water

MDPI

Review

Nutrient Retention in Ecologically Functional Floodplains: A Review

Brad A. Gordon [1,*], Olivia Dorothy [1] and Christian F. Lenhart [2]

[1] American Rivers, 1101 14th Street NW, Suite 1400, Washington, DC 20005, USA; odorothy@americanrivers.org
[2] Department of Bioproducts and Biosystems Engineering, University of Minnesota, 1390 Eckles Ave., St. Paul, Minneapolis, MN 55108, USA; lenh0010@umn.edu
* Correspondence: bradgordon18@gmail.com; Tel.: +1-651-272-3991

Received: 11 August 2020; Accepted: 28 September 2020; Published: 4 October 2020

Abstract: Nutrient loads in fresh and coastal waters continue to lead to harmful algal blooms across the globe. Historically, floodplains—low-lying areas adjacent to streams and rivers that become inundated during high-flow events—would have been nutrient deposition and/or removal sites within riparian corridors, but many floodplains have been developed and/or disconnected. This review synthesizes literature and data available from field studies quantifying nitrogen (N) and phosphorus (P) removal within floodplains across North America and Europe to determine how effective floodplain restoration is at removing nutrients. The mean removal of nitrate-N (NO_3^--N), the primary form of N in floodplain studies, was 200 (SD = 198) kg-N ha^{-1} year^{-1}, and of total or particulate P was 21.0 (SD = 31.4) kg-P ha^{-1} year^{-1}. Based on the literature, more effective designs of restored floodplains should include optimal hydraulic load, permanent wetlands, geomorphic diversity, and dense vegetation. Floodplain restorations along waterways with higher nutrient concentrations could lead to a more effective investment for nutrient removal. Overall, restoring and reconnecting floodplains throughout watersheds is a viable and effective means of removing nutrients while also restoring the many other benefits that floodplains provide.

Keywords: floodplain restoration; nitrogen; phosphorus; nutrient retention; algal blooms

1. Introduction

Despite global efforts to reduce nutrient loads, harmful algal blooms (HABs) and hypoxic zones are occurring more often, in new places, for longer durations, and at different toxicities [1]. Most are caused by nutrient pollution [1]. The impacts of these algal blooms can be devastating for human and animal health, wildlife habitat, and economics. Nutrient loads have been increasing, especially from nitrogen (N) and phosphorus (P) fertilizer use, river alterations, field and bank erosion, human and animal waste, and expanding aquaculture practices [1–9]. Common forms of nutrients in waterways are nitrite (NO_2^-), nitrate (NO_3^-), ammonium (NH_4^+), organic N, particulate P, and soluble reactive phosphorus (SRP). However, the proportions of each form of these nutrients are in constant flux, and the occurrence of each algal bloom depends on the algal species, environmental conditions, other organisms creating those conditions, and the nutrient concentrations and forms present [1]. Each form of N and different ratios of N:P lead to the growth of different algal species and varying levels of toxicity from some species [10–14]. In addition to toxicity from HABs, high concentrations of NO_3^- in drinking water are a risk to human health and NO_3^- has been prioritized for nutrient removal efforts in many areas, especially in the Mississippi River Basin [4,15–17].

Efforts have been made to reduce nutrient loads from point sources across the globe but some of those sources of nutrients have increased, for example, where populations have increased but sewage treatment has not improved [1]. Furthermore, there is still a significant need for reducing non-point sources of nutrients. Current attempts to reduce non-point nutrient pollution include practices such as riparian buffers, cover crops, contour farming, fertilizer management, wetlands, woodchip bioreactors, and others [18–25]. However, there need to be more options in the riparian corridor that are highly effective at removing nutrients while providing other needed benefits such as water storage, carbon sequestration, fish and wildlife habitat, and recreational benefits. [26].

1.1. Floodplain Processes Driving Nutrient Cycling

Ecologically functional floodplains are connected to the river without barriers that prevent flooding, and they are capable of supporting plant and animal communities native to that region. They can either be naturally occurring or restored via human action to reconnect the floodplain and restore the natural ecosystem features. Ecologically functional floodplains provide multiple processes that improve nutrient retention (Figure 1). Many of these processes are similar to those driving nutrient cycling in treatment wetlands [27].

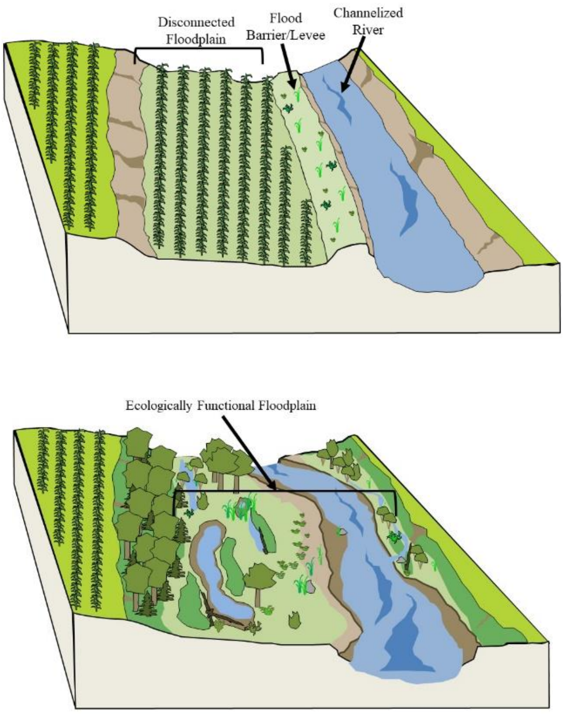

Figure 1. This is a depiction of a disconnected and non-functional versus an ecologically functional floodplain. Ecologically functional floodplains flood regularly without constructed barriers or incised main channels that prevent water from entering the floodplain during high-flow events. They are also capable of supporting plant and animal communities native to that region.

The best permanent removal mechanism for N is usually denitrification if N is in the form of NO_3^- or first converted to it [28]. Denitrification is a microbially facilitated reduction of dissolved NO_3^- to a gaseous form of N usually under anaerobic conditions. Most denitrifying microbes use NO_3^- as an electron acceptor for respiration when oxygen is depleted, although some microbes prefer NO_3^- over oxygen [27]. When soil is saturated for extended periods of time, the oxygen becomes depleted and microbes switch to using NO_3^- instead of oxygen. Several other conditions are necessary for denitrification including abundant carbon, warm temperatures, readily available NO_3^-, and a sufficient population of microbes [29]. If water levels fluctuate to the extent that oxygen is replenished, then most microbes will switch back to oxygen and could even increase the NO_3^- concentration by converting other forms of N to NO_3^- [27].

Because particulate P is the primary form of P in large rivers including the Mississippi River [4], sedimentation, soil accretion, and burial are the primary means of removing P throughout those basins [30]. Floodplains are thought to be net sinks for sediment [31]. However, in basins where SRP is the primary form, biological uptake is likely the primary means of removing P. Generally, the dissolved form of SRP is the form that is most available for uptake by plants and algae and contributes disproportionally to algal blooms [32]. It can exchange between the dissolved or particulate phases depending on the conditions and through processes such as decomposition, biological uptake, redoximorphic release, or sedimentation and accretion [18]. Dissolved forms are often released from soil during anaerobic conditions; changes to pH can also drive release of P from soils that fluctuate between wet and dry conditions. Soil properties that favor retention of P through iron and other chemical complexes strongly influence the ability of floodplain soils to retain P [33].

1.2. Limitations to the Nutrient Cycle in Floodplains

With extensive alterations to riparian corridors associated with human settlement along rivers, many limitations have impeded the cycling of nutrients in floodplains. Floodplains have historically been some of the most nutrient-rich soil for agriculture and provide land with easy access to the riverfront. Rivers have undergone drastic changes to accommodate transportation, urban development, and agriculture. Some of the earliest activities responsible for changes to the floodplain include timber harvesting, agriculture, and river impoundment. For example, surveys from the Mississippi River above its confluence with the Illinois River indicate that 56% of the floodplain was forested and 41% was prairie before European settlement. The floodplains were reduced to 35% forest cover by 1938 and 6% prairie by 1994 [34]. Llewellyn [35] estimates that only 20,000 km^2 of the formerly 85,000 km^2 of bottomland hardwood forests along the Mississippi River remain. By the 1990s, an estimated 90% of the entire Mississippi River floodplain had been disconnected from the main channel due to levee construction [36] and up to 90% of floodplains across North America are cultivated and thus have lost most, if not all, ecological functionality [37]. In the Upper Mississippi River Basin alone, there are over 8000 miles of known levees disconnecting floodplains from their river channels [38].

Floodplains typically become wider and provide more flood storage moving downstream as streamflow and river size increase. Lower river reaches tend to have a longer floodwater retention, promoting nutrient removal. Floodplain soils are dynamic, as sand, silt and clay are deposited, or resuspended, by the river. Floodplain soil is often layered as the sediment is deposited at different rates that depend on grain size during each flood event. Larger grain sizes (typically sands or gravel) deposit first as swift moving water slows. Silt and clay drop out of the water column later, as velocities are reduced, typically further away from the main channel boundary [30]. Phosphorus retention is highest in floodplain soils that have more silt and clay because P can bond with the chemically reactive surfaces of clay and silt-sized organic matter particles. Soils with abundant iron, aluminum and manganese often form insoluble phosphate compounds that remove P from more mobile, soluble forms. Depending on pH and oxygen levels, P may dissolve and become available again, as SRP, particularly during anaerobic conditions [39–41].

1.3. Selection Criteria for Literature Search

With the extent of floodplain acres currently disconnected from the Mississippi River and other rivers across the globe, nutrient removal by floodplains has been greatly reduced. However, attempts to quantify and project these potential reductions are still limited. Restored and constructed wetlands have been thoroughly reviewed several times already [42–45] and a small collection of floodplain studies have been reviewed [18]. This paper synthesizes the literature and data available from field studies quantifying N and P removal within floodplains across the globe. Floodplains from rivers and streams of all sizes were included. Laboratory experiments, mesocosm studies, and modeling projects were not included.

The review started with a systematic search using literature database search engines (Google Scholar, Worldcat, Web of Science, etc.) and university library databases to find peer-reviewed publications and technical papers as well as reference lists within other papers. The searches used terms including "phosphorus," "nitrogen," "nutrients," and each nutrient form combined with terms including "floodplain," "riparian," "wetland," and synonyms of these terms. The initial list included over 200 peer-reviewed articles, technical papers, and graduate student papers. Of those sources, 40 provided new empirical data from in-field measurements (Figure 2). Those 40 sources measured NO_3^- removal in 29 floodplains and total or particulate P in 42 floodplains—10 of which were particulate P only.

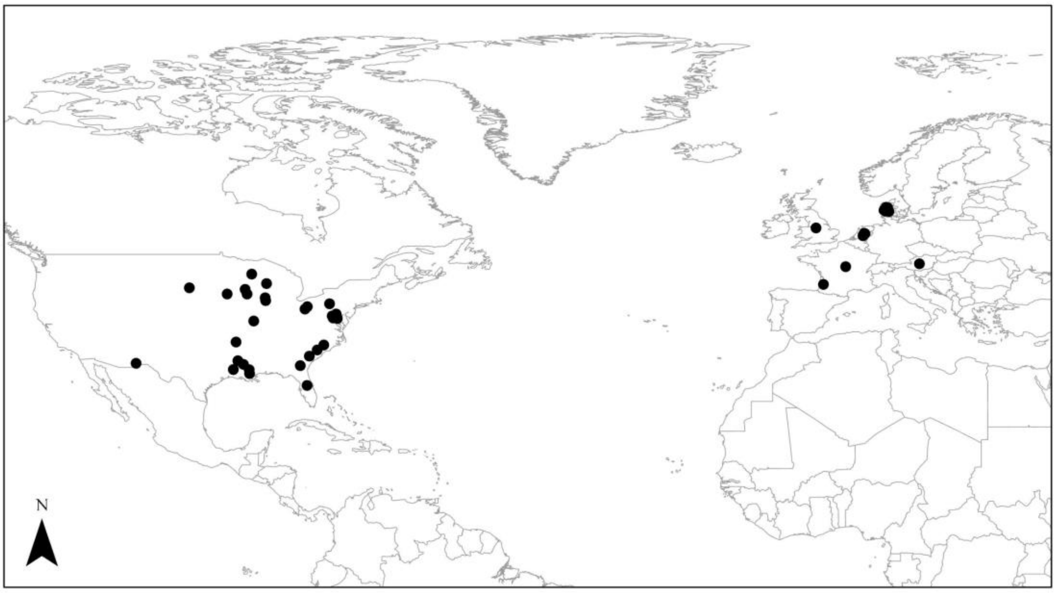

Figure 2. This is a map of the 40 floodplains included in the statistical review. Some reviewed studies included multiple floodplains within one publication.

The definition of a floodplain varied throughout the literature reviewed, but in this review an ecologically functional and reconnected floodplain is defined as a low-lying area adjacent to a river that becomes inundated regularly during high flows or floods, and it is capable of supporting plant and animal communities native to that region. High areas adjacent to rivers that are not flooded by the river in its current flow regime, known as terraces, were not included in this analysis. Connected floodplains containing unnatural surfaces void of vegetation, constructed treatment wetlands receiving no stream flow, and two-stage ditches were a few systems that were also excluded from this review since their connectivity was not fully restored or they were very small in areal extent. Restored oxbow wetlands and floodplains reconnected using large, riparian water pumps were included in the review.

1.4. Evaluating Nutrient Removal from Previous Studies

Nutrient removal studies occurring specifically in floodplains varied in the metrics and methods they used, thus complicating cross-study comparisons. The original goal was to collect data from floodplain studies for comparisons using the plug-flow area-based first-order model [46,47], a model that has become widely used to represent removal kinetics, but too few studies provided all the variables needed for the equation. The review therefore used simpler evaluations of the data. The studies most commonly expressed results of removal effectiveness as changes in concentration, mass balances, mass accumulations, percent removal, or denitrification measurements. The review of these results focused primarily on studies that summarized mass balances within floodplains (i.e., mass in versus mass out or mass removed). Studies measuring changes in concentration from upstream to downstream points in the main channel were excluded due to nutrient spiraling being complex and the difficulty in assuming reductions took place due to proximity to the adjacent floodplain. Some studies measured nutrient cycling in vegetation and the impact of plant communities on nutrient removal [18,48–52]. From each study, removal values and potential controlling factors were collected for statistical analyses.

Using the Spearman rank correlation test, this review compared the loads and removal rates of N and P among wetland sizes, nutrient loads, and hydraulic load. Statistical analyses included ANOVA (normal distribution) or the Mann–Whitney U test (non-normalized distribution) to compare categories of floodplains and their removal rates based on flooding frequency. When analyzing flooding frequencies, the floodplains were grouped into categories of permanent inundation or seasonal inundation. Permanent inundation did not mean the entire floodplain was impounded, but it meant there was standing water somewhere in the floodplain year-round. Seasonal inundation meant the floodplain dried at some time each year. Analyses were performed in Rstudio® and Microsoft® Excel [53].

1.5. Evaluation Considerations

1.5.1. Seasonal Variations and Climate

Most floodplains are inundated seasonally rather than the entire year. In line with this hydrological variation, studies varied in the time frame over which they presented their nutrient removal rates. Many results were presented as mass removed per day, but this does not explain how much total mass those floodplains could remove annually. Therefore, the daily reductions from a study were converted to annual reductions by multiplying the daily reductions by the average number of days per year of inundation reported in that respective study. If a study did not mention the number of days of inundation per year, it was not included in these comparisons. This conversion to annual retention also aided in accounting for varying climates. All studies included in the statistical evaluation were from the northern hemisphere, but some of the floodplains were located far enough south to remain unfrozen year-round. In these cases, the floodplains were usually inundated for more days each year, but the mass retentions would still be multiplied by the number of days of inundation per year reported in the study.

1.5.2. Nutrient Forms

Nitrogen and P both appear in multiple forms in the environment. However, NO_3^- was the most prevalent form of N in removal studies, and TP and particulate P were the most common forms of P reported. Although the other forms of these nutrients are important to study as part of their cycles in floodplains (i.e., NH_4^+, organic N, and SRP), they were not evaluated in this review because there were too few studies on those nutrients. While TP includes SRP, TP and particulate P were analyzed together in this review due to the majority of TP being particulate and removal being through deposition.

1.5.3. Size

There were large variations in sizes of floodplains studied. Larger rivers typically have larger floodplains [54]. Although floodplain-to-watershed area ratios would have been helpful for assessing nutrient removal efficiency, the watershed area was seldom mentioned. In order to reconcile the variation in floodplain size, the mass removal rates were divided by the floodplain area.

2. Nutrient Removal Results

Mass reductions in NO_3^--N in floodplains ranged from 2.35 to 962 kg-N ha^{-1} year^{-1}, with a mean reduction of 200 (SD = 198) kg-N ha^{-1} year^{-1}. Total P reductions ranged from a net release of 14.6 to net retention of 130 kg-P ha^{-1} year^{-1}, with a mean retention of 21.0 (SD = 31.4) kg-P ha^{-1} year^{-1}. Only 1 of the 41 floodplains in the studies had a net release of P rather than net retention (Table 1). The average concentration of NO_3^--N entering floodplains in 14 studies was 2.3 mg L^{-1} (SD = 1.7).

Table 1. Mass removal of nutrients in floodplains. Annual mass reductions in nitrate-N (NO_3^--N) and total or particulate phosphorus (TP or particulate P) loads in floodplains from literature reviewed.

	Load Reduction				
	25th (kg ha^{-1} year^{-1})	75th (kg ha^{-1} year^{-1})	Mean (kg ha^{-1} year^{-1})	Median (kg ha^{-1} year^{-1})	N
NO_3^--N	77.1	260	200	137	28
TP or Particulate P	2.58	22.9	21.0	8.99	41

As the hydraulic load increased, the mass removal of NO_3^- also increased (y = 5.6568x + 79.61, ρ = 0.79, p = 0.004, Figure 3) but the percent removal of NO_3^- decreased, i.e., they were inversely related (y = −8.255ln(x) + 84.542, ρ = −0.76, p = 0.007, Figure 4). For TP, there was a positive relationship between the hydraulic load (m year^{-1}) and TP mass removal (kg ha^{-1} year^{-1}; y = 5.8913ln(x) + 4.8823, ρ = 0.75, p = 0.003, Figure 5).

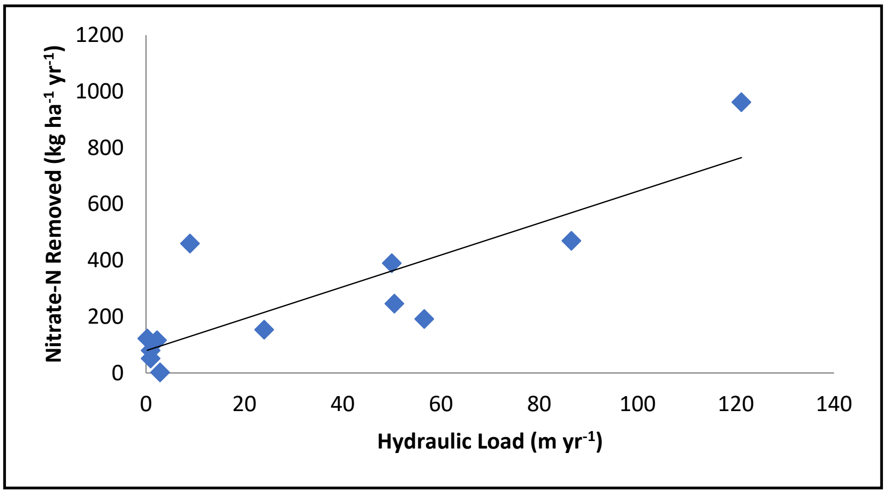

Figure 3. Relationship between the hydraulic load entering each floodplain and nitrate-N (NO_3^--N) mass removed (y = 5.6568x + 79.61, ρ = 0.79, p = 0.004).

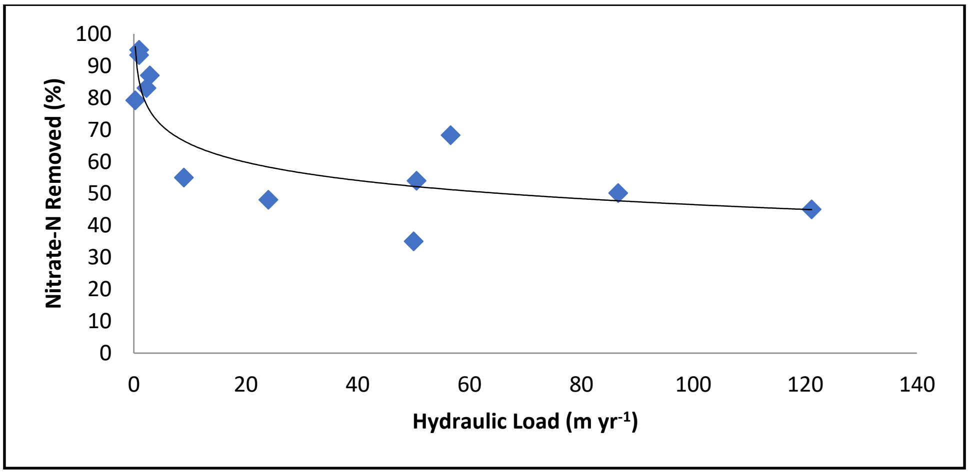

Figure 4. Relationship between the hydraulic load entering each floodplain and percent of nitrate-N (NO_3^--N) removed (y = −8.255ln(x) + 84.542, ρ = −0.76, p = 0.007).

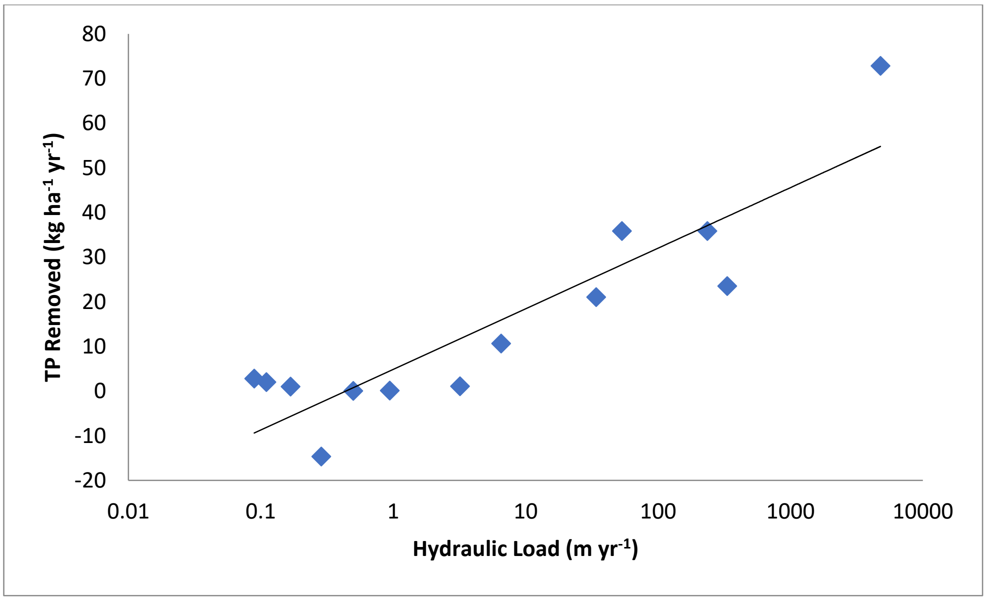

Figure 5. Relationship between the hydraulic load entering each floodplain and total phosphorus (TP) mass removed (y = 5.8913ln(x) + 4.8823, $\rho = 0.75$, $p = 0.003$).

In this review, there was a significant difference ($p < 0.001$) in NO_3^--N removal between floodplains with a permanent inundation landscape feature and those which dried completely for some time each year (Figure 6). The mean NO_3^--N removal in floodplains with permanent inundation somewhere on the floodplain was 312 kg-N ha^{-1} year^{-1} (SD = 267, n = 15), whereas removal in floodplains which eventually dry each year removed an average of 43.2 kg-N ha^{-1} year^{-1} (SD = 41, n = 7).

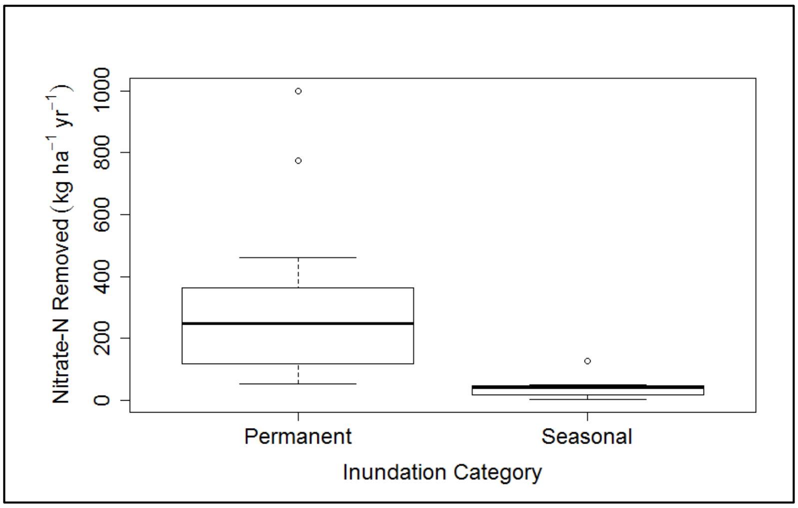

Figure 6. Nitrate-N (NO_3^--N) mass reductions in floodplains with areas of permanent inundation and floodplains with seasonal inundation. Lines from bottom to top: minimum, Q1, median, Q3, and maximum. Points: outliers.

Although there was no significant relationship between the inflow concentration and retention in this review, mass removal increased as the NO_3^--N load into the floodplain increased (y = 0.2946x + 115.67, $\rho = 0.70$, $p = 0.004$; Figure 7). The mean percent reduction in NO_3^--N in floodplains in this review was 64.2% (SD = 19.7%), and the mean percent reduction in TP in floodplains was 26.5% (SD = 24.8%; Table 2). There was a negative relationship between NO_3^--N load and percent removal. As the NO_3^--N load increased, the percent removal decreased exponentially

($y = 68.75^{e^{-2 \times 10-4\ x}}$, ρ = −0.66, p = 0.009; Figure 8). For TP, there was no correlation between the TP input load and percent removal, but the mass removed did increase as the input load increased (y = 0.0326x + 11.936, ρ = 0.63, p = 0.01; Figure 9).

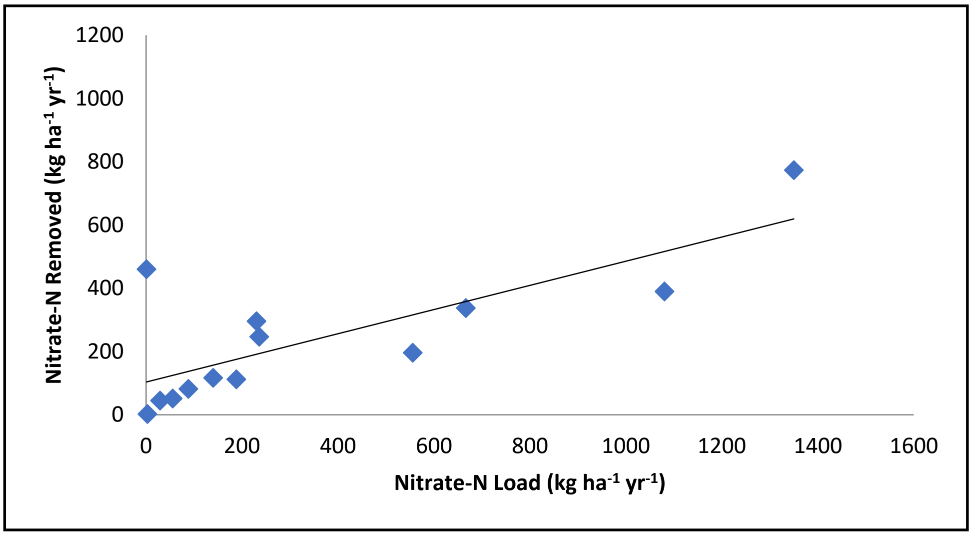

Figure 7. Increase in nitrate-N (NO_3^--N) mass removed as the NO_3^--N load into the floodplain increased (y = 0.2946x + 115.67, ρ = 0.70, p = 0.004).

Table 2. Percent removal of nutrients in floodplains. Percent reductions in nitrate-N (NO_3^--N) and total or particulate phosphorus (TP or particulate P) loads in floodplains from literature reviewed.

	Load Reduction				
	25th (%)	**75th (%)**	**Mean (%)**	**Median (%)**	**N**
NO_3^--N	50.0	79.1	64.2	62.7	21
TP or Particulate P	6.0	43.5	26.5	13.0	21

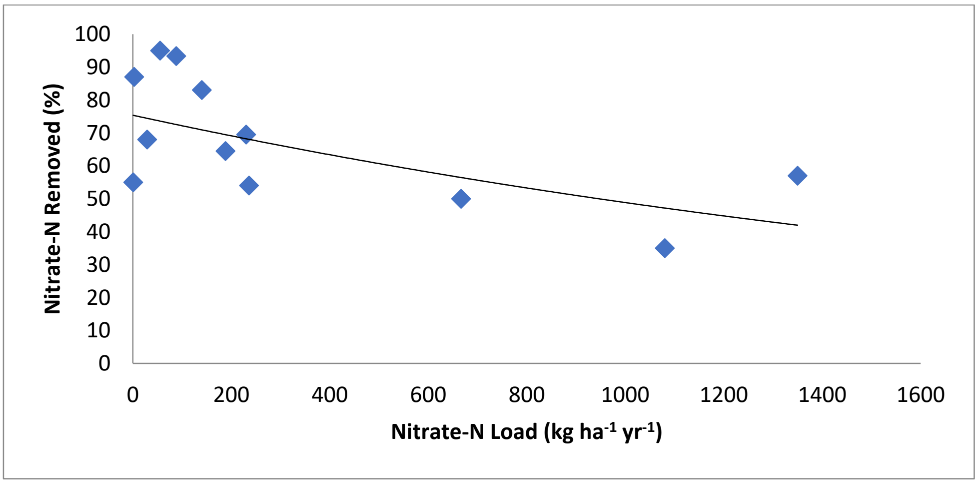

Figure 8. Decrease in the percent removal of nitrate-N (NO_3^--N) mass entering each floodplain as the load of NO_3^--N increased ($y = 68.75^{e^{-2 \times 10-4\ x}}$, ρ = −0.66, p = 0.009).

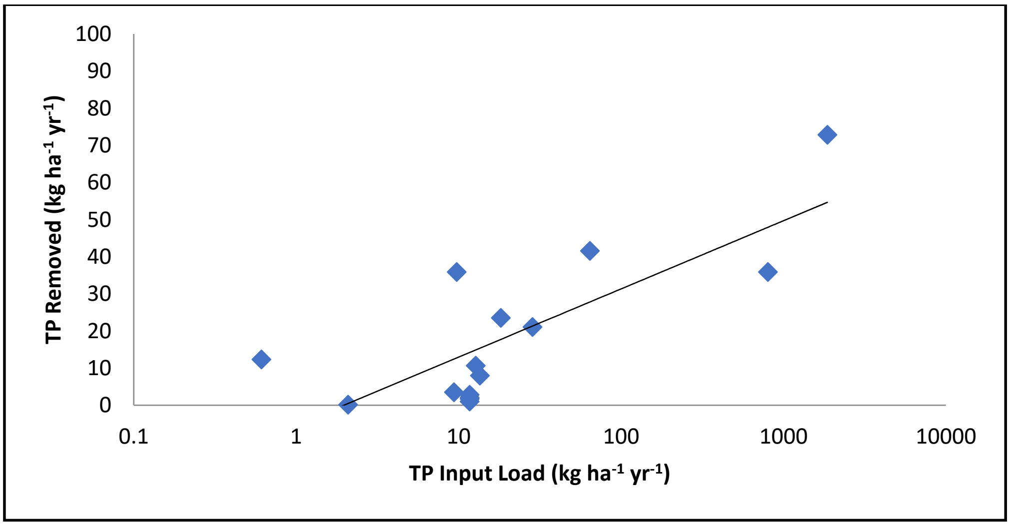

Figure 9. Increase in total phosphorus (TP) mass removed as the TP load increased (y = 0.0326x + 11.936, ρ = 0.63, *p* = 0.01).

Vegetation harvest was not a common practice for removing N from floodplains because it removed such a small portion of the N load. There were, however, some studies about harvesting P in floodplain vegetation. From the 8 studies that included vegetation harvest, the mean P removal was 8.8 kg-P ha^{-1} $year^{-1}$ in the floodplain (Table 3). With the mean TP load in the studies reviewed being 15.8 kg-P ha^{-1} $year^{-1}$ (SD = 16.1, 2 outliers removed), this could be a significant portion of the P load, but vegetation has a limit for how much it assimilates into tissue. There are many other biomass studies in constructed or highly managed wetlands, and two of those measured P removal exceeding 15 kg-P ha^{-1} $year^{-1}$ [55] and 20–60 kg-P ha^{-1} $year^{-1}$ [56] by harvesting *Typha* sp. stands. The site with the greatest P assimilation in floodplains included alfalfa, switchgrass, and cottonwood trees that were harvested every four years [48].

Table 3. Mean phosphorus mass removed from floodplains through vegetation harvest. Ranges reflect the range of means from multiple floodplains in the study listed.

Source	Phosphorus Harvested (kg-P ha^{-1} $year^{-1}$)	Plant Community Type
[18]	4.5–15	Reed marshes
[48]	25	Alfalfa, switchgrass, and cottonwood
[49]	3.9	Reeds, grasses, and sedges
[50]	11	Sedge meadow
[51]	3.3–19	Sedge meadow
[52]	3.8	Riparian forest
[57]	3–5	Riparian forest
[58]	1.7	Riparian forest

3. Discussion

Results from this review have major implications for floodplain reconnection initiatives, nutrient reduction strategies, and floodplain restoration designs. Restored and reconnected floodplains can remove significant masses of nutrients from rivers and streams. When properly restored, these systems also provide habitat for fish and wildlife, floodwater storage, recreation, and other benefits. The NO_3^- removal results were much lower in this review than those found by Dee et al. [59], but P removal estimates in this review were similar. Some of the key findings from this review can be described in the following categories.

3.1. Hydrology

Reconnecting floodplains to rivers and designing them with the correct flow regime is key for properly restoring the function of floodplains. Hydraulic load and water depth play significant roles in nutrient removal effectiveness in wetlands and floodplains, and there are flow rate recommendations for effective treatment wetlands [27,60]. Because nutrient removal increases as hydraulic load increases, restoring floodplains with greater connectivity to the river could prove a more effective investment for floodplain reconnection projects. However, that effectiveness decreases as the hydraulic load becomes too great. Optimal hydraulic load from floodplains in this review was roughly less than 20 m $year^{-1}$ for NO_3^- removal and less than 50 m $year^{-1}$ for P removal. While the volume of water entering the floodplain was an important factor for nutrient removal, the duration of inundation was also important.

Floodplains with year-round inundation somewhere on the floodplain (i.e., restored wetlands within the floodplain) removed significantly more N each year than floodplains that would dry completely for any period of time each year. This follows the expectations that flow regimes will impact nutrient removal effectiveness, but the important implication is that flooding lawns, crop fields, or other areas as temporary floodways would not be as effective at removing nutrients as permanently restoring and reconnecting floodplains. Restoring ecological functionality of floodplains would be more effective at nutrient removal, but restoring some permanently inundated wetlands would provide even more benefit. These results could be related to permanent wetlands containing sustained denitrifying microbe populations in the wet areas, organic-rich soils, or fringes of the wetlands [61–66]. Some have raised concerns about connecting floodplains on small-order streams because flood duration is shorter on smaller streams and therefore may not provide enough time for impactful nutrient removal or may even lead to a greater release of nutrients [67]. However, the floodplains on many small-order streams may be lacking a permanent wetland due to channelization and land use change along stream buffers. Floodplain reconnections and restorations along these smaller streams may require a permanently inundated or saturated wetland restoration component for the floodplain to effectively remove nutrients.

3.2. Microbes

In research investigating the role of microbes in floodplains, Argiroff et al. [61] measured a greater genetic potential for denitrification in floodplains that were more frequently flooded than those that were only occasionally flooded. Hernandez and Mitsch [68] measured greater denitrification in permanently flooded marshes than in occasionally flooded areas. Tomasek, Staley, et al. [62] concluded that more frequent inundation is likely to lead to greater denitrification and greater denitrifying microbe abundance. However, a higher population of denitrifying bacteria does not always lead to a correlatively higher denitrification rate, and sometimes the greatest denitrifier gene abundance is in the littoral zone or periodically inundated area rather than in the main stream channel or permanently inundated sediments where the greatest denitrifying potential was measured [63–65]. It is possible that these fringe habitats with fluctuating wet and dry conditions could be denitrification hot spots where denitrifying microbes are most active after a flood pulse [66]. Restoring more areas of permanent inundation in floodplains would create more fringe areas in return, especially if there is a high ratio of fringe to pooled areas.

More topographic and geomorphic diversity in the floodplain could capture the greater denitrification potential found in permanently inundated areas and enhance populations of denitrifying microbes in more fringe areas. It is important, however, to avoid creating permanent pools that have limited fringe zones and are too deep. These deep pools can limit vegetative growth, the organic matter provided by the vegetation, and the bacteria dependent on organic matter [61]. Furthermore, areas of inundation should have slow flow through them to avoid flushing out too much organic carbon and microbes attached to the sediment [69]. Residence times in the floodplain should be at least five days to maximize denitrification [27]. Restoring floodplains with swale wetlands in them that sit below the water table coupled with ridges that remain above the water table may therefore help to maximize denitrification.

It is also important to design floodplains so the majority of sedimentation occurring in the 50 m closest to the river does not fill swales that were meant to be permanent denitrification wetlands [70,71]. However, creating ridges that will slow down the flow and allow for better sedimentation will provide better P retention. While vegetation can remove a large portion of P from the soil, sedimentation and the subsequent burial of particulate P is likely the best means of permanently retaining large loads of P in the floodplain [72].

3.3. Vegetation

While sedimentation and accretion are likely the best means for retaining P in floodplains, vegetation will provide the organic matter in the soil to drive denitrification and assimilate some of the nutrients into plant tissue. Vegetation type varied among the studies, so this review was not able to determine the best community for nutrient removal. Therefore, the best plant community for floodplain restorations should depend on location and historic communities more than optimal nutrient removal at this time. However, as explained more thoroughly by Dee et al. [59], each community type may establish more or less successfully based on the flood regime and be more or less successful at removing nutrients [72]. Herbaceous communities with dense ground cover, like grasses and rushes, are better at slowing flow velocity, increasing retention time, promoting sedimentation, and burying P [19,73]. Forested floodplains are likely to be better at assimilating nutrients more permanently, establishing growth better in less-frequently flooded areas, tolerating larger floods and sediment loads, and removing more NO_3^- from groundwater [58,74,75].

Harvesting vegetation can have a positive impact on removing P from the floodplain, but there are still unknowns about its impact on the plant community and if there can be a market for selling harvested vegetation. If the plant community is already overwhelmed by invasive species, such as *Typha* sp., a harvesting regime could improve diversity [76,77]. Although highly productive species such as *Typha* sp. were highly effective at assimilating P, they are usually invasive in North America and would be a detriment to the native biodiversity if planted specifically for nutrient removal. If vegetation is harvested, there still needs to be an investment in the proper harvesting equipment, someone who wants to purchase the biomass, and good timing to be able to access the floodplain when it is not flooded. If vegetation is not harvested, some of it can decompose and release the nutrients it assimilated the previous season. However, the organic material is oftentimes accreted into the soil layers where more sediment deposits on top and buries it into storage [72].

3.4. Nutrient Loading

Some of the most commonly referenced and most effective NO_3^- removal wetlands in the Midwest United States are the Iowa Conservation Reserve Enhancement Program (CREP) wetlands [78–80]. When calculating the mass of N removed for each hectare converted to wetland pooled area and buffer of 79 CREP wetlands, they removed an average of 450–500 kg-N ha^{-1} $year^{-1}$ through 2016 [78]. Of the 16 sites consistently monitored, they were significantly more effective than all the floodplains in this study (mean = 224 kg-N ha^{-1} $year^{-1}$, $p = 0.002$), but they were not significantly different than floodplains that had permanent inundation (mean = 312 kg-N ha^{-1} $year^{-1}$, $p = 0.1$). Some smaller NO_3^- treatment wetlands in Illinois and Minnesota ranged from an average of 166 to 619 kg-N ha^{-1} $year^{-1}$ [81,82]. This difference between load removals is possibly related to the incoming NO_3^--N concentration and load to each wetland or floodplain. The average concentration entering the Iowa CREP wetlands was 14 mg L^{-1}, the Illinois wetlands was 8.5–13.0 mg L^{-1}, and the Minnesota wetland was 15.6 mg L^{-1}. Concentrations of NO_3^--N in tile drainage discharge can be very high, oftentimes exceeding 20 mg-N L^{-1} [24,83,84] or even 40 mg-N L^{-1} in some areas [27]. The mean concentration entering the floodplains in this review was 2.3 mg L^{-1} (SD = 1.7). The lack of a correlation between inflowing concentration in floodplains and retention may have been due to a limited range of and difference among inflow concentrations, but there was a linear correlation between NO_3^--N load and NO_3^--N removal effectiveness. As the NO_3^--N load increased, the mass removed increased, but the percent removal decreased. Similarly for TP, as the load increased, the mass removal increased. In the same way, as the hydraulic load increased, so did the masses of N and P removal increase while percent removal decreased. Therefore, prioritizing floodplain reconnections along rivers or streams with higher concentrations may lead to greater masses of N and P removed without sacrificing percent removal effectiveness. Although P removal increased as load increased, floodplains had a maximum capacity for P removal, but floodplains did not lose nutrients as the flow increased.

3.5. Nutrient Release from Floodplains

Due to the dynamic nature of floodplains, most processes of nutrient storage are not permanent. However, contrary to any concerns that may arise about floodplains releasing nutrients or scouring currents resuspending them during the next flood, this release did not exceed retention in the literature reviewed. Only one TP study measured a net release of P [85], and one study measured an increase in both nutrients in the river channel next to a floodplain during some seasons of the year [86]. However, SRP contributes disproportionally to downstream algal blooms and is an important consideration in some situations [32]. Over long time-scales sediment and nutrients are returned to the river via the process of lateral stream migration and streambank erosion which removes part of the floodplain each year [30,40]. While sediment and nutrient resuspension from the floodplain are a part of the nutrient cycling process, it was not a common observation to see release be greater than retention in these studies.

3.6. Future Research

There need to be more studies on the nutrient removal effectiveness of floodplains. This review is necessary to provide initial estimates of floodplain removal effectiveness and better understand some of the trends and factors impacting that effectiveness, but the calculations are based on fewer than 50 studies. With the limited number of studies included in the statistical analyses, some trends could be influenced by outliers or could change if more values are included. Because P removal in floodplains is likely mostly from deposition [30], particulate P and TP were grouped together in this analysis, but they should be separated in future studies in order to better distinguish particulate and soluble P removal. Furthermore, as results from more empirical studies become available about nutrient cycling in floodplains, a review using the areal rate constant would be beneficial. It has become a widely used model to represent nutrient removal kinetics, but not enough studies provided all the variables needed for the equation at this time [46,47]. There also need to be more studies on the retention of other forms of N and P in floodplains. The most common forms studied, by far, were NO_3^- and total or particulate P. However, recent studies have revealed that NH_4^+ and ratios of nutrient forms to each other may play larger roles in influencing which algae are present in HABs and how toxic the bloom may be compared to understandings of HABs in decades past [1,13,14]. Furthermore, studies from only North America and Europe were included in this review due to the difficulty finding studies from elsewhere, written in English and meeting the selection criteria. More research is needed on nutrient removal in other parts of the globe, particularly tropical rivers in the southern hemisphere.

4. Conclusions

Hypoxic zones, HABs, and accelerated eutrophication continue to be problems in bodies of water across the globe as nutrient loads remain at high levels. While many efforts are underway to reduce those nutrient loads, especially from point sources and agriculture, more options need to become available and be proven effective. Restoring and reconnecting floodplains to rivers and streams throughout watersheds is a viable and cost-effective practice for removing nutrients. The following design considerations could improve removal effectiveness:

- Engineer the floodplain to optimize hydraulic load. Although more flow across the floodplain could lead to a greater total mass of nutrients removed, the floodplain will lose effectiveness, (the percent removal) as flow rates increase.
- Incorporate a permanently inundated wetland in the floodplain area to improve NO_3^--N removal.
- Ensure geomorphic diversity across the floodplain to increase both N and P removal due to improved microbial habitat for denitrification and more areas for sedimentation and accretion of P.
- Restore dense vegetation to improve nutrient removal by providing organic matter for denitrifying microbes and slow water flow for better sedimentation and accretion.
- Harvest vegetation from floodplains where feasible to aid in P removal, but caution should be taken due to the unknown impact on native plant communities.
- Restore floodplains along waterways with higher concentrations of nutrients to increase the load of nutrients into the floodplain. Limit flow to maximize nutrient removal.

Reconnecting floodplains to their respective rivers or streams and restoring their geomorphology and vegetation would greatly reduce nutrient loads. Floodplains are dynamic systems that provide the benefits of nutrient retention, water storage, fish and wildlife habitat, aquifer recharge, and recreation. As more empirical studies are conducted on nutrient removal in floodplains, especially on retention of other forms of N and P, more design considerations could be incorporated into these projects to achieve better nutrient removal results across the globe.

Author Contributions: Conceptualization, B.A.G. and O.D.; methodology, B.A.G.; software, B.A.G.; validation, B.A.G., O.D. and C.F.L.; formal analysis, B.A.G.; investigation, B.A.G.; resources, O.D.; data curation, B.A.G.; writing—original draft preparation, B.A.G.; writing—review and editing, O.D. and C.F.L.; visualization, B.A.G.; supervision, C.F.L.; project administration, B.A.G.; funding acquisition, O.D. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the Anthony A. Lapham Fellowship and the McKnight Foundation through American Rivers.

Acknowledgments: The Lapham family generously provided funding for a two-year post-graduate fellowship at American Rivers to better understand the impact of floodplain restoration on nutrient cycling in rivers. We especially thank Eileen Shader for her supervision and guidance throughout the fellowship, the Lapham family for the opportunity, and American Rivers for their support.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, or in the decision to publish the results.

References

1. Glibert, P.M.; Burford, M.A. Globally changing nutrient loads and harmful algal blooms: Recent advances, new paradigms, and continuuing challenges. *Oceanography* **2017**, *30*, 58–69. [CrossRef]
2. Galloway, J.N.; Cowling, E.B. Reactive nitrogen and the world: 200 years of change. *AMBIO J. Hum. Environ.* **2002**, *31*, 64–71. [CrossRef]
3. Howarth, R.W. Coastal nitrogen pollution: A review of sources and trends globally and regionally. *Harmful Algae* **2008**, *8*, 14–20. [CrossRef]
4. Goolsby, D.A.; Battaglin, W.A.; Lawrence, G.B.; Artz, R.S.; Aulenbach, B.T.; Hooper, R.P.; Keeney, D.R.; Stensland, G.J. *Flux and Sources of Nutrients in the Mississippi-Atchafalaya River Basin Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico*; National Centers for Coastal Ocean Science: Silver Spring, MD, USA, 1999.
5. Alexander, R.B.; Smith, R.A.; Schwarz, G.E.; Boyer, E.W.; Nolan, J.V.; Brakebill, J.W. Differences in Phosphorus and Nitrogen Delivery to The Gulf of Mexico from the Mississippi River Basin. *Environ. Sci. Technol.* **2008**, *42*, 822–830. [CrossRef] [PubMed]
6. Christianson, L.E.; Harmel, R.D.; Smith, D.; Williams, M.R.; King, K. Assessment and synthesis of 50 years of published drainage phosphorus losses. *J. Environ. Qual.* **2016**, *45*. [CrossRef] [PubMed]

7. King, K.W.; Williams, M.R.; Macrae, M.L.; Fausey, N.R.; Frankenberger, J.; Smith, D.R.; Kleinman, P.J.A.; Brown, L.C. Phosphorus transport in agricultural subsurface drainage: A review. *J. Environ. Qual.* **2015**, *44*, 467–485. [CrossRef] [PubMed]
8. Robertson, D.M.; Saad, D.A. SPARROW Models Used to Understand Nutrient Sources in the Mississippi/Atchafalaya River Basin. *J. Environ. Qual.* **2013**, *42*, 1422. [CrossRef]
9. Smith, D.R.; King, K.W.; Johnson, L.; Francesconi, W.; Richards, P.; Baker, D.; Sharpley, A.N. Surface runoff and tile drainage transport of phosphorus in the midwestern United States. *J. Environ. Qual.* **2015**, *44*, 495–502. [CrossRef]
10. Hardison, D.R.; Sunda, W.G.; Litaker, R.W.; Shea, D.; Tester, P.A. Nitrogen limitation increases brevetoxins in Karenia brevis (Dinophyceae): Implications for bloom toxicity. *J. Phycol.* **2012**, *48*, 844–858. [CrossRef]
11. Hardison, D.R.; Sunda, W.G.; Shea, D.; Litaker, R.W. Increased toxicity of Karenia brevis during phosphate limited growth: Ecological and evolutionary implications. *PLoS ONE* **2013**, *8*, e58545. [CrossRef]
12. Granéli, E.; Flynn, K.J. Chemical and physical factors influencing toxin content. In *Ecology of Harmful Algae*; Granéli, E., Turner, J.T., Eds.; Springer: Berlin/Heidelberg, Germany, 2005; pp. 229–241.
13. Wilhelm, C.; Büchel, C.; Fisahn, J.; Goss, R.; Jakob, T.; LaRoche, J.; Lavaud, J.; Lohr, M.; Riebesell, U.; Stehfest, K.; et al. The regulation of carbon and nutrient assimilation in diatoms is significantly different from green algae. *Protist* **2006**, *157*, 91–124. [CrossRef] [PubMed]
14. Glibert, P.M. Margalef revisited: A new phytoplankton mandala incorporating twelve dimensions including nutritional physiology. *Harmful Algae* **2016**, *55*, 25–30. [CrossRef] [PubMed]
15. Ward, M.H.; deKok, T.M.; Levallois, P.; Brender, J.; Gulis, G.; Nolan, B.T.; VanDerslice, J. Workgroup report: Drinking-water nitrate and health-Recent findings and research needs. *Environ. Health Perspect.* **2005**, *113*, 1607–1614. [CrossRef] [PubMed]
16. Committee on Environment and Natural Resources. *Scientific Assessment of Hypoxia in U.S. Coastal Waters*; Committee on Environment and Natural Resources: Washington, DC, USA, 2010.
17. Deacon, J.R.; Lee, C.J.; Norman, J.; Reutter, D. Nutrient and Pesticide Data Collected from the USGS National Water Quality Network and Previous Networks, 1963–2016. Available online: https://dx.doi.org/doi:10.5066/F7BK19GZ (accessed on 1 November 2019).
18. Hoffmann, C.C.; Kjaergaard, C.; Uusi-Kämppä, J.; Hansen, H.C.B.; Kronvang, B. Phosphorus Retention in Riparian Buffers: Review of Their Efficiency. *J. Environ. Qual.* **2009**, *38*, 1942–1955. [CrossRef]
19. Mayer, P.M.; Reynolds, S.K.; McCutchen, M.D.; Canfield, T.J. Meta-analysis of nitrogen removal in riparian buffers. *J. Environ. Qual.* **2007**, *36*, 1172–1180. [CrossRef]
20. Feyereisen, G.W.; Wilson, B.N.; Sands, G.R.; Strock, J.S.; Porter, P.M. Potential for a rye cover crop to reduce nitrate loss in southwestern Minnesota. *Agron. J.* **2006**, *98*, 1416. [CrossRef]
21. Kaspar, T.C.; Jaynes, D.B.; Parkin, T.B.; Moorman, T.B.; Singer, J.W. Effectiveness of oat and rye cover crops in reducing nitrate losses in drainage water. *Agric. Water Manag.* **2012**, *110*, 25–33. [CrossRef]
22. Gassman, P.W.; Osei, E.; Saleh, A.; Rodecap, J.; Norvell, S.; Williams, J. Alternative practices for sediment and nutrient loss control on livestock farms in northeast Iowa. *Agric. Ecosyst. Environ.* **2006**, *117*, 135–144. [CrossRef]
23. Christianson, L.E.; Harmel, R.D. 4R water quality impacts: An assessment and synthesis of forty years of drainage nitrogen losses. *J. Environ. Qual.* **2015**, *44*, 1852–1860. [CrossRef]
24. Crumpton, W.; Stenback, G.; Miller, B.; Helmers, M. Potential benefits of wetland filters for tile drainage systems: Impact on nitrate loads to Mississippi River subbasins. In *Agricultural and Biosystems Engineering Technical Reports White Papers*; Iowa State University: Ames, IA, USA, 2006; Volume 8.
25. Addy, K.; Gold, A.J.; Christianson, L.E.; David, M.B.; Schipper, L.A.; Ratigan, N.A. Denitrifying bioreactors for nitrate removal: A meta-analysis. *J. Environ. Qual.* **2016**, *45*, 873–881. [CrossRef]
26. Sparks, R.E. Need for ecosystem management of large rivers and their floodplains. *Bioscience* **1995**, *45*, 168–182. [CrossRef]
27. Kadlec, R.H.; Wallace, S.D. *Treatment Wetlands*, 2nd ed.; CRC Press: Boca Raton, FL, USA, 2008; ISBN 9781566705264.
28. Xue, Y.; David, M.B.; Gentry, L.E.; Mulvaney, R.L.; Kovacic, D.A.; Lindau, C.W. In situ measurements of denitrification in constructed wetlands. *J. Environ. Qual.* **1999**, *28*, 263–269. [CrossRef]
29. Philippot, L.; Hallin, S.; Schloter, M. Ecology of denitrifying prokaryotes in agricultural soil. *Adv. Agron.* **2007**, *96*, 249–305. [CrossRef]

30. Knighton, D. *Fluvial Forms and Processes: A New Perspective*; Routledge: England, UK, 2014.
31. Walling, D.E.; He, Q.; Blake, W.H. River flood plains as phosphorus sinks. *IAHS Publ. Int. Assoc. Hydrol. Sci.* **2000**, *263*, 211–218.
32. Jarvie, H.P.; Johnson, L.T.; Sharpley, A.N.; Smith, D.R.; Baker, D.B.; Bruulsema, T.W.; Confesor, R. Increased soluble phosphorus loads to Lake Erie: Unintended consequences of conservation practices? *J. Environ. Qual.* **2017**, *46*, 123–132. [CrossRef]
33. Batjes, N.H. Global Distribution of Soil Phosphorus Retention Potential. In *ISRIC-World Soil Inf., No. 2011/0*; ISRIC: Wageningen, The Netherlands, 2011.
34. Nelson, J.C.; Redmond, A.; Sparks, R.E. Impacts of settlement on floodplain vegetation at the confluence of the Illinois and Mississippi Rivers. *Trans. Ill. State Acad. Sci.* **1994**, *87*, 117–133.
35. Llewellyn, D.W.; Shaffer, G.P.; Craig, N.J.; Creasman, L.; Pashley, D.; Mark, S.; Brown, C. A decision-support system for prioritizing restoration sites on the Mississippi River alluvial plain. *Conserv. Biol.* **1996**, *10*, 1446–1455. [CrossRef]
36. Delaney, R.L.; Craig, M.R. *Longitudinal Changes in Mississippi River Floodplain Structure*; U.S. Geological Survey: Onalaska, WI, USA, 1997.
37. Tockner, K.; Stanford, J.A. Riverine flood plains: Present state and future trends. *Environ. Conserv.* **2002**, *29*, 308–330. [CrossRef]
38. Galloway, G.E. Learning from the Mississippi flood of 1993: Impacts, management issues, and areas for research. In Proceedings of the U.S.—Italy Research Workshop on the Hydrometeorology, Impacts, and Management of Extreme Floods, Perugia, Italy, 13–17 November 1995; pp. 1–29.
39. Magilligan, F.J. Sedimentology of a fine-grained aggrading floodplain. *Geomorphology* **1992**, *4*, 393–408. [CrossRef]
40. Lenhart, C.F.; Titov, M.L.; Ulrich, J.S.; Nieber, J.L.; Suppes, B.J. The role of hydrologic alteration and riparian vegetation dynamics in channel evolution along the lower Minnesota River. *Trans. ASABE* **2013**, *56*, 549–561. [CrossRef]
41. Brady, N.C.; Weil, R.R. *The Nature and Properties of Soils*; Pearson Prentice Hall: Upper Saddle River, NJ, USA, 2007.
42. Vymazal, J. The use of hybrid constructed wetlands for wastewater treatment with special attention to nitrogen removal: A review of a recent development. *Water Res.* **2013**, *47*, 4795–4811. [CrossRef] [PubMed]
43. Kadlec, R. Large constructed wetlands for phosphorus control: A review. *Water* **2016**, *8*, 243. [CrossRef]
44. Vymazal, J. Removal of nutrients in various types of constructed wetlands. *Sci. Total Environ.* **2007**, *380*, 48–65. [CrossRef] [PubMed]
45. Crumpton, W.; Van Der Valk, A.; Hoyer, W.; Osterberg, D.; Van Der Valk, A.G. *Wetland Restoration in Iowa: Challenges and Opportunities*; The Iowa Policy Project: Iowa City, IA, USA, 2012.
46. Rozema, E.R.; VanderZaag, A.C.; Wood, J.D.; Drizo, A.; Zheng, Y.; Madani, A.; Gordon, R.J. Constructed wetlands for agricultural wastewater treatment in northeastern North America: A review. *Water* **2016**, *8*, 14. [CrossRef]
47. Newcomer Johnson, T.A.; Kaushal, S.S.; Mayer, P.M.; Smith, R.M.; Sivirichi, G.M. Nutrient retention in restored streams and rivers: A global review and synthesis. *Water* **2016**, *8*, 28. [CrossRef]
48. Kelly, J.M.; Kovar, J.L.; Sokolowsky, R.; Moorman, T.B. Phosphorus uptake during four years by different vegetative cover types in a riparian buffer. *Nutr. Cycl. Agroecosyst.* **2007**, *78*, 239–251. [CrossRef]
49. Koerselman, W.; Bakker, S.A.; Blom, M. Nitrogen, phosphorus and potassium budgets for two small fens surrounded by heavily fertilized pastures. *J. Ecol.* **1990**, *78*, 428. [CrossRef]
50. Hoffmann, C.C.; Dahl, M.; Kamp-Nielsen, L.; Stryhn, H. *Vandog Stofbalance i en Natureng*; Miljøstyrelsen: Copenhagen, Denmark, 1993.
51. Venterink, H.O.; Pieterse, N.M.; Belgers, J.D.M.; Wassen, M.J.; de Ruiter, P.C. N, P, and K budgets along nutrient availability and productivity gradients in wetlands. *Ecol. Appl.* **2002**, *12*, 1010–1026. [CrossRef]
52. Fail, J.L.; Haines, B.L.; Todd, R.L. Riparian forest communities and their role in nutrient conservation in an agricultural watershed. *Am. J. Altern. Agric.* **1987**, *2*, 114. [CrossRef]
53. *R Core Team R: A Language and Environment for Statistical Computing*; R Founding for Statistical Computing: Vienna, Austria, 2013.

54. Lauer, J.W.; Echterling, C.; Lenhart, C.; Belmont, P.; Rausch, R. Air-photo based change in channel width in the Minnesota River basin: Modes of adjustment and implications for sediment budget. *Geomorphology* **2017**, *297*, 170–184. [CrossRef]
55. Dubbe, D.R.; Garver, E.G.; Pratt, D.C. Production of cattail (Typha spp.) biomass in Minnesota, USA. *Biomass* **1988**, *17*, 79–104. [CrossRef]
56. Grosshans, R.; Grieger, L.; Ackerman, J.; Gauthier, S.; Swystun, K.; Gass, P.; Roy, D. *Cattail Biomass in a Watershed-Based Bioeconomy: Commercial-Scale Harvesting and Processing for Nutrient Capture, Biocarbon and High-Value Bioproducts*; International Institute for Sustainable Development: Winnipeg, MB, Canada, 2014.
57. Correll, D.L.; Weller, D.E. Factors limiting processes in freshwater wetlands: An agricultural primary stream riparian forest. In *Freshwater Wetlands and Wildlife*; Sharitz, R.R., Gibbons, J.W., Eds.; United States Department of Energy: Oak Ridge, TN, USA, 1989.
58. Peterjohn, W.T.; Correll, D.L. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* **1984**, *65*, 1466–1475. [CrossRef]
59. Dee, M.M.; Tank, J.L.; Mahl, U.H.; Powers, S.M. *Estimating the Impact of Floodplain Restoration on Nutrient and Sediment Export from the Wabash River Watershed: A Historical Perspective*; Indiana Water Resources Research Center: Marsteller St., IN, USA, 2014.
60. USDA NRCS Part 651 agricultural waste management system component design. In *National Engineering Handbook*; USDA: Washington, DC, USA, 2009; p. 216.
61. Argiroff, W.A.; Zak, D.R.; Lanser, C.M.; Wiley, M.J. Microbial community functional potential and composition are shaped by hydrologic connectivity in riverine floodplain soils. *Microb. Ecol.* **2017**, *73*, 630–644. [CrossRef] [PubMed]
62. Tomasek, A.; Staley, C.; Wang, P.; Kaiser, T.; Lurndahl, N.; Kozarek, J.L.; Hondzo, M.; Sadowsky, M.J. Increased denitrification rates associated with shifts in prokaryotic community composition caused by varying hydrologic connectivity. *Front. Microbiol.* **2017**, *8*, 2304. [CrossRef] [PubMed]
63. Ligi, T.; Truu, M.; Truu, J.; Nõlvak, H.; Kaasik, A.; Mitsch, W.J.; Mander, Ü. Effects of soil chemical characteristics and water regime on denitrification genes (nirS, nirK, and nosZ) abundances in a created riverine wetland complex. *Ecol. Eng.* **2013**, *72*, 47–55. [CrossRef]
64. Wang, C.; Zhu, G.; Wang, Y.; Wang, S.; Yin, C. Nitrous oxide reductase gene (nosZ) and N2O reduction along the littoral gradient of a eutrophic freshwater lake. *J. Environ. Sci.* **2013**, *25*, 44–52. [CrossRef]
65. Tomasek, A.; Kozarek, J.L.; Hondzo, M.; Lurndahl, N.; Sadowsky, M.J.; Wang, P.; Staley, C. Environmental drivers of denitrification rates and denitrifying gene abundances in channels and riparian areas. *Water Resour. Res.* **2017**. [CrossRef]
66. Tomasek, A.A.; Hondzo, M.; Kozarek, J.L.; Staley, C.; Wang, P.; Lurndahl, N.; Sadowsky, M.J. Intermittent flooding of organic-rich soil promotes the formation of denitrification hot moments and hot spots. *Ecosphere* **2019**, *10*, e02549. [CrossRef]
67. Scott, D.T.; Gomez-Velez, J.D.; Jones, C.N.; Harvey, J.W. Floodplain inundation spectrum across the United States. *Nat. Commun.* **2019**, *10*, 1–8. [CrossRef]
68. Hernandez, M.E.; Mitsch, W.J. Denitrification in created riverine wetlands: Influence of hydrology and season. *Ecol. Eng.* **2007**, *30*, 78–88. [CrossRef]
69. Tomasek, A.A.; Barman, T.D.; Wang, P.; Kozarek, J.L.; Staley, C.; Sadowsky, M.J.; Hondzo, M. The effects of turbulence and carbon amendments on nitrate uptake and microbial gene abundances in stream sediment. *J. Geophys. Res. Biogeosci.* **2018**, *123*, 1289–1301. [CrossRef]
70. Brunet, R.C.; Pinay, G.; Gazelle, F.; Roques, L. Role of the floodplain and riparian zone in suspended matter and nitrogen retention in the adour river, south-west France. *Regul. Rivers Res. Manag.* **1994**, *9*, 55–63. [CrossRef]
71. Kronvang, B.; Andersen, I.K.; Hoffmann, C.C.; Pedersen, M.L.; Ovesen, N.B.; Andersen, H.E. Water exchange and deposition of sediment and phosphorus during inundation of natural and restored lowland floodplains. *Water. Air. Soil Pollut.* **2007**, *181*, 115–121. [CrossRef]
72. Noe, G.B.; Hupp, C.R. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. *Ecol. Appl.* **2005**, *15*, 1178–1190. [CrossRef]
73. Osborne, L.L.; Kovacic, D.A. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshw. Biol.* **1993**, *29*, 243–258. [CrossRef]

74. Lowrance, R.; Todd, R.; Fail, J.; Hendrickson, O.; Leonard, R.; Asmussen, L. Riparian forests as nutrient filters in agricultural watersheds. *Bioscience* **1984**, *34*, 374–377. [CrossRef]
75. Jordan, T.; Thomas, E.; Correl, D.; Weller, D. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *J. Environ. Qual.* **1993**, *22*, 467–473. [CrossRef]
76. Zedler, J.B. Feedbacks that might sustain natural, invaded and restored states in herbaceous wetlands. In *New Models for Ecosystem Dynamics and Restoration*; Hobbs, R.J., Suding, K.N., Eds.; Society for Ecological Restoration International: Washington, DC, USA, 2009; pp. 236–258.
77. Lishawa, S.C.; Lawrence, B.A.; Albert, D.A.; Tuchman, N.C. Biomass harvest of invasive Typha promotes plant diversity in a Great Lakes coastal wetland. *Restor. Ecol.* **2015**, *23*, 228–237. [CrossRef]
78. Crumpton, W.; Stenback, G. Iowa Conservation Reserve Enhancement Program. In *Annual Performance Report*; CREP: London, UK, 2016; Volume 16.
79. Iovanna, R.; Hyberg, S.; Crumpton, W. Treatment wetlands: Cost-effective practice for intercepting nitrate before it reaches and adversely impacts surface waters. *J. Soil Water Conserv.* **2008**, *63*, 14A–15A. [CrossRef]
80. Hyberg, S.; Iovanna, R.; Crumpton, W.; Richmond, S. The cost effectiveness of wetlands designed and sited for nitrate removal: The effect of increased efficiency, rising easement costs, and lower interest rates. *J. Soil Water Conserv.* **2015**, *70*, 30A–32A. [CrossRef]
81. Kovacic, D.A.; David, M.B.; Gentry, L.E.; Starks, K.M.; Cooke, R.A. Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. *J. Environ. Qual.* **2000**, *29*, 1262. [CrossRef]
82. Lenhart, C.; Gordon, B.; Gamble, J.; Current, D.; Ross, N.; Herring, L.; Nieber, J.; Peterson, H. Design and hydrologic performance of a tile drainage treatment wetland in Minnesota, USA. *Water* **2016**, *8*, 549. [CrossRef]
83. Kovacic, D.A.; David, M.B.; Gentry, L.E. Grassed detention buffer strips for reducing agricultural nonpoint-source pollution from tile drainage systems. In *Proceedings of the Research on Agricultural Chemicals in Illinois Groundwater: State and Future Directions*; Davis, M., Ed.; Southern Illinois University: Carbondale, IL, USA, 1996; pp. 88–97.
84. Carlson, B.; Vetsch, J.; Randall, G. *Nitrates in Drainage Water in Minnesota*; University of Minnesota Extension Publication: Minneapolis, MN, USA, 2013.
85. Hoffmann, C.C.; Kronvang, B.; Audet, J. Evaluation of nutrient retention in four restored Danish riparian wetlands. *Hydrobiologia* **2011**, *674*, 5–24. [CrossRef]
86. Brunet, R.C.; Astin, K.B. A 12-month sediment and nutrient budget in a floodplain reach of the River Adour, southwest France. *Regul. Rivers Res. Manag.* **2000**, *16*, 267–277. [CrossRef]

Publisher's Note: MDPI stays neutral with regard to jurisdictional claims in published maps and institutional affiliations.