NICHES

Nature's integration in cities' hydrologies, ecologies and societies

D2.1 Module on ES provisioning

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1 Preface

This background of this report lies in developing a link between an aquatic ecosystem model and a quantitative approach to the delivery of ecosystem services. Doing so will deliver a toolbox capable of estimating ES delivery in aquatic systems under future scenarios. Predicting possible future ES delivery is key in the urban waterscape in relation to sewage overflow events and extreme rainfall, as they are expected to change markedly in the future. Simultaneously, as combined sewage systems are technical systems, they offer the ability to be adapted in such a way that the impacts of CSOs on aquatic ecosystems and the ecosystems services they provide are mitigated.

2 Summary

NICHES advances scientific knowledge on restorative NBS through the application and testing of impact assessments, models and transitional governance models for improved urban drainage in five cases within and beyond Europe. The project hypothesizes and aims to demonstrate that sustainable transformations of cities based on restorative NBS which enhance water retention capacities in urban areas could widely mitigate impacts from combined sewers on aquatic ecosystems. As the urban catchment is part of a multi-owner landscape with associated stakeholder conflicts linked through teleconnections and multi-scale governance structures, the involvement of diverse stakeholders and their values from the NICHES core cities is vital to co-design the impact assessment and ES module design and to ensure maximal applicability. This deliverable describes the development of an aquatic ecosystem services module for the Rotterdam case study. In short, we build on an existing framework (Seelen et al., 2021) where ecosystem service delivery is determined based on threshold values of water quality and ecological variables. Rather than determining these variables from field-based measurements we retrieve them from an ecosystem model, PCLake+, which is widely used within the Netherlands by water management. In doing so, the ecosystem service provisioning may be estimated not only under the current conditions but also under scenarios of future conditions. We validated and exemplify its use by applying it to the waterscape of Rotterdam where we test the impact of increasingly intense rainfall events. In conclusion, the model is well suited to model changes in ES delivery due to increasing CSO events. By taking a quantitative approach to we lay the groundwork for a risk-based assessment of CSO impacts on ecosystem.

3 List of abbreviations

EU	European Union
ES	Ecosystem Service
CSO	Combined Sewage Overflow
AEM	Aquatic Ecosystem Model
WFD	EU Water Framework Directive

4 Development of an Ecosystem Services (ESs) provisioning model

Healthy freshwater ecosystems can provide vital ecosystem services (ESs), but this capacity may be hampered due to water quality deterioration and climate change. In the urban waterscape combined sewer overflows (CSOs) form a direct threat to the quality of aquatic ecosystems and the species that inhabit them. Additionally, many of the services that the urban populace depends on (e.g., recreational fishing, swimming, carbon and nutrient retention) are threatened by loss of ecological quality caused by CSOs. CSO events are expected to increase with increasing intensity in rainfall (van der Werf et al., 2023) and hence there is a need to understand how increased CSO events will impact both ecological functioning as well as service provisioning.

There is an urgent need to identify new solutions for reducing the impact of increased precipitation both on sewage systems and aquatic ecosystems. Nature-Based solutions (NBS) offer an alternative to the existing technical stormwater management systems, having the potential to alleviate pressure during high rainfall events while also providing wider societal and environmental benefits. Societal uptake remains may be hampered due to lacking evidence, approaches and targeted guidance that take the wider social-ecological-technological system (SETS) into account. NICHES aims to fill this gap by defining a holistic SETS framework for understanding restorative NBS for urban runoff mitigation and the resultant reduction of impacts on aquatic systems and resulting ecosystem services.

In the context of CSOs, there are characteristics of the sociological, ecological and technological urban waterscape systems that determine vulnerability to CSO events of receiving water bodies, its exposure to CSO events, and its capacity to adapt to CSO events (Figure 1). Ecosystem services provide a link between the ecological and the sociotechnological system, and a quantitative understanding of ES provisioning under CSO events will provide us a deeper understanding of SETS in the context of urban waterscapes.

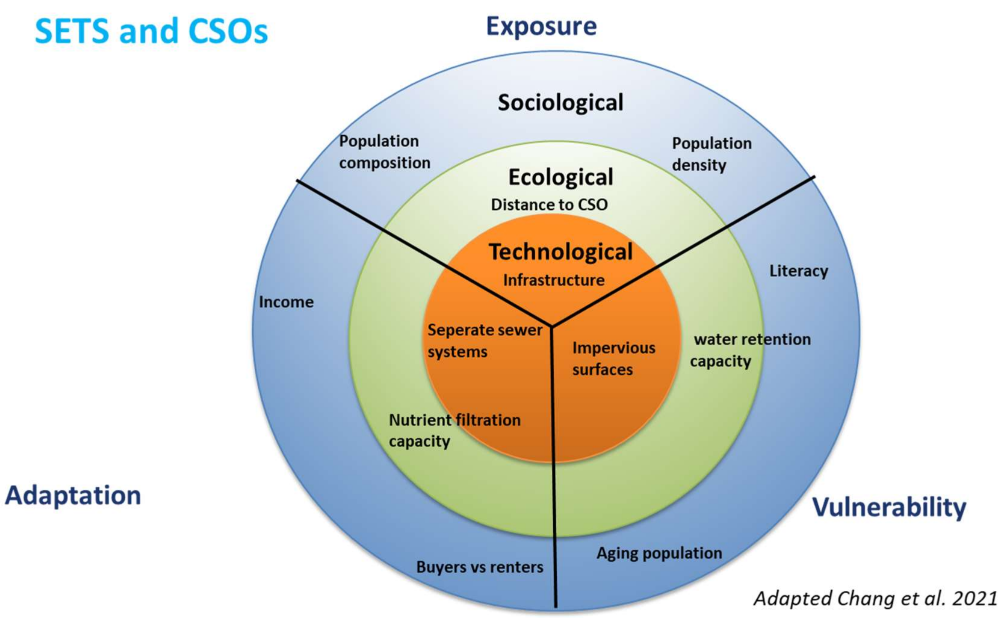

Figure 1. Scheme illustrating how the three systems of SETs (sociological, ecological and technological) relate to CSO events and how different aspects of the systems are influencing aspects present in a risk framework approach. Figure adapted based on (Chang et al., 2021). Increasing the nutrient filtration capacity of urban waterscapes through constructed wetlands and wadis could be viewed as a Nature Based Solution application.

4.1 The need to link aquatic ecosystem models to ecosystem services provisioning estimation

Quantifying ecosystem services can be instrumental in recognizing the benefits humans receive from ecosystems, providing stronger arguments for ecological restoration (Grizzetti et al., 2019; Guerry et al., 2015). Conveying restoration impacts in terms of the loss or gain of ESs can facilitate effective communication of restoration outcomes to policy-makers and river basin authorities responsible for implementing restoration measures (Wortley et al., 2013). While modeling terrestrial ecosystem services often focuses on mapping ESs provisioning through spatial variations of catchment attributes (e.g., land use, topography, lithology) (Nelson et al., 2009), the non-linear dynamics of water quantity and quality necessitate a more explicit consideration in aquatic ecosystem service modeling (Grizzetti et al., 2016).

There is increasing evidence that freshwater ecosystem services provisioning is closely linked to the ecological quality (or ecological state) of different aquatic environments, including shallow lakes (Janssen et al., 2021), deep lakes (Seelen et al., 2022), rivers, and coastal waters (Grizzetti et al., 2019). Based on data reported under the European Water Framework Directive (WFD), Grizzetti et al. (2019) demonstrated that higher provisioning of ESs is mostly correlated with more desirable ecological states (i.e. clear, submerged plant dominated waters), particularly for regulating services (e.g., water purification, erosion retention, flood protection) and cultural services (e.g., recreation). However, current modeling tools for water-related services primarily focus on water quantity (Grizzetti et al., 2016), with limited integration of services closely related to water quality (Keeler et al., 2012).

Water quality dynamics are mediated by complex interactions among a myriad of ecosystem processes, which are often oversimplified in large-scale modeling frameworks. For instance, one widely-used ecosystem service model, InVEST, simplifies by using nutrient loading as a proxy for determining the availability of lake-related ESs (Nelson et al., 2009; Polasky et al., 2011), assuming simple linear responses of ecosystems to nutrient loading. This approach contradicts the resistance theory of (Gómez-Baggethun & Ruiz-Pérez, 2011; Ibelings et al., 2007), which supports threshold-type ecosystem responses to pressures. Consequently, the assessment of management actions often relies on variables collected at the landscape scale (Burkhard et al., 2012), which may be inaccurate due to the aforementioned nonlinear responses or ill-fitting when assessing the impacts of in-lake restoration measures (Lürling & Mucci, 2020). Keeler et al. (2012) proposed a conceptual framework linking ecological-related services with corresponding water quality variables based on a review of existing ES models, emphasizing the importance of this link in assessing management actions. Given the long-history of development of AEMs (Janssen et al., 2015), linking water quality variable outcomes of these models to ESs provisioning approaches is a logical next step to capture the full dynamics of how water quality dynamics impacts ESs.

Water management and policy require tools to estimate how robust their current measures for restoring ES provisioning are in light of future climate conditions. To effectively addressing this need requires a model capable of capturing non-linearity in aquatic ecological responses as well as how ecological outcomes translate into ESs provisioning. To this end, we propose the use of an existing aquatic ecosystem model (AEM) and development of an ESs module. AEMs have a long history of development and application to practical water management and policy questions (Janssen et al., 2015) making them ideally suited as a foundation for developing a coupled AEM-ES model.

4.2 Expanding the AEM PCLake+ model to include ESs provisioning

In this study, we use the AEM PCLake+. PCLake+ is a process-based ecological model that was developed to simulate water quality and assess the trophic state of lakes based on ecological interactions (Janse, 2005, p. 200; Janssen et al., 2019). The model is a 0D model and assumes either a fully mixed water column connected to a sediment layer, or a two-layer water column differentiating between epilimnion and hypolimnion when water is stratified. Water exchanges take place through both diffusion and advective transport of water using simplistic conservative exchange between compartments (concentration differences, direct advective exchange based on flow differences). PCLake+ models nutrient cycling including nitrogen and phosphorus and a simple food web consisting of three functional groups of phytoplankton (cyanobacteria, green algae and diatoms), zooplankton, and fish. The model is widely used to assess effective management strategies for water bodies in the Netherlands and worldwide (Andersen et al., 2020; Janse et al., 2008; Wang et al., 2019). The model is able to capture well the state-shifts that can occur in inland waters, when nutrient loading forces a system to transition from a clear macrophyte-dominated state to a turbid phytoplankton-dominated state. In shallower systems, this state-shift is a step-change happening at a specific nutrient loading, defined as the critical nutrient loading. Importantly, due to a process called hysteresis, this step-change happens at a different transition point from clear to turbid, then from turbid to clear. As such the system can be in two alternative states. In deeper systems, however, this transition happens more gradually.

The model has also been used to estimate impacts on ecological and water quality of climate change and changing socio-economical scenarios (Mooij et al., 2007; Yang et al., 2022). Here we expand the model with threshold-based ecosystem service delivery (Seelen et al., 2021) based on its existing ecological outcomes.

We followed a framework for assessing ecosystem services proposed by Seelen et al. (2021), which links ecosystem state indicators with ecosystem service provisioning through a threshold approach. The threshold values reflect the values that certain water quality parameter required to support the provision of a given service. Threshold values were based on published peer-reviewed literature, a field campaign covering 51 quarry lakes in the south of the Netherlands, and expert judgment (Table 1; see Seelen et al. 2021 for the supporting materials). Per ecosystem service, different aspects of the water quality requirements of the service are considered. For instance, the service of swimming is only suitable when the lake has sufficient transparency, the cyanobacterial biomass is at a safe level, and there is a part of the water column present that is not overgrown with vegetation. In the ES module, the suitability of delivering each ES was expressed by an indicator function ranging between 0-1, with "1" representing a fully suitable provisioning, "0" representing an unsuitable provisioning, and values in between representing a moderate suitability.

In total, nine ESs are modelled with their water quality requirements summarized in Table 1. We followed the Common International Classification of Ecosystem Services (CICES; (Haines-Young et al., 2012)), in which the ESs are divided into four different groups: provisioning (water, materials, energy and others), regulation and maintenance (remediation and regulation of the biophysical environment, flow regulation, regulation of the physic-chemical and biotic environment), cultural (physical or experiential use of ecosystems, intellectual representations of ecosystems), and abiotic (abiotic materials, energy, and space) services. We selected the ecosystem services based on the constraints of the quantitative ecological state indicators able to be computed by PCLake+. The current selection however, can be expanded in the future in further model developments.

Table 1. List of Ecosystem Services, their corresponding ecosystem state indicators and threshold values being included in the modeling framework. Details on the choice of ecosystem state indicator thresholds are given in more detail in Seelen et al. (2021). Some services may be considered in multiple categories depending on cultural context (e.g. fishing may be provisioning or cultural depending on its

Category	*Service*	*CICES Code*	*Ecosystem state indicators*	*Threshold values*

Provisioning	Fishing	1.1.4.1	Steady state fish density (kg/ha)	>100 (suitable), 10-100 (moderate), < 10 (unsuitable)
	Common reed (P*hragmites australis*) production for roof thatching	1.1.5.2	Helophytes shoot biomass (marsh zone, g DW/m^2)	>2500 (suitable)
	Irrigation	4.2.1.2	Cyanobacterial chlorophyll-a (ug/L)	<12 (suitable), 12-75 (moderate), >75 (unsuitable)
Regulation and maintenance	Nutrient (P and N) burial in lake sediment	2.2.4.2	Reduction phosphorus/nitrogen load (%)	>50 (suitable), 20-50 (moderate), <20 (unsuitable)
	Maintenance of habitats for WFD	2.2.4.2	Surface coverage (%) with sufficient light (>4%)	>60 (suitable), 30-60 (moderate), <30 (unsuitable)
	Particle capture between macrophytes	2.1.1.2	Macrophyte biomass (gDW/m^2)	>200 (suitable), 20-200 (moderate), <20 (unsuitable)
Cultural	Swimming	6.1.1.1	Transparency (Secchi depth, m)	>1.5 (suitable), <1.5 (unsuitable)
			Cyanobacterial chlorophyll-a (ug/L)	<12 (suitable), 12-75 (moderate), >75 (unsuitable)
			Plant nuisance: vegetation-free water column (m)	>0.5 (suitable), 0-0.5 (moderate), <0.5 (unsuitable)
	Bird watching	3.1.1.2	Fish density (kg/ha)	<67 (unsuitable), >67 (suitable)
			Helophyte density in littoral zone (g DW/m^2)	<73 (unsuitable), >73 (suitable)
			Transparency (Secchi depth, m)	< 1.5 (unsuitable), 1.5-5 (moderate), >5 (suitable)

Below, we explain the adjustment we made a to the thresholds proposed by Seelen et al., 2021.

4.2.1 Macrophyte habitat provisioning

For evaluation of the ES macrophyte habitats, we used a critical depth with 4% light of the surface light reaching the lake bottom, as an average proxy for the marginal depth that has sufficient light for possible germination, following Dobberfuhl (2007) and Kemp et al. (2004). In PCLake+ light attenuation over depth is calculated by Lambert-Beer equation:

$$I_{(z)} = I_{(0)} \times e^{-kz} \qquad (1)$$

In which $I_{(z)}$ represents light (W m^{-2}) at depth z (m), $I_{(0)}$ represents surface light (W m^{-2}), k represents attenuation coefficient (m^{-1}). With a $I_{(z)}/I_{(0)} = 4\%$, we can calculate the critical depth, denoted by z_{crit}. The zone shallower than this critical depth was considered to be suitable for macrophyte habitats. The corresponding coverage was calculated assuming a simplified lake topography, i.e. linear bank slope. The wetland zone is assumed to be always suitable for macrophyte habitat, and added into the coverage value accordingly. The corresponding equation is illustrated as follows:

$$Coverage = 1 - ((z - z_{crit})/z)^2 \qquad (2)$$

4.2.2 Swimming and irrigation

For services that are dependent on cyanotoxin concentration (swimming and irrigation), as suggested in Seelen et al. (2021), we used cyano-chlorophyll *a* (Chl-*a*) as a proxy instead. PCLake+ is unable to model cyanotoxin, but capable of estimating cyano-chlorophyll *a* based on the directly modelled cyanobacteria biomass. The corresponding threshold values of cyano-chlorophyll *a* were based on Dutch cyanobacteria protocol 2020 (Schets et al., 2020).

4.2.3 Inputs to the model

To run our developed coupled AEM-ES PCLake+ model a number of input parameters are required. While PCLake+ has over 500 parameters, a large part of these parameters does not need to be changed by users as they result from the generic calibration of the model (Janse et al., 2010). Users are primarily required to define the boundary conditions of their own water system in terms of inflows (water and nutrients), climate and meteorology (precipitation, evaporation, irradiance) and lake properties such as depth, lake area and fetch (Figure 2). Water temperature can be estimated based on simple parameters defining variation around a mean temperature, or time series of water temperature from either measurements or physical lake models (e.g. Flake (Kirillin et al., 2011), General Lake Model (Hipsey et al., 2019)) can be used. The required inputs to the model closely align with the type of information available from climate models and the type of knowledge gathered in the construction of river basin management plans in the WFD.

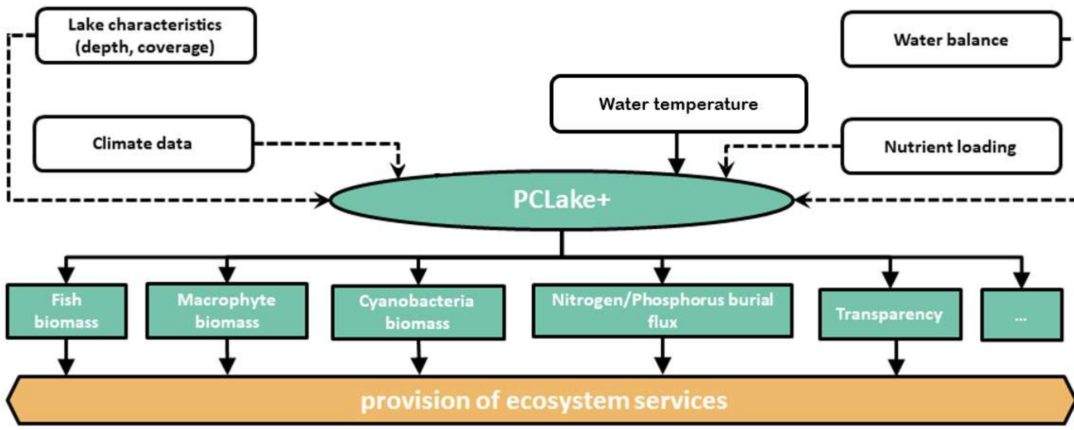

Figure 2. Model chain for ecosystem service modeling. Rectangles denote state variables, ovals denote models, hexagon denotes ecosystem service module, rounded rectangles denote input data, solid arrows denote model input or output, dashed arrows denote data input. (PCLake+ in green, input in white, output in orange).

5 Application of the AEM-ES model to the city of Rotterdam

5.1 CSO events in Rotterdam

The city of Rotterdam is a prime example of an urban area where the presence of CSOs provide a risk for the water quality of the waterscape. The municipality of Rotterdam has monitored the sewage overflow events since 2018 at all known CSO outfalls, leading to a good insight into the current state of CSO events. Data from the municipality reveals that on average CSO events occur 67 times each year across the city (2018-2022), with CSO events occurring at a given location 2.4 times each year on average. However, variation between CSOs and geographic regions is substantial, with some locations overflowing as much as 12 times in a year, whereas others seldom to never exhibit CSO events.

5.1.1 Predicting CSO events

CSO events are linked to rainfall intensity, both on short- and longer-term frequency (hours to multiple days). Hence, we extracted spatially explicit (1000x1000m raster) hourly rainfall data (in mm/h) from the Royal Dutch Meteorological Institute (KNMI) for each of the locations of the CSOs. Subsequently, we used a logistic regression to model the occurrence of CSO events as function of rainfall intensity of the last 24 hours, 48 hours, 72 hours and 7 days. A model including all factors proved to have the best fit (McFadden's Pseudo R^2: 13.8%) as well as the lowest AIC value (3956) compared to models with only a subset of the rainfall intensity predictors. The resulting logistic regression model (figure 3) can subsequently by used to predict the chance of a CSO event occurring given on rainfall intensity.

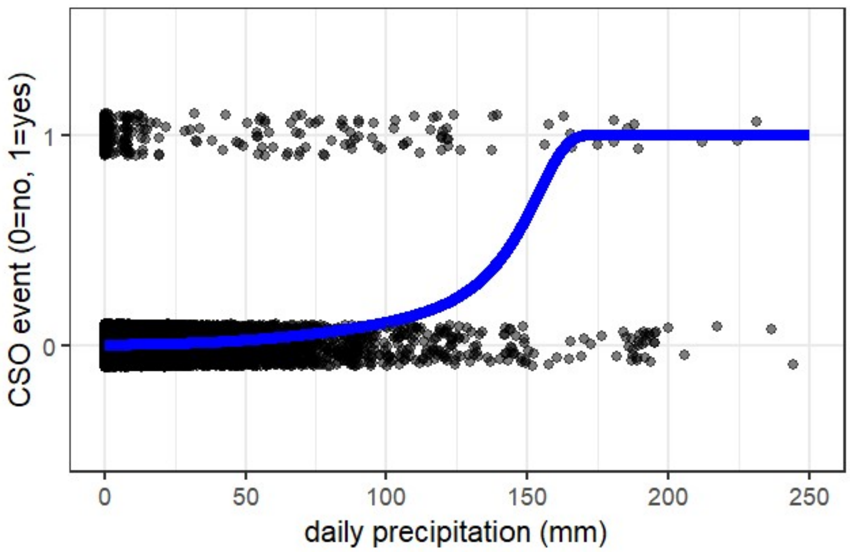

Figure 3. Illustration of the CSO event data collected by the municipality of Rotterdam plotted against the daily precipitation values. In blue, the modelled probability of a CSO event occurring based on daily, two daily, three daily and weekly sums of precipitation at the location of the CSOs.

5.1.1.1 Probabilities of CSO events along a gradient of intensifying rainfall

To examine the impact of increasingly intense rainfall we use our logistic regression model (see above) to predict the chance of CSO event occurrence for a range of rainfall intensities. To do so, we first calculated the number of dry and wet days from 2018 to 2022 (i.e. dry: 0 mm of rainfall, wet >0 mm of rainfall). Subsequently, we calculated the percentile of each daily rainfall intensity within a 10% percentile increments of the entire data (from the lowest 0-10% up to highest 90-100% values in the precipitation data set). We generated a randomly drawn set of the size of the number of wet days given a certain percentile range and shuffled them together with a set of zero values the size of the number of dry days. With this, we created randomized precipitation patterns of increasing intensity. By calculating the 48-hour, 72-hour and 7-day precipitation sums of this artificial data we were able to apply the logistic regression model to calculate a probability of CSO events occurring at each given day. Note that we take a generic, first-order approach here and do not account for specifics of CSO outfalls and their sewage sheds such as infiltration and storage capacity.

5.1.2 Impact of CSO events on ecological state of water

5.1.2.1 Defining a baseline for model runs

We start by defining a baseline for the PCLake+ model to compare the impacts of CSO events against. This baseline represents a relatively wide stagnant water body in the city of Rotterdam, a common water body type encountered in the city in the form of ponds and canals. We assume a water body of 1.5 meters depth with an organic-clayish sediment type and consistent inflow of 22mm per day (inflow of water from upstream sources, including rainfall). Evaporation and water temperature were based on sine waves functions that mimic seasonal dynamics of Dutch waters (Janse et al., 2005). Nutrient loading is integral in determining aquatic ecosystem state, however loading in urban contexts can vary markedly due to its wide range of possible sources (see e.g., Teurlincx et al., 2019). Hence, rather than determine specific loads for each water body in the city we use an exploratory approach aimed at seeing if sewage overflow frequency can cause state shifts in waters in Rotterdam. To do so we first calculated the load-response curve of nutrient loading to vegetation biomass (figure 4). We see an increase in submerged vegetation biomass with increasing loading (an eutrophication effect), up to a point of collapse where phytoplankton replaces vegetation dominance (near-zero vegetation biomass in figure 4). This load response curve shows the critical nutrient loads, the amount of loading where the system switches from clear to turbid, or turbid to clear. Conforming to alternative stable state theory (Scheffer et al., 1993, Kefi et al.,2016), these transition points are not situated at the same nutrient loading but are dependent on the starting condition of the water body (clear vs turbid).

Based on these critical nutrient loads, we choose a nutrient loading below the transition point from turbid back to the clear state (2.02 mgP/m^2/day) and start our model from a clear state. The chosen baseline loading lies outside of the hysteresis zone, making it so that small perturbations to the system should not lead to an abrupt collapse. CSO events will need to be sufficiently impactful to push the system into the hysteretic zone (between 2.3 and 4.7 mgP/m^2/day) or past the critical transition point from clear to turbid (4.7 mgP/m2/day).

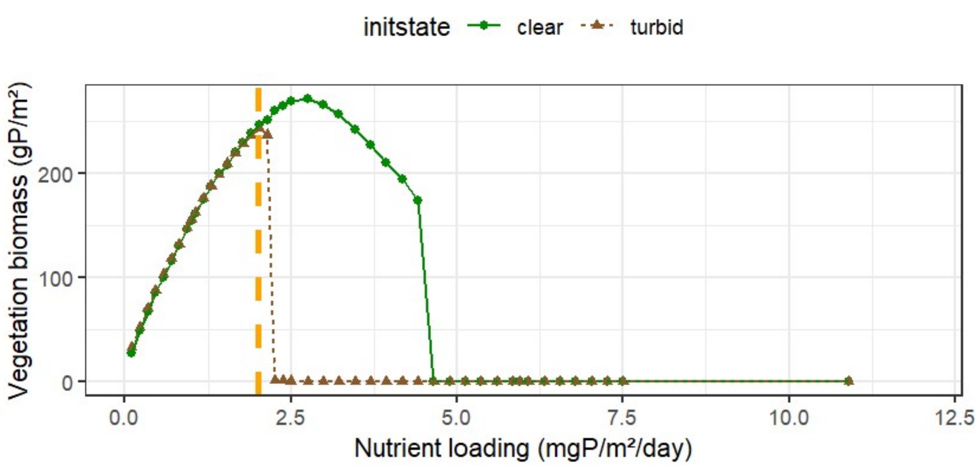

Figure 4. A load-response curve between nutrient loading and submerged vegetation biomass. Each symbol represents a summer (day 150-210) mean of vegetation of a 30-year PCLake+ model run. Models were run across a range of nutrient loading and with two different initial starting conditions (clear and turbid). The yellow dashed line indicates the baseline model nutrient load from which the impacts of CSO events are analysed.

5.1.2.2 A gradient of CSO event occurrences

We defined two factors that influence the incidence of a CSO event occurrence and hence were used to create a gradient of CSO event occurrences, namely as a function of intensity of rainfall (see 5.1.1.1 for more information) and the number of CSO locations connected to a water body. As we randomized dry and wet days, the timing of CSO occurrences could vary, influencing model outcomes. Hence, we repeated all model runs four times with the goal of creating a wider spread of CSO event impacts. Each model starts from a baseline condition (see 5.1.2.1) and runs for 20 years without CSO impacts. After this, 10 years of rainfall with subsequent CSO events were modelled to assess ecological vulnerability. Here, we define ecosystem vulnerability, as the loss of aquatic vegetation and the subsequent increase in turbidity.

In the case of a CSO event, we assume an average CSO flow of 134.4 mm, based on municipality data. For concentrations of nutrients in the sewage overflow water were based on research from RIONED, the Dutch foundation for urban sewage management (P: 3.1 mg/L, N: 12.5 mg/L). To also account for the flow of organic matter to the water systems we took the average organic concentration in sewage overflows (Suspended organic material: 230 mg/L) used in the OxyVal model, a model used to assess anoxia risks by Dutch water management (Tanis et al., 2018). Said concentrations were multiplied with the incoming CSO flow and added to the baseline loading received by the water body in the model. Inflow of water was also added to the base inflow of the model.

5.1.2.3 Impact of gradients of CSO frequency on ecological vulnerability

Next, we aimed to assess the impact of CSO events on ecosystem vulnerability. Figure 5 illustrates that the approach taken leads to a distribution of CSO event occurrences, ranging from 0-120 events per 10 years. Along this gradient of CSO occurrences we see that ecological vulnerability decreases, illustrated by the loss of aquatic vegetation at high CSO event occurrences. Similarly, we see an increase in Chlorophyll-a with increasing CSO event occurrences, showing a shift towards more waters being in a turbid state. Variation present at similar levels of CSO events is due to the temporal occurrence of CSOs within the 10-year time frame. As days of precipitation are randomly distributed over the 10-year time period and given a rainfall intensity (mm) based on the percentile of rainfall for the given scenario, there is a great deal of variation possible in consecutive days in terms of rainfall. The chance of CSO events depends on up to 7 days of cumulative precipitation, leading to an increasing chance of a CSO event when random occurrence leads to multiple consecutive days of rain. Moreover, as days of rain are randomly distributed over a 10-year time period, an increasing occurrence of rainfall near the end of the 10-year period will increase the chance of observing an ecological collapse (lack of vegetation) at the end of said period. In contrast, a situation where most rainfall and thus CSO events are situated in the first years of the 10-year period may exhibit signs of recovery from the CSO events after ten years.

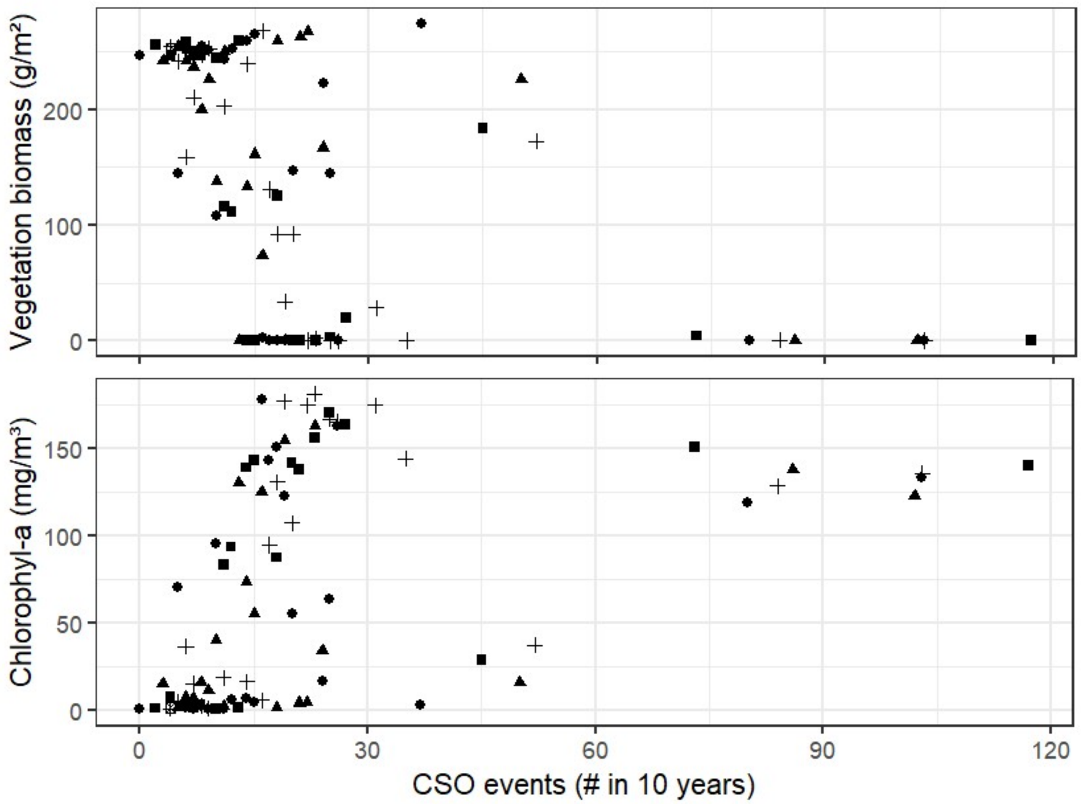

Figure 5. Impact of increasing CSO events on vegetation biomass (top), indicative of a clear state, and chlorophyll-a (bottom), a proxy of phytoplankton biomass indicative of a turbid state.

5.1.2.4 Impact of gradients of CSO frequency on ES provisioning

In Figure 5 the impact that increases in the number of CSO events has on ES provisioning is shown. First, it should be noted that nearly all ESs were diminished at high frequency of CSO events (>7 CSO events per year). We can also observe variation in ES delivery with a limited number of CSO events (<5) where some services exhibit no effect (Birding, Swimming) while others may show limited positive changes (Fishing, N&P sequestration, Particle capture). The latter group of ESs likely benefits from the small positive changes observed in vegetation biomass, where vegetation, when it is able to persist may proliferate given the increased nutrient supply. These results illustrate two major conclusions, namely that CSO events a) are unlikely to impact all ecosystem services similarly, and b) are likely to lead to non-linear loss of ES delivery for some ESs.

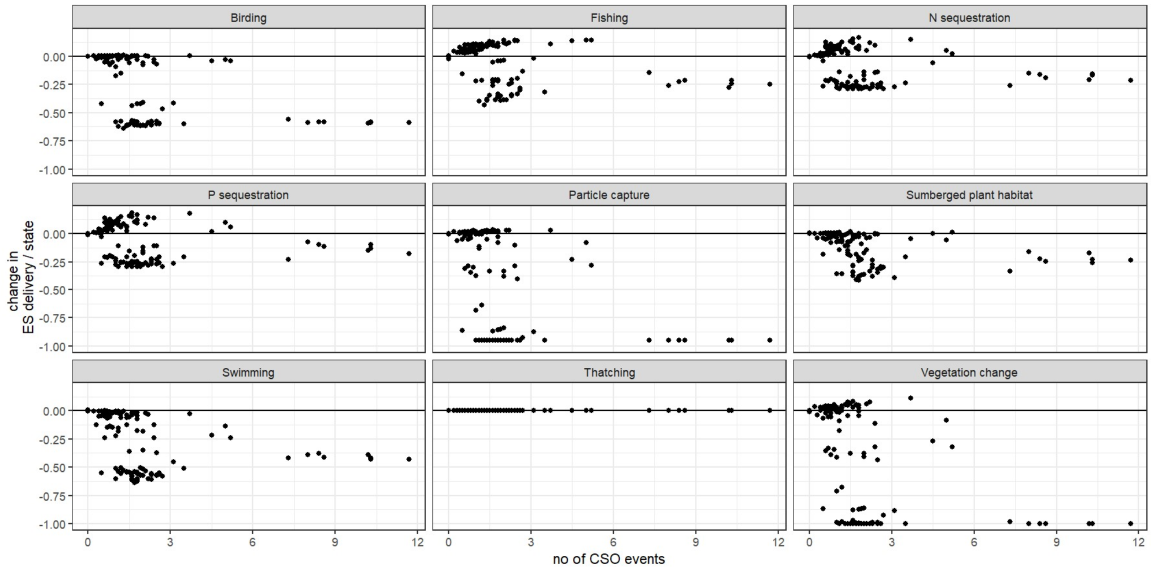

Figure 6. Impact of increasing CSO events (average number of CSO events per year) on the relative change in different ecosystem services (see Table 1). A positive value indicates an increase in ES provisioning, whereas a negative value indicates a decrease. In the bottom right, the relative change in vegetation biomass is shown for reference, corresponding closely to the pattern observed in Figure 5.

5.1.2.5 Embedding results into a spatial risk based assessment

The developed model may offer a solid starting point to cover part of the ecological and societal vulnerability aspects of the envisioned risk assessment of NICHES (see Figure 1). To exemplify this potential, we plotted the vulnerability to CSO events of both the aquatic ecosystem as well as one of the ecosystem services it provides. We used the different neighbourhoods of Rotterdam as the spatial unit to analyse and used their overlap with CSO locations as a baseline. Next, for each neighbourhood we looked at the chance of collapse of vegetation based on our modelled results along the CSO gradient (see Figure 5 and 6). Here we account for the number of CSOs in the neighbourhood (0 to 3), including only model runs with the given number of CSOs. Ecological vulnerability was expressed as the amount of model runs where vegetation decreased by more than 50% relative to the total amount of model runs. Swimming provisioning vulnerability was defined as the number of model runs with a loss of swimming provisioning (given a threshold of 5% decrease to avoid spurious results) relative to the total number of model runs. Figure 7 shows a strong overlap between ecological vulnerability and swimming vulnerability, indicating that systems prone to ecological collapse are similarly prone to losing a potentially valued ecosystem service.

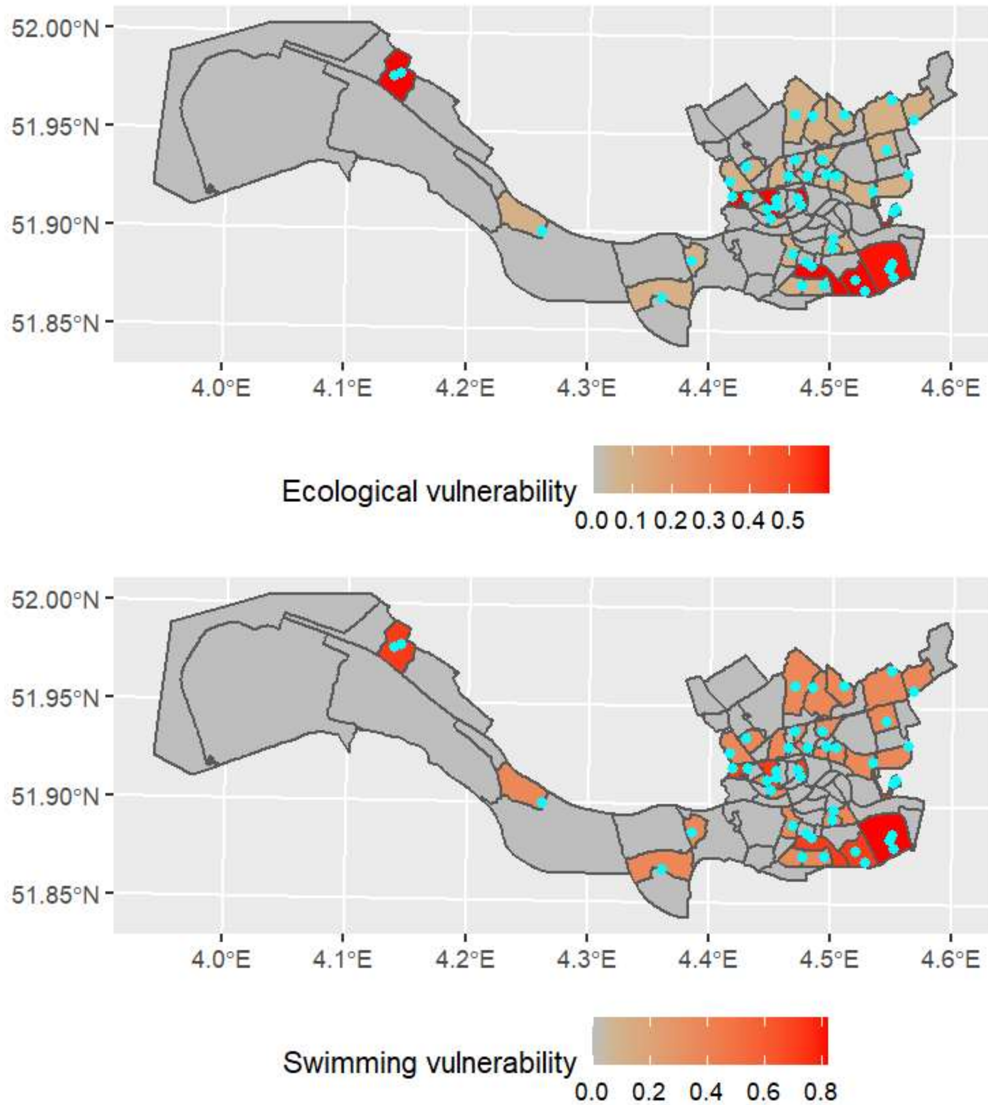

Figure 7: The ecological vulnerability (top) and swimming service provisioning vulnerability (bottom) to CSO events in different neighbourhoods in Rotterdam. Ecological vulnerability is defined as a crash in vegetation biomass (relative decrease compared to baseline of 50%). Loss of swimming service is defined as a decrease in swimming suitability of the water based on relevant thresholds (set at a relative decrease of 5% or more to avoid negligible decrease). In blue, the location of all CSO locations in Rotterdam.

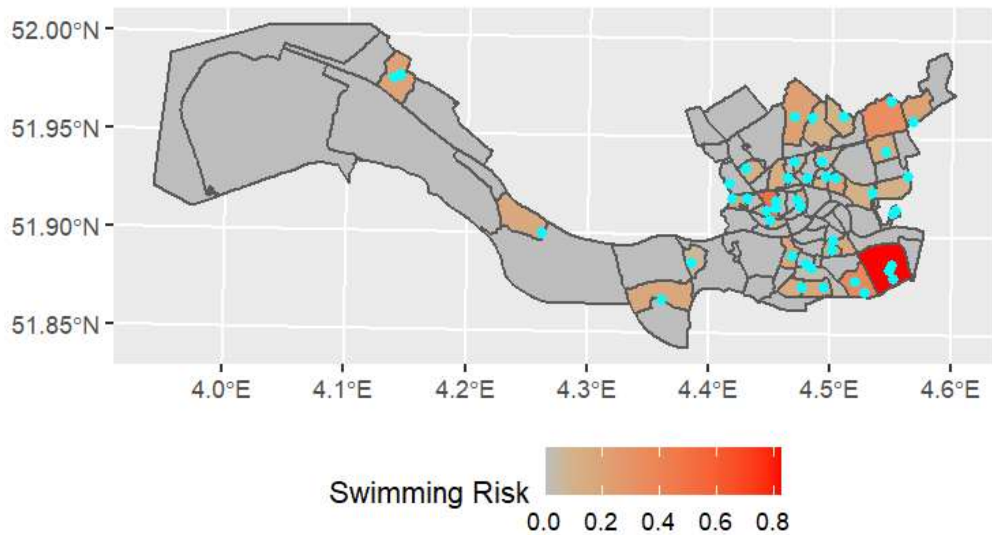

Figure 8: The potential risk of losing swimming waters to CSO events in different neighbourhoods in Rotterdam. In blue, the location of all CSO locations in Rotterdam.

We do not account for the use of swimming service to people when only considering vulnerability though. Hence, we scaled our service provisioning by the number of people living in each neighbourhood (data from: Central Bureau for Statistics, NL) in Figure 8. Comparing swimming vulnerability and risk, we see that some neighbourhoods where there is a large risk of swimming water provisioning being in danger have relatively few people living there (e.g. industrial areas), making the risk less severe (i.e. by having limited exposure). That said, our approach here is but a first step approach and should be seen as illustrative. Service demands may not be spatially bound to neighbourhoods. Furthermore, water quality problems are not necessarily bound to neighbourhood boundaries either as CSO water may be transported downstream and thereby polluted more of the waterscape than only the direct vicinity. We envision more detailed analyses using local information on the waterscape and sewage systems, as well as social context for core cities in NICHES. Furthermore, with the prove of concept of a coupled AEM-ESs model in hand, we can now guide development of ecosystem models based on inclusion of variables relevant to estimate ESs, rather than only on those variables relevant for ecological dynamics themselves. Our results and analysis deliver a first insight into the application domain of a coupled AEM-ESs model, illustrating its potential in assessing vulnerability and risks for both the ecological and societal system in an urban waterscape.

6 References

Andersen, T. K., Nielsen, A., Jeppesen, E., Hu, F., Bolding, K., Liu, Z., Søndergaard, M., Johansson, L. S., & Trolle, D. (2020). Predicting ecosystem state changes in shallow lakes using an aquatic ecosystem model: Lake Hinge, Denmark, an example. *Ecological Applications*, *30*(7), e02160. https://doi.org/10.1002/eap.2160

Burkhard, B., Kroll, F., Nedkov, S., & Müller, F. (2012). Mapping ecosystem service supply, demand and budgets. *Ecological Indicators*, *21*, 17–29. https://doi.org/10.1016/j.ecolind.2011.06.019

Chang, H., Pallathadka, A., Sauer, J., Grimm, N. B., Zimmerman, R., Cheng, C., Iwaniec, D. M., Kim, Y., Lloyd, R., McPhearson, T., Rosenzweig, B., Troxler, T., Welty, C., Brenner, R., & Herreros-Cantis, P. (2021). Assessment of urban flood vulnerability using the social-ecological-technological systems framework in six US cities. *Sustainable Cities and Society*, *68*, 102786. https://doi.org/10.1016/j.scs.2021.102786

Dobberfuhl, D. R. (2007). Light limiting thresholds for submerged aquatic vegetation in a blackwater river. *Aquatic Botany*, *86*(4), 346–352.

Gómez-Baggethun, E., & Ruiz-Pérez, M. (2011). Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography: Earth and Environment*, *35*(5), 613–628. https://doi.org/10.1177/0309133311421708

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., & Cardoso, A. C. (2016). Assessing water ecosystem services for water resource management. *Environmental Science & Policy*, *61*, 194–203. https://doi.org/10.1016/j.envsci.2016.04.008

Grizzetti, B., Liquete, C., Pistocchi, A., Vigiak, O., Zulian, G., Bouraoui, F., De Roo, A., & Cardoso, A. C. (2019). Relationship between ecological condition and ecosystem services in European rivers, lakes and coastal waters. *Science of The Total Environment*, *671*, 452–465. https://doi.org/10.1016/j.scitotenv.2019.03.155

Guerry, A. D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G. C., Griffin, R., Ruckelshaus, M., Bateman, I. J., Duraiappah, A., Elmqvist, T., Feldman, M. W., Folke, C., Hoekstra, J., Kareiva, P. M., Keeler, B. L., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., … Vira, B. (2015). Natural capital and ecosystem services informing decisions: From promise to practice. *Proceedings of the National Academy of Sciences*, *112*(24), 7348–7355. https://doi.org/10.1073/pnas.1503751112

Haines-Young, R., Potschin, M., & Kienast, F. (2012). Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. *Ecological Indicators*, *21*, 39–53. https://doi.org/10.1016/j.ecolind.2011.09.004

Hipsey, M. R., Bruce, L. C., Boon, C., Busch, B., Carey, C. C., Hamilton, D. P., Hanson, P. C., Read, J. S., de Sousa, E., Weber, M., & Winslow, L. A. (2019). A General Lake Model (GLM 3.0) for linking with high-frequency sensor data from the Global Lake Ecological Observatory Network (GLEON). *Geoscientific Model Development*, *12*(1), 473–523. https://doi.org/10.5194/gmd-12-473-2019

Ibelings, B. W., Portielje, R., Lammens, E. H. R. R., Noordhuis, R., van den Berg, M. S., Joosse, W., & Meijer, M. L. (2007). Resilience of Alternative Stable States during the Recovery of Shallow Lakes from Eutrophication: Lake Veluwe as a Case Study. *Ecosystems*, *10*(1), 4–16. https://doi.org/10.1007/s10021-006-9009-4

Janse, J. H. (2005). *Model studies on the eutrophication of shallow lakes and ditches*. Wageningen University.

Janse, J. H., De Senerpont Domis, L. N., Scheffer, M., Lijklema, L., Van Liere, L., Klinge, M., & Mooij, W. M. (2008). Critical phosphorus loading of different types of shallow lakes and the consequences for management estimated with the ecosystem model PCLake. *Limnologica*, *38*(3), 203–219. https://doi.org/10.1016/j.limno.2008.06.001

Janse, J. H., Scheffer, M., Lijklema, L., Van Liere, L., Sloot, J. S., & Mooij, W. M. (2010). Estimating the critical phosphorus loading of shallow lakes with the ecosystem model PCLake: Sensitivity, calibration and uncertainty. *Ecological Modelling*, *221*(4), 654–665. https://doi.org/10.1016/j.ecolmodel.2009.07.023

Janssen, A. B. G., Arhonditsis, G. B., Beusen, A., Bolding, K., Bruce, L., Bruggeman, J., Couture, R.-M., Downing, A. S., Alex Elliott, J., Frassl, M. A., Gal, G., Gerla, D. J., Hipsey, M. R., Hu, F., Ives, S. C., Janse, J. H., Jeppesen, E., Jöhnk, K. D., Kneis, D., … Mooij, W. M. (2015). Exploring, exploiting and evolving diversity of aquatic ecosystem models: A community perspective. *Aquatic Ecology*, *49*(4), 513–548. https://doi.org/10.1007/s10452-015-9544-1

Janssen, A. B. G., Hilt, S., Kosten, S., de Klein, J. J. M., Paerl, H. W., & Van de Waal, D. B. (2021). Shifting states, shifting services: Linking regime shifts to changes in ecosystem services of shallow lakes. *Freshwater Biology*, *66*(1), 1–12. https://doi.org/10.1111/fwb.13582

Janssen, A. B. G., Teurlincx, S., Beusen, A. H. W., Huijbregts, M. A. J., Rost, J., Schipper, A. M., Seelen, L. M. S., Mooij, W. M., & Janse, J. H. (2019). PCLake+: A process-based ecological model to assess the trophic state of stratified and non-stratified freshwater lakes worldwide. *Ecological Modelling*, *396*, 23–32. https://doi.org/10.1016/j.ecolmodel.2019.01.006

Kirillin, G., Hochschild, J., Mironov, D., Terzhevik, A., Golosov, S., & Nützmann, G. (2011). FLake-Global: Online lake model with worldwide coverage. *Environmental Modelling & Software*, *26*(5), 683–684. https://doi.org/10.1016/j.envsoft.2010.12.004

Lürling, M., & Mucci, M. (2020). Mitigating eutrophication nuisance: In-lake measures are becoming inevitable in eutrophic waters in the Netherlands. *Hydrobiologia*, *847*(21), 4447–4467. https://doi.org/10.1007/s10750-020-04297-9

Michael Kemp, W., Batleson, R., Bergstrom, P., Carter, V., Gallegos, C. L., Hunley, W., Karrh, L., Koch, E. W., Landwehr, J. M., & Moore, K. A. (2004). Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: Water quality, light regime, and physical-chemical factors. *Estuaries*, *27*, 363–377.

Mooij, W. M., Janse, J. H., De Senerpont Domis, L. N., Hülsmann, S., & Ibelings, B. W. (2007). Predicting the effect of climate change on temperate shallow lakes with the ecosystem model PCLake. In R. D. Gulati, E. Lammens, N. De Pauw, & E. Van Donk (Eds.), *Shallow Lakes in a Changing World* (pp. 443–454). Springer Netherlands. https://doi.org/10.1007/978-1-4020-6399-2_40

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., Chan, K. M., Daily, G. C., Goldstein, J., Kareiva, P. M., Lonsdorf, E., Naidoo, R., Ricketts, T. H., & Shaw, Mr. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. https://doi.org/10.1890/080023

Polasky, S., Nelson, E., Pennington, D., & Johnson, K. A. (2011). The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Environmental and Resource Economics*, *48*(2), 219–242. https://doi.org/10.1007/s10640-010-9407-0

Schets, F. M., van der Oost, R., & van de Waal, D. B. (2020). *Cyanobacteria protocol 2020*.

Seelen, L. M. S., Teurlincx, S., Armstrong, M. R., Lürling, M., van Donk, E., & de Senerpont Domis, L. N. (2021). Serving many masters at once: A framework for assessing ecosystem services delivered by quarry lakes. *Inland Waters*, *0*(0), 1–17. https://doi.org/10.1080/20442041.2021.1944765

Seelen, L. M. S., Teurlincx, S., Armstrong, M. R., Lürling, M., van Donk, E., & de Senerpont Domis, L. N. (2022). Serving many masters at once: A framework for assessing ecosystem services delivered by quarry lakes. *Inland Waters*, *12*(1), 121–137. https://doi.org/10.1080/20442041.2021.1944765

van der Werf, J. A., Kapelan, Z., & Langeveld, J. (2023). Real-time control of combined sewer systems: Risks associated with uncertainties. *Journal of Hydrology*, *617*, 128900. https://doi.org/10.1016/j.jhydrol.2022.128900

Wang, M., Strokal, M., Burek, P., Kroeze, C., Ma, L., & Janssen, A. B. G. (2019). Excess nutrient loads to Lake Taihu: Opportunities for nutrient reduction. *Science of The Total Environment*, *664*, 865–873. https://doi.org/10.1016/j.scitotenv.2019.02.051

Wortley, L., Hero, J.-M., & Howes, M. (2013). Evaluating Ecological Restoration Success: A Review of the Literature. *Restoration Ecology*, *21*(5), 537–543. https://doi.org/10.1111/rec.12028

Yang, J., Strokal, M., Kroeze, C., Ma, L., Bai, Z., Teurlincx, S., & Janssen, A. B. G. (2022). What is the pollution limit? Comparing nutrient loads with thresholds to improve water quality in Lake Baiyangdian. *Science of The Total Environment*, *807*, 150710. https://doi.org/10.1016/j.scitotenv.2021.150710

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