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Research papers

Importance of spatial heterogeneity of nutrient loading on the ecological status of lake Tana, Ethiopia

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ABSTRACT

Understanding spatial variability of nutrient loading and transport in large lakes, and its effect on the eutrophication status are often lacking and hinder effective and sustainable management of lakes. Yet knowledge on the extent of spatial variability of eutrophication in large shallow tropical lakes hinders sustainable management. Understanding how eutrophication varies across lakes helps spatially targeted nutrient load mitigation strategies. This modelling study assesses how spatial heterogeneity in nutrient loads may drive spatial variability in in-lake eutrophication effects for Lake Tana, a large tropical lake. We applied a novel method of coupling a 2D application of the flow model Duflow with the ecosystem model PCLake+ (a zero-dimensional food-web model). We defined different impact zones of major tributaries with a tracer model and simulated ecological processes and food web relations for each impact zone. Furthermore, we defined critical nitrogen and phosphorus loads for each impact zone and the whole lake. Subsequently, we analysed the spatial variability of phytoplankton and aquatic vegetation dynamics among the impact zones, as well as the differences in critical loadings. The model results indicate different ecological conditions in the impact zones and the importance of spatial heterogeneity of eutrophication reflected in different critical loading. The model shows that the north impact zone and northwest impact zone are mainly vegetation-dominated, and southeast impact zone and southwest impact zone are phytoplankton-dominated. Simulated chlorophyll-a concentrations ranged from 0 to 59 $\mu g\ l^{-1}$ and the coverage of the vegetation ranged from 1 to 127 g d.w.m^{-2}, aligning with the range of observed values in the lake. We demonstrate that this model approach can help to identify spatial heterogeneity in hydrology and eutrophication, as such lakes cannot be regarded as completely mixed systems. Our new approach demonstrates how spatially variable nutrient loading correspond to variability in eutrophication effects in a large shallow tropical lake, and will aid in setting more spatially-targeted lake management strategies.

1. Introduction

Eutrophication is one of the most widespread environmental problems of inland water bodies. It has undesired effects on ecosystem state and services, public health, and socioeconomic activities. Some of these environmental effects in lakes include risk of algal blooms leading to a decrease of water clarity, hypoxia and fish kills, altering biogeochemical processes, and disrupting the aquatic food web (Scheffer, 1998). Eutrophication in lakes is caused by excessive loads of nutrients delivered by rivers (Kane et al., 2014; Lürling and van Oosterhout, 2013). The extent of eutrophication could increase as a result of increased nutrient inputs from the intensification of agriculture and the increase of the human population.

The management and restoration of lakes are often based on reducing the nutrient input to below a critical level. Critical loads represent an ecological threshold at which there can be an abrupt change in an ecosystem state, property, or phenomenon, or where small changes in an environmental driver produce large responses in the ecosystem (Scheffer and Carpenter, 2003; Groffman et al., 2006; Xu et al., 2015). It is documented that in shallow lakes eutrophication can lead to an abrupt shift of conditions, with alternative stable states either with submerged vegetation or phytoplankton dominated (Scheffer, 1998; Janssen et al., 2014). However, field data of lakes that actually experience alternative stable states are scarce (Capon et al 2015). This

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might be due to lakes being in transition for long periods, or they switch back and forth easily (Van Geest et al, 2007; Arani et al, 2021). Still, some empirical evidence for regime shifts in lakes is found (Ibelings et al. 2007; Kosten et al. 2012). The concept of regime shifts and critical loads in lakes is valuable in defining lake restoration plans and setting nutrient loading targets (Scheffer and Carpenter, 2003; Janssen et al, 2014).

Critical nutrient inputs can be derived from aquatic ecosystem models. Several lake models were reviewed by (Janssen et al., 2019a). They conclude that to study state transitions in lakes a mechanistic dynamic model is required that include the aquatic food web. In this way, non-linear processes and the relevant feedback are accounted for. PCLake meets the above criteria (Janse et al., 2010) and is widely used to explore the transition between alternative stable states and to estimate critical nutrient loads.

Many aquatic models often assume lakes are completely mixed systems. Natural lakes (especially large lakes) have clear spatial differences, both for hydrology and chemistry. This is due to factors like complex morphology, the different depth and spatially distributed inflow of water and nutrients. In shallow lakes, these are mainly horizontal differences. In deep lakes vertical gradients also exist due to stratification. Lake characteristics and internal and external connectivity can drive spatial variability (Janssen et al., 2017; Liu and Qiu, 2007; Rahm and Danielsson, 2007; Yu et al., 2008; Fragoso et al., 2008). Consequently, we hypothesize that there are different zones in large shallow lakes based on lake hydrology and distributed inflows of rivers ('impact zones'). These impact zones have their own characteristics with different critical nutrient loads, which can differ from the critical load when we assume complete mixing.

The spatial heterogeneity of eutrophication has implications for lake management. The effective and sustainable management of eutrophication and lake restoration benefits from a clear understanding of spatial heterogeneity in nutrient loading and hydrology (Janssen et al., 2019a). In this way, the effort of reducing nutrient loads in specific subbasins can be focused on the parts of the lake that are most vulnerable.

The effects of spatial heterogeneity on the different biological components of an aquatic ecosystem include seasonal succession of phytoplankton, macrophyte growth, invasive alien species, and lake turbidity (Ding et al., 2015; Pringle, 1990; Rychtecký and Znachor, 2011; Soares et al., 2012; Xu et al., 2020). Moreover, different residence times influences spatial heterogeneity, which is important for a deeper understanding of ecological processes in freshwater lakes. It is also a mechanism for maintaining the species diversity of phytoplankton communities.

Most applications of ecological models regard lakes as one complete mixed system. This approach disregards spatial differences, e.g. rivers that discharge on the lake will only affect the lake part near the river mouth. Littoral zones are more shallow than the pelagic zones, resulting in different conditions for vegetation growth. In this paper, we modelled spatial differences in Lake Tana, a large tropical lake, to estimate the spatial heterogeneity of eutrophication in a large tropical lake which might facilitate the design of more regional or locally targeted measures.

The main research questions of this study are:

1. What are the different impact zones of the major tributary rivers of Lake Tana? 2. Is there spatial variability of eutrophication among the impact zones, expressed in nutrient and chlorophyll concentrations? 3. What are the critical N and P loads for each impact zone? To address these questions we use a novel modelling approach. First, we define different impact zones of major tributaries in Lake Tana by using a two-dimensional application of the flow and quality model Duflow (Clemmens et al., 1993). Secondly, we determine critical nutrient loads for these impact zones and Lake Tana separately with the ecosystem model PCLake+ (Janssen et al., 2019).

2. Methodology

2.1. Study area

Lake Tana is a shallow (maximum 14 m, average 8 m deep), occasionally stratifying, and large tropical lake found at an altitude of 1800 m a s l. It is the largest lake in Ethiopia, accounting for 50% of the surface water volume of the freshwater resource of the country. It has a surface area of ca. 3111 km^2, 28.4 km^3 volume, and has a maximum length of 90 km and width of 65 km. The Tana basin has a total drainage area of 16,500 km^2 (Fig. 1). More than six medium- to large-sized tributary rivers and>40 ephemeral streams drain into Lake Tana. Among the rivers, the Gilgel Abay, Dirma, Gumara, Gelda, Rib, and Megech rivers contribute>90% of the inflow (Sirak, 2008). Lake Tana is the source of the Blue Nile, which is the only natural surface outflow of the lake. In addition, there is substantial abstraction of water from the lake for hydropower generation (the Tana-Beles hydropower plant).

Rainfall is maximum during July and August at 250–330 mm per month. Mean annual rainfall is nearly 1280 mm (Abebe and Minale, 2017). The rainy season (July–September) receives about two-thirds of the annual rainfall, while the dry season receives 2%; the pre-rainy season (April–June) receives 25% and the post-rainy season 8% of annual rainfall. The average seasonal air temperature reaches its maximum of 21.1 °C in the pre-rainy season and its minimum (18.4 °C) in the rainy season, with a large diurnal but small seasonal change.

The population of the basin was projected to be 4.5 million in 2015 (CSA, 2007), with a population density of 228 persons per km^2 in 2007 (Anteneh, 2017), and 70% of the basin is agricultural land (Abebe and Minale, 2017). In this study, we mainly focused on the largest subbasins: Gilgel Abay, Dirma, Megech, Gumara, Gelda, Rib, Arno, and Garno (Fig. 1).

Lake Tana is an example of a large tropical lake that cannot be considered completely mixed because of multiple inflowing rivers with different discharges and nutrient concentrations. Spatial variations are observed in phytoplankton, nutrient concentrations, and vegetation conditions, that can affect spatial patterning of critical nutrient loads in Lake Tana. The measured and secondary data about the current trophic conditions of lake Tana show spatial variability among the shore, open water, and river mouths of the lake. Most of the river mouths and a significantly smaller portion of the lakeshore area show eutrophic conditions, and the majority of the lake exhibited more oligotrophic conditions (Goraw Goshu, unpublished).

2.2. Research approach

We developed a method to facilitate the analysis of spatial variability of eutrophication in a heterogenous lake (summarized in Fig. 2). We combined two modelling approaches with modifications for our study area based on the models Duflow (Clemmens et al., 1993) and PCLake+ (Janse et al. 2010; Janssen et al., 2019). We used a two-dimensional application of the flow and water quality model Duflow, to perform unsteady flow computations and to simulate tracer distribution in the lake to define so-called impact zones. Using real tracers in field studies is practically impossible in such large systems like Lake Tana with multiple inflowing rivers. Thereafter, we ran the PCLake+ model for each impact zone separately and for Lake Tana as one mixed system to simulate ecosystem conditions (e.g. nutrient concentrations, phytoplankton, vegetation) and to give indications of critical nutrient loads for ecosystem transitions (Janse et al., 2010). Since PCLake+ has been applied in comparable lakes, and is based on fundamental ecological and chemical concepts, we apply the model in this observational data-limited lake to assess eutrophication effects likely vary across space. This analysis provides an example of how to identify the relative contribution of variable nutrient loadings in data-limited shallow tropical lakes to prioritize areas for management action, but determining specific target levels will require collection of more thorough

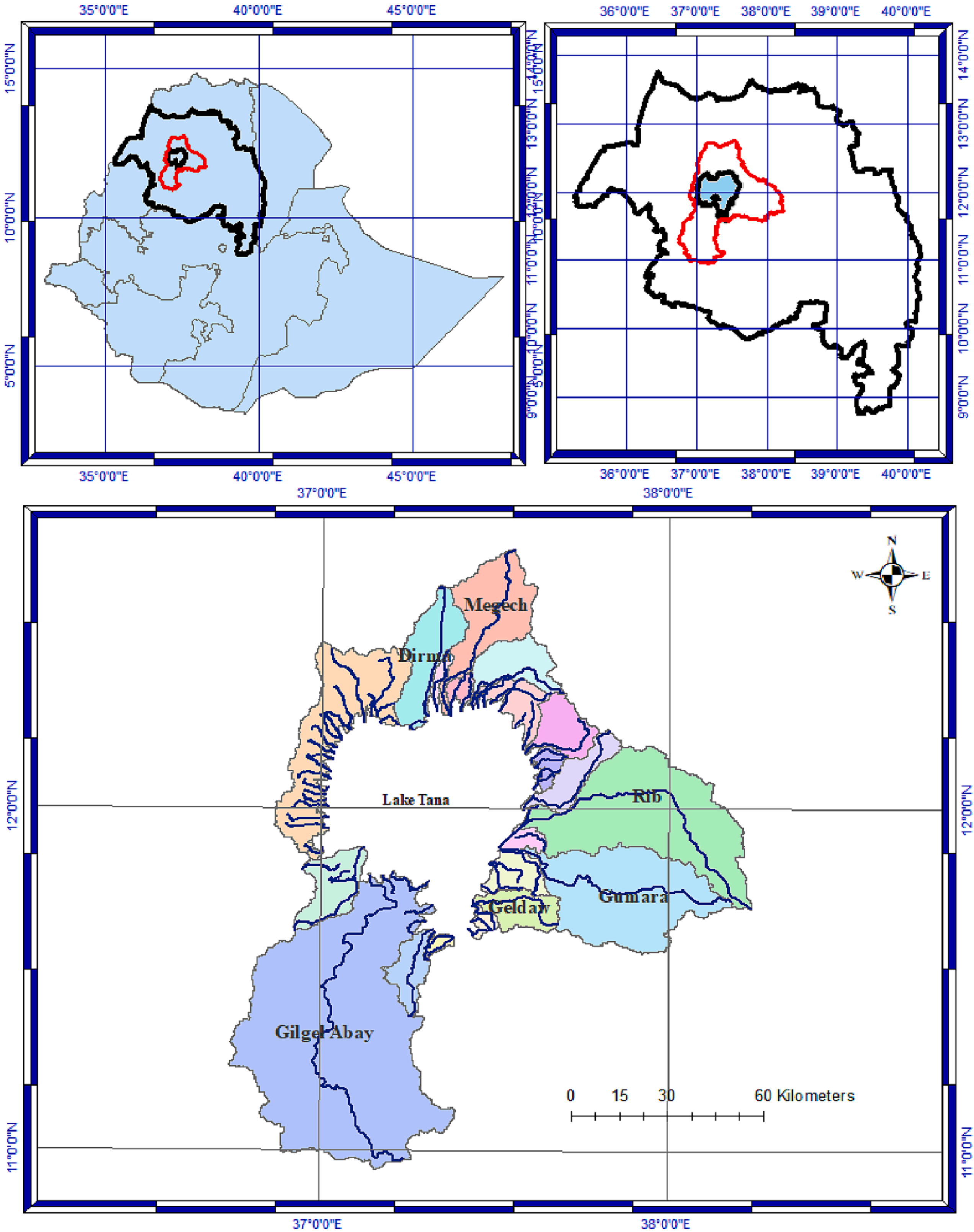

Fig. 1. Ethiopia and the Amhara region (upper left); the Amhara region, the Lake Tana basin and Lake Tana (upper right); the drainage area of the Lake Tana basin and the sub-basins draining into the lake (lower panel).

observational data.

2.3. Modelling lake Tana with a tracer model in Duflow

2.3.1. Network schematization

Duflow is designed as a 1D unsteady flow and quality model. For this

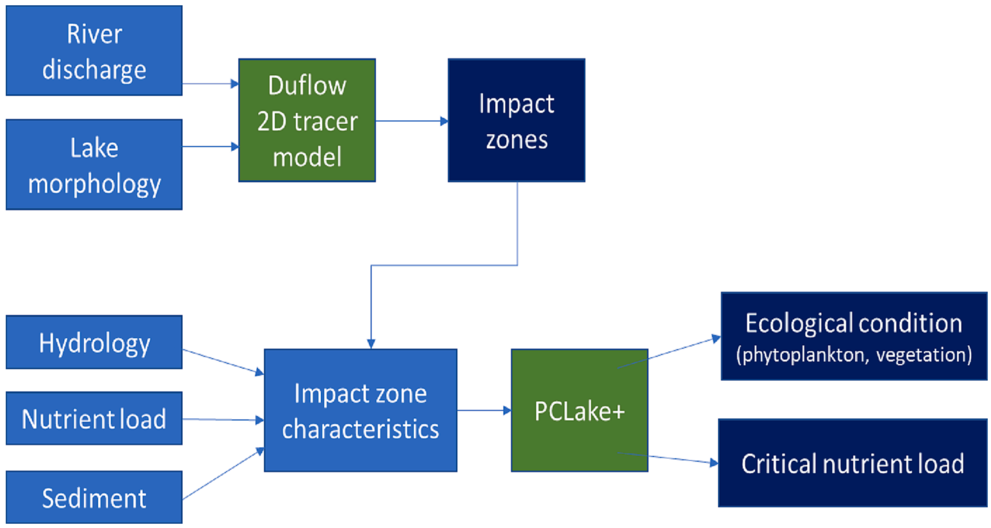

Fig. 2. Conceptual diagram of model coupling of Duflow and PCLake+ (blue: input, green: model, dark blue: result).

study, we set up a 2D application by constructing a dense network of sections in multiple directions to be able to simulate water flow and solute transport in the lake. Duflow solves the transport equation (advection–dispersion) and the user can add specific physical and (bio) chemical processes. We defined a simple tracer model with only transport and no processes. As a first step in the configuration of the Duflow model we classified the lake into four depth zones: 0–2 m, 2–6 m, 6–10 m, and 10–14 m. The 0–2 m deep zone is defined as shore (4%), 2–6 is littoral (12%), 6–10 is sublittoral (21%), and 10–14 m is classified as a pelagic zone (62%). We built the network on a geographical background by adding the bathymetric map of Lake Tana (Wale et al., 2008). The length of the vertical and horizontal cross-sections (internode distance) ranges from 0.5 km mostly in the shore area to 5 km in off-shore zones. We defined the width and length of the sections in such a way that the sum of the section areas corresponded with the total lake area.

2.3.2. Flow model

After network schematization, we defined initial and boundary conditions for the hydrology and configured the calculation. We defined the boundary conditions, i.e. inflow of a tributary river, at the edge of the lake system, based on discharge monitoring data that were aggregated to monthly discharges. Furthermore, we defined water abstraction to the hydropower plant (Tana-Beles) and the Blue Nile outlet as negative discharge and the precipitation surplus (total rainfall - evaporation on the lake area) as negative or positive depending on the net value. We configured the model with a calculation time step of one hour and a one-day output step. Once the flow calculation was verified, we added the quality model.

To explore the spatial distribution of solutes coming from separate inflows, we defined a multiple tracer quality model. The model definition was without chemical processes to simulate the transport of a conservative substance. We simulated four different tracers with a starting concentration of 100 mg l^{-1} in the main inflowing rivers. These are the gauged rivers; no tracer concentration was defined at other tributary rivers. Depending on the hydraulic conditions, the model will show that the tracers are spread over the lake and possibly transported to the Blue Nile outlet and Tana-Beles abstraction.

2.3.3. Delineation of the ‘impact zones’ of major incoming rivers with the 2D tracer model

The impact zone is the area of the lake that is under the specific influence of the feeding tributary river. We focused on the major tributary rivers, and we grouped the major tributary rivers into four zones: south-west (Gilgel Abay), south-east (Gelda, Gumara, Rib), north-east (Arno-Garno), and north (Dirma, Megech). Grouping of tributaries was done as some rivers are close to each other and difficult to separate.

To delineate impact zones of the rivers in the lake, we used two indicators from the tracer model: the concentration of the four tracers in all sections of the lake and the fraction of one tracer from the sum of all tracer concentrations (**eq** (1)):

$$Fract_i = \frac{100*\boldsymbol{Tracer_i}}{\boldsymbol{Tracer_1} + \boldsymbol{Tracer_2} + \boldsymbol{Tracer_3} + \boldsymbol{Tracer_4}} \quad (1)$$

where, $Fract_i$ is the fraction of tracer 1, 2, 3, or 4, and $Tracer_i$ is the concentration of tracer 1, 2, 3, or 4.

For delineation of the impact zones, we used the criteria that $Fract_i >$ 90% and $Tracer_i >$ 10 mg l^{-1}.

2.4. PCLake+ model configuration for the lake Tana impact zones

The modelling of N and P critical loads to Lake Tana with PCLake+ encompassed the hydraulic and nutrient loads from major tributaries to the lake. Only the major tributary rivers which are gauged were accounted for. PCLake+ is a zero-dimensional ecosystem model that disregards spatial heterogeneity. It is a complex ecological model used to assess the nutritional status of stratified and non-stratified freshwater lakes (Janssen et al., 2019b) The model includes biological components (phytoplankton, submerged vegetation, zooplankton, planktivorous fish and piscivorous fish) in a simplified food web, and non-biological components (transparency, oxygen, inorganic carbon and nutrients) (Janse et al., 2010). The model is built in a sequence from primary producers to top predators. N and P cycles in the water column and sediments combine with higher levels in the food web (Janse et al., 2010). The Marsh area is a PCLake + module, involving a simplified growth model of *Phragmites australis*, an emergent species of macrophytes, combined with vegetative processes in the sediment and the water column of the marsh area (Sollie and Verhoeven, 2008). This area with emergent plants is connected to the open water by a flow of water between them (Janssen et al., 2019b). There are four types of lake-specific settings to parameterize PCLake+: specific lake characteristics, nutrients input, lake hydrology, and sediment type. The specific impact zone characteristics mean depth, impact zone area, the main wind direction were computed in this study. We used Arc GIS to calculate the fetch, and length of the impact zone in the main wind direction. The mean depth of each impact zone was estimated in ArcGIS using a weighted average technique from the bathymetric map of Lake Tana. Marsh area, a fraction of the lake area, was retrieved by taking into account all wetland areas within the catchment of Lake Tana

(Aynalem et al., 2017). The mean fetch of each impact zone was calculated by the square root of the surface area. Sediment characteristics are parameterized using the data from previous studies. The Lake Tana sediment composition and texture class, is predominately clay (67 %), sand (17 %), and silt(16%). The average organic matter content of the sediment is 16 g kg^{-1}, and the sediment available phosphorus was 19 mg kg^{-1} (Kebedew et al., 2020).

We set up the models for the year 2017. We quantified the monthly N and P loads from the six major sub-basins of Lake Tana from discharge and nutrient data. The dynamic simulation of the aquatic ecosystem condition in each impact zone consisted of two parts. First, the current condition was simulated, for which the model was initialized and run for 50 years to reach equilibrium (Janssen et al., 2019a). The results in the last 2 years of the simulation were taken as the final output and chlorophyll-a concentrations were compared with the measurements of 2016 and 2017.

Secondly, a bifurcation analysis was done for each impact zone separately and for Lake Tana as one mixed system. For this, PCLake + is run sequentially with increasing and decreasing nutrient inputs, to find the critical nutrient loads for a shift from phytoplankton dominance to macrophyte dominance and vice versa. Each run for the bifurcation analysis was done for 30 years, and the average of the May-September period in the last 2 years of the simulation was taken as the result. In this study, the ratio between the external N and P loads was measured in 2017. We took this as a reference for the bifurcation analysis with a constant N:P ratio. The critical nutrient loads were used to identify the required reductions of external nutrient loading to restore the foreseen clear stable state in the lake. Or, in case the clear state is present, it is used to identify the maximum loads before it possibly switches to the turbid state (Janse et al, 2008).

2.5. *Hydrology and water quality*

2.5.1. *Water balance of lake Tana*

The water balance is needed to calculate the hydraulic load of the impact zones and the whole lake as input for PCLake+. The water balance was validated with literature data before implementing it. The water balance was calculated for the year 2017, as it is the most complete year for the discharge data to and from the lake and nutrient concentrations (needed for calculating nutrient fluxes). The monthly water balance of Lake Tana was calculated using equation (2).

$$\frac{\Delta S}{\Delta T} = P + Qgauged + Qungauged - E_0 - Q_{HP} - Q_{Dam} \tag{2}$$

Where,

$\frac{\Delta S}{\Delta T}$ Storage change (closure term) [L^3 T^{-1}],

P Precipitation over Lake Tana [L^3 T^{-1}].

Qgauged Surface water discharge from gauge reading near lake inlet [L^3 T^{-1}].

Qungauged Surface water discharge from ungauged towards lake Tana [L^3 T^{-1}].

E_0 Penman open water evaporation [L^3 T^{-1}].

Q_{HP} Surface water discharge towards Hydropower plant Belles [L^3 T^{-1}].

Q_{Dam} Surface water discharge towards Blue Nile river [L^3 T^{-1}].

The surface-water flow to Lake Tana includes six major sub-basins (Table 1) which are gauged, and 14 sub-basins that are not gauged. Groundwater flow within the Lake Tana Basin takes place in a heterogeneous aquifer system with the groundwater flow converging to Lake Tana, however, groundwater flow also takes place from the Lake Tana Basin towards the adjacent Beles River basin (Mamo et al., 2016; Nigate et al., 2017). Furthermore, as concluded by Mamo et al. (2016), estimated groundwater input to the lake was found to be of minor importance and therefore groundwater flow was excluded from the water balance for this research. The detailed computation of the water balance components is described in the supplementary materials.

Table 1
Sub-basins of the Lake Tana basin and the percentage of the surface area of the specific sub-basin within the total Lake Tana Basin. ().

Sub-basin	Surface area sub-basin (km^2)	Percentage surface area of the total basin (%)	Gauged (yes/no)
Gilgel Abay	3866	34.4	Yes
Arno-Garno	269	2.4	Yes
Dirma	461	4.1	Yes
Gelda	261	2.3	Yes
Gumara	1376	12.2	Yes
Megech	659	5.9	Yes
Rib	1727	15.4	Yes

Source Goshu et al., 2020

2.5.2. *Hydraulic loadings*

The hydraulic loadings of tributary rivers in each impact zone and the Lake Tana were computed by dividing the sum of the discharges of the tributary rivers in the impact zone by the area of the impact zone, estimated based on the tracer model study (eq.3). Water discharge data of a tributary for each month in 2017 were obtained from the Ethiopian Ministry of Water, Electric and Irrigation.

$$\boldsymbol{HLR} = \frac{\sum_{i=1}^{m} Q_i}{A} \tag{3}$$

Where,

HLR is the hydraulic loading rate (m s^{-1}),

Q_i is the discharge at the gauging stations at a site I (m^3 s^{-1}),

A is the area of the impact zone (m^2),

m is the number of discharge measurements.

2.5.3. *Nutrient loadings*

The water sampling and analysis of the different nitrogen and phosphorus species are described in detail in Goshu et al. (2020). The DIN or DIP load to the impact zone and Lake Tana (g s^{-1}) is a product of DIN or DIP concentration (g m^{-3}) and discharge (m^3 s^{-1})(see eq.4). Based on our data set and because daily discharge and monthly nutrient concentrations were available for the year 2017, the average load to the impact zone was calculated by multiplying the average discharge and the average concentration (De Vries and Klavers, 1994), providing:

$$\boldsymbol{L} = \frac{\sum_{i=1}^{m} Q_i.}{m} . \frac{\sum_{j=1}^{n} Cj.}{n} \tag{4}$$

Where,

L is a load of DIN or DIP (g s^{-1}),

Q_i is the discharge at the gauging stations at the time I (m^3 s^{-1}),

C_j is the concentration at different times j (g m^{-3}),

m is the number of discharge measurements, n is the number of concentration measurements.

Chlorophyll-a was measured as an indicator for the phytoplankton concentration. During three periods (August 2016, December 2016 and March 2017) water samples for chlorophyll-a analysis were taken at 143 locations at Lake Tana in a spatial grid of 5 km by 5 km, and chlorophyll-a samples were also collected from the rivers monthly for 2017. The chlorophyll-a concentration was determined using the acetone extraction procedure (Wetzel and Likens, 1991). From the analysis, a map of the chlorophyll-a over the lake was constructed using the Inverse Distance Weighted (IDW) interpolation technique in ArcGIS.

3. Results

3.1. Current ecological condition of the lake based on measurements

3.1.1. Current status of eutrophication

The effect of eutrophication in Lake Tana clearly showed spatial variability among the shore, littoral, pelagic, and river mouth stations of the lake. Most of the river mouths and a small part of the lakeshore and littoral areas showed relatively high nutrient and chlorophyll-a concentrations, especially after the rainy season (Jun-Sep). The majority of the lake exhibited oligotrophic conditions based on chlorophyll-a concentration (see Fig. 3).

3.1.2. Chlorophyll-a in littoral, pelagic, and river mouths of lake Tana

The chlorophyll-a concentration in Lake Tana shows high spatial and temporal variability and ranged from < 0.5 to 191 μg l^{-1} (Fig. 3).

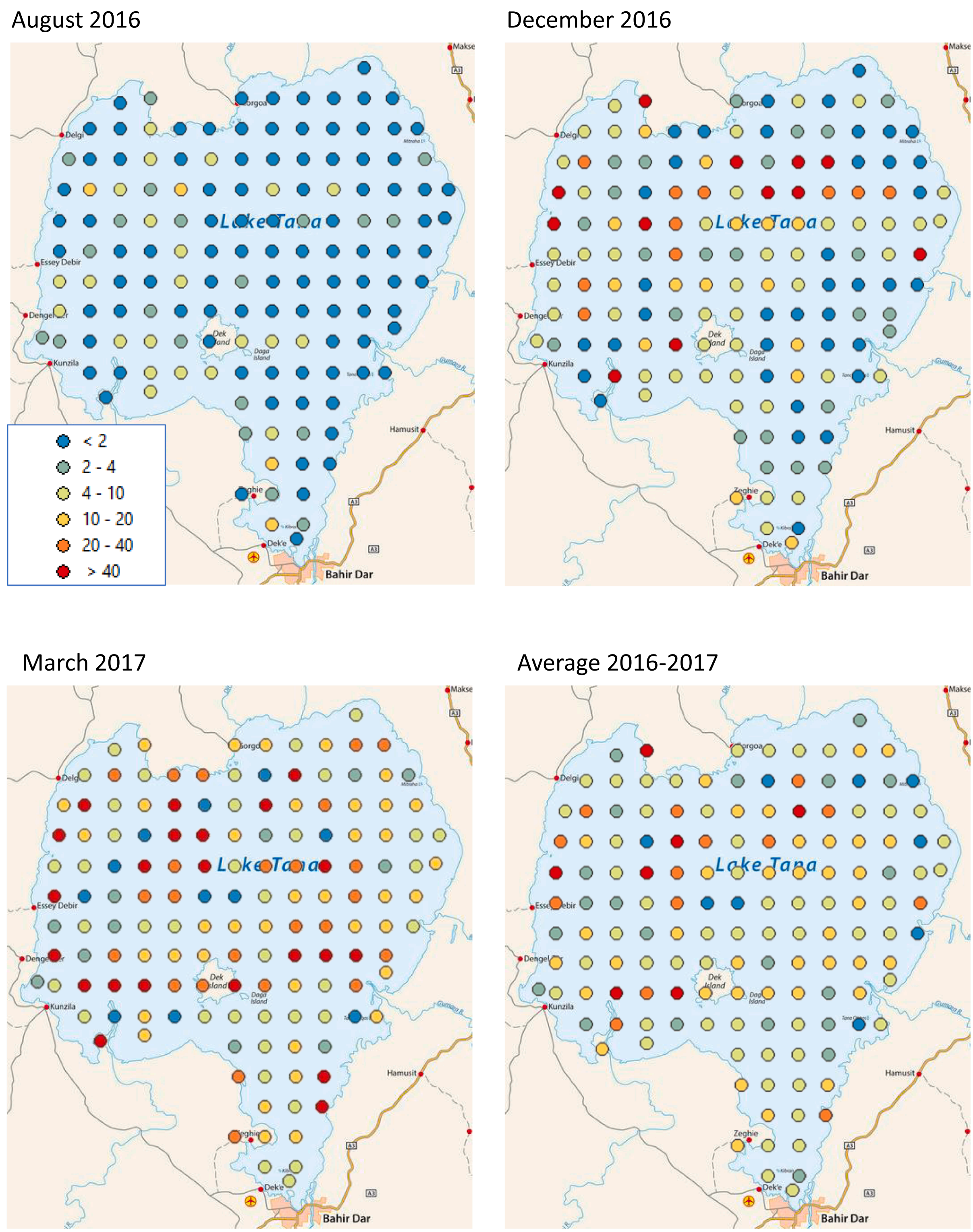

Fig. 3. Spatial Maps of measured Chlorophyll-a (μg l^{-1}) for littoral, pelagic, and river mouths of lake Tana for months of the year 2016/2017.

Table 2
The minimum and maximum Chl-a concentration ($\mu g\ l^{-1}$) of the river mouths of major tributary rivers of lake Tana in 2016/2017 (n = 12).

Statistic	Dirma mouth	Gilgel Abay mouth	Gelda mouth	Gumara mouth	Rib mouth
Min	27	1	15	8	23
Max	94	78	259	185	352

Generally, the chlorophyll-a concentrations were highest in the river mouths (average 87 $\mu g\ l^{-1}$) and lowest in the pelagic zone (average 14 $\mu g\ l^{-1}$) except for some inshore and littoral areas where temporally low chlorophyll-a concentrations were found. This might be caused by the presence of water hyacinth infestation and its ecological effect of depleting nutrients and shading (Gezie et al., 2018).

The three sampling periods (August 2016, December 2017 and

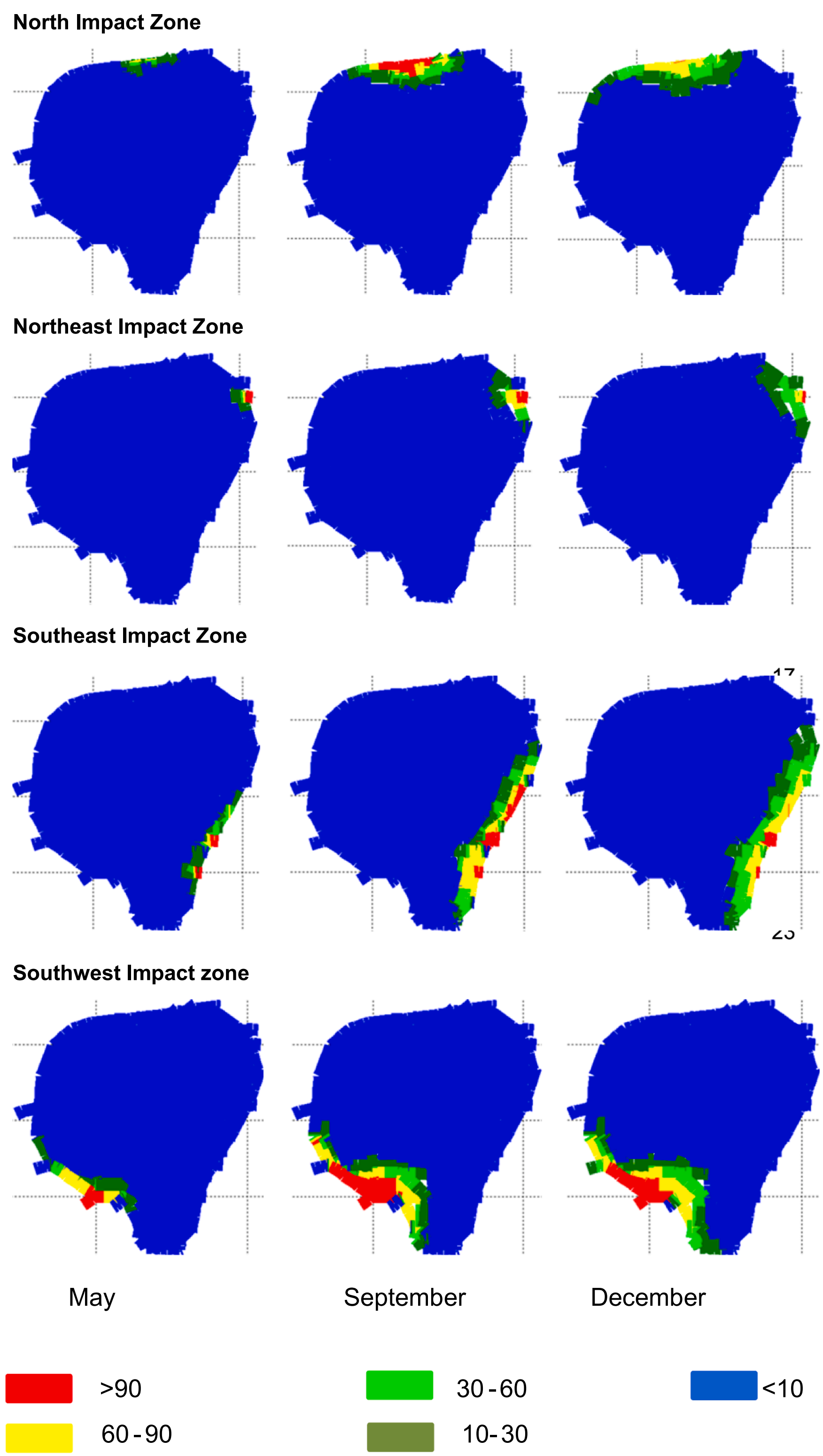

Fig. 4. Spatial coverage of the different impact zones in Lake Tana defined by the tracer study, in May, September and December of the year 2017. (colours indicate tracer concentration (mg l^{-1}); initial concentration in the inflows was 100 mg l^{-1}).

March 2017) show clear seasonal variability in chlorophyll-a concentrations. During the rainy season (August) >85% of Lake Tana had chlorophyll-a concentrations below 4 μg l^{-1}. In December 2016 (post rainy season), the chlorophyll-a concentrations ranged from < 0.5 μg l^{-1} to 148 μg l^{-1}, and more than half of the lake (56 %) had chlorophyll-a concentration between 4 and 27 μg l^{-1}. Finally, in March 2017 (dry season), the chlorophyll-a concentrations ranged from < 0,5 μg l^{-1} to 191 μg l^{-1}, with 59 % of the lake having concentrations above 20 μg l^{-1}.

Spatial and temporal variations of chlorophyll-a concentrations were observed also in the river mouths of Dirma, Gilgel Abay, Gelda, Gumara, and Rib (Table 2). The chlorophyll-a concentrations in the river mouths were generally higher than in the lake (22.8–351.5 μg l^{-1}) probably because of low flow velocities and high nutrient concentrations in these areas. Spatially, the minimum chlorophyll-a concentration was noted in the Gilgel Abay river mouth, and the maximum was measured in the Rib river mouth. Nevertheless, there were no significant spatial and temporal differences in chlorophyll-a concentrations among the river mouths ($P > 0.05$, Repeated Measures ANOVA, $n = 60$).

Based on the monthly measurements in 2017, the minimum chlorophyll-a was noted in the rainy season and the maximum was noted in post-rainy and pre-rainy seasons for most of the river mouths.

Table 3
Impact zones defined by tracer study, and their area in May, Sep., and Dec. of the year 2017; including mean fetch, marsh area, and mean depth. North Impact Zone-Dirma and Megech; North East Impact Zone-Arno-Garno; South East Impact Zone-Rib, Gumara and Gelda, and South West Impact Zone-GilgelAbay.

Impact zone	May	Sep	Dec
Area (km^2)			
North	43.6	213	308
North East	30	88	150
South East	102	362	490
South West	134	365	475
Fetch (km)			
North	7	15	18
North East	5	9	12
South East	10	19	22
South West	12	19	22
Marsh Area (km^2)			
North	0.10	0.50	0.72
North East	0.07	0.21	0.35
South East	0.24	0.84	1.14
South West	0.31	0.85	1.11
Mean depth (m)			
North	2.00	2.79	2.39
North East	0.21	2.00	1.6
South East	2.64	4.43	4.05
South West	1.63	3.41	3.03

3.2. Delineation of the 'impact zones' of major incoming rivers

The contours of the impact zones were defined by the tracer study for May, September, and December of the year 2017 (Fig. 4). We observed a direct relationship between the river basin area and the impact zone area. The larger the river basin area and the subsequent river discharge, the larger the impact zone. The southwest impact zone has the largest area in all months study except December. The northeast impact zone has the smallest area in all months of the model study. Generally, the impact zones have the smallest area in the dry season (May), and the largest in the post rainy season (December). Surprisingly, the impact zones are mainly confined in the shore and littoral parts of the lake and do not cover the pelagic part (Fig. 4).

Based on the impact zones delineation we computed the different characteristics of the separate zones needed for the PCLake modelling. These include the fetch, marsh area and mean depth (Table 3), and the monthly N and P load (Table 5).

3.3. Hydrology and water balance

The water balance study of Lake Tana shows that the water level drops during dry and pre rainy seasons, and increases in the rainy season (Fig. 5, Table 4). This water level change is well considered in the Duflow modelling. Fig, 5 displays the seasonal variation in water

Table 4
Calculated water balance of Lake Tana for the year 2017.

Water balance components	mm year^{-1}	Mm^3 year^{-1}
Qgauged	+2002	+6229
Q_{HP}	+1064	−3311
Q_{Dam}	−319	−994
P	+1561	+4857
E_0	−1584	−4928
Qunguaged	+582	+1810
$\frac{\Delta S}{\Delta T}$	+1178	+3664

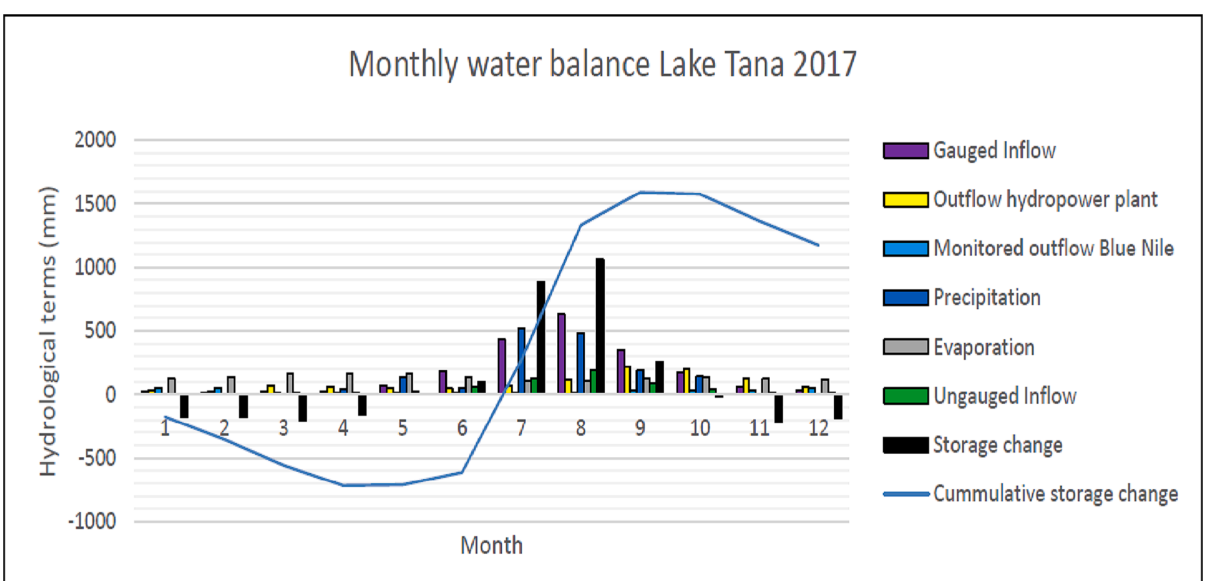

Fig. 5. Monthly water balance of Lake Tana for the year 2017.

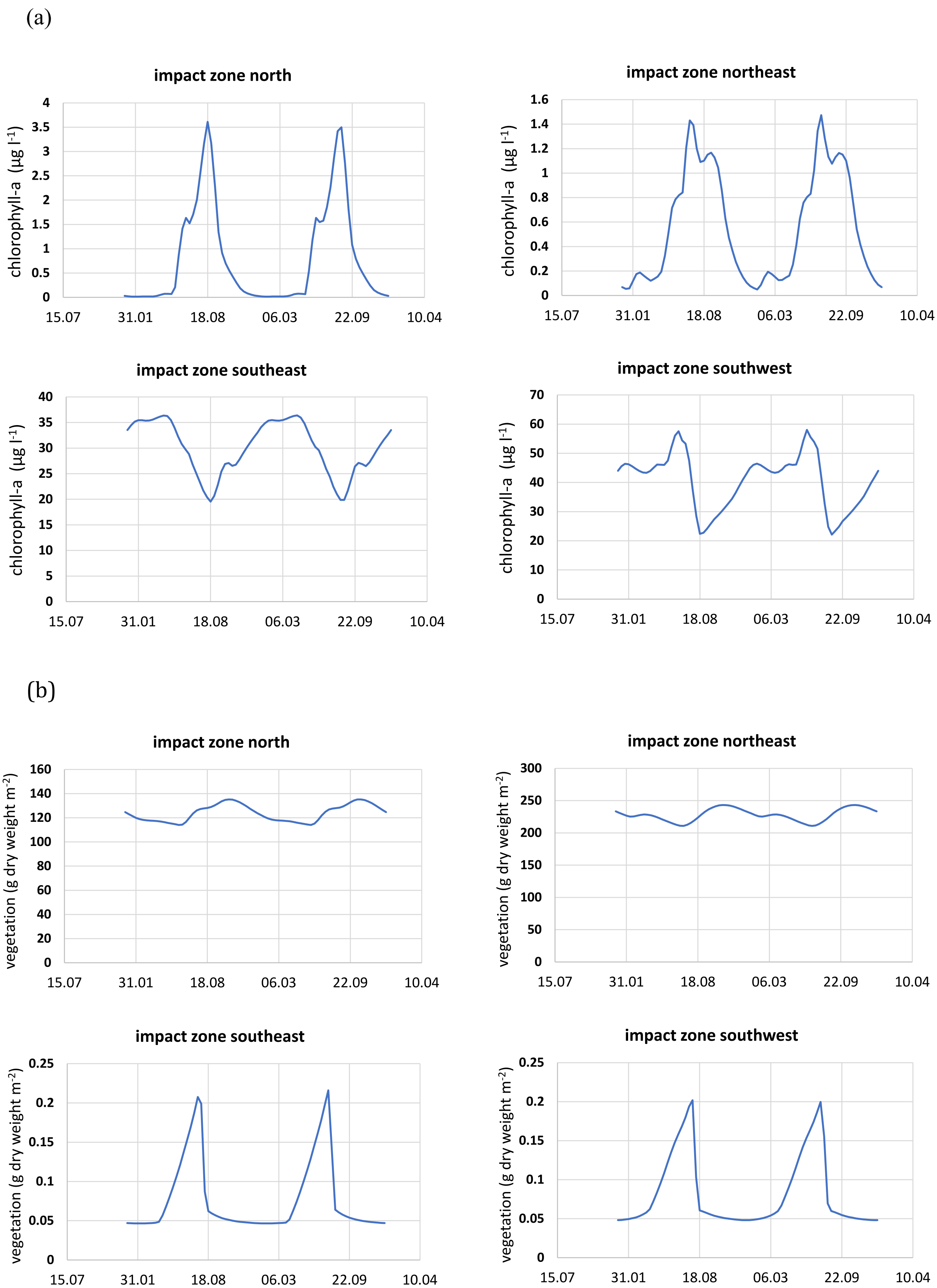

Fig. 6. Simulation results of chlorophyll-a (µg l^{-1}, panel a) and vegetation (g dry weight m^{-2}, panel b) of the four impact zones in Lake Tana in 2017, during the last 2 years of the simulation period.

balance of Lake Tana for 2017. Table 4 gives the calculated annual water balance terms of Lake Tana. The main inflow of water to Lake Tana is coming from the gauged sub-basins. Furthermore, evaporation exceeds precipitation on an annual basis (Table 4). The lake exhibits water storage from May until September, with the largest positive storage for July and August. Despite the outflow flowing into the hydropower plant Tana Beles, and the monitored part of the Blue Nile, there is a large cumulative closure term in the water balance of Lake Tana at the end of the year, which should be compensated by an additional loss term from the lake. This large closure term might be because the largest part of the outflow towards the Blue Nile River is not monitored.

3.4. Simulation of ecological indicators and critical nutrient loads of the four impact zones and lake Tana

The simulation of phytoplankton showed chlorophyll-a dominance in the southeast and southwest zones (maximum chlorophyll-a 40 and 70 μg l^{-1}), compared with the north and northeast zones (maximum chlorophyll-a 3,5 and 1.6 μg l^{-1}) (see Fig. 6a). In contrast, vegetation was more dominant in the north and northeast impact zones (maximum vegetation = 135 and 245 g d.w. m^{-2}) than the southeast and southwest impact zones (maximum vegetation = 0.2 and 0.2 g d.w. m^{-2}) (Fig. 6b). The simulation results of TN, TP, and Secchi depth are shown in Annex 2.

The bifurcation analyses of the four impact zones and Lake Tana as one mixed system are presented in Fig. 7 and Fig. 8. This bifurcation run was done with a changing P-load throughout the year, coupled to the N-load with an TN/TP ratio based on the incoming nutrient fluxes. This approach reflects the effect of a combined reduction of N- and P- loads to the impact zones. The average TN/TP ratio was 4.9, however with a substantial variation among the different the impact zones (2.2 to 8.8, Table 5). All impact zones show a clear deviation from the Redfield N/P ratio of 16 (Tett et al, 1985), indication relative high P loads. The bifurcation analysis clearly indicates critical loadings, except for northeast zone, where a linear increase of chlorophyll with increasing P-load is found. The north and southwest impact zones display hysteresis (2 critical loadings with alternative stable states). Furthermore, Figs. 7 and 8 show that for the two impact zones that are in the turbid state (high chlorophyll-a) the external nutrient loading should be reduced considerably (to around 10% of the current load) to switch back to the clear vegetation state. In contrast, the current nutrient loads for the two impact zones that exhibit vegetation dominance (clear state) are predicted to be below the critical values.

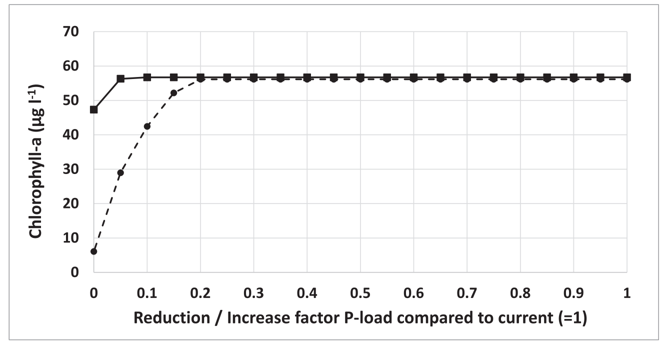

Fig. 8. Bifurcation analysis for the whole lake Tana when modelled as one mixed system. On the x-axes the reduction factor (<1) or increase factor (>1) of the P-load; on the y-axes average chlorophyll concentration in the last 2 years (of 30-years run) in the months May – September. Dotted line is transition from clear to turbid state, solid line is transition from turbid to clear state.

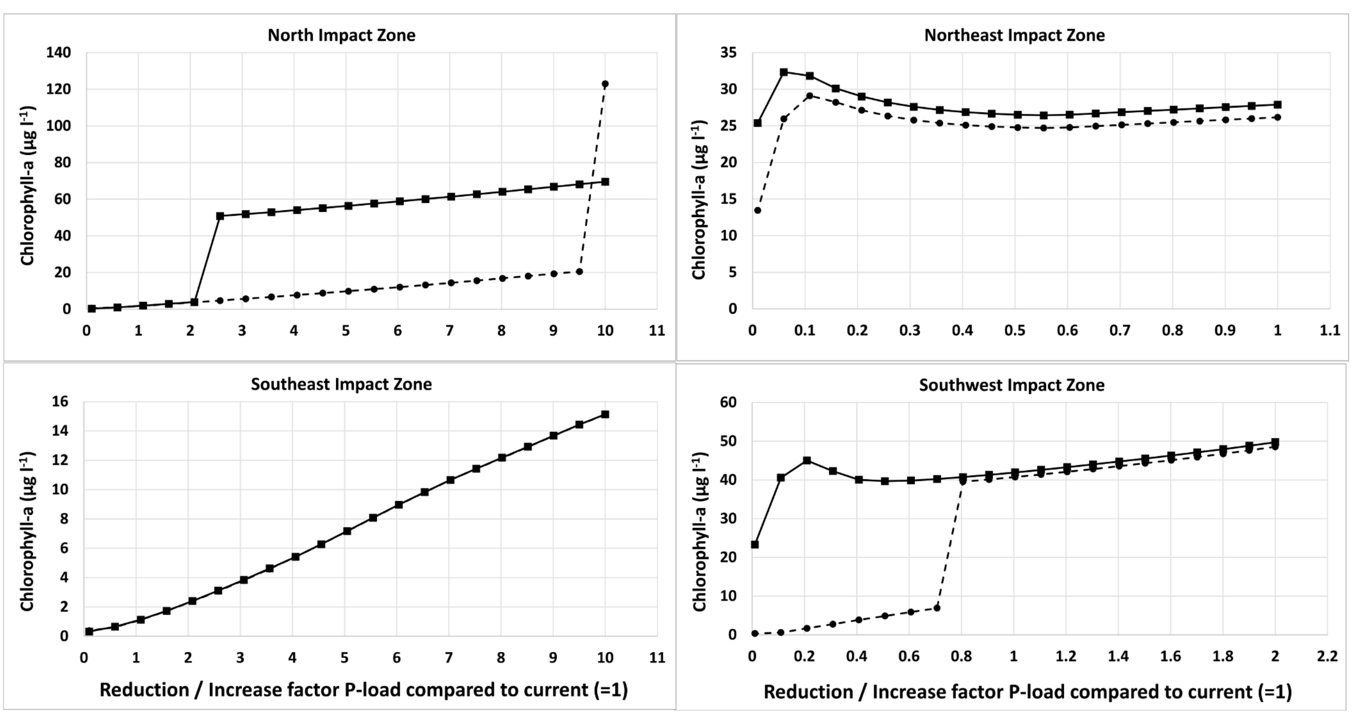

Fig. 7. Bifurcation analysis results of the four impact zones in Lake Tana. On the x-axes are the reduction factor (<1) or increase factor (>1) of the P-load; on the y-axes average chlorophyll concentration in the last 2 years (of 30-years run) in the months May – September. Dotted line is transition from clear to turbid state, solid line is transition from turbid to clear state.

Table 5
N and P loads of the different impact zones for the months of the year 2017.

	North zone		Northeast zone		Southeast zone		Southwest zone	
	Nload	Pload	Nload	Pload	Nload	Pload	Nload	Pload
Months	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}	g m^{-2} d^{-1}
Jan	0.0001	0.0000	0.0002	0.0002	0.0003	7.14E-05	0.0008	0.0003
Feb	0.0024	0.0002	0.0035	0.0027	0.0009	4.10E-04	0.0062	0.0037
Mar	0.0001	0.0001	0.0015	0.0006	0.0028	2.52E-03	0.0017	0.0012
Apr	0.0012	0.0004	0.0000	0.0003	0.0025	6.07E-03	0.0017	0.0012
May	0.0026	0.0002	0.0100	0.0010	0.0054	1.02E-02	0.0311	0.0046
Jun	0.0250	0.0100	0.0195	0.0008	0.0143	1.20E-02	0.1009	0.0752
July	0.0211	0.0080	0.0985	0.0040	0.0241	1.25E-02	0.0439	0.0365
Aug	0.0271	0.0150	0.1138	0.0060	0.0433	2.09E-02	0.0352	0.0000
Sep	0.0083	0.0018	0.0161	0.0033	0.0299	6.42E-02	0.0426	0.0345
Oct	0.0015	0.0012	0.0057	0.0008	0.0063	6.39E-04	0.0200	0.0177
Nov	0.0005	0.0001	0.0012	0.0003	0.0004	4.08E-04	0.0024	0.0001
Dec	0.0002	0.0000	0.0006	0.0001	0.0003	2.02E-04	0.0011	0.0003
Average	0.008	0.003	0.023	0.002	0.011	0.011	0.024	0.015
Average N/P ratio	4.8		8.8		2.2		3.7	

4. Discussion

4.1. Spatial heterogeneity

With the approach presented here we show the importance of spatial variability in analysing the ecological condition and determining critical loads in a large lake. Variability in lake conditions is strongly related to the locations and magnitude of the in flowing rivers. The results for the separate impact zones show clear differences, among each other and also when compared with modelling the lake as one mixed system.

The simulated chlorophyll-a concentrations deviate to some extent from the observed chlorophyll-a concentrations (Fig. 3 and Fig. 6a), although in line with the observations, especially if we consider the average chlorophyll-a concentrations in the impact zones. We found that the order of the impact zones for chlorophyll-a concentrations is the same for the model and the observations with highest average concentrations in the southwest and lowest in the northeast (Table 6). The temporal variation of higher concentrations in March 2017 compared with August 2016 is clear and reflects the actual observations.

According to the model there are vegetation dominance and low chlorophyll-a concentrations in impact zones north and northeast, and the other way around in southeast and southwest impact zones. This finding is consistent with previous studies in Lake Tana (Dersseh et al. 2019; Wondie et al. 2012). In these studies the spatial potential of aquatic vegetation infestation was estimated based on a multivariate analysis with total phosphorus, total nitrogen, temperature, pH, salinity, and depth as main parameters. The researchers concluded that the north and northeast regions have the highest potential for vegetation dominance, which was later confirmed with satellite image observations (Dersseh et al. 2020).

Our finding of spatial variability of eutrophication in Lake Tana is also reported in other tropical and sub-tropical large lakes of the world. Lake Taihu (Southeast China) is a good example of a large shallow lake with high spatial variation; macrophytes are established at the shores and in the bays, whereas they are absent in the lake's centre due to strong wind (Janssen et al., 2014; Xu et al., 2015b; Zhao et al., 2013).

Table 6
Average chlorophyll-a concentrations of the impact zones, from measurements and from the model (μg l^{-1}). Between brackets [ ] the ranking of the impact zones based on the average concentrations.

	average chlorophyll-a concentration (μg l^{-1})	
Impact zone	measurements	model
North zone	9.8 [2]	3.1 [2]
Northeast zone	7.6 [1]	1.2 [1]
Southeast zone	12.6 [3]	29.0 [3]
Southwest zone	19.1 [4]	36.9 [4]

The study by Soares et al. (2012) reported that the tropical Funil Reservoir (Brazil) demonstrated spatial heterogeneity that could affect the occurrence and distribution of algal blooms along the reservoir, the fluvial, intermediate, and lentic compartments. This supports the conclusion that high spatial heterogeneity in macrophytes and phytoplankton abundance suggests variation in the responses to eutrophication within the lake. Differences in lake morphology can already result in spatial variation, with water depth being an important morphological factor regulating macrophyte abundance. In Lake Tana most of the studied littoral zones have comparable depths but still differ in macrophyte cover, which can be attributed to different nutrient inputs. This variation results in spatial patterning of critical nutrient loads in Lake Tana.

4.2. Critical loads and bifurcation analysis

The critical nutrient loads for the impact zones and the lake as a whole are summarized in Table 7. These are derived from the average current loads (Table 5) and the reduction factors (multipliers) from Figs. 7 and 8. The presented reduction factors are an annual average and given the variability of the current monthly loads these can only be used indicatively. However, the critical loads estimated here enable a comparison with other studies in lakes where critical loads were estimated.

From Table 7 it is evident that current nutrient loads, to southeast and southwest impact zones are higher than the estimated critical loads to these impact zones. On the other hand, we found a different ecological condition in the north and northeast impact zone, where the

Table 7
The current average P-loads, reduction/increase factors (Turbid to clear and Clear to Turbid), and critical loads (Turbid to Clear and Clear to Turbid) of the separate impact zones and Lake Tana.

	current average P-load	multiplier Turbid to Clear	multiplier Clear to Turbid	critical load Turbid to Clear	critical load Clear to Turbid
	(mg m^{-2} d^{-1})	(-)	(-)	(mg m^{-2} d^{-1})	(mg m^{-2} d^{-1})
North Zone	3	2.5	9.5	7.5	28.5
Northeast Zone	2	0.05	0.1	0.1	0.2
South East Zone	11	undetermined	undetermined		
South west Zone	15	0.2	0.7	3	10.5
Lake Tana	3.9	0.05	0.2	0.2	0.78

phytoplankton is less dominating.

Lake Tana is a large tropical lake with a mean depth of 8 m, but the impact zones are generally only 2 m deep. This is comparable to shallow Dutch and Chinese lakes where PCLake was applied. In Janse et al. (2008), critical P-loadings were determined that can be considered representative of Dutch shallow lakes. It was observed that for a default lake (see exact settings in Janse, et al., 2008), critical bifurcation points of the lake, turning from a turbid to clear, and clear to the turbid state were 0.9 mg P m^{-2} d^{-1} and 3 mg P m^{-2} d^{-1} respectively. The general numbers for lake Tana are similar (Table 6), but the range is larger (0.1 to 28.5 mg m^{-2} d^{-1}), possibly due to the spatial heterogeneity of the impact zones with different characteristics. Studies with PCLake + for deeper lakes, such as described in (Li et al., 2019) for Lake Dianchi with an average depth of 4.9 m, showed critical loadings of 0.34 mg m^{-2} d^{-1} and 0.38 mg m^{-2} d^{-1}. Furthermore, lakes with larger depths will have lower critical P-loadings because less submerged macrophytes will be present, due to less favourable light conditions in deeper lakes (Janse, 2005; Janse et al., 2008). This causes more nutrients available for the growth of phytoplankton causing more turbid conditions at lower nutrient loadings.

4.3. Model uncertainties

Though PCLake was originally parameterized for shallow temperate lakes (Janse et al 2010), the ecological and chemical concepts can be applied to Lake Tana to provide a comparative first-cut assessment of relative spatial variability. This is supported by studies where the model in its present form has been applied successfully in several tropical and subtropical lakes (Janssen et al., 2017; Li et al., 2019; Kong et al., 2017; Fragoso et al., 2008). A multi-lake study showed that the original PCLake model was able to predict reasonably well the dominance of macrophytes or phytoplankton in approximately 80 large lakes from around the world (Janssen et al., 2014). Still, the model results should be considered with care, and the focus should be on patterns rather than on absolute values of the output. We acknowledge that the derived critical loads for lake Tana contain uncertainties, but comparison with critical loads in other regions shows that the values for lake Tana can be plausible. Lake Tana is one of the first attempts of applying PCLake to African lakes. Future monitoring and modelling in tropical lakes can contribute to reducing the uncertainties, specifically frequent observations of N and P concentrations, phytoplankton, transparency and vegetation cover and densities. In addition, although the processes and feedbacks in the model are generic, process parameters and environmental conditions have to be determined on a local scale.

4.4. Implications

Studies conducted in China and Brazil have shown the importance of spatial heterogeneity of shallow lakes (Janssen et al., 2017; Fragoso et al., 2008). For ecological modelling as well as defining nutrient management options, large tropical lakes like Lake Tana are traditionally treated as completely mixed systems. Because of this, the traditional approach of nutrient management has only limited success. Our research in this lake has clearly shown that importance of spatial heterogeneity in terms of eutrophication effects and critical nutrient loads.

Our spatial modelling provides new insights into exploring the effects of eutrophication in a large shallow tropical lake. We demonstrate that lake Tana is a heterogeneous lake and the conditions that drive this are likely to be existing also in other lakes. The modelled impact zones clearly show spatial differences in N and P critical loadings, implying that each zone has a different nutrient reduction target. Therefore, in impact zones like the southeast and southwest parts of Lake Tana, priority should be given to nutrient reduction measures. Lake managers could aim to reduce the present nutrient load to below the lowest critical nutrient load in these impact zones. The major N and P sources of lake Tana are animal manure, synthetic fertilizers, and human waste (Goshu et al., 2020). Therefore, nutrient reduction measures from the above sources should be practiced in sub-basins of prioritized impact zones of Lake Tana. Measures should be within the social and economic possibilities of the regions.

5. Conclusions

This study showed that there are clear distinct impact zones in lake Tana, related to the main tributaries, with different characteristics. Both the measurements and the modelling with PCLake + show that the current ecological status has spatial variability over the lake and among the impact zones. Critical nutrient loadings vary among the impact zones, implying that each zone has a different reduction target. For lake Tana it is clear that in north impact zone and northeast impact zone, the current level of P load is below the critical loading. In the other two zones, the current loading is above the critical loading, so in the southeast and southwest parts of the lake, the river loading should be decreased substantially. The critical loadings of N and P are quite different when we take Lake Tana as one mixed system. Our findings demonstrate the importance of accounting for spatial heterogeneity of eutrophication to determine sub-catchment critical nutrient loads in a relatively large lake. We acknowledge that the derived critical loads for lake Tana contain uncertainties. Comparison with critical loads in other regions has limited value, but it shows that the values for lake Tana can be plausible. Future monitoring and modelling in tropical lakes can contribute to reducing the uncertainties. Nevertheless, we believe that the general conclusion of importance of spatial heterogeneity in large shallow lakes is justified here.

CRediT authorship contribution statement

Goraw Goshu: Conceptualization, Methodology, Validation, Writing – original draft, Funding acquisition. **Micha Veenendaal:** Methodology, Validation, Writing – review & editing. **Jeroen de Klein:** Conceptualization, Methodology, Validation, Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2023.129815.

References

Abebe WB, Minale AS. Land Use and Watershed Management Practices in Lake Tana Basin. Social and Ecological System Dynamics. Springer, 2017, pp. 479-521.

Anteneh, M., 2017. Demographic Characteristics of the Lake Tana Basin. Social and Ecological System Dynamics. Springer, pp. 283–292.

Arani, B. M. S., S. R. Carpenter, L. Lahti, E. H. Van Nes, and M. Scheffer.. Exit time as a measure of ecological resilience. Science 2021, 372 eaay4895.

Aynalem S, Goshu G, Wondie A. Wetlands of the Lake Tana watershed. Social and Ecological System Dynamics. Springer, 2017, pp. 245-256.

Capon, S.J., Lynch, A.J.J., Bond, N., Chessman, B.C., Davis, J., Davidson, N., Finlayson, M., Gell, P.A., Hohnberg, D., Humphrey, C., Kingsford, R.T., Nielsen, D., Thomson, J.R., Ward, K., Nally, R.M., 2015. Regime shifts, thresholds and multiple stable states in freshwater ecosystems; a critical appraisal of the evidence. Sci. Total Environ. 534, 122–130.

Clemmens, A.J., Holly, F.M., Schuurmans, W., 1993. Description and evaluation of program: Duflow. J. Irrig. Drain. Eng. 119 (4), 724–734.

De Vries, A., Klavers, H., 1994. Riverine fluxes of pollutants: monitoring strategy first, calculation methods second. Eur. Water Manag. 4, 12–17.

Dersseh, M.G., Kibret, A.A., Tilahun, S.A., Worqlul, A.W., Moges, M.A., Dagnew, D.C., et al., 2019. Potential of water hyacinth infestation on lake tana, ethiopia: A prediction using a gis-based multi-criteria technique. Water 11, 1921.

Dersseh, M.G., Tilahun, S.A., Worqlul, A.W., Moges, M.A., Abebe, W.B., Mhiret, D.A., Melesse, A.M., 2020. Spatial and Temporal Dynamics of Water Hyacinth and Its Linkage with Lake-Level Fluctuation: Lake Tana, a Sub-Humid Region of the Ethiopian Highlands. Water 12, 1435.

Ding, J., Cao, J., Xu, Q., Xi, B., Su, J., Gao, R., Huo, S., Liu, H., 2015. Spatial heterogeneity of lake eutrophication caused by physiogeographic conditions: An analysis of 143 lakes in China. J. Environ. Sci. 30, 140–147.

Fragoso, C.R., Marques, D.M.L.M., Collischonn, W., Tucci, C.E.M., van Nes, E.H., 2008. Modelling spatial heterogeneity of phytoplankton in Lake Mangueira, a large shallow subtropical lake in South Brazil. Ecol. Model. 219 (1-2), 125–137.

Gezie, A., Assefa, W.W., Getnet, B., Anteneh, W., Dejen, E., Mereta, S.T., 2018. Potential impacts of water hyacinth invasion and management on water quality and human health in Lake Tana watershed, Northwest Ethiopia. Biol. Invasions 20 (9), 2517–2534.

Goshu, G., Strokal, M., Kroeze, C., Koelmans, A.A., de Klein, J.J.M., 2020. Assessing seasonal nitrogen export to large tropical lakes. Sci. Total Environ. 731, 139199.

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9 (1), 1–13.

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.

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

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 (3-4), 203–219.

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. Ecol. Model. 221 (4), 654–665.

Janssen, A.B.G., Teurlincx, S., An, S., Janse, J.H., Paerl, H.W., Mooij, W.M., 2014. Alternative stable states in large shallow lakes? J. Great Lakes Res. 40 (4), 813–826.

Janssen, A.B.G., de Jager, V.C.L., Janse, J.H., Kong, X., Liu, S., Ye, Q., Mooij, W.M., 2017. Spatial identification of critical nutrient loads of large shallow lakes: implications for Lake Taihu (China). Water Res. 119, 276–287.

Janssen, A.B., Janse, J.H., Beusen, A.H., Chang, M., Harrison, J.A., Huttunen, I., Kong, X., Rost, J., Teurlincx, S., Troost, T.A., van Wijk, D., Mooij, W.M., 2019b. How to model algal blooms in any lake on earth. Curr. Opin. Environ. Sustain. 36, 1–10.

Janssen, A.B., Teurlincx, S., Beusen, A.H., Huijbregts, M.A., Rost, J., Schipper, A.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.

Janssen, A.B.G., van Wijk, D., van Gerven, L.P.A., Bakker, E.S., Brederveld, R.J., DeAngelis, D.L., Janse, J.H., Mooij, W.M., 2019a. Success of lake restoration depends on spatial aspects of nutrient loading and hydrology. Sci. Total Environ. 679, 248–259.

Kane, D.D., Conroy, J.D., Peter Richards, R., Baker, D.B., Culver, D.A., 2014. Re-eutrophication of Lake Erie: Correlations between tributary nutrient loads and phytoplankton biomass. J. Great Lakes Res. 40 (3), 496–501.

Kebedew, M.G., Tilahun, S.A., Zimale, F.A., Steenhuis, T.S., 2020. Bottom sediment characteristics of a tropical lake: Lake Tana. Ethiopia. Hydrology 7, 18.

Kong, X., He, Q., Yang, B., He, W., Xu, F., Janssen, A.B.G., Kuiper, J.J., van Gerven, L.P.A., Qin, N., Jiang, Y., Liu, W., Yang, C., Bai, Z., Zhang, M., Kong, F., Janse, J.H., Mooij, W.M., 2017. Hydrological regulation drives regime shifts: evidence from paleolimnology and ecosystem modeling of a large shallow Chinese lake. Global Change Biol. 23 (2), 737–754.

Kosten, SARIAN, Vernooij, MARTINE, Van nes, E.H., Sagrario, MARÍA.DE.L.Á.G., Clevers, J.G.P.W., Scheffer, MARTEN, 2012. Bimodal transparency as an indicator for alternative states in South American lakes. Freshw Biol. 57 (6), 1191–1201.

Li, X., Janssen, A.B.G., de Klein, J.J.M., Kroeze, C., Strokal, M., Ma, L., Zheng, Y.i., 2019. Modeling nutrients in Lake Dianchi (China) and its watershed. Agric Water Manag 212, 48–59.

Liu, W., Qiu, R., 2007. Water eutrophication in China and the combating strategies. J. Chem. Technol. Biotechnol.International Research in Process, Environmental & Clean Technology 82 (9), 781–786.

Lürling, M., Oosterhout, F.V., 2013. Controlling eutrophication by combined bloom precipitation and sediment phosphorus inactivation. Water Res. 47 (17), 6527–6537.

Mamo, S., Ayenew, T., Berehanu, B., Kebede, S., 2016. Hydrology of the Lake Tana Basin, Ethiopia: Implication to Groundwater-Surface Waters Interaction. J. Environ. Earth Sci 6, 54–66.

Nigate, F., Ayenew, T., Belete, W., Walraevens, K., 2017. Overview of the Hydrogeology and Groundwater Occurrence in the Lake Tana Basin, Upper Blue Nile River Basin. Social and ecological system dynamics. Springer 77–91.

Pringle, C.M., 1990. Nutrient spatial heterogeneity: effects on community structure, physiognomy, and diversity of stream algae. Ecology 71, 905–920.

Rahm, L., Danielsson, Å., 2007. Spatial heterogeneity of nutrients in the Baltic Proper, Baltic Sea. Estuar. Coast. Shelf Sci. 73 (1-2), 268–278.

Rychtecký, P., Znachor, P., 2011. Spatial heterogeneity and seasonal succession of phytoplankton along the longitudinal gradient in a eutrophic reservoir. Hydrobiologia 663 (1), 175–186.

Scheffer, M., Carpenter, S.R., 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends Ecol. Evol. 18 (12), 648–656.

Scheffer, M., Ecology of shallow lakes,1998.Chapman and Hall, London, 0–357.

Sirak, T., 2008. Watershed Modelling of Lake Tana Basin Using SWAT. Arbaminch, Ethiopia.

Soares, M.C.S., Marinho, M.M., Azevedo, S.M.O.F., Branco, C.W.C., Huszar, V.L.M., 2012. Eutrophication and retention time affecting spatial heterogeneity in a tropical reservoir. Limnologica 42 (3), 197–203.

Sollie, S., Verhoeven, J.T.A., 2008. Nutrient cycling and retention along a littoral gradient in a Dutch shallow lake in relation to water level regime. Water Air Soil Pollut. 193 (1-4), 107–121.

Tett, P., Droop, M., Heaney, S., 1985. The Redfield Ratio and Phytoplankton Growth Rate. J. Mar. Biol. Assoc. U. K. 65 (2), 487–504. https://doi.org/10.1017/S0025315400050566.

Van Geest, G.J., Coops, H., Scheffer, M., van Nes, E.H., 2007. Long Transients Near the Ghost of a Stable State in Eutrophic Shallow Lakes with Fluctuating Water Levels. Ecosystems 10 (1), 37–47.

Wale, A., Rientjes, T., Dost, R., Gieske, A., 2008. Hydrological Balance of Lake Tana Upper Blue Nile Basin. Ethiopia, ITC The Netherlands.

Wetzel, R.G., Likens, G.E., 1991. Primary productivity of phytoplankton. In: Wetzel, R.G., Likens, G.E. (Eds.), Limnological Analyses. Springer New York, New York, NY, pp. 207–226.

Wondie A, Seid A, Molla E, Goshu G, Gkidan W, Shibabaw A, et al. Preliminary assessment of water hyacinth (Eichornia crassipes) in Lake Tana. Proceedings of the National Workshop (Biological Society of Ethiopia), Addis Ababa, Ethiopia, 2012.

Xu T, Yang T, Xiong M. Time scales of external loading and spatial heterogeneity in nutrients-chlorophyll a response: Implication on eutrophication control in a large shallow lake. Ecological Engineering 2020; 142: 105636.

Xu, H., Paerl, H.W., Qin, B., Zhu, G., Hall, N.S., Wu, Y., 2015. Determining critical nutrient thresholds needed to control harmful cyanobacterial blooms in eutrophic Lake Taihu. China. Environmental science & technology 49 (2), 1051–1059.

Yu, Y., Yan, Q., Feng, W., 2008. Spatiotemporal heterogeneity of plankton communities in Lake Donghu, China, as revealed by PCR–denaturing gradient gel electrophoresis and its relation to biotic and abiotic factors. FEMS Microbiol. Ecol. 63, 328–337.

Zhao D, Lv M, Jiang H, Cai Y, Xu D, An S. Spatio-temporal variability of aquatic vegetation in Taihu Lake over the past 30 years. PLoS One 2013; 8: e66365.