; Durant et al., 2008). At very high grazing intensities, arthropod species may even benefit from fenced buffer strips along waterways (Cole et al., 2012).

Water table fluctuations may additionally enhance habitat conditions and increase habitat heterogeneity for wetland arthropods, birds and amphibians through nutrient inputs and by exposing bare ground (Porej and Hetherington, 2005; Smart et al., 2006; Datry et al., 2014; Lafage and Petillon, 2016; Žmihorski et al., 2016; Daniel et al., 2019). A natural

hydrological regime may even be more important than vegetation management because extreme events of both high and low flood events often favor habitat generalist species and not those of highest conservation value (Bonn et al., 2002; Lambeets et al., 2008; Lafage and Petillon, 2016). Many terrestrial species are wetland specialists, even though most species are poorly adapted to floods and need to recolonize when the water subsides (Plum, 2005; Rothenbacher and Schaefer, 2006; Lambeets et al., 2008; Batzer and Wu, 2020). This harsh environment leads to the exclusion of ground-nesting species such as ants and many bees (Batzer and Wu, 2020).

Mosquito larvae develop in stagnant and standing waters (Becker et al., 2010) and wetland establishment therefore benefits mosquitoes (Yadav et al., 2012). Even though most biodiversity is viewed positively by landowners, worries about mosquito infestations may reduce willingness to create wetlands (Hanford et al., 2019). However, mosquitoes are not a homogeneous group (Schäfer et al., 2004) and most species in temperate regions occur in permanent waters where their numbers are regulated by predator populations and rarely reach nuisance numbers (Becker et al., 2010), but some species may vector mosquito borne infections (Kampen and Walther, 2018). The current main worries instead concern flood water mosquitoes (e.g., *Aedes sticticus*), that lay eggs on regularly flooded soils and may reach extreme densities in some areas with high summer floods, but where grazing and other management that reduce vegetation height seem to reduce their abundance (Lindström et al., 2021).

In summary, heterogeneity within wetlands and at the wetlandscape level is generally considered to increase species diversity. Seasonal floods and droughts at an appropriate frequency and intensity seem to be of key importance for many wetland specialist species, whereas wetlands with too high or too low flooding experience a shift to species assemblages with a higher proportion of habitat generalists. Grazing is also positive at intermediate levels by creating spatial heterogeneity in vegetation height with an abundance of resources for pollinators and other arthropods, whereas high grazing intensities lead to very short grass that may benefit some wading birds but few arthropods. Finally, it seems evident that fish presence generally reduces the diversity of other species groups and methods to control fish abundance may be necessary for maintaining high diversity in small wetlands. In either case, future evaluations of biodiversity values should include a larger range of taxa than birds, amphibians and fish, because different groups are favored by different wetland properties.

2.5. Cultural ecosystem services

Wetlands provide benefits to human mental and physical wellbeing, and support important economic activities such as tourism. Development and management of cultural ecosystem services (CES) are therefore often important objectives for wetland re-establishment (Table 1), particularly when located in urban or peri-urban areas close to human populations (Blicharska and Johansson, 2016; Johansson et al., 2019b; Pedersen et al., 2019). Consequently, considerations regarding wetland accessibility for human visitation, including proximity to urban centers, rights of access, and availability and condition of visitation infrastructure such as paths, signs and car parking, are important for many wetland re-establishment projects (Ghermandi and Fichtman, 2015; Jensen et al., 2019; Johansson et al., 2019b). Education is also linked to accessibility, especially visitation infrastructure such as information boards and signs as well as the presence of guides and organized tours (Ghermandi and Fichtman, 2015; Margaryan et al., 2018), but educational CES have also been linked to the proximity of urban centers and to the history of usage for educational purposes (Moore and Hunt, 2012).

While many types of CES may be indirectly linked to structural wetland properties, via properties such as biodiversity and the presence of specific taxa, direct relationships are little studied. Wetland birds provide perhaps the most prominent example, and are frequently linked to recreation, inspiration, education, sense of place and cultural heritage (Manuel, 2003; Söderqvist, 2003; Green and Elmberg, 2014; Blicharska and Rönnbäck, 2018). Similarly, recreational fishing and hunting are often linked to

specific groups of target species or taxa, and thereby indirectly to structural elements and biodiversity assemblages supporting these species.

Although recreation and aesthetics are arguably the most directly perceived and most studied CES (e.g., Plieninger et al., 2013), less is known regarding how wetland properties influence these services (Alikhani et al., 2021). Stretches of open water are often perceived as positive, particularly for recreational activities such as ice-skating, bird-watching and hunting (Wahlroos et al., 2015; Margaryan et al., 2018), and, along with a relatively complex shoreline and fluctuating water levels, are often associated with higher aesthetic appreciation (Manuel, 2003; Johansson et al., 2019b; Pedersen et al., 2019). Similarly, good water quality is important for several recreational pursuits (Ghermandi and Fichtman, 2015), and algal soups caused by high nutrient loads may reduce aesthetic appeal. The relationship between emergent vegetation and aesthetic value is less clear. Whereas preferences can be related to mirroring water surfaces (Margaryan et al., 2018), emergent vegetation may also contribute to the relative complexity of the wetland landscape, resulting in higher aesthetic value (Pedersen et al., 2019). Additionally, the perceived naturalness or wildness of a wetland, including the presence of natural sounds and scents, is key to its aesthetic appeal (Manuel, 2003; Wahlroos et al., 2015). The presence of modern infrastructure to facilitate accessibility may therefore decrease the aesthetic benefits (Pueyo-Ros et al., 2019), whereas wetland management using traditional methods such as hay-cutting and shore-grazing instead increase both aesthetic and cultural heritage service provision (Naturvårdsverket, 2019). The properties of areas surrounding wetlands, such as riparian forests, may similarly improve the provision of recreational services (Scholte et al., 2016) by an aesthetic appeal or by providing shaded conditions, and peatlands may represent important historical archives on past climates and vegetation.

Beyond recreational activities and aesthetics, wetlandscapes are often perceived as sources of inspiration (Blicharska and Rönnbäck, 2018; Margaryan et al., 2018). Sense of place, local identity and place attachment are linked to more natural wetlands compared to constructed wetlands (Alikhani et al., 2021), and to more complex and diverse wetlands (Wahlroos et al., 2015), however there is little research covering these issues.

3. Tradeoffs and synergies within individual wetlands

In this section, we first identify and discuss potential tradeoffs (opposing signs along columns in Table 1) and synergies (same signs) between wetland properties and objectives for wetland re-establishment. We then review studies that explicitly explored the existence of tradeoffs or synergies within and between objectives through observational studies or experiments at relevant scales, and finally suggest approaches for better understanding tradeoffs and synergies acting either through chains of causal relationships or because processes interact directly (Fig. 1).

Table 1 identifies potential tradeoffs between objectives aimed at reducing GHG emissions and those promoting biodiversity conservation, and, thus, indirectly, with several cultural services that are strongly linked to biodiversity or to the availability of specific taxa. Even though production processes vary between GHGs, emissions are generally higher in flood zones within wetlands and increase due to shore grazing and nutrient availability, whereas these same properties at intermediate levels tend to increase the abundance and species richness of plant and animal taxa. Traditional management methods such as shore grazing and haymaking are linked to provision of aesthetic and cultural heritage services whereas processes to promote biodiversity conservation may negatively impact on GHG mitigation because animals grazing on aquatic vegetation may increase CH₄ emissions (e.g., Winton and Richardson, 2017).

Tradeoffs between objectives to reduce GHG and those to reduce recipient nutrient loads may occur because higher nutrient loads in wetlands typically increase CH₄ and N₂O emissions. However, the net effect on climate mitigation may be positive because of increased C sequestration in nutrient rich wetlands. High nutrient loads can also have both positive and negative impacts on biodiversity, depending on spatial scale (Chase and

Leibold, 2002). For instance, the very high nutrient concentrations achieved in efficient nutrient retention wetlands may cause fast succession towards less biodiversity-friendly vegetation states, both in the form of algal soups in the water and dominance by large emergent plants along shorelines, and indirectly affect several forms of aquatic recreation negatively. One approach to balance this effect may be to distribute the nutrient loads over more wetlands, but that may reduce the efficiency of each wetland, decrease cost-efficiency and remove more productive land.

A tradeoff is also apparent between biodiversity conservation objectives and the interest of landowners to maintain fish populations for recreational fishing, because many taxa are vulnerable to fish predation and competition and because fish may increase resuspension and internal P loading. Finally, the issue of water table fluctuations is generally contentious. A natural hydrological regime with seasonal flooding seems essential for many taxa, including arthropods and birds, but may cause problems for landowners, who respond by constructing ponds with steep slopes without flood zones. A natural hydrological regime may also not be optimal for water retention and flood peak mitigation.

Even though tradeoffs are common, evidence of synergies and co-production of ecosystem services are also apparent (Table 1). Partial synergies are linked to the mostly positive impact of emergent and submerged vegetation at intermediate densities on many key objectives, including nutrient retention, reduction of GHG, and biodiversity conservation. Similarly, properties relating to wetland area appear to have largely positive impacts on flood control, water storage, nutrient retention, reduction of CH₄, and biodiversity conservation objectives. Given the many indirect impacts of biodiversity on cultural ecosystem services such as recreation and inspiration, it seems likely that increases in wetland area may support increased provision of these services. Shoreline complexity is similarly shown to have mostly synergetic impacts on both biodiversity conservation and cultural services.

Although our study indicates that tradeoffs and synergies between objectives for wetland re-establishment may be common, as implicated in previous papers, few studies provide quantitative data (for overview see Table 2) and patterns have mainly been treated phenomenologically. Jessop et al. (2015) provided perhaps the most comprehensive analysis by estimating proxies for nutrient retention, carbon sequestration, water storage and biodiversity (birds and amphibians) in 30 mitigation wetlands across Illinois (USA). Their study confirms one finding from Table 1, that carbon storage (measured as soil organic matter content) and nitrogen retention (measured as denitrification potential) are positively related, whereas both are negatively related to bird abundance and perhaps to amphibian abundance because high primary production leads to high emergent vegetation and algal production that result in reduced opportunities for foraging and reproduction. The study did not connect these objectives and wetland properties, making conclusions about underlying mechanisms difficult. However, they estimated that land use in the surrounding landscape, especially greater proportions of agricultural land use, seemed to decrease both amphibian and bird abundances. Other studies are narrower in scope, focusing on fewer objectives. For instance, Boughton et al. (2019) examined tradeoffs between flood control and biodiversity objectives in Florida (USA). They showed that improved water retention to reduce flood risks downstream also increased emergent vegetation, decreased amphibian densities and that aquatic macroinvertebrates were most abundant at intermediate water tables. Similarly, Herring et al. (2021) showed that reduced water use in rice paddies negatively affected waterbirds by decreasing the time available for chick developments.

Our review indicates that current knowledge of tradeoffs and synergies is insufficient for firm conclusions. Several key properties are not sufficiently studied to develop more precise predictions on when and where tradeoffs and synergies are more or less likely. This lack of studies is particularly evident for wetland cultural ecosystem services, where many knowledge gaps exist. Additionally, relationships may be complicated and non-linear, which points to the necessity for additional studies where multiple response variables are estimated simultaneously in large-scale observational or experimental studies. Some processes, such as grazing, may be

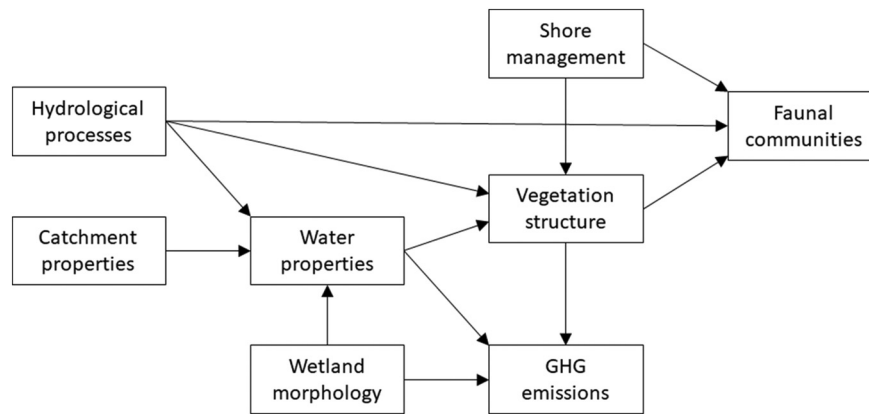


Fig. 1. Flow chart showing tentative causal relationships between structural and hydrological properties of wetlands and measures of biodiversity and greenhouse gas (GHG) emissions.

possible to manipulate by fencing parts of a wetland whereas other processes, e.g., hydrological functions, necessitate that entire wetlands are manipulated. Such large scale manipulations may for instance be performed in constructed wetlands where in- and outlets can be controlled, and may be needed to observe effects on animal densities. Such studies would also benefit from adopting a transdisciplinary approach, given the need to integrate different types of knowledge concerning wetlands functions, not only across multiple scientific disciplines but also practical knowledge concerning land-uses from a diversity of potential beneficiaries (as suggested by Cimon-Morin et al., 2021). Dynamic simulation modelling may similarly play an important role in developing the knowledge base, and lead to quantitative predictions on interactive effects, but requires the careful development of theories concerning causal relationships and underlying mechanisms (Fig. 1). Finally, it is apparent that measurements of some processes in previously performed studies were quite coarse, which made evaluations imprecise. Future studies would benefit from carefully considering the type of measures for estimated processes.

4. Multifunctional wetlandscapes – conclusions

Our findings show that, despite increasing demands, the ability of individual wetlands to deliver multiple societal objectives is typically limited by inherent tradeoffs. We therefore argue that a more effective strategy, offering several potential advantages, may be to aim for multifunctional wetlandscapes featuring an array of potentially interconnected wetlands with each wetland configured to optimize delivery of specific objectives based on its unique set of properties and position in the landscape (Fig. 2). First, control of larger floods is best accomplished by large wetlands or lakes situated low in the landscape yet upstream of urban populations and infrastructure where demand for flood control is highest, because wetlands higher up in the catchment are more often saturated and less able to buffer high water flows (Acreman and Holden, 2013). However, small

wetlands dispersed through the catchment may also be important for control of smaller and more localized floods (Dadson et al., 2017) and to reduce the rate at which water moves through the landscape during severe flood events (Bullock and Acreman, 2003). Second, nutrient leakage is highest downstream of major nutrient sources such as productive agricultural fields, which are typically found at relatively low elevation in the landscape. Retention leading to less runoff into streams, lakes and oceans may therefore be best accomplished by wetlands clustered close to nutrient sources, upstream of outlet points into large waterbodies, and along major water flow paths (Quin et al., 2015; Thorslund et al., 2017). Third, the demand for reducing GHG leakage caused by human disturbance is highest in peatlands with high amounts of stored C. Therefore, rewetting drained peatlands high up in the catchment would most efficiently decrease CO₂ emissions and maintaining a stable water table just below the peat surface (10–30 cm) would minimize CH₄ emissions (Evans et al., 2021). Finally, current wetland restorations with the aim to promote biodiversity have a strong focus on nutrient rich wetlands in agricultural areas because of their value for birds and amphibians, a value that is often promoted by allowing cattle to graze along shorelines. While this focus is commendable, maximized regional diversity may be best achieved by increasing heterogeneity through a mixture of wetland types (Datry et al., 2014; Swartz and Miller, 2021), and by restoring a natural hydrological regime (Lafage and Petillon, 2016).

Evaluating multifunctionality at the landscape scale relies on proxies for objectives, often combined into multicriteria scores (Cimon-Morin and Poulin, 2018) and underlying decision support tools such as systematic conservation planning (Cimon-Morin et al., 2021). As an example of the type of statistics and analytical tools needed for such analyses, Odgaard et al. (2017) used publicly available data (including proxies for biodiversity conservation, nutrient mitigation, public use and flood risks) at a 15 ha catchment scale, through a combination of scenario testing and hotspot analysis, to analyze optimal and realized distributions of wetland reconstruction

Table 2
Empirical studies examining relationships between different types of objectives.

Objective 1	Objective 2	Relation	Mechanism	Reference
Nutrient retention	Waterbird abundance	–	Higher nutrient availability leads to high vegetation that negatively affect birds.	(Jessop et al., 2015)
Carbon storage	Waterbird abundance	–	Higher nutrient availability leads to high plant production that negatively affect birds	(Jessop et al., 2015)
Waterbird abundance	Nutrient retention	–	Waterbirds increase nutrient transport from cropland to water	(Kitchell et al., 1999)
Nitrogen retention	Carbon storage	+	Higher nutrients increase gross primary production	(Jessop et al., 2015)
Hunting intensity	Waterbird abundance	0		(Hagy et al., 2017)
Water retention to reduce flooding	Emergent vegetation	+	Less drought increases vegetation growth	(Boughton et al., 2019)
Water retention to reduce flooding	Amphibian abundance	–	Unclear mechanism	(Boughton et al., 2019)
Water retention to reduce flooding	Macroinvertebrate abundance	+ / –	Maximum abundance at intermediate maximum water depths	(Boughton et al., 2019)
Water retention to conserve water	Waterbird abundance	–	Water saving methods in rice farming reduce pond availability	(Herring et al., 2021)

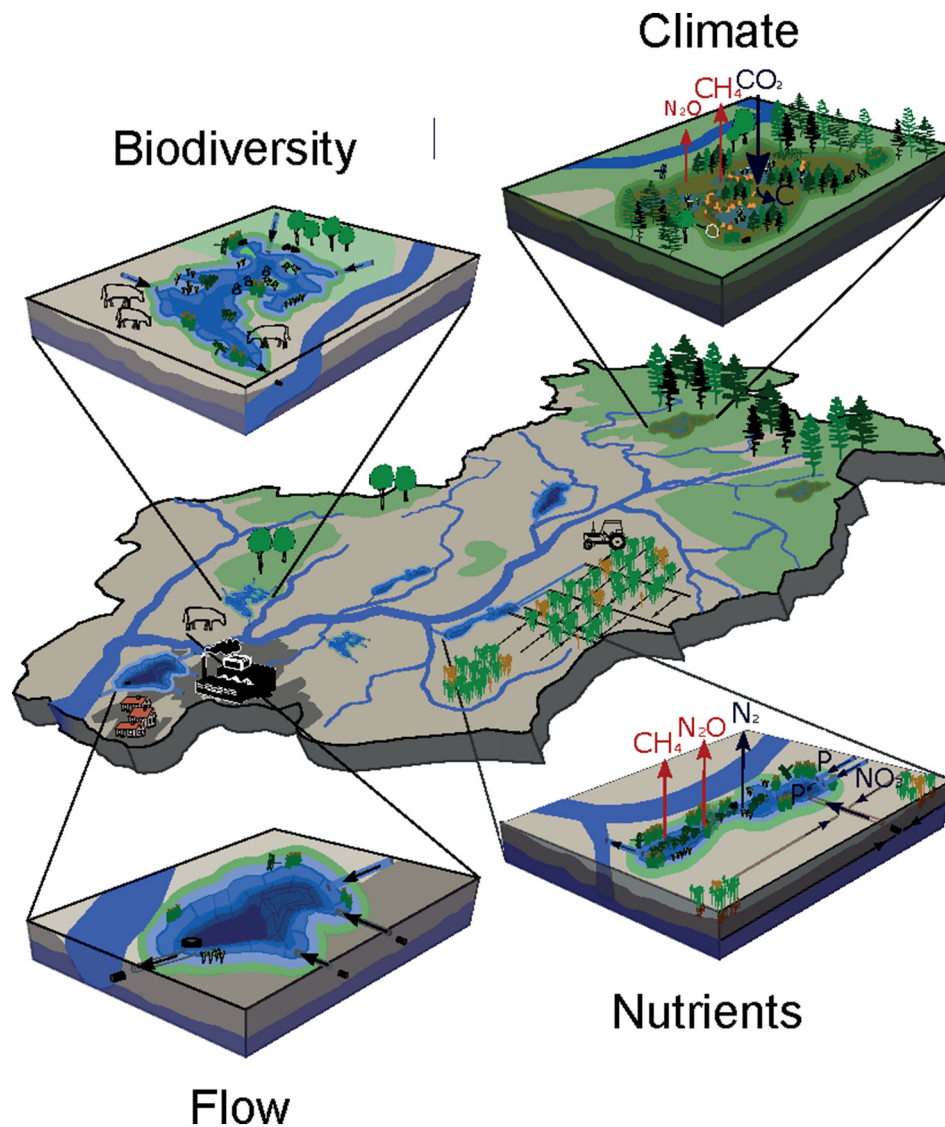


Fig. 2. The placement for wetland re-establishments depends on objectives. Net reduction of GHG radiative forcing is maximized when rewetting C-rich peatlands to a state with stable hydrological regimes, most easily achieved high up in the wetlandscape. Nutrient retention is maximized when wetlands are downstream from high nutrient loads in the agricultural landscape. Flood control is maximized by the placement of large wetlands low in the catchment and cultural values (not explicitly shown in figure) are largest when wetlands are accessible from urban areas. Biodiversity conservation is place-dependent because wetlands in different parts of a wetlandscape house different species, but restoration efforts mainly target nutrient rich wetlands where vegetation is kept short through grazing in order to promote bird diversity. Illustration: L.D. Schneider.

sites (see also Rapinel et al., 2016 for a case using satellite data). An alternative approach involves quantitative modelling with the same goal of identifying spatial mismatches (Thorslund et al., 2017), although such an approach is only possible for a subset of objectives. For instance, Brandt et al. (2009) used quantitative hydrological modelling to show that many wetlands are poorly placed for high nutrient retention, which necessitates the development of novel tools for modelling water and nutrient load in order to estimate wetland area and potential nutrient retention as developed by Djodjic et al. (2020). Moreover, analyses of nutrient retention in particular should focus not only on current sources but also on legacy nutrients that have loaded soil and huge groundwater volumes over decades (Basu et al., 2022) where they slowly leak to recipients in a diffuse manner causing failed attempts to reduce nutrient loads (Destouni et al., 2021). When constructing wetlands, such legacy effects can be reduced by removing topsoil (Audet et al., 2020), but wetland re-establishment can also specifically target legacy-sources identified from water quality monitoring data (Destouni and Jarsjö, 2018). These analyses indicate the need of not only evaluating the effects of single restoration objects but also to relate

effects and spatial configurations to key demands from society (Cimon-Morin and Poulin, 2018). In particular, re-establishments of wetlands should be planned to complement already present landscape features and may involve both restoration efforts and preservation of pristine sites (Goyette et al., 2021).

Beyond new knowledge and tools, the development and management of multifunctional wetlandscapes over time also implies an engagement with the socio-political dimension of wetlandscapes and an application of collaborative governance approaches that identify benefits and priorities among stakeholders (e.g., Doyle, 2020). Where wetland re-establishment is mainly based on voluntary landowner interests, catchment officers acting as advisors in wetland constructions assist in this respect; an assessment of the efficiency of existing wetlands in a given catchment can help identify which additional wetland types could optimize multifunctionality at the wetlandscape scale. Additionally, existing social and governance arrangements define the range of actions that may be taken, including how they are financed and implemented, e.g. through different policy instruments (Lindahl and Söderqvist, 2004), as well as determine the degree to which

various actors may participate in decision-making processes. New actors, e.g. NGOs, can provide important new perspectives and resources for wetlands re-establishment, but their participation may be limited due to the hierarchical structure of existing governance arrangements (e.g., Dawson et al., 2017; Dawson et al., 2021). Finally, governance and management of the complex social-ecological systems at the wetlandscape scales requires additional resources (time, money, human) and multi-competent leaders but also careful developments of long-term learning-focused collaborations among diverse groups of land-owners and government bodies separately involved in specific objectives. For this reason, programs and other policy instruments operating at the wetlandscape scale are likely to be more challenging to implement than the current suite of policy instruments supporting local wetland re-establishment.

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CRedit authorship contribution statement

PAH, JJ and GH conceived the idea for the paper, PAH together with LD, PG, JJ, IK, MP and MB wrote the first draft of the paper, all authors were involved in the discussion at the workshop and provided comments and suggestion to both the text and the table and LDS made the artwork for Fig. 2.

Data availability

No data was used for the research described in the article.

Declaration of competing interest

There are no financial or personal interests affecting the contents of this paper.

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References

- Acreman, M., Holden, J., 2013. How wetlands affect floods. *Wetlands* 33, 773–786. <https://doi.org/10.1007/s13157-013-0473-2>.
- Acreman, M.C., Harding, R.J., Lloyd, C., McNamara, N.P., Mountford, J.O., Mould, D.J., et al., 2011. Trade-off in ecosystem services of the Somerset levels and Moors wetlands. *Hydrol. Sci. J.* 56, 1543–1565. <https://doi.org/10.1080/02626667.2011.629783>.
- Adamus, P.R., Clairain, E.J., Smith, R.D., Young, R.E., 1987. *Wetland Evaluation Technique (WET). Volume II. Methodology.* US Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi.
- Åhlén, I., Hambäck, P., Thorslund, J., Frampton, A., Destouni, G., Jarsjö, J., 2020. Wetlandscape size thresholds for ecosystem service delivery: evidence from the Norrström drainage basin, Sweden. *Sci. Total Environ.* 704, 135452. <https://doi.org/10.1016/j.scitotenv.2019.135452>.
- Åhlén, I., Vigouroux, G., Destouni, G., Pietron, J., Ghajarnia, N., Anaya, J., et al., 2021. Hydroclimatic changes of wetlandscapes across the world. *Sci. Rep.* 11, 2754. <https://doi.org/10.1038/s41598-021-81137-3>.
- Åhlén, I., Thorslund, J., Hambäck, P., Destouni, G., Jarsjö, J., 2022. Wetland position in the landscape: impact on water storage and flood buffering. *Ecology* e2458. <https://doi.org/10.1002/eco.2458>.
- Åkerblom, S., Bishop, K., Björn, E., Lambertsson, L., Eriksson, T., Nilsson, M.B., 2013. Significant interaction effects from sulfate deposition and climate on sulfur concentrations

- constitute major controls on methylmercury production in peatlands. *Geochim. Cosmochim. Acta* 102, 1–11. <https://doi.org/10.1016/j.gca.2012.10.025>.
- Alikhani, S., Nummi, P., Ojala, A., 2021. Urban wetlands: a review on ecological and cultural values. *Water* 13, 3301. <https://doi.org/10.3390/w13223301>.
- Andersen, L.H., Nummi, P., Rafn, J., Frederiksen, C.M.S., Kristjansen, M.P., Lauridsen, T.L., et al., 2021. Can reed harvest be used as a management strategy for improving invertebrate biomass and diversity? *J. Environ. Manag.* 300, 113637. <https://doi.org/10.1016/j.jenvman.2021.113637>.
- Andersson, K., 2012. *Multifunctional Wetlands and Stakeholder Engagement: Lessons From Sweden.* SEI Working Paper No. 2012-08.
- Audet, J., Zak, D., Bidstrup, J., Hoffmann, C.C., 2020. Nitrogen and phosphorus retention in Danish restored wetlands. *Ambio* 49, 324–336. <https://doi.org/10.1007/s13280-019-01181-2>.
- Basu, N.B., Van Meter, K.J., Byrnes, D.K., Van Cappellen, P., Brouwer, R., Jacobsen, B.H., et al., 2022. Managing nitrogen legacies to accelerate water quality improvement. *Nat. Geosci.* 15, 97–105. <https://doi.org/10.1038/s41561-021-00889-9>.
- Batzer, D.P., Wu, H., 2020. Ecology of terrestrial arthropods in freshwater wetlands. *Annu. Rev. Entomol.* 65, 101–119. <https://doi.org/10.1007/s11069-006-9003-3>.
- Baxter, C.V., Fausch, K.D., Saunders, W.C., 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. *Freshw. Biol.* 50, 201–220. <https://doi.org/10.1111/j.1365-2427.2004.01328.x>.
- Beaulieu, J.J., DelSontro, T., Downing, J.A., 2019. Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. *Nat. Commun.* 10, 1375. <https://doi.org/10.1038/s41467-019-09100-5>.
- Becker, N., Petric, D., Zgomba, M., Dahl, C., Madon, M., Kaiser, A., 2010. *Mosquitoes and their control.* Springer Verlag.
- Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. *Ecol. Lett.* 12, 1394–1404. <https://doi.org/10.1111/j.1461-0248.2009.01387.x>.
- Blackwell, M.S.A., Pilgrim, E.S., 2011. Ecosystem services delivered by small-scale wetlands. *Hydrol. Sci. J.* 56, 1467–1484. <https://doi.org/10.1080/02626667.2011.630317>.
- Blicharska, M., Johansson, F., 2016. Urban ponds for people and by people. In: Francis, R., Millington, J., Chadwick, M.A. (Eds.), *Urban Landscape Ecology: Science, Policy and Practice.* Routledge, pp. 164–180.
- Blicharska, M., Rönnbäck, P., 2018. What factors enable or hinder engagement of civil society in ecosystem management? The case of 'pike factories' and wetland restoration in Sweden. *J. Environ. Plan.* 61, 950–969. <https://doi.org/10.1080/09640568.2017.1350145>.
- Bonn, A., Hagen, K., Wohlgemuth-Von, Reiche D., 2002. The significance of flood regimes for carabid beetle and spider communities in riparian habitats - a comparison of three major rivers in Germany. *River Res. Appl.* 18, 43–64. <https://doi.org/10.1002/rra.632>.
- Boughton, E.H., Quintana-Ascencio, P.F., Jenkins, D.G., Bohlen, P.J., Fauth, J.E., Engel, A., et al., 2019. Trade-offs and synergies in a payment-for-ecosystem services program on ranchlands in the Everglades headwaters. *Ecosphere* 10. <https://doi.org/10.1002/ecs2.2728>.
- Bradford-Lawrence, T., Finch, T., Bradbury, R.B., Buchanan, G.M., Midgley, A., Field, R.H., 2021. The potential contribution of terrestrial nature-based solutions to a national 'net zero' climate target. *J. Appl. Ecol.* 58, 2349–2360. <https://doi.org/10.1111/1365-2664.14003>.
- Brandt, M., Arheimer, B., Gustavsson, H., Pers, C., Rosberg, J., Sundström, M., et al., 2009. Uppföljning av effekterna av anlagda våtmarker i jordbrukslandskapet på belastning av kväve och fosfor (In Swedish with English abstract). Swedish Environmental Protection Agency, Stockholm, Sweden Report 6309.
- Braskerud, B.C., Tonderski, K.S., Wedding, B., Bakke, R., Blankenberg, A.G.B., Ulén, B., et al., 2005. Can constructed wetlands reduce the diffuse phosphorus loads to eutrophic water in cold temperate regions? *J. Environ. Qual.* 34, 2145–2155. <https://doi.org/10.2134/jeq2004.0466>.
- Bridgman, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C., 2006. The carbon balance of North American wetlands. *Wetlands* 26, 889–916. [https://doi.org/10.1672/0277-5212\(2006\)26\[889:TCBONA\]2.0.CO;2](https://doi.org/10.1672/0277-5212(2006)26[889:TCBONA]2.0.CO;2).
- Brody, S.D., Highfield, W.E., Ryu, H.C., Spaniel-Weber, L., 2007. Examining the relationship between wetland alteration and watershed flooding in Texas and Florida. *Nat. Hazards* 40, 413–428. <https://doi.org/10.1007/s11069-006-9003-3>.
- Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. *Hydrol. Earth Syst. Sci.* 7, 358–389. <https://doi.org/10.5194/hess-7-358-2003>.
- Cattin, M.F., Blandinier, G., Banasek-Richter, C., Bersier, L.F., 2003. The impact of mowing as a management strategy for wet meadows on spider (Araneae) communities. *Biol. Conserv.* 113, 179–188. [https://doi.org/10.1016/s0006-3207\(02\)00297-5](https://doi.org/10.1016/s0006-3207(02)00297-5).
- Cazzanelli, M., Warming, T.P., Christoffersen, K.S., 2008. Emergent and floating-leaved macrophytes as refuge for zooplankton in a eutrophic temperate lake without submerged vegetation. *Hydrobiologia* 605, 113–122. <https://doi.org/10.1007/s10750-008-9324-1>.
- Chasar, L.C., Scudder, B.C., Stewart, A.R., Bell, A.H., Aiken, G.R., 2009. Mercury cycling in stream ecosystems. 3. Trophic dynamics and methylmercury bioaccumulation. *Environ. Sci. Technol.* 43, 2733–2739. <https://doi.org/10.1021/es8027567>.
- Chase, J.M., Leibold, M.A., 2002. Spatial scale dictates the productivity-biodiversity relationship. *Nature* 416, 427–430. <https://doi.org/10.1038/416427a>.
- Chen, Y.Y., Destouni, G., Goldenberg, R., Prieto, C., 2021. Nutrient source attribution: quantitative typology distinction of active and legacy source contributions to waterborne loads. *Hydrol. Process.* 35. <https://doi.org/10.1002/hyp.14284>.
- Cheng, F.Y., Van Meter, K.J., Byrnes, D.K., Basu, N.B., 2020. Maximizing US nitrate removal through wetland protection and restoration. *Nature* 588, 625–630. <https://doi.org/10.1038/s41586-020-03042-5>.
- Cimon-Morin, J., Poulin, M., 2018. Setting conservation priorities in cities: approaches, targets and planning units adapted to wetland biodiversity and ecosystem services. *Landscape Ecol.* 33, 1975–1995. <https://doi.org/10.1007/s10980-018-0707-z>.

- Cimon-Morin, J., Goyette, J.O., Mendes, P., Pellerin, S., Poulin, M., 2021. A systematic conservation planning approach to maintaining ecosystem service provision in working landscapes. *PLANNING* 6, 1570–1600. <https://doi.org/10.1139/plan-2020-010>.
- Cole, L.J., Brocklehurst, S., McCracken, D.I., Harrison, W., Robertson, D., 2012. Riparian field margins: their potential to enhance biodiversity in intensively managed grasslands. *Insect Conserv. Divers.* 5, 86–94. <https://doi.org/10.1111/j.1752-4598.2011.00147.x>.
- Cole, L.J., Brocklehurst, S., Robertson, D., Harrison, W., McCracken, D.I., 2015. Riparian buffer strips: their role in the conservation of insect pollinators in intensive grassland systems. *Agric. Ecosyst. Environ.* 211, 207–220. <https://doi.org/10.1016/j.agee.2015.06.012>.
- Dacey, J.W., Klug, M.J., 1979. Methane efflux from lake sediments through water lilies. *Science* 203, 1253–1255. <https://doi.org/10.1126/science.203.4386.1253>.
- Dadson, S.J., Hall, J.W., Murgatroyd, A., Acreman, M., Bates, P., Beven, K., et al., 2017. A re-statement of the natural science evidence concerning catchment-based 'natural' flood management in the UK. *Proc. Roy. Soc. A-Math. Phys.Sci.* 473. <https://doi.org/10.1098/rspa.2016.0706>.
- Daniel, J., Gleason, J.E., Cottenie, K., Rooney, R.C., 2019. Stochastic and deterministic processes drive wetland community assembly across a gradient of environmental filtering. *Oikos* 128, 1158–1169. <https://doi.org/10.1111/oik.05987>.
- Datry, T., Corti, R., Belletti, B., Piegay, H., 2014. Ground-dwelling arthropod communities across braided river landscape mosaics: a Mediterranean perspective. *Freshw. Biol.* 59, 1308–1322. <https://doi.org/10.1111/fwb.12350>.
- Davidson, N.C., 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. *Mar. Freshw. Res.* 65, 934–941. <https://doi.org/10.1071/Mf14173>.
- Davidson, N.C., Fluet-Chouinard, E., Finlayson, C.M., 2018. Global extent and distribution of wetlands: trends and issues. *Mar. Freshw. Res.* 69, 620–627. <https://doi.org/10.1071/Mf17019>.
- Davidson, N.C., van Dam, A.A., Finlayson, C.M., McInnes, R.J., 2019. Worth of wetlands: revised global monetary values of coastal and inland wetland ecosystem services. *Mar. Freshw. Res.* 70, 1189–1194. <https://doi.org/10.1071/Mf18391>.
- Dawson, L., Elbakidze, M., Angelstam, P., Gordon, J., 2017. Governance and management dynamics of landscape restoration at multiple scales: learning from successful environmental managers in Sweden. *J. Environ. Manag.* 197, 24–40. <https://doi.org/10.1016/j.jenvman.2017.03.019>.
- Dawson, L., Elbakidze, M., Schellens, M., Shkaruba, A., Angelstam, P., 2021. Bogs, birds, and berries in Belarus: the governance and management dynamics of wetland restoration in a state-centric, top-down context. *Ecol. Soc.* 26, 8. <https://doi.org/10.5751/ES-12139-260108>.
- Declerck, S., De Bie, T., Ercken, D., Hampel, H., Schrijvers, S., Van Wichelen, J., et al., 2006. Ecological characteristics of small farmland ponds: associations with land use practices at multiple spatial scales. *Biol. Conserv.* 131, 523–532. <https://doi.org/10.1016/j.biocon.2006.02.024>.
- Delaney, J.M., Wilkins, R.M., 1995. Toxicity of Microcystin Lr, isolated from *Microcystis-aeruginosa*, against various insect species. *Toxicol.* 33, 771–778. [https://doi.org/10.1016/0041-0101\(95\)00018-H](https://doi.org/10.1016/0041-0101(95)00018-H).
- Destouni, G., Jarsjö, J., 2018. Zones of untreatable water pollution call for better appreciation of mitigation limits and opportunities. *Wiley Interdisc. Rev. Water* 5. <https://doi.org/10.1002/wat2.1312>.
- Destouni, G., Cantoni, J., Kalantari, Z., 2021. Distinguishing active and legacy source contributions to stream water quality: comparative quantification for chloride and metals. *Hydro. Process.* 35. <https://doi.org/10.1002/hyp.14280>.
- Devlin, S.P., Saarenheimo, J., Syyvänta, J., Jones, R.I., 2015. Top consumer abundance influences lake methane efflux. *Nat. Commun.* 6. <https://doi.org/10.1038/ncomms9787>.
- Dixon, M.J.R., Loh, J., Davidson, N.C., Beltrame, C., Freeman, R., Walpole, M., 2016. Tracking global change in ecosystem area: the wetland extent trends index. *Biol. Conserv.* 193, 27–35. <https://doi.org/10.1016/j.biocon.2015.10.023>.
- Djordjic, F., Geranmayeh, P., Markensten, H., 2020. Optimizing placement of constructed wetlands at landscape scale in order to reduce phosphorus losses. *Ambio* 49, 1797–1807. <https://doi.org/10.1007/s13280-020-01349-1>.
- Doherty, J.M., Miller, J.F., Prellwitz, S.G., Thompson, A.M., Loheide, S.P., Zedler, J.B., 2014. Hydrologic regimes revealed bundles and tradeoffs among six wetland services. *Ecosystems* 17, 1026–1039. <https://doi.org/10.1007/s10021-014-9775-3>.
- Downing, J.A., Cole, J.J., Middelburg, J.J., Striegl, R.G., Duarte, C.M., Kortelainen, P., et al., 2008. Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century. *Glob. Biogeochem. Cycles* 22. <https://doi.org/10.1029/2006gb002854>.
- Doyle, C., 2020. *Wetland Management in Ontario: A Social Multi-criteria Evaluation of Niagara Falls' Slough Forests*. University of Waterloo, Waterloo, Canada.
- Drescher, M., Epstein, G.B., Warriner, G.K., Rooney, R.C., 2019. An investigation of the effects of conservation incentive programs on management of invasive species by private landowners. *Conserv. Sci. Pract.* 1. <https://doi.org/10.1111/csp2.56>.
- Durant, D., Tichit, M., Kerneis, E., Fritz, H., 2008. Management of agricultural wet grasslands for breeding waders: integrating ecological and livestock system perspectives - a review. *Biodivers. Conserv.* 17, 2275–2295. <https://doi.org/10.1007/s10531-007-9310-3>.
- EPCN, 2007. Developing the pond manifesto. *Ann. Limnol.-Int. J. Lim.* 43, 221–232. <https://doi.org/10.1051/limn:2007001>.
- Evans, C.D., Peacock, M., Baird, A.J., Artz, R.R.E., Burden, A., Callaghan, N., et al., 2021. Overriding water table control on managed peatland greenhouse gas emissions. *Nature* 593, 548–552. <https://doi.org/10.1038/s41586-021-03523-1>.
- Everard, M., 2008. Selection of taxa as indicators of river and freshwater wetland quality in the UK. *Aquat. Conserv.* 18, 1052–1061. <https://doi.org/10.1002/aqc.896>.
- Fahrig, L., 2020. Why do several small patches hold more species than few large patches? *Glob. Ecol. Biogeogr.* 29, 615–628. <https://doi.org/10.1111/geb.13059>.
- Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.L., Frame, D., et al., 2021. The Earth's energy budget, climate feedbacks, and climate sensitivity. In: Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Péan, C., Berger, S., et al. (Eds.), *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. CUP.
- Fossey, M., Rousseau, A.N., 2016. Assessing the long-term hydrological services provided by wetlands under changing climate conditions: a case study approach of a Canadian watershed. *J. Hydrol.* 541, 1287–1302. <https://doi.org/10.1016/j.jhydrol.2016.08.032>.
- Fossey, M., Rousseau, A.N., Savary, S., 2016. Assessment of the impact of spatio-temporal attributes of wetlands on stream flows using a hydrological modelling framework: a theoretical case study of a watershed under temperate climatic conditions. *Hydro. Process.* 30, 1768–1781. <https://doi.org/10.1002/hyp.10750>.
- Gann, G.D., McDonald, T., Walder, B., Aronson, J., Nelson, C.R., Jonson, J., et al., 2019. International principles and standards for the practice of ecological restoration. Second edition. *Restor. Ecol.* 27, S1–S46. <https://doi.org/10.1111/rec.13035>.
- Gedney, N., Cox, P.M., Huntingford, C., 2004. Climate feedback from wetland methane emissions. *Geophys. Res. Lett.* 31, L20503. <https://doi.org/10.1029/2004gl020919>.
- Geranmayeh, P., Johannesson, K.M., Ulén, B., Tonderski, K.S., 2018. Particle deposition, re-suspension and phosphorus accumulation in small constructed wetlands. *Ambio* 47, 134–145. <https://doi.org/10.1007/s13280-017-0992-9>.
- Ghermandi, A., Fichtman, E., 2015. Cultural ecosystem services of multifunctional constructed wetlands and waste stabilization ponds: time to enter the mainstream? *Ecol. Eng.* 84, 615–623. <https://doi.org/10.1016/j.ecoleng.2015.09.067>.
- Gilbert, P.J., Taylor, S., Cooke, D.A., Deary, M.E., Jeffries, M.J., 2021. Quantifying organic carbon storage in temperate pond sediments. *J. Environ. Manag.* 280, 111698. <https://doi.org/10.1016/j.jenvman.2020.111698>.
- Glover, J.B., Domino, M.E., Altman, K.C., Dillman, J.W., Castleberry, W.S., Eidson, J.P., et al., 2010. Mercury in South Carolina fishes, USA. *Ecotoxicology* 19, 781–795. <https://doi.org/10.1007/s10646-009-0455-6>.
- Gomez-Creutzberg, C., Lagisz, M., Nakagawa, S., Brockerhoff, E.G., Tylianakis, J.M., 2021. Consistent trade-offs in ecosystem services between land covers with different production intensities. *Biol. Rev.* <https://doi.org/10.1111/brv.12734>.
- Goyette, J.O., Cimon-Morin, J.M., Mendes, P., Thériault, M., Pellerin, S., Poulin, M., 2021. Planning wetland protection and restoration for the safeguard of ecosystem service flows to beneficiaries. *Landsc. Ecol.* 36, 2691–2706. <https://doi.org/10.1007/s10980-021-01267-x>.
- Graversgaard, M., Jacobsen, B.H., Hoffmann, C.C., Dalgaard, T., Odgaard, M.V., Kjaergaard, C., et al., 2021. Policies for wetlands implementation in Denmark and Sweden - historical lessons and emerging issues. *Land Use Policy* 101, 105206. <https://doi.org/10.1016/j.landusepol.2020.105206>.
- Green, A.J., Elmerg, J., 2014. Ecosystem services provided by waterbirds. *Biol. Rev.* 89, 105–122. <https://doi.org/10.1111/brv.12045>.
- Greig, H.S., Kratina, P., Thompson, P.L., Palen, W.J., Richardson, J.S., Shurin, J.B., 2012. Warming, eutrophication, and predator loss amplify subsidies between aquatic and terrestrial ecosystems. *Glob. Chang. Biol.* 18, 504–514. <https://doi.org/10.1111/j.1365-2486.2011.02540.x>.
- Griscom, B.W., Adams, J., Ellis, P.W., Houghton, R.A., Lomax, G., Miteva, D.A., et al., 2017. Natural climate solutions. *PNAS USA* 114, 11645–11650. <https://doi.org/10.1073/pnas.1710655114>.
- Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koesch, F., et al., 2020. Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nat. Commun.* 11, 1644. <https://doi.org/10.1038/s41467-020-15499-z>.
- Hagy, H.M., Horath, M.M., Yetter, A.P., Hine, C.S., Smith, R.V., 2017. Evaluating tradeoffs between wetland sanctuary for migrating waterbirds and recreational opportunities in a restored wetland complex. *Hydrobiologia* 804, 103–118. <https://doi.org/10.1007/s10750-016-2711-0>.
- Hanford, J.K., Webb, C.E., Hochuli, D.F., 2019. Habitat traits associated with mosquito risk and aquatic diversity in urban wetlands. *Wetlands* 39, 743–758. <https://doi.org/10.1007/s13157-019-01133-2>.
- Hansson, L.A., Annadotter, H., Bergman, E., Hamrin, S.F., Jeppesen, E., Kairesalo, T., et al., 1998. Biomaniipulation as an application of food-chain theory: constraints, synthesis, and recommendations for temperate lakes. *Ecosystems* 1, 558–574. <https://doi.org/10.1007/s100219900051>.
- Hansson, L.A., Brönmark, C., Nilsson, P.A., Åbjörnsson, K., 2005. Conflicting demands on wetland ecosystem services: nutrient retention, biodiversity or both? *Freshw. Biol.* 50, 705–714. <https://doi.org/10.1111/j.1365-2427.2005.01352.x>.
- Hansson, A., Pedersen, E., Weisner, S.E., 2012. Landowners' incentives for constructing wetlands in an agricultural area in south Sweden. *J. Environ. Manag.* 113, 271–278. <https://doi.org/10.1016/j.jenvman.2012.09.008>.
- Herring, M.W., Robinson, W.A., Zander, K.K., Garnett, S.T., 2021. Increasing water-use efficiency in rice fields threatens an endangered waterbird. *Agric. Ecosyst. Environ.* 322. <https://doi.org/10.1016/j.agee.2021.107638>.
- Hsu, C.-B., Hsieh, H.-L., Yang, L., Wu, S.-H., Chang, J.-S., Hsiao, S.-C., et al., 2011. Biodiversity of constructed wetlands for wastewater treatment. *Ecol. Eng.* 37, 1533–1545. <https://doi.org/10.1016/j.ecoleng.2011.06.002>.
- Hugelius, G., Loisel, J., Chadburn, S., Jackson, R.B., Jones, M., MacDonald, G., et al., 2020. Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. *PNAS USA* 117, 20438–20446. <https://doi.org/10.1073/pnas.1916387117>.
- Jaramillo, F., Desormeaux, A., Hedlund, J., Jawitz, J.W., Clerici, N., Piemontese, L., et al., 2019. Priorities and interactions of sustainable development goals (SDGs) with focus on wetlands. *Water* 11. <https://doi.org/10.3390/w11030619>.
- Jensen, A.K., Uggeldahl, K.C., Jacobsen, B.H., Jensen, J.D., Hasler, B., 2019. Including aesthetic and recreational values in cost-effectiveness analyses of land use change based nitrogen abatement measures in Denmark. *J. Environ. Manag.* 240, 384–393. <https://doi.org/10.1016/j.jenvman.2019.03.076>.
- Jessop, J., Spyreas, G., Pociask, G.E., Benson, T.J., Ward, M.P., Kent, A.D., et al., 2015. Tradeoffs among ecosystem services in restored wetlands. *Biol. Conserv.* 191, 341–348. <https://doi.org/10.1016/j.biocon.2015.07.006>.

- Johansson, A.E., Gustavsson, A.M., Öquist, M.G., Svensson, B.H., 2004. Methane emissions from a constructed wetland treating wastewater—seasonal and spatial distribution and dependence on edaphic factors. *Water Res.* 38, 3960–3970. <https://doi.org/10.1016/j.watres.2004.07.008>.
- Johansson, F., Bini, L.M., Coiffard, P., Svanbäck, R., Wester, J., Heino, J., 2019a. Environmental variables drive differences in the beta diversity of dragonfly assemblages among urban stormwater ponds. *Ecol. Ind.* 106, 105529. <https://doi.org/10.1016/j.ecolind.2019.105529>.
- Johansson, M., Pedersen, E., Weisner, S., 2019b. Assessing cultural ecosystem services as individuals' place-based appraisals. *Urban For. Urban Green.* 39, 79–88. <https://doi.org/10.1016/j.ufug.2019.02.011>.
- Johnston, R., Cools, J., Liersch, S., Morardet, S., Murgue, C., Mahieu, M., et al., 2013. WEtwin: a structured approach to evaluating wetland management options in data-poor contexts. *Environ. Sci. Policy* 34, 3–17. <https://doi.org/10.1016/j.envsci.2012.12.006>.
- Kačergytė, I., Arlt, D., Berg, Å., Žmihorski, M., Knap, J., Rosin, Z.M., et al., 2021a. Evaluating created wetlands for bird diversity and reproductive success. *Biol. Conserv.* 257. <https://doi.org/10.1016/j.biocon.2021.109084>.
- Kačergytė, I., Petersson, E., Arlt, D., Hellström, M., Knap, J., Spens, J., et al., 2021b. Environmental DNA metabarcoding elucidates patterns of fish colonisation and co-occurrences with amphibians in temperate wetlands created for biodiversity. *Freshw. Biol.* 66, 1915–1929. <https://doi.org/10.1111/fwb.13800>.
- Kadlec, R.H., 2008. The effects of wetland vegetation and morphology on nitrogen processing. *Ecol. Eng.* 33, 126–141. <https://doi.org/10.1016/j.ecoleng.2008.02.012>.
- Kampen, H., Walther, D., 2018. In: Benelli, G. (Ed.) *Mosquito-borne diseases*. Parasitology Research Monographs, Springer.
- Kingsford, R.T., Basset, A., Jackson, L., 2016. Wetlands: conservation's poor cousins. *Aquat. Conserv.* 26, 892–916. <https://doi.org/10.1002/aqc.2709>.
- Kitchell, J.F., Schindler, D.E., Herwig, B.R., Post, D.M., Olson, M.H., Oldham, M., 1999. Nutrient cycling at the landscape scale: the role of diel foraging migrations by geese at the Bosque del Apache National Wildlife Refuge, New Mexico. *Limnol. Oceanogr.* 44, 828–836. https://doi.org/10.4319/lo.1999.44.3_part_2.0828.
- Knight, T.M., McCoy, M.W., Chase, J.M., McCoy, K.A., Holt, R.D., 2005. Trophic cascades across ecosystems. *Nature* 437, 880–883. <https://doi.org/10.1038/nature03962>.
- Knutson, M.G., Richardson, W.B., Reineke, D.M., Gray, B.R., Parmelee, J.R., Weick, S.E., 2004. Agricultural ponds support amphibian populations. *Ecol. Appl.* 14, 669–684. <https://doi.org/10.1890/02-5305>.
- Kosten, S., Pineiro, M., de Goede, E., de Klein, J., Lamers, L.P.M., Ettwig, K., 2016. Fate of methane in aquatic systems dominated by free-floating plants. *Water Res.* 104, 200–207. <https://doi.org/10.1016/j.watres.2016.07.054>.
- Kronberg, R.M., Tjerngren, I., Drott, A., Björn, E., Skyllberg, U., 2012. Net degradation of methyl mercury in alder swamps. *Environ. Sci. Technol.* 46, 13144–13151. <https://doi.org/10.1021/es303543k>.
- Kyllmar, K., Forsberg, L.S., Andersson, S., Mårtensson, K., 2014. Small agricultural monitoring catchments in Sweden representing environmental impact. *Agric. Ecosyst. Environ.* 198, 25–35. <https://doi.org/10.1016/j.agee.2014.05.016>.
- Kynkäänniemi, P., Ulén, B., Torstensson, G., Tonderski, K.S., 2013. Phosphorus retention in a newly constructed wetland receiving agricultural tile drainage water. *J. Environ. Qual.* 42, 596–605. <https://doi.org/10.2134/jeq2012.0266>.
- Lafage, D., Petillon, J., 2016. Relative importance of management and natural flooding on spider, carabid and plant assemblages in extensively used grasslands along the Loire. *Basic Appl. Ecol.* 17, 535–545. <https://doi.org/10.1016/j.baae.2016.04.002>.
- Lambeets, K., Vandegehuchte, M.L., Maelfait, J.P., Bonte, D., 2008. Understanding the impact of flooding on trait-displacements and shifts in assemblage structure of predatory arthropods on river banks. *J. Anim. Ecol.* 77, 1162–1174. <https://doi.org/10.1111/j.1365-2656.2008.01443.x>.
- Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., et al., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environ. Evid.* 5. <https://doi.org/10.1186/s13750-016-0060-0>.
- Le Gall, M., Fournier, M., Chaput-Bardy, A., Huste, A., 2018. Determinant landscape-scale factors on pond odonate assemblages. *Freshw. Biol.* 63, 306–317. <https://doi.org/10.1111/fwb.13065>.
- Lehikoinen, P., Lehikoinen, A., Mikkola-Roos, M., Jaatinen, K., 2017. Counteracting wetland overgrowth increases breeding and staging bird abundances. *Sci. Rep.* 7, 41391. <https://doi.org/10.1038/srep41391>.
- Lemmens, P., Mergeay, J., De Bie, T., Van Wichelen, J., De Meester, L., Declerck, S.A.J., 2013. How to maximally support local and regional biodiversity in applied conservation? Insights from pond management. *Plos One* 8. <https://doi.org/10.1371/journal.pone.0072538>.
- Lewis-Phillips, J., Brooks, S.J., Sayer, C.D., Patmore, I.R., Hilton, G.M., Harrison, A., et al., 2020. Ponds as insect chimneys: restoring overgrown farmland ponds benefits birds through elevated productivity of emerging aquatic insects. *Biol. Conserv.* 241. <https://doi.org/10.1016/j.biocon.2019.108253>.
- Lindahl, T., Söderqvist, T., 2004. Building a catchment-based environmental programme: a stakeholder analysis of wetland creation in Scania, Sweden. *Reg. Environ. Chang.* 4, 132–144. <https://doi.org/10.1007/s10113-004-0072-0>.
- Lindström, A., Eklöf, D., Lilja, T., 2021. Different hatching rates of floodwater mosquitoes *Aedes sticticus*, *Aedes rossicus* and *Aedes cinereus* from different flooded environments. *Insects* 12. <https://doi.org/10.3390/insects12040279>.
- Ma, Z.J., Cai, Y.T., Li, B., Chen, J.K., 2010. Managing wetland habitats for waterbirds: an international perspective. *Wetlands* 30, 15–27. <https://doi.org/10.1007/s13157-009-0001-6>.
- Maclean, I.M.D., Boar, R.R., Lugo, C., 2011. A review of the relative merits of conserving, using, or draining *Papyrus* swamps. *Environ. Manag.* 47, 218–229. <https://doi.org/10.1007/s00267-010-9592-1>.
- Maltby, E., 2018. Functional assessment of wetlands. In: Finlayson, C.M., Everard, M., Irvine, K., McInnes, R.J., Middleton, B.A., van Dam, A.A., et al. (Eds.), *The Wetland Book*. Springer, Dordrecht, pp. 1729–1739.
- Mander, Ü., Tournebize, J., Espenberg, M., Chaumont, C., Torga, R., Garnier, J., et al., 2021. High denitrification potential but low nitrous oxide emission in a constructed wetland treating nitrate-polluted agricultural run-off. *Sci. Total Environ.* 779, 146614. <https://doi.org/10.1016/j.scitotenv.2021.146614>.
- Manuel, P.M., 2003. Cultural perceptions of small urban wetlands: cases from the Halifax regional municipality, Nova Scotia, Canada. *Wetlands* 23, 921–940. [https://doi.org/10.1672/0277-5212\(2003\)023\[0921:Cposuw\]2.0.Co;2](https://doi.org/10.1672/0277-5212(2003)023[0921:Cposuw]2.0.Co;2).
- Margaryan, L., Prince, S., Ioannides, D., Röslmaier, M., 2018. Dancing with cranes: a humanist perspective on cultural ecosystem services of wetlands. *Tour. Geogr.*, 1–22. <https://doi.org/10.1080/14616688.2018.1522512>.
- Martikainen, P.J., Nykänen, H., Crill, P., Silvola, J., 1993. Effect of a lowered water-table on nitrous-oxide fluxes from northern peatlands. *Nature* 366, 51–53. <https://doi.org/10.1038/366051a0>.
- May, V., McBarron, E.J., 1973. Occurrence of bluegreen alga, *Anabaena circinalis* Rabenh, in New South Wales and toxicity to mice and honey bees. *J. Aust. I. Agr. Sci.* 39, 264–266.
- McCauley, L.A., Anteau, M.J., van der Burg, M.P., Wiltermuth, M.T., 2015. Land use and wetland drainage affect water levels and dynamics of remaining wetlands. *Ecosphere* 6. <https://doi.org/10.1890/es14-00494.1>.
- Mendes, L.R.D., 2021. Nitrogen removal from agricultural subsurface drainage by surface-flow wetlands: variability. *Processes* 9, 156. <https://doi.org/10.3390/pr9010156>.
- Mitra, S., Wassmann, R., Vlek, P.L.G., 2005. An appraisal of global wetland area and its organic carbon stock. *Curr. Sci.* 88, 25–35.
- Moore, T.L.C., Hunt, W.F., 2012. Ecosystem service provision by stormwater wetlands and ponds - a means for evaluation? *Water Res.* 46, 6811–6823. <https://doi.org/10.1016/j.watres.2011.11.026>.
- Nakano, S., Kawaguchi, Y., Taniguchi, Y., Miyasaka, H., Shibata, Y., Urabe, H., et al., 1999. Selective foraging on terrestrial invertebrates by rainbow trout in a forested headwater stream in northern Japan. *Ecol. Res.* 14, 351–360. <https://doi.org/10.1046/j.1440-1703.1999.00315.x>.
- Natuhara, Y., 2013. Ecosystem services by paddy fields as substitutes of natural wetlands in Japan. *Ecol. Eng.* 56, 97–106. <https://doi.org/10.1016/j.ecoleng.2012.04.026>.
- Naturvårdsverket, 2019. Myllrande våtmarker - underlagsrapport till den fördjupade utvärderingen av miljömålen 2019. Rapport. 6873.
- Odgaard, M.V., Turner, K.G., Bøcher, P.K., Svenning, J.C., Dalgaard, T., 2017. A multi-criteria, ecosystem-service value method used to assess catchment suitability for potential wetland reconstruction in Denmark. *Ecol. Ind.* 77, 151–165. <https://doi.org/10.1016/j.ecolind.2016.12.001>.
- Ollivier, Q.R., Maher, D.T., Pitfield, C., Macreadie, P.I., 2019. Punching above their weight: large release of greenhouse gases from small agricultural dams. *Glob. Chang. Biol.* 25, 721–732. <https://doi.org/10.1111/gcb.14477>.
- Pacheco, F.S., Roland, F., Downing, J.A., 2014. Eutrophication reverses whole-lake carbon budgets. *Inland Waters* 4, 41–48. <https://doi.org/10.5268/iw-4.1.614>.
- Peacock, M., Ridley, L.M., Evans, C.D., Gauci, V., 2017. Management effects on greenhouse gas dynamics in fen ditches. *Sci. Total Environ.* 578, 601–612. <https://doi.org/10.1016/j.scitotenv.2016.11.005>.
- Peacock, M., Audet, J., Bastviken, D., Cook, S., Evans, C.D., Grinham, A., et al., 2021. Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide. *Glob. Chang. Biol.* 27, 5109–5123. <https://doi.org/10.1111/gcb.15762>.
- Pearman, P.B., 1993. Effects of habitat size on tadpole populations. *Ecology* 74, 1982–1991. <https://doi.org/10.2307/1940841>.
- Pedersen, E., Weisner, S.E.B., Johansson, M., 2019. Wetland areas' direct contributions to residents' well-being entitle them to high cultural ecosystem values. *Sci. Total Environ.* 646, 1315–1326. <https://doi.org/10.1016/j.scitotenv.2018.07.236>.
- Peimer, A.W., Krzywicka, A.E., Cohen, D.B., Van den Bosch, K., Buxton, V.L., Stevenson, N.A., et al., 2017. National-level wetland policy specificity and goals vary according to political and economic indicators. *Environ. Manag.* 59, 141–153. <https://doi.org/10.1007/s00267-016-0766-3>.
- Persson, J., Somes, N.L.G., Wong, T.H.F., 1999. Hydraulics efficiency of constructed wetlands and ponds. *Water Sci. Technol.* 40, 291–300. [https://doi.org/10.1016/S0273-1223\(99\)00448-5](https://doi.org/10.1016/S0273-1223(99)00448-5).
- Plieninger, T., Dijk, S., Oteros-Rozas, E., Bieling, C., 2013. Assessing, mapping, and quantifying cultural ecosystem services at community level. *Land Use Policy* 33, 118–129. <https://doi.org/10.1016/j.landusepol.2012.12.013>.
- Plum, N., 2005. Terrestrial invertebrates in flooded grassland: a literature review. *Wetlands* 25, 721–737. [https://doi.org/10.1672/0277-5212\(2005\)025\[0721:Tiiifga\]2.0.Co;2](https://doi.org/10.1672/0277-5212(2005)025[0721:Tiiifga]2.0.Co;2).
- Porej, D., Hetherington, T.E., 2005. Designing wetlands for amphibians: the importance of predatory fish and shallow littoral zones in structuring of amphibian communities. *Wetl. Ecol. Manag.* 13, 445–455. <https://doi.org/10.1007/s11273-004-0522-y>.
- Pueyo-Ros, J., Ribas, A., Fraguell, R.M., 2019. A cultural approach to wetlands restoration to assess its public acceptance. *Restor. Ecol.* 27, 626–637. <https://doi.org/10.1111/rec.12896>.
- Qian, J., Tang, S.J., Wang, P.F., Lu, B.H., Li, K., Jin, W., et al., 2021. From source to sink: review and prospects of microplastics in wetland ecosystems. *Sci. Total Environ.* 758. <https://doi.org/10.1016/j.scitotenv.2020.143633>.
- Quin, A., Jaramillo, F., Destoumi, G., 2015. Dissecting the ecosystem service of large-scale pollutant retention: the role of wetlands and other landscape features. *Ambio* 44 (Suppl. 1), S127–S137. <https://doi.org/10.1007/s13280-014-0594-8>.
- Rannap, R., Kaart, M.M., Kaart, T., Kill, K., Uuemaa, E., Mander, U., et al., 2020. Constructed wetlands as potential breeding sites for amphibians in agricultural landscapes: a case study. *Ecol. Eng.* 158. <https://doi.org/10.1016/j.ecoleng.2020.106077>.
- Rapinel, S., Hubert-Moy, L., Clement, B., Maltby, E., 2016. Mapping wetland functions using Earth observation data and multi-criteria analysis. *Environ. Monit. Assess.* 188. <https://doi.org/10.1007/s10661-016-5644-1>.

- Raymond, P.A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., et al., 2013. Global carbon dioxide emissions from inland waters. *Nature* 503, 355–359. <https://doi.org/10.1038/nature12760>.
- Richardson, C.J., 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228, 1424–1427. <https://doi.org/10.1126/science.228.4706.1424>.
- Rosentreter, J.A., Borges, A.V., Deemer, B.R., Holgerson, M.A., Liu, S.D., Song, C.L., et al., 2021. Half of global methane emissions come from highly variable aquatic ecosystem sources. *Nat. Geosci.* 14, 225–+. <https://doi.org/10.1038/s41561-021-00715-2>.
- Rothenbuecher, J., Schaefer, M., 2006. Submersion tolerance in floodplain arthropod communities. *Basic Appl. Ecol.* 7, 398–408. <https://doi.org/10.1016/j.baec.2006.05.005>.
- Rouquette, J.R., Posthumus, H., Morris, J., Hess, T.M., Dawson, Q.L., Gowing, D.J.G., 2011. Synergies and trade-offs in the management of lowland rural floodplains: an ecosystem services approach. *Hydrol. Sci. J.* 56, 1566–1581. <https://doi.org/10.1080/02626667.2011.629785>.
- Sanchez-Zapata, J.A., Anadon, J.D., Carrete, M., Gimenez, A., Navarro, J., Villacorta, C., et al., 2005. Breeding waterbirds in relation to artificial pond attributes: implications for the design of irrigation facilities. *Biodivers. Conserv.* 14, 1627–1639. <https://doi.org/10.1007/s10531-004-0534-1>.
- Santamaría, L., 2002. Why are most aquatic plants widely distributed? Dispersal, clonal growth and small-scale heterogeneity in a stressful environment. *Acta Oecol.* 23, 137–154. [https://doi.org/10.1016/S1146-609X\(02\)01146-3](https://doi.org/10.1016/S1146-609X(02)01146-3).
- Saunois, M., Stavert, A.R., Poulter, B., Bousquet, P., Canadell, J.G., Jackson, R.B., et al., 2020. The global methane budget 2000–2017. *Earth Syst. Sci. Data* 12, 1561–1623. <https://doi.org/10.5194/essd-12-1561-2020>.
- Schaefer, J.K., Kronberg, R.M., Björn, E., Skjellberg, U., 2020. Anaerobic guilds responsible for mercury methylation in boreal wetlands of varied trophic status serving as either a methylmercury source or sink. *Environ. Microbiol.* 22, 3685–3699. <https://doi.org/10.1111/1462-2920.15134>.
- Schäfer, M.L., Lundström, J.O., Pfeffer, M., Lundkvist, E., Landin, J., 2004. Biological diversity versus risk for mosquito nuisance and disease transmission in constructed wetlands in southern Sweden. *Med. Vet. Entomol.* 18, 256–267. <https://doi.org/10.1111/j.0269-283X.2004.00504.x>.
- Scheffer, M., van Geest, G.J., Zimmer, K., Jeppesen, E., Sondergaard, M., Butler, M.G., et al., 2006. Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. *Oikos* 112, 227–231. <https://doi.org/10.1111/j.0030-1299.2006.14145.x>.
- Scholte, S.S.K., Todorova, M., van Teeffelen, A.J.A., Verburg, P.H., 2016. Public support for wetland restoration: what is the link with ecosystem service values? *Wetlands* 36, 467–481. <https://doi.org/10.1007/s13157-016-0755-6>.
- Sebacher, D.I., Harriss, R.C., Bartlett, K.B., 1985. Methane emissions to the atmosphere through aquatic plants. *J. Environ. Qual.* 14, 40–46. <https://doi.org/10.2134/jeq1985.00472425001400010008x>.
- Semlitsch, R.D., Peterman, W.E., Anderson, T.L., Drake, D.L., Ousterhout, B.H., 2015. Intermediate pond sizes contain the highest density, richness, and diversity of pond-breeding amphibians. *PLoS One* 10, e0123055. <https://doi.org/10.1371/journal.pone.0123055>.
- Shulse, C.D., Semlitsch, R.D., Trauth, K.M., Gardner, J.E., 2012. Testing wetland features to increase amphibian reproductive success and species richness for mitigation and restoration. *Ecol. Appl.* 22, 1675–1688. <https://doi.org/10.1890/11-0212.1>.
- Smart, J., Gill, J.A., Sutherland, W.J., Watkinson, A.R., 2006. Grassland-breeding waders: identifying key habitat requirements for management. *J. Appl. Ecol.* 43, 454–463. <https://doi.org/10.1111/j.1365-2664.2006.01166.x>.
- Söderqvist, T., 2003. Are farmers prosocial? Determinants of the willingness to participate in a Swedish catchment-based wetland creation programme. *Ecol. Econ.* 47, 105–120. <https://doi.org/10.1016/j.ecolecon.2002.09.004>.
- Søvik, A.K., Augustin, J., Heikkinen, K., Huttunen, J.T., Necki, J.M., Karjalainen, S.M., et al., 2006. Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. *J. Environ. Qual.* 35, 2360–2373. <https://doi.org/10.2134/jeq2006.0038>.
- St Louis, V.L., Rudd, J.W.M., Kelly, C.A., Beaty, K.G., Flett, R.J., Roulet, N.T., 1996. Production and loss of methylmercury and loss of total mercury from boreal forest catchments containing different types of wetlands. *Environ. Sci. Technol.* 30, 2719–2729. <https://doi.org/10.1021/es950856h>.
- Stewart, I., Seawright, A.A., Shaw, G.R., 2008. Cyanobacterial poisoning in livestock, wild mammals and birds - an overview. *Adv. Exp. Med. Biol.* 619, 613–637. https://doi.org/10.1007/978-0-387-75865-7_28.
- Strand, J.A., Weisner, S.E.B., 2001. Dynamics of submerged macrophyte populations in response to biomanipulation. *Freshw. Biol.* 46, 1397–1408. <https://doi.org/10.1046/j.1365-2427.2001.00746.x>.
- Strand, J.A., Weisner, S.E.B., 2013. Effects of wetland construction on nitrogen transport and species richness in the agricultural landscape-experiences from Sweden. *Ecol. Eng.* 56, 14–25. <https://doi.org/10.1016/j.ecoleng.2012.12.087>.
- Strandin Pers, A., 2012. Mark i marginalen: Drivkrafter, pionjärer & myrödlingslandskap. vol. 142.
- Ström, L., Tagesson, T., Mastepanov, M., Christensen, T.R., 2012. Presence of *Eriophorum scheuchzeri* enhances substrate availability and methane emission in an Arctic wetland. *Soil Biol. Biochem.* 45, 61–70. <https://doi.org/10.1016/j.soilbio.2011.09.005>.
- Swartz, T.M., Miller, J.R., 2021. The American Pond Belt: an untold story of conservation challenges and opportunities. *Front. Ecol. Environ.* 19, 501–509. <https://doi.org/10.1002/fee.2381>.
- Thiere, G., Milenkovski, S., Lindgren, P.E., Sahlén, G., Berglund, O., Weisner, S.E.B., 2009. Wetland creation in agricultural landscapes: biodiversity benefits on local and regional scales. *Biol. Conserv.* 142, 964–973. <https://doi.org/10.1016/j.biocon.2009.01.006>.
- Thorslund, J., Jarsjö, J., Jaramillo, F., Jawitz, J.W., Manzoni, S., Basu, N.B., et al., 2017. Wetlands as large-scale nature-based solutions: status and challenges for research, engineering and management. *Ecol. Eng.* 108, 489–497. <https://doi.org/10.1016/j.ecoleng.2017.07.012>.
- Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., et al., 2020. A comprehensive quantification of global nitrous oxide sources and sinks. *Nature* 586, 248–256. <https://doi.org/10.1038/s41586-020-2780-0>.
- Tjerngren, I., Karlsson, T., Björn, E., Skjellberg, U., 2012a. Potential Hg methylation and MeHg demethylation rates related to the nutrient status of different boreal wetlands. *Biogeochemistry* 108, 335–350. <https://doi.org/10.1007/s10533-011-9603-1>.
- Tjerngren, I., Meili, M., Björn, E., Skjellberg, U., 2012b. Eight boreal wetlands as sources and sinks for methyl mercury in relation to soil acidity, C/N ratio, and small-scale flooding. *Environ. Sci. Technol.* 46, 8052–8060. <https://doi.org/10.1021/es300845x>.
- Vretare, V., Weisner, S.E.B., Strand, J.A., Granéli, W., 2001. Phenotypic plasticity in *Phragmites australis* as a functional response to water depth. *Aquat. Bot.* 69, 127–145. [https://doi.org/10.1016/S0304-3770\(01\)00134-6](https://doi.org/10.1016/S0304-3770(01)00134-6).
- Vymazal, J., Březinová, T., 2015. The use of constructed wetlands for removal of pesticides from agricultural runoff and drainage: a review. *Environ. Int.* 75, 11–20. <https://doi.org/10.1016/j.envint.2014.10.026>.
- Wahlroos, O., Valkama, P., Mäkinen, E., Ojala, A., Vasander, H., Väänänen, V.M., 2015. Urban wetland parks in Finland: improving water quality and creating endangered habitats. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 11, 46–60. <https://doi.org/10.1080/21513732.2015.1006681>.
- Walton, R.E., Sayer, C.D., Bennion, H., Axmacher, J.C., 2021. Improving the pollinator pantry: restoration and management of open farmland ponds enhances the complexity of plant-pollinator networks. *Agric. Ecosyst. Environ.* 320. <https://doi.org/10.1016/j.agee.2021.107611>.
- Webb, J.R., Hayes, N.M., Simpson, G.L., Leavitt, P.R., Baulch, H.M., Finlay, K., 2019. Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink. *PNAS USA* 116, 9814–9819. <https://doi.org/10.1073/pnas.1820389116>.
- Webb, J.R., Clough, T.J., Quayle, W.C., 2021. A review of indirect N₂O emission factors from artificial agricultural waters. *Environ. Res. Lett.* 16, 043005. <https://doi.org/10.1088/1748-9326/abed00>.
- Weilhoefer, C.L., Williams, D., Nguyen, I., Jakstis, K., Fischer, C., 2017. The effects of reed canary grass (*Phalaris arundinacea* L.) on wetland habitat and arthropod community composition in an urban freshwater wetland. *Wetl. Ecol. Manag.* 25, 159–175. <https://doi.org/10.1007/s11273-016-9507-x>.
- Weisner, S.E.B., Strand, J.A., 2002. *Ecology and management of plants in aquatic ecosystems. Handbook of Ecological Restoration*, pp. 242–256.
- Weisner, S.E.B., Strand, J.A., Sandsten, H., 1997. Mechanisms regulating abundance of submerged vegetation in shallow eutrophic lakes. *Oecologia* 109, 592–599. <https://doi.org/10.1007/s004420050121>.
- Weisner, S.E.B., Johannesson, K., Thiere, G., Svengren, H., Ehde, P.M., Tonderski, K.S., 2016. National large-scale wetland creation in agricultural areas-potential versus realized effects on nutrient transports. *Water* 8, 544. <https://doi.org/10.3390/w8110544>.
- Winton, R.S., Richardson, C.J., 2017. Top-down control of methane emission and nitrogen cycling by waterfowl. *Ecology* 98, 265–277. <https://doi.org/10.1002/ecy.1640>.
- Xu, X., Jiang, B., Tan, Y., Costanza, R., Yang, G., 2018. Lake-wetland ecosystem services modeling and valuation: progress, gaps and future directions. *Ecosyst. Serv.* 33, 19–28. <https://doi.org/10.1016/j.ecoser.2018.08.001>.
- Xu, J.Y., Liem-Nguyen, V., Buck, M., Bertilsson, S., Björn, E., Bravo, A.G., 2021. Mercury methylating microbial community structure in boreal wetlands explained by local physicochemical conditions. *Front. Environ. Sci.* 8. <https://doi.org/10.3389/fenvs.2020.518662>.
- Yadav, P., Foster, W.A., Mitsch, W.J., Grewal, P.S., 2012. Factors affecting mosquito populations in created wetlands in urban landscapes. *Urban Ecosyst.* 15, 499–511. <https://doi.org/10.1007/s11252-012-0230-y>.
- Zedler, J.B., 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. *Front. Ecol. Environ.* 1, 65–72. [https://doi.org/10.1890/1540-9295\(2003\)001\[0065:WAYSRI\]2.0.CO;2](https://doi.org/10.1890/1540-9295(2003)001[0065:WAYSRI]2.0.CO;2).
- Zedler, J.B., Doherty, J.M., Miller, N.A., 2012. Shifting restoration policy to address landscape change, novel ecosystems, and monitoring. *Ecol. Soc.* 17. <https://doi.org/10.5751/es-05197-170436>.
- Žmihorski, M., Pärt, T., Gustafson, T., Berg, Å., 2016. Effects of water level and grassland management on alpha and beta diversity of birds in restored wetlands. *J. Appl. Ecol.* 53, 587–595. <https://doi.org/10.1111/1365-2664.12588>.