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Combination of Measures to Restore Eutrophic Urban Ponds in The Netherlands

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Abstract: Urban ponds provide the most important public contact with surface waters, implying that good water quality is crucial to the quality of urban life. Three eutrophic urban ponds in the south of The Netherlands with a long history of eutrophication-related nuisance were studied and subjected to mitigating measures. The external nutrient load from a mixed sewer overflow to one of the ponds had already been dismantled prior to the study, in a second pond it was dismantled during, while in the third pond the major nutrient source (stormwater run-off from impervious surfaces) was left untouched. In order to rehabilitate the ponds, all were dredged to reduce the internal loading, the fish biomass was reduced, the banks were softened, macrophytes were planted, users were advised to minimize the feeding of the fish and waterfowl, and the external nutrient load was reduced in two of the ponds. The two ponds in which the major external load was reduced showed strongly improved water quality after the additional in-pond measures. In contrast, the pond with ongoing external loading from stormwater run-off showed only marginally improved water quality. This study underpins that stormwater run-off can be polluting and that mitigating measures should only be implemented when the system analysis has revealed their feasibility.

Keywords: biomanipulation; dredging; eutrophication control; internal loading; pond restoration

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1. Introduction

Ponds are small-sized (<5 ha), shallow (maximum 5 m deep), man-made or natural water bodies with <30% coverage by emergent vegetation and may permanently contain water or seasonally dry up [1–3]. Ponds are ubiquitous and estimates range between 5.47×10^8 and 3.2×10^9 ponds [4] existing worldwide. Ponds are common in urban areas, where they may provide multiple benefits to human society, such as offering recreation opportunities, serving as water storage, receiving sewer overflows and surplus rainwater, regulating the micro climate, delivering amenity and cultural values, and contributing to biodiversity [5–11]. In urban areas, more people come into contact with these ponds than in rural areas [12]. The embedding in the urban landscape, however, also implies that urban ponds experience high anthropogenic pressures. Nutrient run-off, fish stocking, bird feeding, along with how they are constructed, for instance with hard banks, as opposed to soft, nature-friendly ones, often results in poor water quality in urban ponds [13].

Eutrophication is one of the key pressures urban ponds face [6,14]. In the urban watershed, nutrients may enter the ponds via multiple routes, such as run-off from impervious surfaces, from gardens and construction sites, sewer overflows, leaf litter, feeding birds and fish, sewage dump from boats, and atmospheric deposition [13,15]. The eutrophication of ponds may cause a transition from clear water with submerged macrophytes to turbid water with high concentrations of phytoplankton [16], which often are mostly cyanobacteria [17,18]. Those cyanobacterial blooms can result in fish kills, are potentially toxic to humans and animals, reduce biodiversity, may accumulate as unpleasant surface scums, and may produce malodors [18]. The nuisance caused by cyanobacterial blooms is widespread in eutrophic urban ponds in The Netherlands [19] and in many other countries [20–22]. These cyanobacterial blooms in urban ponds may impede water use, hamper economic activities, and cause real estate depreciation [23], and in general, phytoplankton-dominated systems provide fewer ecosystem services than macrophyte-dominated ones [24]. Hence, the authorities have strong incentives to improve the water quality in urban ponds such that ecosystem services are no longer impaired.

To improve the water quality in urban ponds, preferably each case has to be studied before tailor-made restoration measures can be applied [25]. The rationale behind such pre-intervention diagnosis is that nutrients may come from various sources [13] and that during eutrophication the system may have changed [26]. This water body-specific diagnosis (system analysis) will indicate the water balance and nutrient budget [27], the extent to which the aquatic community deviates from the desired composition, and therewith indicate the buttons to be pressed to realize the desired water quality improvement. Here, we report on such an approach where water authorities made a diagnosis of three ponds in the south of The Netherlands and subsequently implemented several measures to reduce the nutrient load and to increase the water transparency. The aim of this study is to report on the effectiveness of the implemented combined rehabilitation measures in improving the water quality in the ponds. To this end, seven (ponds Dongen and Eindhoven) and nine years (Pond Heesch) post-intervention monitoring results were used. In line with the expectations, water quality strongly improved in ponds Heesch and Dongen, but only meagerly in Pond Eindhoven due to the ongoing high external nutrient load.

2. Materials and Methods

2.1. Study Sites

The three urban ponds studied are located in the province Noord-Brabant in The Netherlands (Figure 1). Each pond falls under the jurisdiction of a different water authority (WA): Pond Dongen—WA Brabantse Delta, Pond Eindhoven—WA De Dommel and Pond Heesch—WA Aa en Maas. These WAs share responsibility for the water quality of all surface waters in their jurisdiction, while the ponds are owned by the municipality and the local fishing associations have legal fishing rights.

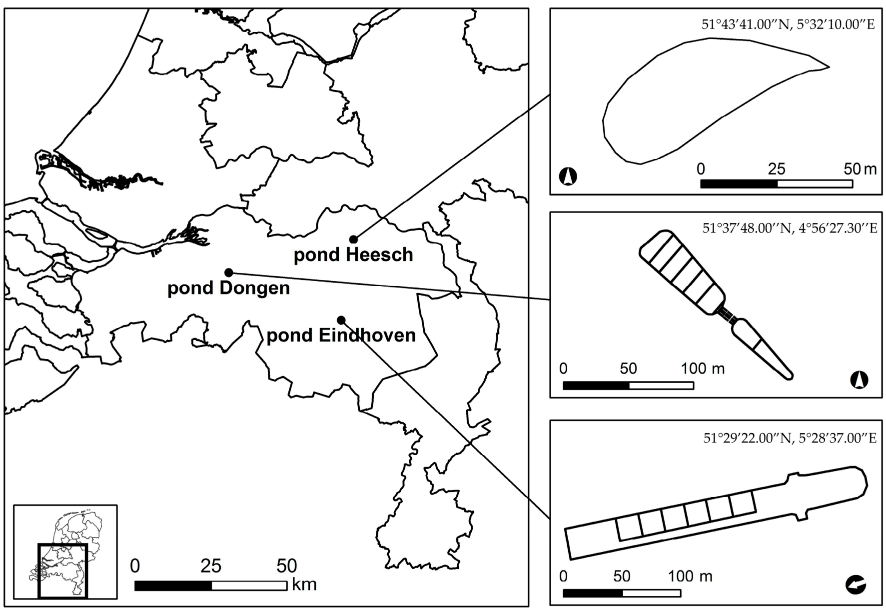

Figure 1. Location of the three ponds Dongen, Eindhoven, and Heesch in The Netherlands, including a schematic drawing of the ponds and the compartments that were constructed in two of them prior to the rehabilitation measures for testing promising in-pond measures.

Pond Dongen (51°37′48.00″ N, 4°56′27.30″ E) was created in 1970 and consists of two connected parts (Figure 1). It is an isolated pond with a surface area of 2500 m^2, a mean depth of 0.7 m and without connection to other surface water. The pond is characterized by infiltration. In dry periods, the water level is maintained by a supply of pumped groundwater. During wet periods, excess water is discharged through the sewer system of the adjacent residential area. From 1970 to 2000, the pond received mixed sewage overflow (Table 1).

Table 1. Characteristics of the three urban ponds (Dongen, Eindhoven, Heesch).

			
Pond→ **Characteristic↓**	**Dongen**	**Eindhoven**	**Heesch**
Surface area	2500 m^2	6500 m^2	1600 m^2
Depth	0.7 m	1.5 m	1.0 m
Constructed	1970	1994	1974
Inflow	no	no	no
Outflow	no	yes	no
Water in	precipitation, (pumped) groundwater	precipitation, stormwater run-off	precipitation, groundwater
Water out	evaporation, infiltration, over-flow	evaporation, over-flow	evaporation, infiltration
Main use	recreation, fishing sewage overflow till 2000	water storage, recreation, fishing	recreation, fishing sewage overflow till 2009
Banks	wooden revetment	wooden revetment	wooden revetment
Vegetation	no helo-/macrophytes	no helo-/macrophytes	no helo-/macrophytes
Fish	1331 kg ha^{-1}	1254 kg ha^{-1}	1444 kg ha^{-1}
sediment resuspending	85%	77%	85%

Pond Eindhoven (51°29′22.00″ N, 5°28′37.00″ E) was created in 1994. This pond has a surface area of 6500 m^2, a mean depth of 1.5 m and is connected to watercourses which drain the excess of water from the pond. There is neither seepage nor infiltration as a 30 cm thick clay layer was used during construction to seal the pond from the underlying soil. The major supply of water comes from stormwater run-off collected from impervious surfaces in the adjacent residential area entering the pond by separated sewer overflows. The major discharge is the outlet of superfluous water to adjacent watercourses (Table 1).

Pond Heesch (51°43′41.00″ N, 5°32′10.00″ E) was created in 1974 and is a small isolated pond with a surface area of 1600 m^2, a mean depth of 1.0 m and without connection to other surface water. The pond is strongly influenced by seepage during wet periods and by infiltration during dry periods. It is presumed that a mixed sewage overflow has regularly flushed polluted water into the pond during periods of intensive rainfall. In 2009, the sewage overflow was disconnected (Table 1).

The ponds are used predominantly for angling, although other water-related activities occur such as boating, feeding water birds, and use as a dog outlet. The ponds had hard edges and lacked submerged macrophytes (Table 1). Cyanobacteria blooms and surface scums were annually reoccurring in these ponds where cyanotoxins (microcystins) in the water column could reach up to 77 µg L^{-1}, and in scums up to 64,000 µg L^{-1} [19]. Prior to the restoration measures presented in this study, only Pond Dongen had undergone a previous restoration measure, as after 30 years a mixed sewer overflow system was disconnected in 2000.

2.2. System Analysis

A system analysis aims to identify the cause(s) of the water quality problem. For eutrophication issues, it usually includes the water and nutrient balance, often the phosphorus (P) balance of the waterbody, the biological characteristics of the system, and its use/function(s) [28]. The water and P balance was made by the respective WAs for Pond Dongen and Pond Eindhoven and has been reported in detail in [29]. For pond Heesch, the responsible WA considered the sewer overflow as the main source of external pollution that, together with the relatively small external sources such as feeding fish and waterfowl, had resulted over time in a relatively large amount of pollution. Hence, this WA focused on getting insight on the internal load. The corresponding P fluxes (mg P m^{-2} d^{-1}) from different sources to the ponds Dongen and Eindhoven are given in [29] and have been included in Figure 2 to compare with pond Heesch (Figure 2).

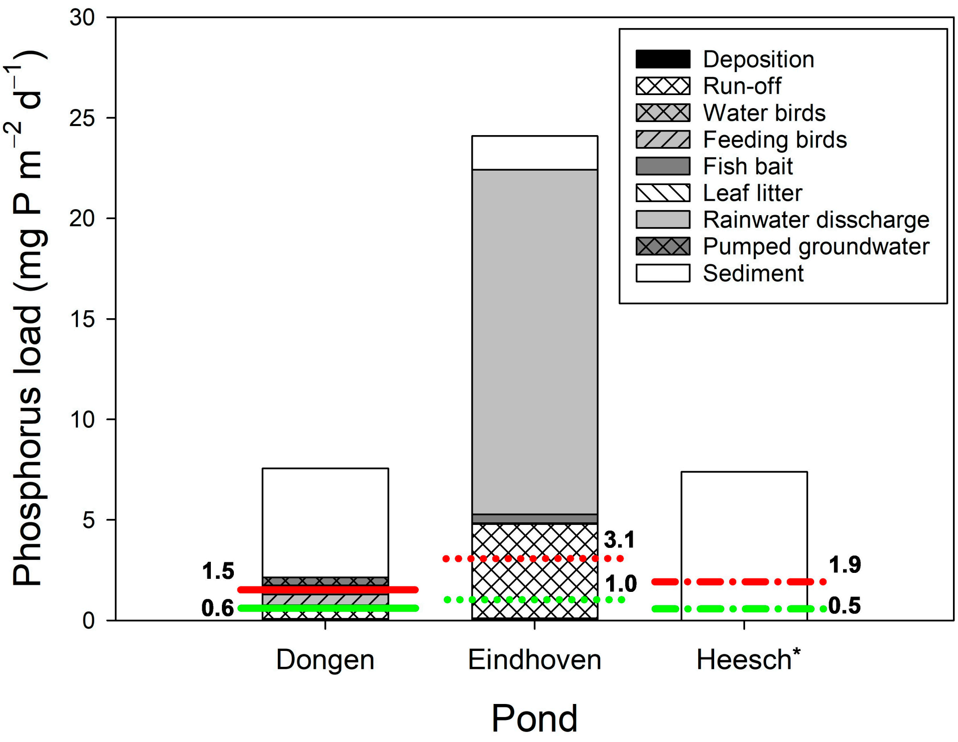

Figure 2. Phosphorus loadings (mg P m^{-2} d^{-1}) from external and internal sources for ponds Dongen, Eindhoven (based on [28]), and Heesch. The critical P loading thresholds for clear to turbid (red lines) and turbid to clear (green lines) were determined with the PCLake Metamodel [30]. * Only internal source included in analysis of drivers of movement of P in pond Heesch.

Estimates for the P loading at which a transition from a clear to turbid state and from a turbid to clear water state could occur were determined with the PCLake Metamodel [30] and have been included in Figure 2.

In each pond, a fish stock assessment was performed that revealed a very high fish biomass over 1000 kg ha^{-1} (Table 1, [31–33]) with a majority (77–85%) being sediment-resuspending fish, such as carp and bream

The ponds were void of macrophytes, whereas the wooden revetments prohibited the growth of helophytes, such as reed. All three ponds were used for angling, as a dog outlet, for walking, and for occasional boating (Table 1).

2.3. Mitigating Measures

In all three ponds, prior to implementing mitigating measures, experiments were conducted to test the efficacy of potential measures to reduce the internal P load. In ponds Dongen and Eindhoven, biomanipulation (fish removal plus introduction of macrophytes), biomanipulation plus dredging, and biomanipulation plus a P binder was tested [29], while in pond Heesch, dredging and/or the addition of a P binder was studied [31]. Based on the system diagnostics, the above-mentioned estimated P loads, critical loads, and the experiments described in [29,31], the WAs Brabantse Delta and Aa en Maas decided various mitigating measures (Table 2) to reduce the P load to at least below the upper critical load and to restructure the ponds so they would have the highest chance for clear water with macrophytes. The interventions were executed in autumn 2011/spring 2012 (Pond Dongen; WA Brabantse Delta) and autumn/winter 2009 (Pond Heesch; WA Aa en Maas). WA De Dommel, due to agreements made prior to the experiments, decided to implement not all the recommended measures in Pond Eindhoven, although the outcome of the system diagnostics indicated low chance for success. The various mitigating measures in Pond Eindhoven were effected in autumn 2011/spring 2012 (Table 2).

Table 2. Measures implemented in the three urban ponds to mitigate eutrophication nuisance.

Pond Dongen	Pond Eindhoven	Pond Heesch
Dredging *	Dredging and deepening pond #	Excavation ‡
Creating soft banks	Creating soft banks	Creating soft banks
Planting macrophytes	Planting macrophytes	Planting macrophytes
Prohibiting dog walking	Informing citizens about dog walking (remove faeces)	Removing sewer overflow
Informing citizens about feeding ducks/fish	Informing citizens about feeding ducks/fish	Informing citizens about feeding ducks/fish
No carp, less baiting	Less carp, less baiting	---
Allow water level fluctuation (less groundwater pumping)	---	---
---	---	Tree harvesting/pruning (prevent leaves in pond
Fish stock manipulation	Fish stock manipulation	Fish stock manipulation

Notes: * dredging = removal of sediment without drawdown, # deepening = removing more material than just the top sediment to increase water depth more, ‡ excavation = removal of overlying water followed by scraping off the upper layer of the lake bed.

Prior to the other measures, fish were removed from the three ponds by a professional fishing company (Visserijbedrijf P. Kalkman, Moordrecht, The Netherlands). In Pond Dongen this was performed on 16th September 2011, preceding the dismantling of the compartments by seine-haul fishing using a 35 m net (mesh size 8–12 mm) that was pulled twice through each compartment. The data derived from the seine-haul fishing (75 m net) combined with the electrofishing (5 kW) performed before the compartmentation on 7 April 2009 [32] were used to estimate the original fish biomass in the pond (Table 3). In Pond Eindhoven, a net was constructed on 11 October 2011 that split the pond into two parts: the fish were removed by combined seine-haul fishing (twice in each part with a 225 m net with 8–12 mm mesh) and followed by electrofishing (5 kW) along the banks; on 8th December 2011, when the water level was lowered for the planned dredging and deepening, the remaining fish were removed by four trawls and electrofishing, showing the dominance of bream (*Abramis brama*) and carp (*Cyprinus carpio*) (Table 3; [33]). In Pond Heesch, one trawl with a 75 m seine-haul net (8–12 mm mesh) was conducted on 6 April 2009 combined with electrofishing (5 kW) along the banks. All fish were identified and weighed [34]. The fish stock manipulation implied the restocking of the ponds after mitigation measures had been carried out with a much lower fish biomass, i.e., 90 kg ha^{-1} in Pond Dongen, 77 kg ha^{-1} in Pond Eindhoven, and 50 kg ha^{-1} in pond Heesch (Table 3). The different species restocked were based on the requests of the local angling societies and views of the three WAs.

Table 3. Original fish community composition (fresh weight kg ha^{-1}) and the restocked fish community after fish removal as one of the rehabilitation measures in the three ponds examined in this study.

Fish Species	Fish Stock (kg ha^{-1}) Pond Dongen Original [1]	Pond Dongen Restocked	Pond Eindhoven Original [2]	Pond Eindhoven Restocked	Pond Heesch Original [3]	Pond Heesch Restocked
Bleak	---	16	---	---	---	5
Bream	22.3	---	207.7	---	3.4	5
Carp	1109.9	---	573.8	---	1011.5	10
Catfish	---	---	---	---	47.1	---
Gibel-carp	16.3	---	180.0	---	218.0	---
Grass-carp	---	---	9.2	---	---	---
Gudgeon	---	---	---	---	0.1	---
Ide	---	---	---	---	0.4	---
Perch	---	16	6.3	---	2.6	5
Pike	---	6	70.8	31	---	5
Pumpkinseed	---		0.5	---	0.8	---
Roach	178.1	24	189.2	46	30.2	5
Rudd	2.9	12	3.1	---	5.0	10
Silver bream	1.8	---	---	---	---	---
Silver carp	---	---	---	---	124.6	---
Tench	---	16	13.8	---	---	5
Total	1331	90	1254	77	1444	50

Note: [1] [32], [2] [33], [3] [34].

2.4. Water Quality Sampling and Analysis

All three ponds were sampled biweekly from March 2009 to August 2011 and thereafter 8–10 times a year until December 2018. In Pond Dongen, the pond was sampled from March to August 2009 and from October 2011 to December 2018, while from September 2009 until 30 August 2011, the control compartment was sampled. In each pond, a variety of water quality variables were measured, such as the dissolved oxygen saturation (OxyGuard Handy Polaris, OxyGuard International A/S, Farum, Denmark), pH (WTW-pH320, WTWGmbH & Co. KG, Weilheim, Germany) and Secchi disc depth. Two-liter water samples were taken from the ponds with a sampling tube (1 m Perspex tube, diameter 5 cm that can be closed with rubber stoppers). In these samples, the total and cyanobacterial chlorophyll-*a* concentrations were measured using a PHYTO-PAM phytoplankton analyzer (Heinz Walz GmbH, Effeltrich, Germany). The turbidity was measured with a Hach 2100P Turbidity meter (Hach Nederland, Tiel, The Netherlands). The unfiltered samples were analyzed on total phosphorus (TP) and total nitrogen (TN) concentrations using a Skalar SAN++ segmented flow analyzer (Skalar Analytical BV, Breda, The Netherlands) following the Dutch standard protocols [35,36].

The water quality variables (total and cyanobacterial chlorophyll-*a* concentrations, TN, TP, Secchi-disc depth, turbidity, dissolved oxygen concentrations, and pH) were plotted as time series before and after the interventions using the SigmaPlot program (version 14.5, Systat Software Inc., San Jose, CA, USA). The water quality variables were subjected to a "before-after" analysis assuming that, were it not for the interventions, the dynamics and magnitudes of the water quality variables would not have changed [37]. Since each pond did not have a non-manipulated control, the total chlorophyll-*a* concentrations derived from a summer snapshot monitoring program in the urban ponds in the region of the three restored ponds were used to check for an overall pattern (Table A1). The preferred before-after analysis was a t-test; however, in most cases the normality test (Shapiro–Wilk) or equal variance test (Brown-Forsythe) failed (i.e., $p < 0.05$) and the non-parametric Mann–Whitney Rank Sum Test was run instead using the SigmaPlot program (version 14.5). The water quality variables were also examined for the presence of trends using the Pearson Product Moment Correlation (not assuming a form of response).

Water quality variables were related to the corresponding Water Framework Directive classification for water type M11, a small (<50 ha), shallow (<3 m), buffered (1–4 meq L^{-1}) water body (Figure A1, [38]). The longer-term treatment effects were compared to a dataset of 32 ponds during the same years (Figure A2).

The permissions to execute the interventions were issued by the legislators of the water authorities Brabantse Delta (Dongen), De Dommel (Eindhoven) and Aa en Maas (Heesch).

3. Results

3.1. Chlorophyll-a Concentrations

In Pond Dongen, before the intervention, the mean total chlorophyll-*a* was 344 µg L^{-1} (standard error, SE 51 µg L^{-1}), after the intervention it was 16 µg L^{-1} (SE 2 µg L^{-1}) without a trend (Table A3; Figure 3a). In Pond Eindhoven, before the intervention, the mean total chlorophyll-*a* was 95 µg L^{-1} (SE 8 µg L^{-1}), after the intervention it was 36 µg L^{-1} (SE 4 µg L^{-1}), with weak evidence of both a negative trend before and after the intervention (Table A3; Figure 3b). In pond Heesch, before the intervention, the mean total chlorophyll-*a* was 262 µg L^{-1} (SE 20 µg L^{-1}), after the intervention it was 31 µg L^{-1} (SE 3 µg L^{-1}) without trend (Table A3; Figure 3c).

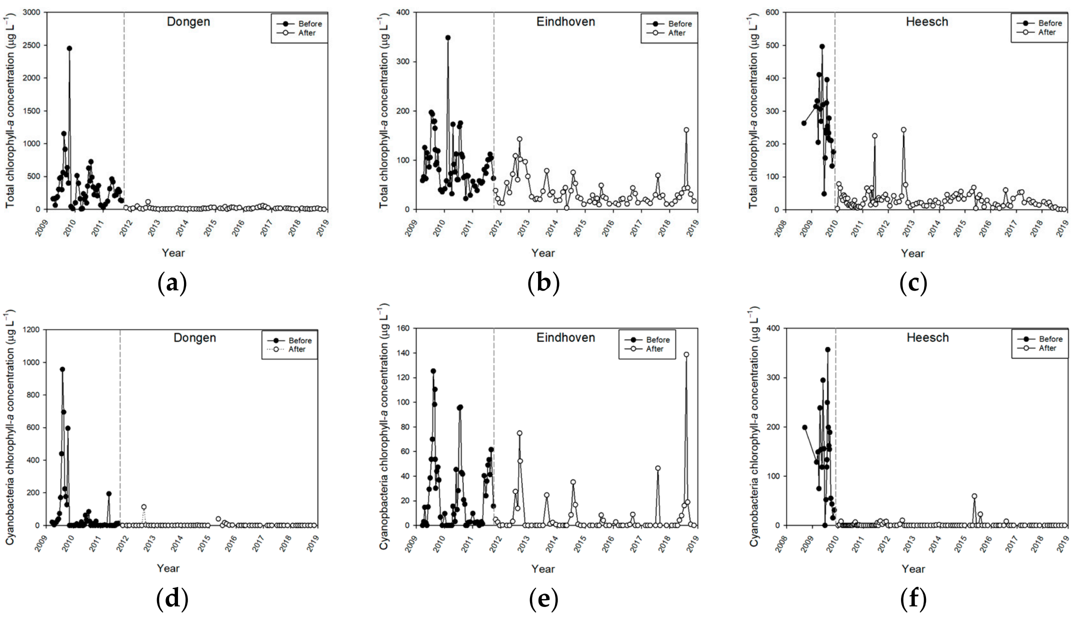

Figure 3. The course of total chlorophyll-*a* concentrations (µg L^{-1}) in Pond Dongen (**a**), Pond Eindhoven (**b**) and Pond Heesch (**c**) as well as the cyanobacterial chlorophyll-*a* concentrations (µg L^{-1}) in Pond Dongen (**d**), Pond Eindhoven (**e**), and Pond Heesch (**f**) before the restoration intervention (black symbols) and after the intervention (open symbols). The dotted vertical lines indicate the moment of intervention in each pond.

In all three ponds, there was very strong evidence for the median total chlorophyll-*a* concentrations to be higher before the intervention than after (Table 4). In a dataset of 32 ponds, the summer snapshot total chlorophyll-*a* concentrations did not show differences over the years (Figure A2).

Table 4. Overview of before-after comparison of total and cyanobacterial chlorophyll-*a* concentrations in the ponds in periods before restoration interventions took place and after restoration.

Total Chlorophyll-*a* Concentrations			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney	U = 149.0	U = 443.5	U = 32.0
Rank Sum Test	$T_{53,66} = 4780$ $p < 0.001$	$T_{56,68} = 4960.5$ $p < 0.001$	$T_{23,103} = 2613$ $p < 0.001$
Cyanobacteria Chlorophyll-*a* Concentrations			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney	U = 824.5	U = 892	U = 68.0
Rank Sum Test	$T_{53,66} = 3945.5$ $p < 0.001$	$T_{56,66} = 4400$ $p < 0.001$	$T_{23,103} = 2577$ $p < 0.001$

In Pond Dongen, the mean cyanobacterial chlorophyll-*a* concentration was 78 μg L^{-1} (SE 26 μg L^{-1}) before the intervention, while it was 3 μg L^{-1} (SE 2 μg L^{-1}) after the intervention (Figure 3d). There was a negative trend before the intervention and no trend after the intervention (Table A3). In Pond Eindhoven, before the intervention, the mean cyanobacterial chlorophyll-*a* was 28 μg L^{-1} (SE 4 μg L^{-1}), while it was 8 μg L^{-1} (SE 3 μg L^{-1}) after the intervention and without a trend (Table A3). The pond had occasional blooms both before as well as after the intervention (Figure 3e). In pond Heesch, before the intervention, the mean cyanobacterial chlorophyll-*a* concentration was 142 μg L^{-1} (SE 19 μg L^{-1}), after intervention it was 2 μg L^{-1} (SE 1 μg L^{-1}) (Figure 3f). No trend in the cyanobacterial chlorophyll-*a* concentrations was found (Table A3).

In all three ponds, there was very strong evidence that the median cyanobacterial chlorophyll-*a* concentrations were higher before the intervention than after (Table 4).

3.2. Nutrients

In Pond Dongen, the mean total phosphorus concentration was 537 μg P L^{-1} (SE 55 μg P L^{-1}) before intervention and 33 μg P L^{-1} (SE 3 μg P L^{-1}) after intervention (Figure 4a). No trends were detected (Table A3). In Pond Eindhoven, the mean total phosphorus concentrations before the intervention was 99 μg P L^{-1} (SE 12 μg P L^{-1}), after the intervention it was 56 μg P L^{-1} (SE 8 μg P L^{-1}) (Figure 4b). Before as well as after the intervention, a negative trend in the total phosphorus concentration was observed (Table A3). In Pond Heesch, the mean total phosphorus concentration before the intervention was 425 μg P L^{-1} (SE 90 μg P L^{-1}), while it was 75 μg P L^{-1} (SE 6 μg P L^{-1}) after the intervention (Figure 4c), with a negative trend after the intervention (Table A3).

In all three ponds, there was very strong evidence that the median total phosphorus concentrations were higher before the intervention than after (Table 5).

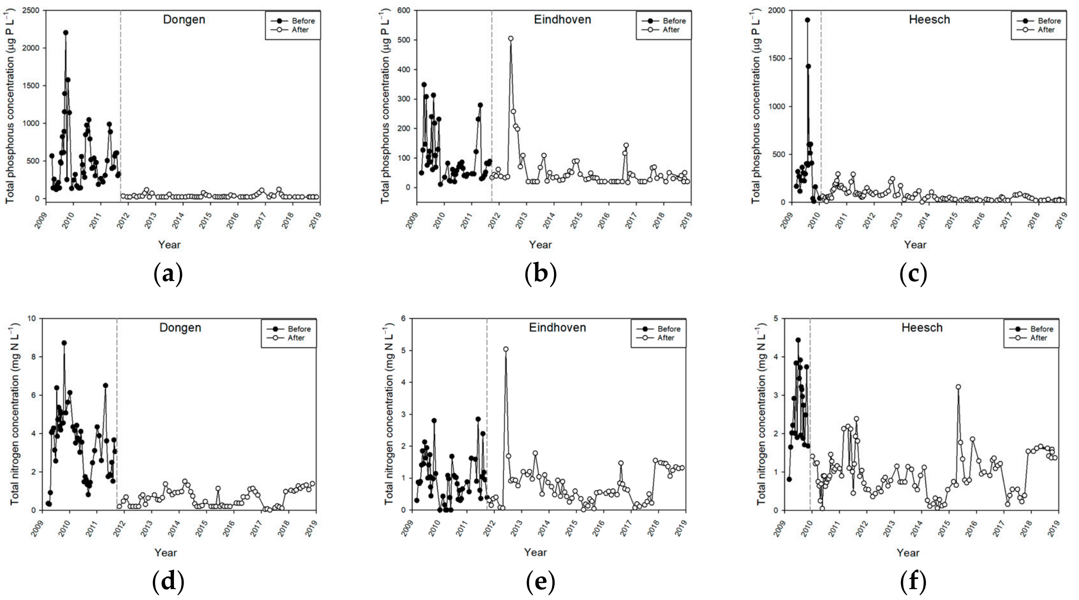

Figure 4. Course of the total phosphorus concentrations (µg P L^{-1}) in Pond Dongen (**a**), Pond Eindhoven (**b**), and Pond Heesch (**c**) as well as the total nitrogen concentrations (mg N L^{-1}) in Pond Dongen (**d**), Pond Eindhoven (**e**), and Pond Heesch (**f**) before the restoration intervention (black symbols) and after the intervention (open symbols). The dotted vertical lines indicate the moment of intervention in each pond.

Table 5. Overview of before-after comparison of total phosphorus and total nitrogen concentrations in the ponds in periods before restoration interventions took place and after restoration.

Total Phosphorus Concentrations			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney	U = 1.0	U = 865	U = 317
Rank Sum Test	$T_{56,65} = 5235$ $p < 0.001$	$T_{51,67} = 3878$ $p < 0.001$	$T_{24,100} = 2383$ $p < 0.001$
Total Nitrogen Concentrations			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney	U = 135	U = 1332	U = 118
Rank Sum Test	$T_{51,65} = 4506$ $p < 0.001$	$T_{51,67} = 3411$ $p = 0.041$	$T_{22,100} = 2335$ $p < 0.001$

The mean total nitrogen concentration in Pond Dongen was 3.45 mg N L^{-1} (SE 0.24 mg N L^{-1}) before intervention, while it was 0.63 mg N L^{-1} (SE 0.05 mg N L^{-1}) after the intervention (Figure 4d). There was weak evidence of trends in the total nitrogen concentrations before and after the intervention (Table A3). In Pond Eindhoven, the mean total nitrogen concentration was 1.00 mg N L^{-1} (SE 0.10 mg N L^{-1}) before the intervention and 0.77 mg N L^{-1} (SE 0.09 mg N L^{-1}) after the intervention (Figure 4e), without trends (Table A3). In Pond Heesch, the mean total nitrogen concentration before the intervention was 2.66 mg N L^{-1} (SE 0.20 mg N L^{-1}), while it was 0.97 mg N L^{-1} (SE 0.06 mg N L^{-1}) after the intervention (Figure 4f), without trends (Table A3). For ponds Dongen and Heesch the data revealed very strong evidence that the total nitrogen concentrations before the intervention were higher than after, while for Pond Eindhoven, the data revealed moderate evidence (Table 5).

3.3. Water Clarity

The water transparency, as determined by the Secchi disc depth, gradually increased in Pond Dongen over time (Figure 5a). The Secchi disc depth was on average (±1 SE) 0.27 m (±0.02 m) before the intervention and 0.62 m (±0.02 m) after the intervention. There was no trend before intervention and a positive trend after intervention (Table A3). In Pond Eindhoven, clear seasonal patterns in the Secchi disc depth were observed with high values in winter and low values in summer (Figure 5b). The Secchi disc depth was 0.46 m (±0.03 m) before the intervention and 0.58 m (±0.03 m) after the intervention, with no trend before and a positive trend after the intervention (Table A3). In Pond Heesch, the Secchi disc depth was 0.21 m (±0.02 m) before the intervention and 0.74 m (±0.02 m) after the intervention (Figure 5c). There was weak evidence of trends (Table A3). There was very strong evidence that the Secchi disc depths increased after the intervention in all three ponds (Table 6).

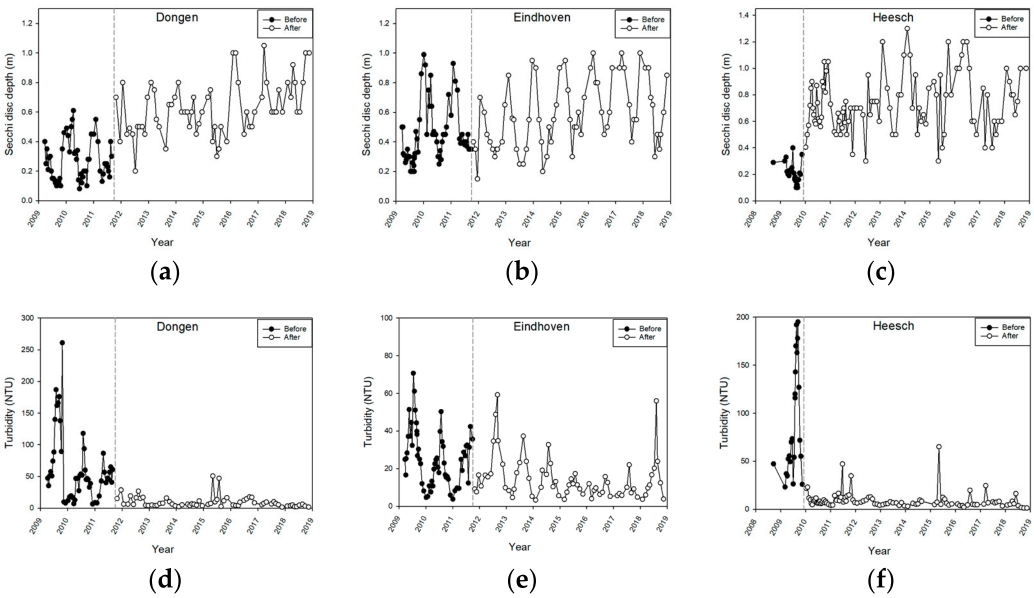

Figure 5. Course of the Secchi disc depth (m) in Pond Dongen (**a**), Pond Eindhoven (**b**), and Pond Heesch (**c**) as well as the turbidity (NTU) in Pond Dongen (**d**), Pond Eindhoven (**e**), and Pond Heesch (**f**) before the restoration intervention (black symbols) and after the intervention (open symbols). The dotted vertical lines indicate the moment of intervention in each pond.

Table 6. Overview of before-after comparison of the Secchi disc depth and turbidity in the ponds in periods before restoration interventions took place and after restoration.

Secchi Disc Depth			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney Rank Sum Test	U = 197.5 $T_{53,63} = 1628.5$ $p < 0.001$	U = 1296 $T_{58,68} = 3007$ $p < 0.001$	U = 10.0 $T_{23,100} = 286$ $p < 0.001$
Turbidity			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney Rank Sum Test	U = 236 $T_{52,65} = 4522$ $p < 0.001$	U = 972 $T_{57,67} = 4500$ $p < 0.001$	U = 20.5 $T_{23,99} = 2532.5$ $p < 0.001$

In Pond Dongen, before the intervention, the mean turbidity of the water was 61.8 (±7.4) NTU and 9.5 (±1.1) NTU after the intervention (Figure 5d) with moderate and weak evidence of negative trends before and after the intervention, respectively (Table A3). In Pond Eindhoven, the mean turbidity was 25.9 (±2.0) NTU before the intervention and 14.8 (±1.4) NTU after the intervention (Figure 5e). There was moderate evidence of negative trends (Table A3). In Pond Heesch, before the intervention, the mean turbidity was 90.5 (±12.1) NTU and after the intervention it was 9.2 (±0.9) NTU (Figure 5e) with moderate evidence for a negative trend before the intervention and no trend after (Table A3).

The data revealed very strong evidence that the turbidity had decreased after the intervention in all three ponds (Table 6).

3.4. Dissolved Oxygen Saturation and pH

In Pond Dongen, before the intervention, the mean dissolved oxygen saturation was 84.2% (±5.8%, 1 SE), after the intervention it was 95.3% (±3.0%). Although there was no evidence the medians before and after the intervention differed (Table A2), strong fluctuations as observed before the intervention, especially in the dissolved oxygen saturation <50% did no longer occur after the intervention (Figure A3a). Moderate evidence for a positive trend was found before the intervention, but not after (Table A3). In Pond Eindhoven, the mean dissolved oxygen saturation showed similar patterns before and after the intervention (Figure A3b) with average (±1 SE) dissolved oxygen saturations of 104.2% (±3.5%) and 102.4% (±3.1%) before and after the intervention, respectively, without trends (Table A3). Likewise, in Pond Heesch, the mean dissolved oxygen saturations were similar (Table A2) before (97.8 ± 8.0%) and after (96.6 ± 2.8 %) the intervention (Figure A3c), and without a trend (Table A3).

The pH of the water in Pond Dongen showed more variability before than after the intervention (Figure A3d), the mean pH (±1 SE) was 7.58 (±0.14) before the intervention and 7.78 (±0.08) after the intervention with only weak evidence that the pH values differed (Table A2). There was a positive trend before, but no trend in pH after the intervention (Table A3). In Pond Eindhoven, the pH of the water showed typical seasonality with higher pH values in summer than in winter with a tendency of more variability before than after the intervention (Figure A3e). The mean pH value (±1 SE) was 8.38 (±0.08) before the intervention and 8.05 (±0.06) after the intervention with a *t*-test supporting the strong evidence that pH values after the intervention were on average lower than before the intervention (Table A2). There was moderate evidence for a positive trend before the intervention, but no trend was detected after the intervention (Table A3). In Pond Heesch, the mean pH (±1 SE) was 7.68 (±0.13) before the intervention and 7.89 (±0.05) after the intervention (Figure A3f) with only weak evidence that the pH values differed (Table A2) and without trends (Table A3).

3.5. Water Framework Directive Scores

The water quality variables were compared to the corresponding WFD classifications, which showed moderate to strong improved status based on the selected water quality variables (Table 7).

Table 7. The mean annual chlorophyll-a concentrations (CHLa, in μg L^{-1}), total phosphorus concentrations (TP, in mg P L^{-1}), and total nitrogen concentrations (TN, in mg N L^{-1}) in Pond Dongen, Pond Eindhoven, and Pond Heesch related to Water Framework Directive status classification for these water quality variables [37]. Red indicates a bad status, orange = insufficient, yellow = moderate, green = good, and blue = excellent. Values in italics are from before the interventions.

	Pond Dongen			Pond Eindhoven			Pond Heesch		
Year	**CHLa**	**TP**	**TN**	**CHLa**	**TP**	**TN**	**CHLa**	**TP**	**TN**
2009	*499.7*	*0.64*	*4.15*	*110.9*	*0.15*	*1.29*	*262.1*	*0.44*	*2.65*
2010	*285.9*	*0.46*	*2.95*	*93.5*	*0.05*	*0.73*	24.4	0.11	0.91
2011	*199.8*	*0.43*	*2.70*	*65.1*	*0.08*	*1.01*	46.5	0.11	1.35
2012	26.2	0.05	0.44	75.0	0.16	1.20	50.7	0.12	0.61
2013	7.7	0.02	0.84	31.8	0.04	1.08	19.2	0.05	0.85
2014	13.1	0.03	0.67	32.4	0.05	0.57	35.3	0.04	0.31
2015	21.1	0.04	0.19	22.2	0.03	0.27	35.5	0.05	1.32
2016	25.7	0.04	0.72	20.8	0.04	0.62	19.0	0.03	1.12
2017	12.1	0.04	0.22	26.2	0.04	0.38	29.5	0.06	0.52
2018	6.7	0.02	1.17	40.9	0.03	1.34	10.3	0.02	1.52

4. Discussion

The post-intervention water quality monitoring results of ponds Dongen and Eindhoven (seven years), and Pond Heesch (nine years) provided valuable information on the effectiveness of the implemented rehabilitation measures in improving the water quality. As expected, the water quality strongly improved in ponds Heesch and Dongen, but improved only meagerly in Pond Eindhoven. In Pond Dongen and Heesch, the rehabilitation measures reduced the chlorophyll-*a* concentrations strongly and brought the mean cyanobacteria chlorophyll-*a* concentrations to 3.8% and 1.4% of the pre-intervention values, respectively. In the entire post-intervention period, no cyanobacteria blooms occurred in Pond Heesch and there were was no need for the authorities to issue warnings. In Pond Dongen, only in August 2012 a short-lived bloom of cyanobacteria occurred (see minor peak in Figure 3d), which was explained by submerged macrophytes not having developed yet. After that, no cyanobacteria issues occurred and no warnings had to be issued. In contrast, in Pond Eindhoven, the occasional cyanobacteria nuisance remained and regular warnings had to be issued despite the mean post-intervention cyanobacterial chlorophyll-*a* concentration being reduced to 29% of the pre-intervention chlorophyll-*a* concentration.

The measures were expected to improve the water quality in ponds Dongen and Heesch, but only weakly in Pond Eindhoven. The system analysis for Pond Eindhoven had pointed out a high external loading (24.1 mg P m^{-2} d^{-1}), especially from the surface run-off and the rainwater discharge through the separated sewer system [29]. The relatively high P load from the separated sewer system (stormwater runoff) is fully in line with the results obtained elsewhere (e.g., [39]) and urges the authorities to reconsider the general view that the separated sewer system only delivers rain water. Evidently, the rain water picks up quite a lot of nutrients from roofs, streets, and pavements on its way to the receiving pond. The P load for Pond Eindhoven was estimated to drop marginally to 21.6 mg P m^{-2} d^{-1} after the implementation of in-pond measures, which was still about ten times higher than the estimated critical P load. Although these estimated loads come with some uncertainties [40], the load was so high that, without strong reduction of it, the in-pond measures would at best only have a short-lived effect. Hence, the advice given was to reduce the external P load drastically so it would come at least below the upper critical P load of 3.1 mg P m^{-2} d^{-1}, which would increase the chance for a switch to a clear water state [41]. The WA De Dommel and the municipality viewed the needed external load reduction measures as too expensive and decided to stay with their pre-diagnosis agreement to focus on the in-pond measures only (see Table 2).

In Pond Dongen, the total P load was estimated to be lowered from 8.1 mg P m^{-2} d^{-1} before the rehabilitation measures to ~1–1.5 mg P m^{-2} d^{-1} after the intervention. The main part of this load estimate after the restoration was sediment release determined under anaerobic conditions, which could in situ become lower due to the oxygenation of the sediment when macrophytes and helophytes expand during the growing season [42]. In Pond Heesch, already the internal loading (7.40 mg P m^{-2} d^{-1}) exceeded by far the critical loadings, while the estimated P loading to Pond Heesch after the rehabilitation measures was 1.1 mg P m^{-2} d^{-1}. In these two ponds, the estimated P loads after the intervention were predicted to fall below the upper critical load, meaning that clear water with macrophytes is possible when sediment-resuspending fish biomass is low.

Prior to the rehabilitation measures, all three ponds were overstocked with carp (see Table 3), which is often the case in urban ponds [43]. Such excessive fish stock will keep the waters in a turbid state because of sediment resuspension and uprooting of plant sprouts preventing the growth of submerged macrophytes [44–48]. Hence, removing excessive fish from eutrophic urban ponds seems a necessity to improve the water quality [49]. The longevity of such interventions can, however, be undermined by fish recolonization [50,51]. Likewise, in shallow lakes, reduction in the fish biomass can improve the water quality (e.g., [45]), but interventions may have to be repeated regularly for a long-term effect [51,52]. The main reason why combined biomanipulation and nutrient load reduction can be successful lies in the fact that they act along both axes of the nutrient load–turbidity plane [16]. As visualized by [41], once the P loading is well within the critical loadings for the transition from a turbid to clear state and a clear to turbid state, direct food web management (biomanipulation) could be considered. Biomanipulation could be effective in further reducing the in-lake/pond P concentration [53]. Biomanipulation is feasible in ponds and small lakes, but for large shallow lakes, such additional in situ measures—to speed-up recovery after external load management—might not be cost-efficient [54]. The combined sediment dredging and biomanipulation of fish standing stocks has also been proposed for wetland eutrophication abatement [55].

Dredging had also been tested in the three ponds [29,56]. A controlled experiment with the sediment cores taken from large compartments in Pond Dongen revealed that the sediment P release was reduced from 5.4 mg P m^{-2} d^{-1} in non-dredged sediment to 1.0 mg P m^{-2} d^{-1} after dredging [29]. The calculations from the course of the phosphate concentrations in those enclosures in Pond Heesch yielded a sediment P release of 7.4 mg P m^{-2} d^{-1} in non-dredged conditions, which was reduced to about 0.4 mg P m^{-2} d^{-1} by dredging [56]. In contrast, an experiment with cores from Pond Eindhoven showed that dredging did not lower the sediment P release: 1.7 mg P m^{-2} d^{-1} in non-dredged and 1.9 mg P m^{-2} d^{-1} in dredged conditions [29]. The reason for this comparable P release before and after dredging is that with dredging, not only the sediment was removed but also a clay layer that resulted in exposure of the former fertilized agricultural lands on which Pond Eindhoven was created in the 1990s. When such former agricultural soils are exposed to water, a substantial internal P loading may result [57]. Thus, the dredging of P enriched sediment can reduce the internal release of P substantially and accordingly be effective as a measure in eutrophication control [27,58–60]. Dredging will, however, only be effective when the majority of the stored P is being removed, but not when freshly exposed sediment may release P to the overlying water [61]. Hence, in Pond Eindhoven, after external load reduction and fish stock manipulation, use of the tested P binder would have been the first choice to counteract internal P load, not dredging.

A (repeated) combined sediment capping and biomanipulation treatment has been suggested as a likely way forward and promising approach to make the restoration more robust [62]. In all three ponds, multiple measures were implemented that focused on internal load reduction, diffuse external load reduction and the restructuring of the ponds in Dongen and Eindhoven (softening banks, deepening, planting macrophytes, and removing fish), and on-point source control (disconnecting a mixed sewer overflow), diffuse external load reduction, internal load reduction, and restructuring of the pond in Heesch (see Table 2). As a consequence of the measures taken, the water quality was strongly improved in ponds Dongen and Heesch, and to a lesser extent in Pond Eindhoven (see Table 7). These results are in line with the expectation that the reduction in the nutrient load and ecosystem restructuring would lead to improved water quality in the ponds.

The water clarity in ponds Dongen and Heesch improved considerably after the rehabilitation, which seemed to be due to less nutrients available for phytoplankton and less resuspension of the sediment by fish. The removal of bottom resuspending fish such as carp may also have led to consolidation of the sediment [63]. In both ponds Dongen and Heesch, the incidence of a visible bottom increased strongly, which may have further benefitted macrophytes. A strong positive feedback exists between the water clarity and macrophyte abundance in shallow waters [16]. Macrophytes were introduced and visible in these two ponds, but their abundance was not quantified during the course of this study. In Pond Dongen, in 2020 and 2021, macrophytes surveys were performed that revealed a coverage of 80% by submerged macrophytes; *Ceratophyllum demersum* and *Myriophyllum* sp. were dominant [64].

These macrophytes may not only have further stabilized the sediment [65], but could also have provided refuge to zooplankton therewith increasing the grazing pressure on algae [66]. In Pond Eindhoven, the water clarity was improved marginally and no growth of the introduced macrophyte *Elodea nuttallii* occurred, as it clearly did not develop in the pond. Dredging of the pond (removal of ~40 cm sediment) and deepening of the pond to 4 m near the outflow resulted in insufficient light reaching the bottom to support macrophyte development. Using an online tool yielded that at 1.3 m, 4% of the incoming light remained, which is viewed as the boundary to support plant growth (https://onderwaterlicht.nl/nl/uitzicht.html, accessed on 22 May 2023).

The total chlorophyll-*a* concentrations in ponds Eindhoven and Heesch were similar over the post-intervention period (Appendix C), which was mostly a result of filamentous green algae being present in Pond Heesch, but phytoplankton in Pond Eindhoven. This is also supported by lower cyanobacterial chlorophyll-*a* concentrations in Pond Heesch compared to Pond Eindhoven during the post-intervention period (Appendix C). As mentioned before, cyanobacterial chlorophyll-a concentrations in the years after the intervention were low both in ponds Dongen and Heesch without any noticeable issues except in August 2012 in Pond Dongen, while on average they were reduced in Pond Eindhoven, occasional blooms and surface accumulations occurred, prompting the WA to issue warnings in the summers of 2012, 2014, 2016, and 2018 (Figure 6).

Figure 6. Picture of Pond Eindhoven with a massive cyanobacterial bloom comprised of *Microcystis* sp. and *Dolichospermum* sp. and of Pond Heesch that had clear water during the same heatwave period in The Netherlands (August 2018).

The mean TP concentration in Pond Eindhoven after intervention was 57% of the pre-intervention TP concentration, while in ponds Dongen and Heesch the TP concentrations after intervention were reduced to 6% and 17% the pre-intervention TP, respectively. Likewise, the TN concentration was more reduced in ponds Dongen and Heesch (18% and 37% of the pre-intervention TN) than in Pond Eindhoven (77% of the pre-intervention TN). The TN concentrations before rehabilitation in ponds Dongen and Heesch reached values that could lead to the disappearance of macrophytes, driven by shading from filamentous algae, phytoplankton, and periphytes [67]. However, in Pond Eindhoven, the TN concentrations were lower and in a range where macrophytes could grow [67]. In fact, the post-intervention TN concentrations in Pond Eindhoven were similar to those in Pond Dongen (Appendix C) and thus seem not to be the reason for the absence of macrophytes in Pond Eindhoven. The post-intervention Secchi disc depth in Pond Dongen was similar to the Secchi disc depth in Pond Eindhoven (Appendix C), which was caused by Pond Dongen being much shallower and a visible bottom being registered as the Secchi disc depth. The water clarity in Pond Dongen was, however, better than in Pond Eindhoven, which was confirmed by a significantly higher turbidity in Pond Eindhoven than in Pond Dongen (Appendix C). A visible bottom occurred regularly in Pond Dongen, while this was never observed in Pond Eindhoven. Hence, the clearer water favored submerged macrophytes in Dongen, while the deeper and more turbid water of Pond Eindhoven hampered growth of introduced macrophytes [68]. The rehabilitation measures had no influence on the oxygen levels and pH.

In Pond Heesch, the water quality improved greatly and remained as such for at least nine years post-treatment and in Pond Dongen for seven years. These ponds also withstood an extreme climate event; a record heat wave that struck large parts of Europe [69], while Pond Eindhoven developed a massive cyanobacterial bloom (see Figure 6). Those rehabilitated ponds provide evidence that to mitigate the effects of climate change, a strong reduction in the nutrient load combined with restored system resilience is required [70,71]. Lowering the nutrient loads is essential to bring the water body into a state where clear water can be realized, but stimulation of submersed macrophytes seems crucial in maintaining a clear water state and preventing/delaying the effect of fish recolonization [50]. The human vector, however, in fish recolonization should not be underestimated [72]. The pond rehabilitation program included communication with the neighborhood and stimulated the citizens and fishermen to reduce fish stocking drastically, and with it the often daily feeding of the fish [73]. Including the neighborhood and local fishing clubs in the entire process has paid out reasonably well. In both ponds Dongen and Heesch, no (additional) carp have been re-stocked according to the local angling societies, although in January 2018, 16 carp (40–70 cm length) were removed from Pond Dongen [74]. Observations showed that no excessive feeding is taking place and that dog outlets around ponds Dongen and Heesch is minimal. The latter, however, is not the case around Pond Eindhoven, where quite some citizens refuse to collect dog feces and leave it on the banks of the pond. The maintenance of the urban ponds is a shared responsibility of all stakeholders that can prevent degradation of the water quality and prolong the delivery of services of the pond to society and therewith to the quality of urban life.

Citizens (often children) use urban ponds for swimming (at least in ponds Eindhoven and Heesch), angling, boating, and other recreational activities. Thus, regular monitoring of those waters to track their state is recommended, as in general, such waters in the vicinity of urban settlements provide the most important contact citizens have with surface waters [12]. Within the EU Water Framework Directive (WFD), The Netherlands does not consider water bodies less than 50 ha. Taking into consideration that of the 117 million lakes on the planet, 90 million are between 0.2 and 1 ha and 23 million between 1 and 10 ha [75], including only lakes of 50 ha or larger in the WFD survey leaves a blind spot of almost 98% of the stagnant water bodies of which many are close to urbanized areas. Since urban ponds may suffer from cyanobacterial blooms, eutrophication control and reducing cyanobacterial nuisance in urban ponds should be of equal importance to water managers as is eutrophication abatement in the designated WFD waters [19]. The importance of small waters—including ponds—for biodiversity, biogeochemical processes, and ecosystem services, as well as the urgency to include them into monitoring programs and into catchment management has been outlined thoroughly in a recent review [1]. Our study adds to this that the water quality in ponds can be improved, yet that a system analysis should take place at the start of any lake and pond rehabilitation to elucidate the sources underlying the problem and enable tailor-made solutions with the highest chance for success. Combining measures that together act on both the nutrient load and water turbidity are recommended in rapid eutrophication abatement.

Although direct nutrient inputs (sewer overflows, feeding of water fowl, angling bait, et cetera) can often be tackled rather easily, many urban ponds will experience ongoing diffuse nutrient inputs, which means that regular maintenance will be needed. Sediment removal is a standard restoration technique for this type of pond, yet it could also affect the benthic fauna and macrophytes [60], and is rather costly. Dredging is certainly needed once in a while to maintain the water depth, or to remove the suffocating abundance of alien macrophytes [76]. Such maintenance combined with planting an assortment of indigenous plant species has also promoted dragonfly diversity [76]. Biodiversity has not been monitored in the ponds in this study, but given the macrophytes' presence in ponds Dongen and Heesch providing structure, and the absence of massive blooms in these two ponds, biodiversity likely improved after the rehabilitation. In addition, the transition from cyanobacteria-dominated water to clear water might also have lowered the greenhouse gas emissions [77]. Nonetheless, alternatives to rather invasive dredging in the maintenance of the ponds would be welcome to WAs. In that view, a P binder was included in the experiments that preceded the intervention in the ponds to test its efficacy compared to dredging [29,31]. Such geo-engineering amendments could be considered in ponds Dongen and Heesch as cheap and effective measures reducing the frequency of dredging. It should be noted that the accumulated sediment in Pond Dongen and very shallow water depth remaining left the WA no option other than to remove this sediment. A water quality monitoring plan is required to determine when the water quality deteriorates to intervene when needed before problems arise. A regular, cheap maintenance strategy that ensures a good water quality is effective in a German swimming lake [78]. Currently, only WA Brabantse Delta has an ongoing monitoring program in Pond Dongen, while the monitoring of the other two ponds stopped in 2018. Evidently, traditional monitoring of all surface waters is an impossible task, yet smart monitoring systems, online devices, and citizen science might offer feasible alternatives. Including citizens might also be pivotal to ensure prolonged success.

The results of the improved water quality in ponds Dongen and Heesch underpin the strength of combining measures to rehabilitate ponds. The importance of combining measures has also been reported for the Danish shallow lake Kollelev, where a combined treatment with P fixation (aluminum addition) and biomanipulation (fish removal) ensured a rapid improvement in the water clarity, whereas separate treatments were not effective [62]. The evaluation of the restoration of three ponds in the Netherlands provides a blueprint for restoring ponds and lakes in general: perform a system analysis to identify the drivers of the problem, which will probably lead to more than one measure to bring the lake or pond as fast as possible to the desired clear water state.

5. Conclusions

- A system analysis should take place at the start of each pond and lake rehabilitation.
- Mitigating measures should only be implemented when the system analysis has revealed their feasibility.
- Combining measures that together act on both the nutrient load and water turbidity are recommended in rapid eutrophication abatement.

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Appendix A

The chlorophyll-*a* data from the summer snapshot sampling in 32 ponds in the vicinity of the three ponds studied in this study were used for the comparison. The names and locations of these ponds are given in Table A1 and the concentrations per year in Figure A2.

The mean annual chlorophyll-*a*, total phosphorus, and total nitrogen concentrations of each pond were compared to the Water Framework Directive (WFD) status classification [37]. This WFD status is based on the classification for small (<50 ha), shallow (<3 m), and buffered (1–4 meq L^{-1}) stagnant surface waters (Figure A1).

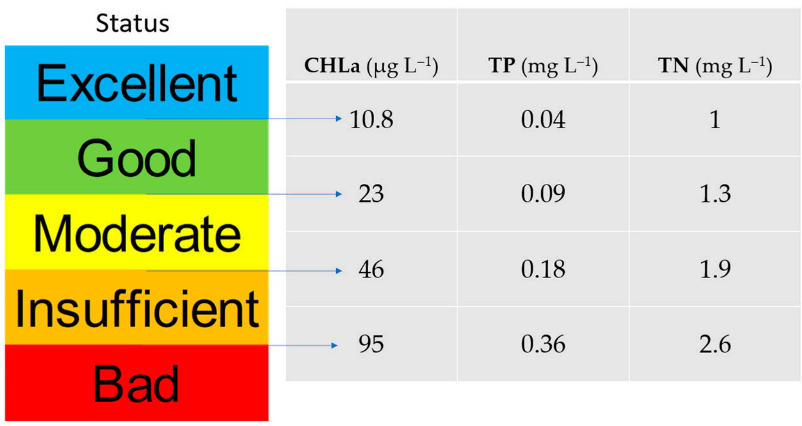

Figure A1. The Water Framework Directive status classification for small (<50 ha), shallow (<3 m), buffered (1–4 meq L^{-1}) stagnant surface waters. Waters with a total chlorophyll-*a* concentration <10.8 µg L^{-1}, total phosphorus concentration (TP) <0.04 mg L^{-1}, and a total nitrogen concentration (TN) <1 mg L^{-1} would be classified as excellent; between 10.8 and 23 µg L^{-1} chlorophyll-*a*, 0.04–0.09 mg TP L^{-1}, and between 1 and 1.3 mg TN L^{-1} as good. The range of chlorophyll-*a* concentrations between 23 and 46 µg L^{-1} classifies as moderate, between 46 and 95 µg L^{-1} as insufficient, while chlorophyll-*a* concentrations exceeding 95 µg L^{-1} will yield a bad status.

Table A1. Overview of 32 ponds (The Netherlands) from snapshot summer sampling.

Place/Location	Latitude	Longitude	Name Water Body
Asten	51°24′18.38″	5°44′53.50″	Vijver in Burg. Ploegmakerspark
Beek and Donk	51°32′03.63″	5°37′41.56″	Vijver aan Otterweg
Bennekom	51°59′43.91″	5°40′11.72″	Vijver aan Kierkamperweg
Boxtel	51°36′03.94″	5°18′54.10″	Vijver aan Parkweg
Boxtel	51°35′53.55″	5°18′22.86″	Essche Heike
Breda	51°36′00.15″	4°46′56.12″	Vijver in Liniepark
Budel	51°14′02.38″	5°35′56.44″	Ringelsven
Deurne	51°26′55.19″	5°47′14.97″	Vijver Burgemeester Roefslaan
Ede (1)	52°01′27.05″	5°38′43.88″	Vijver aan Verenigde naties
Ede (2)	52°02′03.84″	5°38′56.83″	Vijver aan Jachtlaan
Ede (3)	52°02′20.99″	5°38′43.67″	Vijver bij Valkestein
Etten-Leur	51°34′09.41″	4°39′02.74″	Vijver aan Vlaamse Schuur
Grave	51°45′08.24″	5°44′57.30″	Vijver aan Anna van Burenweg
Heesch	51°44′09.00″	5°32′24.65″	Vijver aan Langven
Helmond	51°29′05.16″	5°38′29.25″	Warandevijver
Maarheeze	51°18′22.25″	5°37′04.85″	Vijver aan Poelsnep
Middelrode	51°39′51.47″	5°25′21.73″	Vijver Christinastraat
Roosendaal	51°30′41.82″	4°26′42.02″	Vijver Dadelberg
Roosendaal	51°30′49.33″	4°26′43.82″	Vijver Dubbelberg
Roosendaal	51°30′52.84″	4°26′34.14″	Vijver Enclaveberg
Schijndel	51°37′14.84″	5°26′43.68″	Vijver Renate Rubinsteinstraat
St-Michielsgestel	51°38′44.51″	5°21′49.85″	Vijver Moerschot
Sint-Oedenrode	51°34′27.41″	5°26′58.30″	Visvijver in Park De Kienehoef
Someren	51°22′54.31″	5°42′21.23″	Vijver aan Wilbertshof
Son	51°30′48.98″	5°29′16.10″	Vijver aan Europalaan
Tilburg	51°32′09.34″	5°05′35.02″	Vijver aan de Berglandweg
Tilburg	51°32′21.42″	5°05′05.20″	Vijver aan de Kaukasusweg
Tilburg	51°32′14.66″	5°04′35.74″	Stappegoor
Tilburg	51°32′36.19″	5°06′16.13″	EsscheStroom Vijver
Tilburg	51°35′32.91″	4°59′46.56″	Vijver Hoge Witsie
Valkenswaard	51°20′19.88″	5°28′05.84″	Dragonder
Wageningen	51°58′07.98″	5°40′38.33″	Dreyenvijver

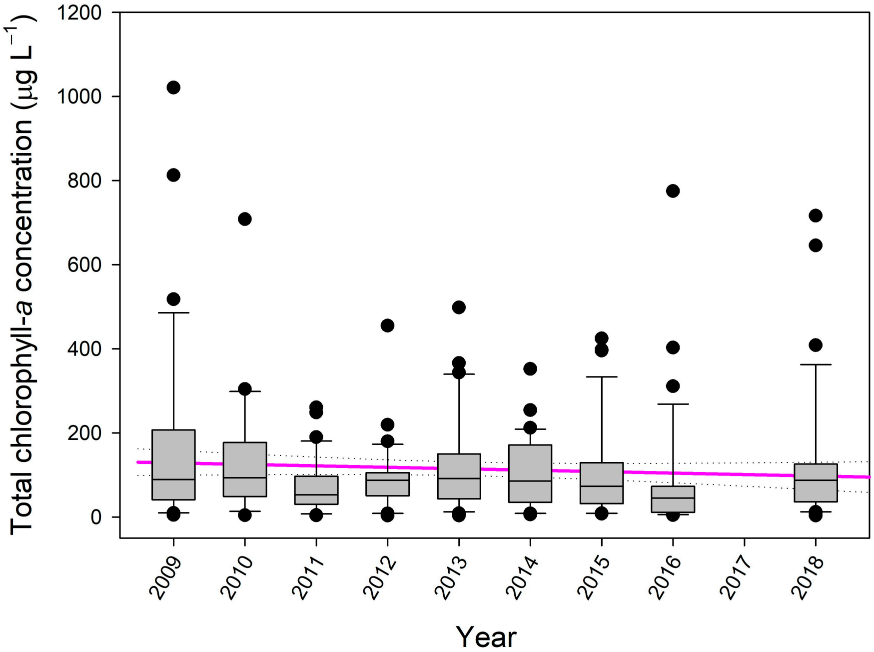

Figure A2. Boxplots of total chlorophyll-*a* concentrations (µg L^{-1}) from summer snapshot samplings in 32 ponds. The pink line indicates linear regression, the dotted lines the 95% confidence interval.

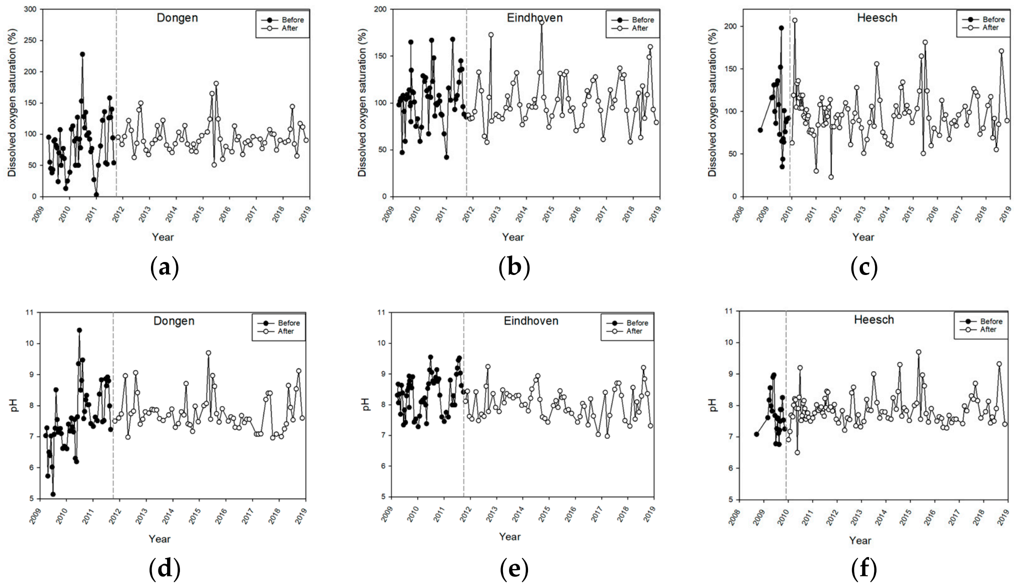

Figure A3. Course of the total dissolved oxygen saturation (%) in Pond Dongen (**a**), Pond Eindhoven (**b**), and Pond Heesch (**c**) as well as the pH in Pond Dongen (**d**), Pond Eindhoven (**e**), and Pond Heesch (**f**) before the restoration intervention (black symbols) and after the intervention (open symbols). The dotted vertical lines indicate the moment of intervention in each pond.

Table A2. Overview of before-after comparison of dissolved oxygen saturation and pH in the ponds in periods before restoration interventions took place and after restoration.

Dissolved Oxygen Saturation			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p < 0.05$	$p < 0.05$
Mann–Whitney	U = 1398.5	U = 1759.5	U = 1104.0
Rank Sum Test	$T_{53,65} = 2829.5$ $p = 0.080$	$T_{57,68} = 3769.5$ $p = 0.376$	$T_{23,102} = 1357$ $p = 0.906$
pH			
Statistical test	Pond Dongen	Pond Eindhoven	Pond Heesch
Normality test	$p < 0.05$	$p = 0.213$	$p < 0.05$
Mann–Whitney/t	U = 1386.5	U = 972	U = 882.5
Rank Sum Test	$T_{53,65} = 2817.5$ $p = 0.069$	$T_{57,67} = 4500$ $p < 0.001$	$T_{23102} = 1158.5$ $p = 0.064$

Appendix B

The Pearson Product Moment Correlation (not assuming a form of response) was used to examine if trends were present in the data before and/or after the interventions (Table A3).

Table A3. Pearson correlation coefficients (corr.), *p*-values of trends and number of samples/measurements (*n*) for selected water quality variables in the period before intervention and after intervention in the three ponds (Dongen, Eindhoven, Heesch).

Variable	**Before**			**After**		
	corr.	***p*−Value**	***n***	**corr.**	***p*−Value**	***n***
Total Chlorophyll						
Dongen	−0.207	0.137	53	−0.127	0.309	66
Eindhoven	−0.263	0.050	56	−0.219	0.073	68
Heesch	−0.321	0.135	23	−0.164	0.099	103

Table A3. *Cont.*

Variable	Before			After		
	corr.	$p-$Value	n	corr.	$p-$Value	n
Cya−Chlorophyll						
Dongen	−0.334	0.014	53	−0.153	0.232	63
Eindhoven	−0.098	0.474	56	0.062	0.619	66
Heesch	−0.262	0.227	23	0.001	0.990	103
Total Phosphorus						
Dongen	−0.048	0.724	56	−0.047	0.710	65
Eindhoven	−0.348	0.013	51	−0.331	0.006	67
Heesch	0.025	0.910	23	−0.555	<0.001	101
Total Nitrogen						
Dongen	−0.266	0.059	51	0.214	0.087	65
Eindhoven	−0.072	0.617	51	−0.010	0.935	67
Heesch	0.259	0.244	22	0.066	0.516	100
Secchi disc depth						
Dongen	0.055	0.697	53	0.428	<0.001	63
Eindhoven	0.212	0.110	58	0.385	0.001	68
Heesch	−0.386	0.069	23	0.181	0.071	100
pH						
Dongen	0.562	<0.001	53	−0.055	0.664	65
Eindhoven	0.293	0.026	58	−0.052	0.675	67
Heesch	−0.147	0.502	23	0.078	0.436	102
Turbidity						
Dongen	−0.341	0.014	52	−0.228	0.068	65
Eindhoven	−0.280	0.035	57	−0.245	0.046	67
Heesch	0.455	0.029	23	−0.164	0.104	99
Oxygen saturation						
Dongen	0.309	0.024	53	−0.012	0.926	65
Eindhoven	0.149	0.267	57	0.111	0.367	68
Heesch	−0.188	0.403	22	0.007	0.941	102

Appendix C

To compare the absolute values of the water quality variables between ponds over the post-intervention periods, a one-way ANOVA on Ranks was performed followed by a multiple comparison to identify which medians differed from each other (Dunn's Method).

The total chlorophyll-*a* concentrations after the intervention were significantly lower in Pond Dongen than in ponds Eindhoven and Heesch (one-way ANOVA on Ranks, H_2 = 34.7; p < 0.001), while the median chlorophyll-a concentrations in the latter were similar (All Pairwise Multiple Comparison Procedures, Dunn's Method).

The cyanobacterial chlorophyll-*a* concentrations after intervention were significantly higher in Pond Eindhoven than in ponds Dongen and Heesch (one-way ANOVA on Ranks, H_2 = 15.4; p < 0.001), while the median cyanobacterial chlorophyll-*a* concentrations in the latter were similar (All Pairwise Multiple Comparison Procedures, Dunn's Method).

The post-intervention TP concentrations were significantly different in each pond (one-way ANOVA on Ranks, H_2 = 37.4; p < 0.001 followed by Dunn's Method), the median TP in Dongen was 20.0 µg P L^{-1}, the median TP in Eindhoven was 33.4 µg P L^{-1}, and the median TP in Heesch was 56.4 µg P L^{-1}.

The post-intervention TN concentrations were significantly higher in Pond Heesch than in ponds Dongen and Eindhoven (one-way ANOVA on Ranks, H_2 = 16.3; p < 0.001), while the median TN concentrations in the latter were similar (All Pairwise Multiple Comparison Procedures, Dunn's Method).

The post-intervention Secchi disc depth was significantly higher in Pond Heesch than in ponds Dongen and Eindhoven (one-way ANOVA on Ranks, $H_2 = 20.9$; $p < 0.001$), while the median Secchi disc depths in the latter were similar (Dunn's Method).

The post-intervention turbidity was significantly higher in Pond Eindhoven than in ponds Dongen and Heesch (one-way ANOVA on Ranks, $H_2 = 22.1$; $p < 0.001$), while the median turbidity in the latter were similar (Dunn's Method).

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