Smart Nutrient Retention Networks

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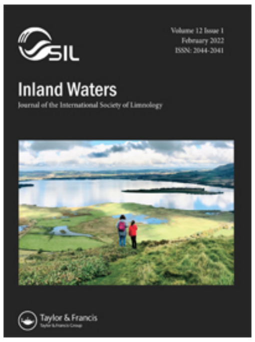

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Smart Nutrient Retention Networks: a novel approach for nutrient conservation through water quality management

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ABSTRACT
Nutrients are essential resources for food production but are used inefficiently, and thereby they pollute inland and coastal waters and are lost into the oceans. Nutrient conservation by retention and consecutive reuse would prevent nutrient losses to the atmosphere and downstream ecosystems. We present Smart Nutrient Retention Networks (SNRNs) as a novel management approach to achieve nutrient conservation across networks of connected waterbodies through strategic water quality management. To present the key features of SNRNs, we review existing knowledge of nutrient retention processes in inland waters, water quality management options for nutrient conservation, and nutrient retention models to develop SNRNs. We argue that successful nutrient conservation, even at a local level, through SNRN management strategies requires clearly formulated goals and catchment-wide system understanding. Waterbody characteristics, such as hydraulic residence time and the presence of macrophytes, shape local nutrient retention with potential network-wide cascading effects of improved water quality and are therefore key targets of SNRN management strategies. Nutrient retention models that include the self-reinforcing feedback loop of ecological water quality, nutrient retention, and nutrient loading in networks of inland waters in relation to management options can support the development of SNRNs. We conclude that SNRNs can contribute to sustainable use of nutrients in human food production.

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ecological feedbacks; hydrological networks; inland waters; models; nutrient reuse; smart management

Introduction

Societal challenge

> Fleets of vessels are dispatched, at great expense, to collect the dung of petrels and penguins at the South Pole, and the incalculable element of opulence which we have on hand, we send to the sea. All the human and animal manure which the world wastes, restored to the land instead of being cast into the water, would suffice to nourish the world. (Victor Hugo in "Les Misérables": Volume V – Jean Valjean, Second Book, 1862.)

Nutrients are essential resources for food production and socioeconomic development, but they pollute inland waters before they are washed to the sea. Currently, about 95 Tg of nitrogen (N) and 16 Tg of phosphorus (P) are applied annually as synthetic fertilizer worldwide (Beusen et al. 2016). Together with other anthropogenic and natural nutrient sources, this input leads to a global nutrient loading into inland waters of 64–253 Tg N and 9–30 Tg P per year (Beusen et al. 2016, Yuan et al. 2018, Smil 2000; Fig. 1), resulting in eutrophication problems such as harmful algal blooms (Heisler et al. 2008) and anoxia (Chislock et al. 2013). Nutrients ultimately flow to the sea where they cause coastal eutrophication before being lost to the sea bed or atmosphere (de Jonge et al. 2002). Thus, although the efficiency of obtaining nutrients has dramatically increased from guano mining in the time Victor Hugo wrote Les Misérables to mining finite deposits of mineral P and the Haber–Bosch process used to produce N today, the critical issue of losing these essential resources into the sea recognized a century and a half ago (Hugo 1862) persists.

Networks of inland waters

Individual waterbodies (e.g., reservoirs, lakes, rivers, and wetlands) can form a network of inland waters, exchanging nutrients and other substances through hydrological connections (Teurlincx et al. 2019) and influencing each

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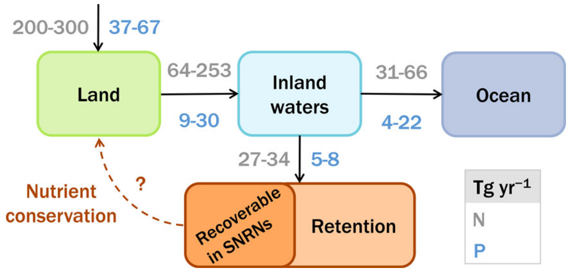

Figure 1. Global nitrogen (N; gray numbers) and phosphorus (P; blue numbers) flows in Tg per year (Smil 2000, Green et al. 2004, Van Drecht et al. 2005, Tysmans et al. 2013, Beusen et al. 2016). Nutrient inputs on land include natural and anthropogenic sources. Considerable amounts of these nutrients end up in inland waters. After entering inland waters, nutrients are retained within waterbodies or transported into the oceans. Note that these numbers represent estimates of global totals. Ratios between the different flows may differ strongly between individual river catchments (Tysmans et al. 2013). The question mark indicates an unknown fraction of retained nutrients that can be recovered in Smart Nutrient Retention Networks (SNRNs) to be reused on land for nutrient conservation. Color version available online.

other's chemical and ecological water quality (Tundisi et al. 1998, Carpenter and Lathrop 2014, Teurlincx et al. 2019). One waterbody can act as a net nutrient source or sink to downstream waterbodies (Zhang et al. 2012). Higher nutrient retention is associated with higher water quality (i.e., clear, submerged macrophyte-dominated vs. turbid, phytoplankton-dominated waters). Water flows through the network influence hydraulic residence times, which also determine nutrient retention (Van Gerven et al. 2017). When nutrient load reduction leads to an ecological regime shift from phytoplankton dominance to submerged macrophyte dominance in one waterbody, nutrient retention could increase locally, resulting in lower nutrient loading to connected waterbodies (discussed later; also see Supplemental Material A). This feedback could cause cascading effects of improved water quality and offer opportunities to benefit from local interventions on a network scale. However, despite the recognized importance of hydrological connections on nutrient flows and retention, water quality is generally assessed for individual waterbodies, and only a few model studies address the potential cascading effects of ecological water quality and nutrient retention in connected inland waters (Hilt et al. 2011, Van Gerven et al. 2017).

A novel approach for nutrient conservation through water quality management

Our aim was to fill this gap in the explicit use of inland water networks in water quality management by presenting what we named Smart Nutrient Retention Networks (SNRNs), a novel management approach for nutrient conservation through water quality management (Fig. 2). Here, we define nutrient conservation as the prevention of nutrient losses to the atmosphere and downstream ecosystems by nutrient retention and consecutive nutrient reuse. Nutrient retention comprises natural internal retention within and natural losses from waterbodies, as well as harvesting by humans. Retained nutrients are only conserved if reused, for example, as organic fertilizer to mitigate synthetic fertilizer production and application. Contrary to traditional water quality management that focuses on nutrient pollution reduction and local remediating interventions (Paerl et al. 2016, Strokal et al. 2020), SNRNs employ the biogeochemical *nutrient retention* potential of networks of inland waters to deal with nutrient pollution and promote nutrient conservation. Thus, we focus on a type of nutrient reuse that is often ignored.

SNRNs aim to restore degraded inland waters, mitigate further ecological degradation, prevent nutrient losses into the ocean, and stimulate on-land reuse of nutrients harvested from inland waters (Fig. 2, bottom right panel) through smart combinations of catchment-specific interventions that account for cascading effects of improved water quality in connected waterbodies. Hence, in SNRNs the *network* of inland waters is managed in a *smart* way, where contextual adaptive management decisions are based on actual data and prior knowledge. This smart management includes manipulation of the system's hydrology and ecological states to enhance nutrient retention. Moreover, nutrients in SNRNs are retained and reused by, for example, harvesting macrophytes, sediment, or fish (Fig. 2, top right panel). Overall, SNRNs could mitigate socioeconomic impacts related to nutrient pollution, which encompass chemical and ecological water quality degradation and unsustainable nutrient resource management.

In this paper, we explore current knowledge to develop SNRN management strategies. We specifically elaborate on (1) natural nutrient retention processes in individual waterbodies and networks of inland waters, (2) water quality management options for nutrient conservation, and (3) nutrient retention models. We focus on the northern temperate zone where strong human impacts on water systems (e.g., a legacy of intense nutrient enrichment) prevail, and regularly refer to examples from (sub)tropical regions where nutrient conservation is more common. Within the northern temperate zone, we expect that the principles of SNRNs can especially improve water quality and nutrient management in regions with highly modified and controlled water systems and many shallow lakes (with potential for macrophyte-dominated vs.

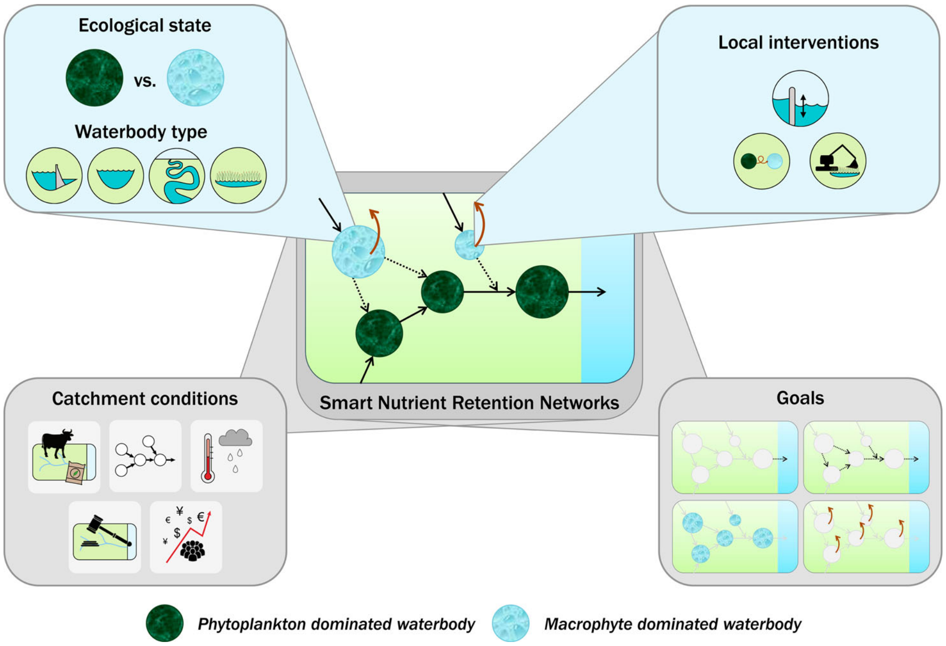

Figure 2. Smart Nutrient Retention Networks (middle panel) have multiple dimensions, that should be considered to achieve the goal(s) set for the network of inland waters: minimized nutrient loss to the oceans, maximized nutrient retention in inland waters, good ecological water quality, or maximized reuse of nutrients retained within the network (bottom right panel). At the level of individual waterbodies, the ecological state and waterbody type influence the potential for nutrient retention (top left panel). Local interventions can influence this nutrient retention potential, for example, by adjusting the hydrology, changing the ecological state, or harvesting and reusing nutrient retaining ecosystem components (top right panel). At the catchment level, conditions apply which are beyond the scope of local water managers: external nutrient loading, configuration of hydrological connections, climate change, catchment-level legislation, and socioeconomic conditions (bottom left panel). Color version available online.

phytoplankton-dominated states), such as lowland western Europe.

Nutrient retention processes in inland waters

In SNRNs, nutrient retention is maximized to benefit the whole catchment. The catchment covers the largest spatial scale of networks of inland waters, including connected waterbodies and the land draining into these waters. At the waterbody level, nutrients are either retained or flow freely with the water in dissolved or particulate forms such as detrital matter or phytoplankton (Teurlincx et al. 2019). Nutrient retention processes include (1) natural internal retention (e.g., long-term storage by sedimentation, burial of biomass, and P bound to mineral particles; Uhlmajnn and Horn 1992, Smolders et al. 2006, Finlay et al. 2013, Kong et al. 2019), (2) natural losses from the waterbody (e.g., denitrification and consumption by migrating waterfowl; Saunders and Kalff 2001, Doughty et al. 2016, Kong et al. 2019), or (3) harvesting by humans (e.g., in the form of macrophytes, sediment, or fish). Water management can influence nutrient retention pathways directly (e.g., harvesting by humans), and indirectly through ecosystem state management (e.g., increased denitrification by bank reshaping). In general, increases in nutrient retention processes could decrease the risk of harmful algal blooms. Some nutrient retention processes counteract nutrient conservation (e.g., N_2 degases by denitrification), which enhances water quality but constitutes a loss process.

Waterbody characteristics

The hydraulic residence time of a waterbody promotes denitrification and sedimentation because it increases sediment–water contact (Ahlgren et al. 1988, Jansson et al. 1994, Saunders and Kalff 2001, Brett and Benjamin 2008, de Klein and Koelmans 2011; Table 1). Waterbodies with a large volume and relatively low water discharge, such as large and dammed reservoirs, have long residence times (Maavara et al. 2015). In the meta-analysis by Saunders and Kalff (2001), rivers had the largest average water discharge rate, followed by lakes and wetlands, whereas total N (TN) retention was largest for wetlands, followed by lakes and rivers. Although the average TN retention differed per waterbody type, it was similar in all waterbody types when

Table 1. Illustrative examples of how waterbody type and system characteristics contribute to higher (+) or lower (–) N and P retention, focused on the northern temperate zone.

Waterbody type	System characteristics	N retention potential	P retention potential	Reference
Dammed reservoir	Hydraulic residence time	+	+	Vörösmarty et al. 2003, Maavara et al. 2015
	Stratification	+	–	Nürnberg 1984, Kõiv et al. 2011, Beaulieu et al. 2014
Lake	Hydraulic residence time	+	+	Ahlgren et al. 1988, Saunders and Kalff 2001, Brett and Benjamin 2008
	Stratification	+	–	Nürnberg 1984, Kõiv et al. 2011, Beaulieu et al. 2014
	Macrophyte/phytoplankton dominance	+/–	+/–	Hilt et al. 2017
River	Hydraulic residence time	+	+	Saunders and Kalff 2001, de Klein and Koelmans 2011
	Size	+	+	Wollheim et al. 2006
	Low-flow zone presence (e.g., with macrophytes)	+	+	Svendsen and Kronvang 1993, Schulz et al. 2003
	Streamflow variability	–	–	Ye et al. 2012
Wetland	Hydraulic residence time	+	+	Jansson et al. 1994, Saunders and Kalff 2001
	Volume to surface area ratio	–	+	Hansson et al. 2005
	Inundation time	+/–	+/–	Sollie et al. 2008, Powers et al. 2012
	Macrophyte stand density	+/–	+/–	Barko and James 1998, Sollie et al. 2008
	Vegetation type	+/–	+/–	Søndergaard et al. 2001

correcting for discharge rates (Saunders and Kalff 2001). Higher water residence times in wetlands may partly be due to dense macrophyte stands that reduce flow velocity and increase sedimentation (Petticrew and Kalff 1992, Benoy and Kalff 1999, Saunders and Kalff 2001). In streams, macrophytes may lower water velocity and provide shelter, stimulating sedimentation (Svendsen and Kronvang 1993, Schulz et al. 2003). The highest nutrient storage potential in rivers was found within and downstream of areas with macrophytes (Svendsen and Kronvang 1993, Schulz et al. 2003). Especially in summer, areas with macrophytes tend to retain more nutrients, but weed cutting and autumn storm flows counteract this temporary storage through resuspension (Svendsen and Kronvang 1993).

Additionally, stratification, relative nutrient processing rates, and the volume to surface area ratio of waterbodies influence the strength of nutrient retention processes (Table 1). Once a lake or reservoir stratifies and the hypolimnion becomes anoxic, it may act as a P source when the water column remixes or the outlet is at the bottom of a dam (Nürnberg 1984, Kõiv et al. 2011). The meta-analysis by Kõiv et al. (2011) of 54 reservoirs and lakes (0–6.6 m deep) showed that stratifying waterbodies are generally deeper, and their P retention capacity decreases with relative depth. This redox-dependent P retention in lake sediment also depends on nitrate, sulfate, and particulate iron concentrations (Andersen 1982, Gächter and Müller 2003). Moreover, the balance, or even tradeoff, between biogeochemical nutrient processing rates and hydraulic residence time may determine net nutrient retention (Höhener and Gächter 1993, Powers et al. 2012, Schmadel et al. 2018), as described by the nutrient spiraling theory for individual streams (Newbold et al. 1981). For example, the combination of shorter hydraulic residence times, relatively invariant reaction times, and larger nutrient loadings during high-flow periods results in a lower N retention efficiency in rivers with higher streamflow variability (Ye et al. 2012). Additionally, direct P adsorption/desorption between water and sediment may be more important than sedimentation in shallow lakes with a relatively large sediment surface area to lake volume (Andersen 1997). Also, shallow wetlands with a large surface area likely retain N through denitrification, whereas those with a smaller surface area more likely retain P by sedimentation (Hansson et al. 2005).

Moreover, the amount of retained N and P tends to increase with nutrient loading (Prairie 1989, Saunders and Kalff 2001, Kõiv et al. 2011, Wang et al. 2020). Saunders and Kalff (2001) found that N loading is an excellent statistical predictor for the magnitude of TN retention in wetlands and lakes. Thus for N retention, water discharge or hydraulic residence time and nutrient loading are important determining factors, although hydraulic residence time's effect was strongest in the global lake dataset of Finlay et al. (2013). Further, P sedimentation in 4 lakes worldwide was found to correlate with P loading and in-lake P concentrations (Prairie 1989). The relation between N and P retention and nutrient loading differs seasonally, however (Hansson et al. 2005). Moreover, increased total P concentrations can increase N retention by stimulating phytoplankton production, settling, and decomposition, which decreases dissolved oxygen concentrations and thereby increases denitrification rates (Finlay et al. 2013). For example, Finlay et al. (2013) showed for a diverse and broadly representative set of lakes that N retention was >7 times higher in P-rich eutrophic lakes than in oligotrophic lakes, and similar trends were found by Donald et al. (2015) for 12 reservoirs in Canada. This dependency of nutrient retention on nutrient loading and seasonal variation may partly be explained by the ecological configuration of the ecosystem in the growing season, with limited retention in oligotrophic systems with little macrophyte growth, increased retention in mesotrophic systems with strong macrophyte growth, and either high or low retention in eutrophic systems, depending on submerged macrophyte or phytoplankton dominance, respectively.

Especially when biological processes are dominant, nutrient retention shows seasonal patterns along with temperature dependencies of process rates (Kadlec and Reddy 2001, de Klein and Koelmans 2011, Wang et al. 2020). At higher temperatures, process rates (e.g., of denitrification) generally increase but oxygen levels tend to drop (Kadlec and Reddy 2001, Jeppesen et al. 2009). Low oxygen levels may result in lower nutrient retention by reducing nitrification and denitrification (Jeppesen et al. 2009, Özen et al. 2010) and enhancing sediment P release (Jensen and Andersen 1992). For example, in Lake Chaohu (China) low summer P retention is due to increased sediment P release and low winter N retention is due to low rates of denitrification (Wang et al. 2020). Reduced nutrient retention under winter conditions can also be explained by nitrate accumulation under ice cover and nutrient release by senescing vegetation (White and Bayley 2001). Moreover, snow changes hydrological conditions as it accumulates and causes long and intense runoff as it melts (German et al. 2003). During snowmelt, this process results in lower nutrient retention efficiencies (German et al. 2003), probably because of shorter hydraulic residence times.

Furthermore, climatic conditions such as precipitation, temperature, and degree of seasonality (Lewis 1996) are important for nutrient retention. Especially in tropical river lakes, seasonal precipitation may strongly affect hydraulic residence time (Lewis 1996) and thereby nutrient retention. In drier climates, less nutrient loading by runoff results in lower in-lake nutrient concentrations (Jeppesen et al. 2009, 2011, Özen et al. 2010). By contrast, in warmer climates more evaporation results in higher in-lake nutrient concentrations and higher chances of harmful algal blooms (Jeppesen et al. 2009, 2011, Özen et al. 2010). Similarly, macrophyte cover and critical nutrient loading levels at which lakes turn from clear to turbid are expected to decrease with warming (Jeppesen et al. 2009, 2011, Özen et al. 2010), and therefore lower nutrient retention is expected at higher temperatures. The balance between the effects of altered hydraulic residence time, nutrient loading, evaporation, and ecological state will determine the effect of different climates on nutrient retention.

The hydrological configuration of networks of inland waters influences local and network-wide nutrient retention. In systems with connected waterbodies, water quality, nutrient concentrations, and nutrient retention may differ depending on surrounding landscape and position in the catchment (Miranda et al. 2008, Schmadel et al. 2018, Teurlincx et al. 2019). For example, in river networks, large rivers retain more N than small rivers because they more effectively retain N per mean length of stream order, and they receive nutrients that are not retained in smaller rivers or that bypass them on land and directly enter the larger river (Wollheim et al. 2006). In general, network-wide nutrient retention is higher with abundant retaining waterbodies, such as lakes (Huttunen et al. 2016; see Supplemental Material B for an example of chains of lakes modeled with PCLake). Additionally, N removal increases with pond roundness and connectivity between ponds and streams (Schmadel et al. 2018). Also, when wetlands and floodplains temporarily connect to rivers by inundation, additional nutrient retention or release may occur (Noe and Hupp 2007). The multi-pond system above Chaohu Lake (China) exemplifies how connected waterbodies can influence nutrient retention (Yin et al. 1993). The ponds retain water, sediment, and nutrients and are used for rice field irrigation, which enhances nutrient recycling and retention on land. In 1993, Yin et al. (1993) reported that from January to September overall, 99% and 98% of N and P loading was retained.

Macrophytes

An intricate balance of direct and indirect processes determines the net effect of macrophytes on nutrient retention (Fig. 3). Macrophytes directly contribute to nutrient retention by assimilation and consecutive burial of plant litter (Granéli and Solander 1988, Jansson et al. 1994, Clarke 2002, Kreiling et al. 2011), enhancing sedimentation and decreasing resuspension (Howard-Williams 1983, Clarke 2002, Zhu et al. 2015). Moreover, macrophytes indirectly contribute to nutrient retention by providing an attachment surface for nutrient consuming epiphytes and denitrifying bacteria (Howard-Williams and Allanson 1981, Weisner et al. 1994) and stimulating both P sorption and (coupled nitrification-) denitrification by sediment oxygenation (Risgaard-Petersen and Jensen 1997, Ottosen et al. 1999, Smolders et al. 2002, Kreiling et al. 2011, Vila-Costa et al. 2016). Macrophytes indirectly stimulate denitrification by (dissolved) organic carbon supply to the water or sediment and sediment nitrate penetration by root water uptake while inhibiting denitrification by competition for N and production and release of oxygen (Weisner et al.

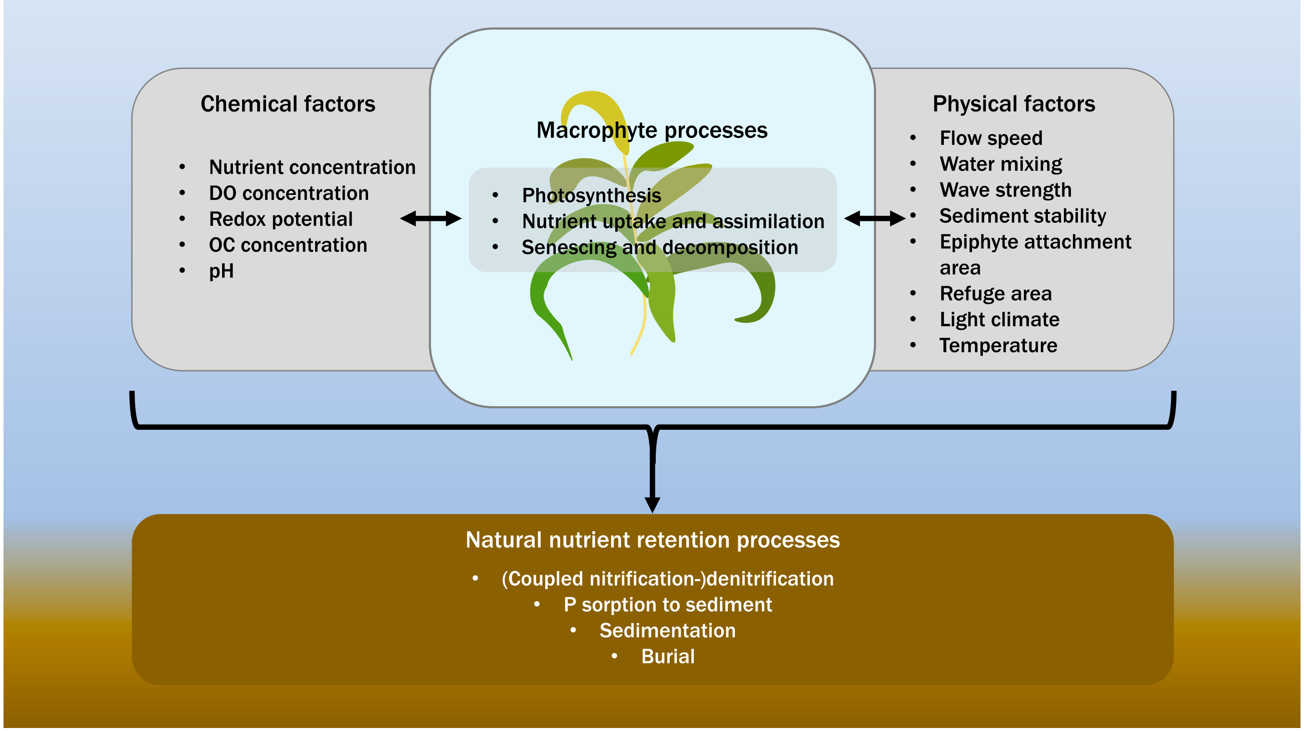

Figure 3. Macrophytes directly and indirectly influence natural nutrient retention processes (white text in brown box) through macrophyte processes (light box). Nutrient uptake and assimilation followed by burial is the most direct route. Indirect influences are found through the effect of macrophytes on both chemical (e.g., dissolved oxygen [DO] and organic carbon [OC] concentration) and physical factors (dark boxes). Note the importance of seasonality, nutrient form (particulate/dissolved), macrophyte functional group, and species for macrophyte and nutrient retention processes. Color version available online.

1994). Moreover, shading by macrophytes may both stimulate and inhibit denitrification because it decreases oxygen production by photosynthesis and lowers the water temperature, respectively (Weisner et al. 1994). Furthermore, the net effect of macrophytes on nutrient retention depends on the time scale of measurement and the nutrient form (Carpenter and Lodge 1986). For example, macrophytes accumulate nutrients during spring and summer but release nutrients during their senescence (Landers 1982). And in general, macrophyte stands contribute to net particulate P retention and net dissolved P release (Carpenter and Lodge 1986).

Macrophyte species and their functional groups may strongly influence their effect on nutrient retention. Larger vegetation types (e.g., helophytes) have more biomass and are therefore expected to contribute more to nutrient retention than low herbaceous vegetation (Sollie et al. 2008). Charophytes are more efficient nutrient sinks than vascular macrophytes because charophytes have lower decomposition rates and take up most of their nutrients from the water (vs. sediment) because of their larger shoot to root ratios (Kufel and Kufel 2002). Whether a macrophyte species is rooting and sessile or non-rooting and (similar to phytoplankton) flowing along with the water (Janssen et al. 2019b) is essential for their contribution to nutrient retention. Also, the functional group strongly influences denitrification rates. Especially, rooted macrophytes can increase denitrification by oxygenating the sediment, thereby enhancing coupled nitrification–denitrification (Risgaard-Petersen and Jensen 1997, Ottosen et al. 1999, Vila-Costa et al. 2016). Closed mats of floating macrophytes may increase denitrification and sediment P release through low dissolved oxygen concentrations (Veraart et al. 2011, Janssen et al. 2020). The net contribution to nutrient retention from other (e.g., submerged) macrophyte species is less evident (Søndergaard et al. 2001). In particular, dense macrophyte stands can seasonally cause a net sediment P release from low oxygen levels (e.g., during decomposition or by constrained water mixing) or increase pH due to high primary production (Søndergaard 1988, Frodge et al. 1991, Barko and James 1998).

Although outcomes among studies vary, they mostly show higher nutrient retention in macrophyte-dominated over phytoplankton-dominated shallow lakes (26 of 40 unique papers on nutrient retention in the review by Hilt et al. 2017). For example, from Veraart et al. (2011) we expect about 10 times more N retention in a vegetated over an unvegetated state (with 12 h light). The relative contribution of assimilation in macrophytes to overall nutrient retention also varies, with 8–77% for N and 12–73% for P (Reddy and De Busk 1985, Kreiling et al. 2011, Veraart et al. 2011, Wang et al. 2013). Vegetation also enhances nutrient retention in streams (Balestrini et al. 2018). Moreover, the nutrient retention capacity of macrophytes explains one of the self-reinforcing (i.e., mathematically positive) feedback loops that self-maintain macrophyte-dominated versus phytoplankton-dominated states in shallow lakes (Scheffer et al. 1993), demonstrated by the ecosystem model PCLake (Supplemental Material A).

Feedback loops involving macrophytes as described by Scheffer et al. (1993) for individual waterbodies can also emerge in hydrological networks. For example, Gillis et al. (2014) showed that mangrove forests and seagrass beds retain nutrients, providing positive interactions with connected ecosystems (e.g., coral reefs) through reduced nutrient loadings. Such spatial effects of local nutrient retention by marine ecosystems could also be expected in networks of inland waters (Teurlincx et al. 2019) through a self-reinforcing feedback loop between nutrient loading, ecological water quality, and nutrient retention in networks of inland waters (Fig. 4). The underlying theory is that nutrient loading reduces water quality (i.e., increases the likelihood of phytoplankton dominance over macrophyte dominance and hence turbidity over clarity; Scheffer et al. 1993). Good water quality itself results in higher nutrient retention by the self-reinforcing feedback loop between macrophytes and water clarity within the waterbody. Finally, on the hydrological network level, the increased nutrient retention within the waterbody decreases nutrient loading to downstream waterbodies, where the feedback chain could repeat itself (i.e., has spatial cascading effects; Klose et al. 2020), resulting in a self-reinforcing feedback loop for the entire network of inland waters.

Water quality management options for nutrient conservation

Hydrological management

Hydrological management strategies to retain, harvest, and reuse more nutrients in and from inland waters can be applied with the ultimate goal to improve nutrient conservation in the entire catchment. Local changes in hydrology and nutrient retention affect downstream hydrology, ecology, and nutrient retention (Hilt et al. 2011, Jenny et al. 2014, Kondolf et al. 2014, Van Cappellen and Maavara 2016, Teurlincx et al. 2019, Maavara et al. 2020). Therefore, Hilt et al. (2011) and Teurlincx et al. (2019) argued for a hydrological network perspective and to tactically use local, upstream interventions. Local hydrological interventions may include water level and flow regulation by pumps, dams, and sluices, and dechannelization (Vörösmarty 1997, Stanley and Doyle 2002, Li et al. 2013, Kong et al. 2017, Fraaije et al. 2019, Maavara et al. 2020). These measures alter the hydraulic residence time and risk of hypoxia (Jenny et al. 2014), thereby influencing nutrient retention in the waterbody (Wang et al. 2020). Also, waterbody types may be altered; for example, dam construction can convert river sections into dammed reservoirs and strongly increase P retention (Tundisi et al. 1998, Vörösmarty et al. 2003). In addition to waterbody-level interventions, management of the hydrological network structure may stimulate net nutrient retention, for example, by increasing lateral connections between rivers and vegetated lakes or wetlands (Mitsch et al. 2008, Kreiling et al. 2011, Newcomer Johnson et al. 2016). More natural reconstruction of hydrological networks may have mutual benefits; for example, increased nutrient retention and flood protection are expected

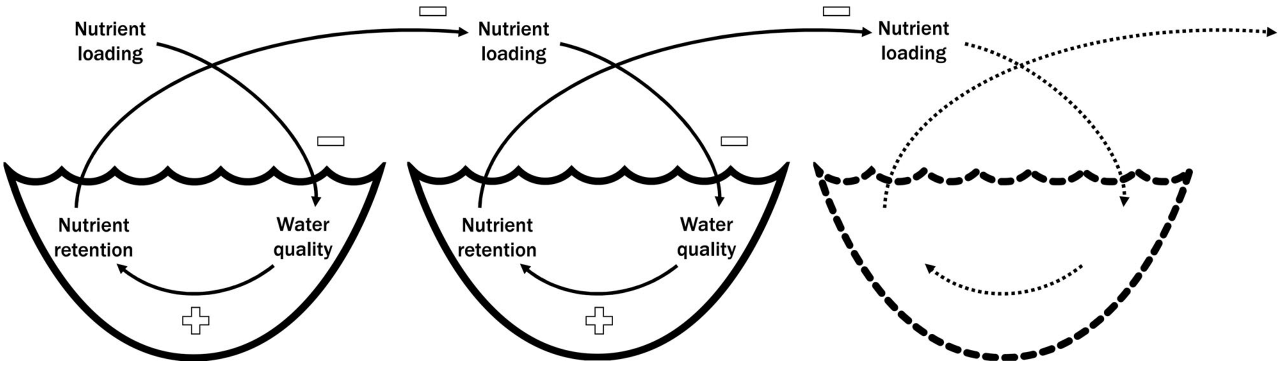

Figure 4. Schematic of the self-reinforcing (i.e., mathematically positive) feedback loop in networks of inland waters. Nutrient loading has a negative feedback on ecological water quality, water quality has a positive feedback on nutrient retention, and nutrient retention has a negative feedback on nutrient loading downstream, overall resulting in a self-reinforcing feedback loop that cascades down the network.

after (re)construction of waterways and flooding areas in the Dutch “Room for the River” and “Living with Water” projects (Van Gerven et al. 2009).

Macrophyte-focused management

Stimulation and preservation of macrophytes (Hilt et al. 2006) may enhance nutrient retention and ecological water quality (e.g., according to the European Water Framework Directive). However, macrophytes may cause problems for drinking water and hydropower production and block waterways for boating (Tundisi et al. 1998, Hilt et al. 2006). Yet the public perception on macrophyte establishment, in for example urban streams, is generally positive or neutral (Larned et al. 2006). Moreover, macrophyte-stimulating interventions may have multiple benefits, such as enhancing biodiversity, recreational value, fish spawning areas, and nutrient retention. At present, the benefits of paludiculture (i.e., wet agriculture/forestry on rewetted peatlands) are being explored (Vroom et al. 2018). Moreover, more vegetated water systems are being promoted or developed to support biodiversity, such as thousands of kilometers of nature-friendly banks in the Netherlands (ter Veld 2014). For lakes specifically, diverse management options to promote macrophyte dominance over phytoplankton dominance exist: flushing with cleaner water, nutrient load reductions beyond the lower critical nutrient load, and biomanipulation by fish removal (Janse et al. 2008, Bernes et al. 2015, Janssen et al. 2019b). Here, we highlight 2 more examples of macrophyte-focused management that can be applied for nutrient conservation: constructed wetlands and an engineering project for macrophyte harvesting and reuse.

Constructed wetlands

Constructed wetlands are wet systems created with macrophytes (and sediment), mostly used to treat wastewater. For example, reed filters in a stream bypass purify and store water at the estate of Lankheet (the Netherlands; Mulder and Querner 2008), and in (sub)tropical regions, common water hyacinth (*Eichhornia crassipes*) and water lettuce (*Pistia stratiotes*) are used to bioremediate multiple wastewater types (Reddy and D’angelo 1990, Kutty et al. 2009, Lu et al. 2010, Akinbile and Yusoff 2012). Howard-Williams (1985) extensively reviewed N and P retention in wetlands and the early developments of constructed wetlands, and Wu et al. (2015) comprehensively reviewed constructed wetland application and recent developments on their sustainable design. The nutrient retention effectiveness of constructed wetlands tends to decrease over time, especially for P (Mitsch et al. 2014), possibly explained by saturation of the soil and accumulation of detritus and plant biomass (Mitsch et al. 2012). To counteract saturation and accumulation effects and to maintain effectiveness of the constructed wetlands, these nutrient-retaining components should be harvested and used as a (nutrient) source elsewhere, thereby enhancing nutrient conservation (e.g., see Reddy and D’angelo 1990). For example, the new “bio-cascade water purification” approach applies knowledge of biogeochemical processes in soil, water, and macrophytes to prevent saturation effects in connected water basins and to conserve nutrients by harvesting helophytes and floating macrophytes (Kwakernaak et al. 2015). However, to meet increasingly strict water quality standards, research and development is still required for appropriate plant harvest and reuse strategies in constructed wetlands (Wu et al. 2015).

Harvesting

More nutrients could be reused by mowing macrophytes (Kuiper et al. 2017), dredging sediment, or fishing. Think of using reed as a building material (Köbbing et al. 2013), lake-dredged materials and decayed or processed water hyacinths as soil amendments (Sigua 2009, Aremu et al. 2012, Masto et al. 2013), or fish as a food source (Edwards et al. 1997, McIntyre et al. 2016, Kim et al. 2019). Such harvests are currently occurring but rarely considered for nutrient conservation or to combat eutrophication problems. For example, Tang and Xie (2000) considered fish catches, like water outflow, a nutrient outflow. Nevertheless, fishing conserved 3–4% and 10% of the N and P loading, respectively. We noted one example of macrophyte harvesting to purposefully remove nutrients from a natural waterbody: an ecological engineering project in subtropical Lake Caohai (China; Wang et al. 2013). Here, seedlings of common water hyacinth were planted in constructed enclosures, harvested after growth, and processed into biogas and organic fertilizer. This process removed 76% of the inflowing TN, with 65% of the overall retained N in the form of macrophyte biomass. Wang et al. (2013) concluded that “large scale utilization of *E. crassipes* for removal of N in the eutrophic lake [Caohai] is practicable,” thus the potential for nutrient conservation exists. However, harvesting should be applied with care to avoid drastic ecosystem disruptions (Van Zuidam and Peeters 2012, Kuiper et al. 2017) and the loss of indirect contributions of macrophytes to nutrient retention. Moreover, risks of contaminating food chains should be carefully assessed before reusing macrophytes and dredged materials (Beyer and Stafford 1993, Aremu et al. 2012).

Catchment-level nutrient management

Most water quality management measures are applied to individual waterbodies but would become more effective if embedded in catchment-wide management strategies, considering potentially cascading effects of improved water quality in networks of inland waters (Fig. 4). At the catchment level, additional measures can be taken by, for example, regional or (inter)national governing authorities (Fig. 2). These measures can target external nutrient loading (i.e., nutrients from diffuse and point sources in the catchment, which may eventually reach target waterbodies) and hydrological connections. These issues may be addressed by landscape-level legislation and enforcement (e.g., fertilizer application limits and wastewater treatment standards) or catchment-level management (e.g., changing hydrological network structure by (re)constructing waterways and adjusting macrophyte mowing schemes) but are beyond the scope of local water managers. Moreover, nutrient loading and retention are influenced by socioeconomic and climatic changes, for example land use change, population and economic growth, increasing temperatures, and changes in precipitation and runoff (Strokal et al. 2016). These (inter)national and global challenges require adjustments at even larger scales.

Nutrient retention models

SNRN management strategies can be designed with the help of nutrient retention models available at various spatial scales (Supplemental Material C). Nutrient retention models simulating individual waterbodies are often process-based (i.e., employing process rates and mechanistic insights to estimate nutrient retention; Van Gerven et al. 2009). For example, PCLake(+) (Janse 2005, Janssen et al. 2019a), PCDitch (Janse and Van Puijenbroek 1997, Janse 2005), and the GLOBIO-Wetlands model (under development; Janse et al. 2019) are process-based models from which nutrient retention processes and balances can be derived (Kong et al. 2019). These models include feedback loops between ecological states and nutrient retention and have been used to explore water quality management options (Janssen et al. 2019b). An example of a partly process-based model to analyze the effect of interventions on water quality is the Dutch KRW-Verkenner (Water Framework Directive Explorer). Users can themselves assign nutrient retention fractions of waterbodies, guided by meta-models for lowland streams, shallow lakes, and rivers that are statistically derived from the mechanistic process-based model AquaVenus (de Klein 2008, Van Gerven et al. 2009). Although some of the meta-models include the effect of macrophytes on nutrient retention, they do not include feedback loops between nutrient loading, ecological water quality, and nutrient retention (Van Gerven et al. 2009). Wetland models most explicitly cover the effect of macrophytes on nutrient retention and are usually more specific, for example, focusing on either N or P, or on a specific wetland type and location (Mitsch and Reeder 1991, Van Dam et al. 2007). Most of these and other waterbody-level nutrient retention models include macrophyte and/or phytoplankton presence (Supplemental Material C).

On the hydrological network level (i.e., global or catchment scale in Supplemental Material C), nutrient retention is often expressed as a fraction of the nutrient flow into the system that is retained and derived by statistical relationships to one or multiple waterbody characteristics. For example, the global model WorldQual derives P retention in waterbodies as a function for the whole catchment depending on hydraulic residence time (Fink et al. 2018). In other global models, the nutrient retention fraction is, in addition to hydraulic residence time, based on denitrification and sedimentation rates (Harrison et al. 2009, Beusen et al. 2016). GlobalNEWS includes denitrification in rivers as a function of water depth and travel time (Seitzinger et al. 2002), nutrient removal by water abstraction for irrigation and other human consumptive water use, and retention in dammed reservoirs as a function of hydraulic residence time and depth (Mayorga et al. 2010). The MARINA model builds on the latter global model but is specified for Chinese river catchments, with multiple subcatchments and channel section-specific nutrient retention (Strokal et al. 2016). In addition to the retention processes in GlobalNEWS, MARINA includes dissolved inorganic P retention in rivers by, for example, sedimentation and accumulation (Strokal et al. 2016). All these and other large-scale nutrient retention models exclude management options for individual waterbodies and generally disregard the effect of ecological state (Supplemental Material C).

To support the development of SNRNs, a model should include management options and the self-reinforcing feedback loop of ecological water quality, nutrient retention, and nutrient loading in networks of inland waters. Hence, existing nutrient retention models for individual waterbodies seem more promising than large-scale models. Especially, global-scale models strongly simplify nutrient retention, which may be justified because they are intended to identify global pollution hotspots and long-term water quality trends, whereas local-scale models are more often designed to assess management options (Tang et al. 2019). Large-scale models disregard alternative ecological states of lakes, which might be explained by their focus on rivers, and neither explicitly consider biogeochemical nutrient retention processes other than denitrification. Most local-scale models are more detailed and process-based but also have their limitations for modeling nutrient retention management at catchment-scale. For example, these models do not focus on nutrient retention, or they focus only on specific nutrients or waterbodies types (Supplemental Material C). The advantage of detailed, process-based models for SNRN management strategy development is that they more often include management options, primary producers, and potentially include feedback loops between ecological states and nutrient retention. By their mechanistic basis, these more process-based local-scale models can serve as a building block to consider cascading effects on a hydrological network scale; alternatively, their insights can be applied in large-scale models.

Toward Smart Nutrient Retention Network Management Strategies

The specific design of any SNRN management strategy depends on the configuration and state of the targeted network of inland waters, including its local waterbody types and their ecological states, and availability and efficacy of local intervention options (Fig. 2). Moreover, successful nutrient conservation through the implementation of SNRN management strategies requires an understanding of catchment conditions and clearly formulated goals (Fig. 2). Catchment conditions include biogeochemical and social system properties such as external nutrient loading, hydrological connections, climate change, catchment-level legislation, and socio-economic conditions. Tailoring case-specific goals and strategies to such cross-sectoral catchment conditions is beyond the scope of water managers alone and requires partnerships of stakeholders and involvement of larger scale governing authorities (see catchment-level nutrient management discussion earlier).

Formulation of clear goals before implementation is a key task for responsible managers and stakeholders. It is crucial that the manager is aware of potential local and regional tradeoffs of interventions and between goals. For example, interventions may have contrary effects on N and P retention, denitrification can enhance water quality but counteract nutrient conservation, and increased flushing may locally improve water quality while deteriorating downstream ecosystems through increased nutrient loading. To avoid unforeseen tradeoffs, more complete and quantitative empirical studies must be conducted on the effect of macrophytes on nutrient retention at the level of networks of inland waters. Empirical examples and studies of effective, efficient, and safe nutrient harvesting and reuse can inform the further development of SNRNs, specifically to manage the risk of undesired changes in ecological state and nutrient retention potential. Ideally, the impact of SNRN management strategies on processes and goals aside from nutrient retention and conservation should be considered, as advocated by integrated water resource management approaches (Al-Jawad et al. 2019). For example, greenhouse gas emissions (Deemer et al. 2016, Chen et al. 2019), biodiversity loss (Dudgeon 2000, Hansson et al. 2005), and sediment starvation (Kondolf et al. 2014) should be considered.

SNRN management strategies can be designed conceptually, based on expert knowledge, or with the help of simulation models. Simulation models can provide outcomes of multiple scenarios for quantitative comparison, but integrated models must be developed to include (1) applications to networks of inland waters; (2) feedback between ecological states, nutrient retention, and nutrient loading; and (3) nutrient retention management options. Networks of inland waters could be modeled using a node-link schematization, building on existing waterbody models, as suggested by Teurlincx et al. (2019). When a wide range of management options is available, such models could be run with optimization algorithms to determine the optimal solution (Al-Jawad et al. 2019, Strokal et al. 2020). While time-consuming and challenging, using an optimization approach could result in novel solutions to water management problems that might not be found intuitively.

Developing complex nutrient retention network models could drive significant improvements in network-level water management for nutrient conservation. Nonetheless, these developments are not necessary for water managers to start to include the core principles of SNRNs. SNRN principles could, for instance, be applied to the “River Basin Management Plans” required by the European Water Framework Directive (Griffiths 2002). Beyond Europe, considering nutrient retention in networks of inland waters may support sustainable water quality and nutrient management.

Human food production was long dependent on, and limited by, nutrient recycling by manure application on agricultural lands. With the onset of the industrial age, feeding a growing world population became increasingly dependent on unsustainable fertilization techniques that exploit mineral P deposits and marginally renewable geological stocks of fossil fuels for N fixation. Maintaining, and ensuring for all, the high living standards that result from such unsustainable agricultural practices, but within the means of the planet, is today's greatest challenge (Raworth 2012). Smart Nutrient Retention Networks can be one component of a sustainable and just future as they are designed around biogeochemical nutrient cycling processes and contribute directly to at least 3 UN Sustainable Development Goals (SDGs): zero hunger (SDG 2), clean water and sanitation (SDG 6), and responsible consumption and production (SDG 12) (United Nations 2015).

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