Journal of Contaminant Hydrology 268 (2025) 104469

Contents lists available at ScienceDirect

Journal of Contaminant Hydrology

journal homepage: www.elsevier.com/locate/jconhyd

Use of process-based coupled ecological-hydrodynamic models to support lake water ecosystem service protection planning at the regional scale

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ARTICLE INFO

Keywords:
QWET
Subalpine lake
Watershed nutrient load
Lake water quality
Eutrophication recovery
WFD monitoring

ABSTRACT

Protection plans of lake waters are based on ecological and/or chemical targets, often simplified in terms of total phosphorus (*TP*) concentrations, customarily the depth-averaged ones at spring mixing for temperate environments. These target lake *TP* concentrations are then commonly employed to determine target external loading through reverse use of Vollenweider-OECD-type steady-state empirical models. Such models are also adopted in their direct form to estimate lake *TP* concentrations following hypothetical external load reductions. However, such approaches suffer from extreme parameterisation and often give inaccurate results. Process-based coupled ecological-hydrodynamic models offer a much wider flexibility and produce an extensive set of information, solving many of the issues of Vollenweider-OECD-type models. However, their application has been up to now restricted to single lakes due to calibration effort and data availability burdens. To overcome these obstacles, in this study we developed a simplified application of the process-based coupled model QWET over 9 lakes in Northern Italy, making use of the ParSAC automatic calibration tool and feeding the models only with general data available from public monitoring. QWET models were calibrated over past observations, simulating nutrient reduction scenarios for the near-future decades. The advantages over traditionally employed models for lake water protection planning at the regional scale were hence identified through a practical application, determining the strengths and limits of the herein-adopted simplified process-based approach over lakes with different features. Obtained results were also analysed considering the specific case study.

1. Introduction

Protection plans of lake waters worldwide are based on ecological and/or chemical targets (EU WFD, 2000; Poikane et al., 2019a, 2019b). Often, concentrations of limiting total phosphorus (*TP*) are then used as condensed reference parameters for simplicity (Stow et al., 2014; Poikane et al., 2019a, 2019b), depth-averaged *TP* concentrations at spring mixing being generally considered for holomictic lakes in temperate zones (Dresti et al., 2023a). The rationale is that the main cause of water quality problems in lakes is eutrophication due to excessive nutrient loads (Hutchinson, 1973; Schindler et al., 2016). Depth-averaged lake *TP* concentrations at spring mixing are in fact a proxy of both lake annual productivity and watershed loading, as given by Vollenweider-OECD (Vollenweider, 1968; Vollenweider and Kerekes, 1982) and similar (Bryhn and Håkanson, 2007) models. These empirical models have been commonly used in water protection plans both in their reverse form, to estimate target *TP* external loads from target lake *TP* concentrations at spring mixing, and in their direct form, to forecast lake *TP* concentrations in response to external load reduction scenarios (Trolle et al., 2011). Vollenweider-OECD-type models are also often used in

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https://doi.org/10.1016/j.jconhyd.2024.104469
Received 20 August 2024; Received in revised form 12 November 2024; Accepted 22 November 2024
Available online 26 November 2024

their reverse form to assess present and past *TP* external loads from the watershed, their estimation being usually problematic. Incoming loads are in fact either theoretically estimated as constant values based on land use and occupation (Rast and Lee, 1983; Behrendt, 1996), regardless of their considerable dependence on annual rainfall (Moatar et al., 2017; Fenocchi et al., 2023), or experimentally evaluated, yet mostly through time-limited campaigns encompassing only major point inflows (Knapp et al., 2020; Fenocchi et al., 2023).

Despite Vollenweider-OECD-type empirical models having generally served as effective conceptual and practical tools for the development of restoration strategies (Steward and Lowe, 2009), they have multiple limitations. In fact, their response is steady-state, they neglect stratification and internal loading, and their model parameters were derived from regressions over groups of sample lakes (Vollenweider and Kerekes, 1982), so that the specific features of single basins cannot be considered (Håkanson, 1999). This has sometimes led to strongly incorrect results, such as repeated overestimations of the efficiency of re-oligotrophication measures (Bryhn and Håkanson, 2007; Lepori et al., 2022). Among the causes of these misestimations are enduring internal load, due to the slow release from sediments of phosphorus collected with the past pollution (Søndergaard et al., 2003; Jensen et al., 2006), and hypolimnetic nutrient accumulation (Rogora et al., 2018, 2021; Fenocchi et al., 2020), caused by weakened mixing in deep lakes with climate warming (Livingstone, 2003; Fenocchi et al., 2018). Parametric descriptions of lake *TP* concentration response times to external load reductions to enhance the information given by Vollenweider-OECD-type steady-state models have been proposed (Rossi et al., 1975; Rossi et al., 1986), yet their accuracy has always been questionable. Furthermore, sometimes target lake *TP* concentrations at spring mixing in water protection plans are defined in relation to hypothetical pristine concentrations from natural external loading, obtained through regressive approaches such as the Morpho-Edaphic Index (*MEI*), calculated as ratio between lake alkalinity and mean depth (Vighi and Chiaudani, 1985; Salerno et al., 2014). This application of multiple parametric approaches may result in significant errors, so that, in some instances, actual lake *TP* concentrations lower than the estimated pristine ones have been observed after external and/or internal load reduction interventions.

Many problems of Vollenweider-OECD-type models can be solved by process-based models. In fact, these numerical models allow interpreting a far higher external factor variability than empirical models due to their mechanistic and dynamic reproduction of processes, thus leading to more reliable forecasts (Bryhn and Håkanson, 2007; Bhagowati and Ahamad, 2019). Process-based models of lake ecological dynamics can be divided into pure ecological models, adopting a dynamic approach for the biogeochemical components and fixing lake physics to single or multiple pre-determined completely mixed layers, and coupled ecological-hydrodynamic models, which also dynamically model the physical processes (Bhagowati and Ahamad, 2019; Dresti et al., 2021). As such, the latter models are more flexible and accurate, especially when considering climate warming, which affects lake water temperatures and stratification and mixing evolution (Dresti et al., 2021). Coupled ecological-hydrodynamic models reproduce daily values at all depths for multiple variables, thus allowing a far more complete evaluation of lake water quality evolution than considering depth-averaged *TP* concentrations at spring mixing alone. Among them, one-dimensional (1D) models over the water column have been proven to be computationally affordable and to suitably reproduce lake stratification and mixing dynamics (Perroud et al., 2009; Dresti et al., 2021), in addition to lake chemistry and primary production (Rinke et al., 2009, 2010; Fenocchi et al., 2019).

Coupled ecological-hydrodynamic models have been so far applied to single basins. Over the regional scale required by water protection plans, it is usually concluded that their mass application is precluded by: (1) lake heterogeneity, preventing the use of a uniform modelling framework; (2) different data availability across basins (Ferreira et al., 2007; Valerio et al., 2022); (3) unsustainable efforts needed to serially calibrate coupled models through user-sensitive manual calibration (Fenocchi et al., 2019; Dresti et al., 2021). Because of these issues, many protection plans still rely on Vollenweider-OECD-type steady-state interpretations (Bhagowati and Ahamad, 2019). Nevertheless, as regards (1), process-based models have a broader applicability range than empirical ones, being much less parametrical. Addressing (2), regulations such as the Water Framework Directive, 2000/60/EC of the European Union (EU WFD, 2000) institutionalised lake monitoring, enabling the creation of a minimum common water quality database for all relevant lakes. Last, referring to (3), the coupled ecological-hydrodynamic 1D model QWET (Nielsen et al., 2017; Nielsen et al., 2021; Schnedler-Meyer et al., 2022) includes the ParSAC (Parallel Sensitivity Analysis and Calibration) multi-objective optimisation tool (Bruggeman and Bolding, 2020), allowing automatic calibration for the many parameters of coupled ecological-hydrodynamic models (Fenocchi et al., 2019; Dresti et al., 2021).

In this work, we hence tested a standardised simplified application of QWET at the regional scale over 9 lakes in Northern Italy with heterogeneous morphometrical and ecological properties. We relied for model setup and calibration only on data of common availability gathered within public monitoring activities. As time series of external loads are rarely available, we employed constant mean annual estimations, obtained from previous studies. Automatic calibration was systematically applied. Through this work, we wanted to: (1) quantify the benefits of simplified process-based modelling over empirical approaches for lake water ecosystem service protection planning at the regional scale; (2) check if general data from public monitoring activities allow implementing solid coupled ecological-hydrodynamic models; (3) discover the limits of employing constant external loads over different types of lakes; (4) verify if consistent results could be obtained through automatic calibration of models.

2. Data and methods

2.1. Case studies

Nine lakes in the prealpine range of Northern Italy, south of the European Alps, are the object of this modelling study (Fig. 1; Table 1). These include three large deep lakes (Lake Como, Lake Garda, Lake Iseo), one mid-sized deep lake (Lake Idro), one small, shallow alpine lake (Lake Endine) and four neighbouring small to mid-sized, shallow to moderately deep moraine lakes (Lake Alserio, Lake Annone East, Lake Annone West, Lake Pusiano). All these lakes experienced a common eutrophication process which started in the 1960s and peaked in the early 1980s (Salmaso and Mosello, 2010). National regulations then demanded the establishment of wastewater treatment plants and the reduction of phosphorus in detergents (Copetti et al., 2019), triggering a recovery process, with different velocities among lakes according to past pollution levels, basin morphometry and intervention efficiency (Salmaso et al., 2020).

The considered deep lakes are all experiencing a reduction in convective mixing frequency due to climate warming (Ambrosetti and Barbanti, 1999; Fenocchi et al., 2018), transitioning towards meromixis, supplemented by a decrease of deep intrusions by tributaries (Dresti et al., 2023b). These effects are exacerbated for Lake Iseo (Ambrosetti and Barbanti, 2005) and especially Lake Idro (Viaroli et al., 2018; Tartari et al., 2021), two meromictic basins that show increasing chemical stability due to intense calcite ($CaCO_3$) precipitation. Water warming, together with mixing weakening, has already caused ecosystem shifting for all considered lakes (Salmaso et al., 2014), hampering re-oligotrophication efforts through cyanobacterial blooms (Morabito et al., 2018) and bottom nutrient accumulation for deep basins (Rogora et al., 2018, 2021).

Water quality targets established by the EU through the WFD prescribe that all lakes should achieve by 2027 the “good” ecological status

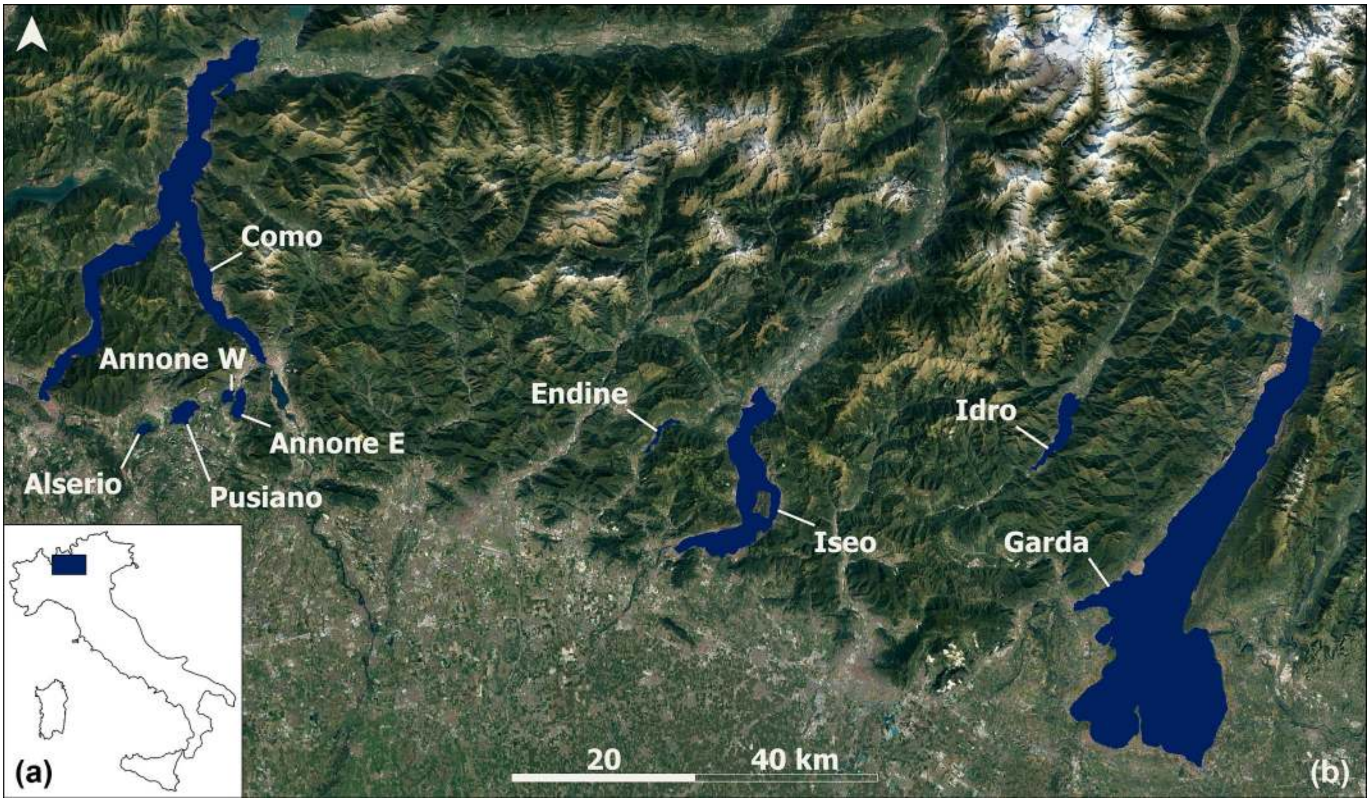

Fig. 1. Position within Italy (a) and map (b) of the lakes considered in the study (orthophoto courtesy of Google).

Table 1
Morphometrical and hydrological properties of the lakes considered in the study (morphometrical information was obtained from available bathymetrical data; * = Regione Lombardia, *unpublished data*; † = Fenocchi et al., 2023; ‡ = Hinegk et al., 2022).

Lake	Surface [km^2]	Volume [10^6 m^3]	Maximum depth [m]	Mean outflowing discharge [m^3 s^{-1}]	Renewal time [a]	Watershed area [km^2]
Alserio	1.3	7.0	8	1.0*	0.2	20.0*
Annone East	3.9	25.8	11	0.6*	1.4	27.3*
Annone West	1.7	6.8	10	0.4*	0.5	12.5*
Como	143.8	25979.4	426	174.0†	4.7	4547.5*
Endine	2.3	12.0	9	1.4*	0.3	36.5*
Garda	368.0	49183.7	350	53.0‡	29.4	2216.0*
Idro	11.3	834.1	124	30.4*	0.9	620.3*
Iseo	61.2	7973.8	252	53.6*	4.7	1800.2*
Pusiano	5.2	70.0	25	3.6*	0.6	93.5*

(EU WFD, 2000). This ecological target is defined in terms of phytoplankton, macrophytes, fish and macrobenthos indicators (EU WFD, 2000; Copetti and Erba, 2024). As nutrient concentrations are key chemical parameters in determining the ecological conditions through the trophic state, depth-averaged *TP* concentrations at spring mixing are often selected as a relevant indicator of the ecological state by EU member states (Poikane et al., 2019a). In fact, for the lakes considered in this paper, lake-specific target *TP* concentrations at spring mixing have been identified by the managing authority Regione Lombardia in past water protection plans by increasing the hypothetical pristine concentrations calculated through the *MEI* index by 25 %. These *TP* concentrations have already been reached for some studied lakes, whereas for others uncertainties on present external nutrient loading do not allow determining if missing compliance is due to either too high external loads or to the slow inertial lake response. The present process-based models shed light on this issue.

2.2. Modelling framework

The QWET (QGIS Water Ecosystem Tool) 1D coupled ecological-hydrodynamic model (Nielsen et al., 2017; Nielsen et al., 2021; Schnedler-Meyer et al., 2022) was employed in this study. QWET combines the 1D hydrodynamic model GOTM (General Ocean Turbulence Model; Burchard and Bolding, 2001) with the zero-dimensional (0D) biogeochemical model PCLake (Janse and van Liere, 1995), through the coupling library FABM (Framework for Aquatic Biogeochemical Models; Bruggeman and Bolding, 2014). The model is distributed as an open-source QGIS (QGIS, 2024) plugin and can be coupled to the ecohydrological model SWAT (Soil & Water Assessment Tool; SWAT, 2024) for integrated lake-watershed assessments on hydrological and nutrient balance. GOTM simulates stratification and mixing phenomena across a user-defined number of layers in which the water column is discretised, employing various user-selectable k-ε formulations for turbulence closure. This differs from the energy budget approach employed by the 1D hydrodynamic models DYRESM (Imberger et al., 1978; Imberger and Patterson, 1981) and GLM (Hipsey et al., 2019). PCLake is instead a process-based 0D biogeochemical model, reproducing ecological cycles and interactions throughout the food web across lake water and sediments. Through PCLake, QWET can potentially reproduce internal loading evolution following nutrient loss from bottom sediments, as it solves the nutrient mass budget inside the sediment layer. This is not completely simulated (Dresti et al., 2023a) by the 1D coupled models DYRESM-CAEDYM (Hamilton and Schladow, 1997) and GLM-AED2 (Hipsey et al., 2013), which parameterise orthophosphate (*P-PO₄*) release by sediments as function of a calibrated maximum flux under reference conditions and of water temperature (T_w) and dissolved oxygen (*DO*) concentration.

The herein-adopted hydrodynamic setup implements lake hydrological balance through the default basic QWET approach. A fixed flow rate, equal to the mean outflowing discharge (Table 1), enters and leaves the lake at the surface. Evaporation is nevertheless included to properly close the heat balance at the surface, the lake elevation being kept constant at the prescribed elevation through a residual input. The present biogeochemical setup follows a minimal NPD (Nutrients – Phytoplankton – Detritus) scheme and includes oxygen, carbon, nitrogen and phosphorus cycles, with a single generic phytoplankton group. In this scheme, phytoplankton is fed on inorganic dissolved nutrients, the organic detritus resulting from phytoplankton mortality being recycled into inorganic nutrients by parameterised mineralisation processes. The grazing effects of the higher trophic levels (Zhang et al., 2022, 2023) on phytoplankton are lumped into phytoplankton mortality, closing the food chain with the minimum model complexity and calibration effort. Constant nutrient concentrations are fed to the model with the main inflow, obtained as ratios between the available mean annual load estimations (Table 2) and the aforementioned discharges (Table 1). This simplified modelling framework, applied identically to all considered lakes, was constrained by data availability across them.

Constant nutrient loads were adopted since: (1) characterisations of total external input load series are missing, except for Lake Como (Fenocchi et al., 2023), Lake Iseo (Valerio et al., 2022) and Lake Pusiano (Copetti et al., 2017), single mean annual values from theoretical estimates or limited experimental campaigns being available for the other basins; (2) this assumption is of interest for the managing authority Regione Lombardia, whose task is determining threshold external load values to meet *TP* concentration targets; (3) when performing future extrapolations in a climate change context, determining reliable discharge and concentration series, including seasonal distribution and interannual rainfall variability, would be problematic and arbitrary (Dresti et al., 2023a). Through the constant-load approximation, the only interannual variations reproduced in model results are hence due to meteorology, thus revealing their impact on biogeochemical phenomena in the simulation results.

For Lake Idro and Lake Iseo, chemical stratification was implemented into the models to properly reproduce the vertical hydrodynamics and the cascading biogeochemical processes. Salinity data were obtained from regressions between sporadic total dissolved ion characterisations and regularly available conductivity data in water samples. Time-invariant vertical salinity distributions averaged over the calibration period were entered into GOTM for these lakes, as simulating their seasonal and interannual variations would have required characterising: (1) salinity, discharge and insertion depth time series of lake inflows; (2) *$CaCO_3$* precipitation, which to the knowledge of the Authors has been only recently included for the first time into a process-based coupled ecological-hydrodynamic model by Many et al. (2024), however still neglecting its feedback on salinity; (3) for Lake Idro, the salinity input of the bottom thermal springs of calcium sulphate (*$CaSO_4$*) (Viaroli et al., 2018; Tartari et al., 2021). The geothermal heat associated to the bottom springs of Lake Idro was nevertheless implemented into its model, fixing in GOTM a constant bottom $T_w = 7.0\ ^\circ$C. This is necessary to reproduce the peculiar thermal structure of this lake, in which the bottom spring input, combined with chemical stability, results in higher water temperatures at the bottom than in the overlying monimolimnion (Tartari et al., 2021). Last, the hypolimnetic withdrawal active since 2008 on Lake Annone East was implemented in the model of this lake at the actual withdrawal depth, entering the daily withdrawn discharge series obtained from the plant manager.

Model calibration was performed with ParSAC (Bruggeman and Bolding, 2020), following the bottom-up approach in Andersen et al. (2020, 2022) and Chou et al. (2021). In this strategy, parameters with increasing mutual dependence are incrementally added to the calibration, to reduce the risk of converging to sub-optimal solutions. The physical, oxygen and carbon, nitrogen, phosphorus and phytoplankton parameters were hence sequentially calibrated. The list of calibration parameters in the water column and in the sediment layer was taken from Regev et al. (2023), leading to 23 physical parameters and 89 biogeochemical parameters. Feedback of phytoplankton-induced turbidity on light extinction was not activated as it was found not to significantly improve GOTM results, so that physical parameters (i.e. meteorological series adjustment, light transmission and turbulence model parameters) were frozen after the relative calibration phase, as opposed to the biogeochemical parameters. Two to five ParSAC optimisation runs with 5000 realisations each were performed for each calibration phase to iteratively identify the optimal range of each parameter manually, hence making the overall calibration procedure actually semi-automatic and to some degree still dependent on expert judgement.

2.3. Employed data

The herein-adopted QWET setup needs as input data: (1) the lake hypsometric curves expressing the depth-area relationship, to compute the volume of each model layer, obtained from available bathymetrical data; (2) the meteorological data series of air temperature, wind velocity, cloud cover, relative humidity and atmospheric pressure, to compute heat fluxes at the lake surface; (3) the inflowing discharges and concentrations of nitrates (*N-NO_3*), ammonium (*N-NH_4*), organic nitrogen (*ON*), orthophosphate (*P-PO_4*) and organic phosphorus (*OP*), which make up external loading; (4) the lake surveyed vertical profiles of T_w and *DO*, aforementioned nutrient and Chlorophyll-*a* (Chl-*a*; used as proxy of total phytoplankton biomass) concentrations, employed as initial conditions and as observations for model calibration.

Regarding meteorological data, publicly available daily series from

Table 2
Assumed mean annual nutrient loads and time periods of the simulations for the lakes considered in the study (* = Regione Lombardia, *unpublished data*; † = Fenocchi et al., 2023; ¶ = estimated through preliminary considerations and simulations; § = Decet and Salmaso (1997); || = Viaroli et al., 2018; # = Scibona et al., 2022).

Lake	Mean annual load [t a^{-1}]					Calibration period	Extension period
	TP	P-PO$_4$	TN	N-NO$_3$	N-NH$_4$		
Alserio	1.0*	0.8¶	38.0¶	26.6¶	7.6¶	2007–2021	2022–2051
Annone East	0.7*	0.6¶	21.0¶	14.7¶	4.2¶	2007–2021	2022–2051
Annone West	1.5*	1.2¶	17.4¶	12.2¶	3.5¶	2007–2021	2022–2051
Como	265.0*	143.0†	7871.0*	4922.0†	576.0†	2004–2018	2019–2048
Endine	1.7*	1.4¶	50.0¶	35.0¶	10.0¶	2009–2021	2022–2047
Garda	97.0*	44.0§	3880.0¶	3395.0§	136.0§	2010–2022	2023–2048
Idro	15.0||	7.5¶	850.0||	595.0¶	42.5¶	2009–2021	2022–2047
Iseo	126.0#	28.0#	2117.0#	1510.0#	104.0#	2009–2021	2022–2047
Pusiano	4.5¶	3.6¶	262.5¶	183.8¶	52.5¶	2007(2013)–2021	2022–2051

the regional environmental protection agencies ARPA Lombardia and ARPA Veneto (for Lake Garda only) were employed. Cloud cover series were computed from measured shortwave radiation data as in Fenocchi et al. (2017). The series of observed meteorological variables were tuned during the physical parameter calibration phase through linear regression models. This allowed obtaining robust model results for T_w even employing weather stations located in the major urban centres close to the lakes, for which longer and more complete series are available, instead of coastal stations. Through meteorological series tuning, in fact, phenomena such as the higher windiness of lake valleys or the air temperature mitigation by lake water masses can be mimicked. Furthermore, the same meteorological data series could be employed for neighbouring basins, adopting different lake-specific tunings.

Constant inflow discharges and nutrient concentrations (computed as ratios between loads and discharges) were obtained from the sources referenced in Tables 1 and 2. For smaller lakes, only *TP* loads are generally available, so that missing *P-PO*$_4$ and nitrogen loads were determined applying nutrient load partition proportions from similar basins for which these data are available, sometimes correcting the estimations by means of the lake concentrations obtained in preliminary simulations. Organic nutrient fractions were obtained from sampled total and dissolved inorganic ones as: *ON* = *TN* - *N-NO*$_3$ - *N-NH*$_4$ and *OP* = *TP* - *P-PO*$_4$.

Lake surveyed vertical profiles were obtained from ARPA Lombardia, which performs monitoring campaigns in application of the WFD, with monthly frequency for larger lakes and bimonthly to quarterly frequency for smaller ones, and for Lake Garda from the Fondazione Edmund Mach di San Michele all'Adige (FEM), which has been performing monthly campaigns synergistically with ARPA Veneto since 1990. For Chl-*a*, integrated samples of the photic layer are available for all lakes at each sampling date.

2.4. Performed simulations

Lake models were calibrated against the whole available series of vertical profiles (for Lake Garda, the calibration period constraint was given by the adopted meteorological series starting in 2010), leading to calibration periods spanning 13–15 years (Table 2). For Lake Pusiano, the calibration period had to be eventually reduced to the last 9 years (see Paragraph 3.1). Simulations were then extended to the mid-XXI century (Table 2), to evaluate the near-future evolution of the studied lakes. This was done by repeating twice the meteorological series observed in the calibration period, hence not considering climate-change effects, which are nevertheless mostly appreciable over longer time scales. Three input-load configurations were considered: (1) constant present phosphorus and nitrogen loads (reference scenario); (2) constant present loads up to 2023, −15 % linear variation in the 2024–2034 period, constant reduced loads afterwards; (3) same as (2), yet with a − 30 % linear variation in the 2024–2034 period.

Annual volume-weighted *TP* concentrations at spring mixing were extracted from the simulations to further evaluate model results against observations and assess near-future trends for this parameter of large interest for lake management. The considered observations were those from the field samplings closest to annual spring-mixing conditions, as inferred from T_w and *DO* profiles. In the simulations, the concentrations on the annual day of spring mixing were considered, this being evaluated as: (1) for holomictic basins (Lake Alserio, Lake Annone East, Lake Annone West, Lake Endine, Lake Pusiano), as the day in which maximum bottom *DO* concentrations are reached before the decrease due to direct stratification; (2) for oligomictic (Lake Como, Lake Garda) and meromictic (Lake Idro, Lake Iseo) basins, as the day in which maximum *DO* concentrations are obtained at a depth which is always reached with some margin by annual spring mixing (60 m for Lake Como, 100 m for Lake Garda, 20 m for Lake Idro, 40 m for Lake Iseo). Exact spring-mixing days were considered for model results rather than the closest sampling days to avoid inconsistencies with the near-future extension period. Volume-weighting of *TP* concentrations was performed for the observations considering the available samples at their sampling depths, whereas in the simulations volume integration was performed across all model layers. This was the only viable choice to avoid inconsistencies with the near-future extension period, since for many lakes monitored by ARPA Lombardia samples are not systematically taken at the same depths at each sampling date. For meromictic Lake Idro and Lake Iseo, *TP* concentrations at spring mixing were also evaluated in the mixolimnion (0–30 m for Lake Idro, 0–60 m for Lake Iseo), to separate the already impacted monimolimnion when evaluating near-future *TP* concentrations.

Among the nine modelled lakes, results will be displayed here for Lake Annone East, Lake Como and Lake Iseo. These pilot basins were chosen as they are deeply meaningful to the evaluation of the adopted modelling approach, being respectively: (1) a small shallow eutrophic lake, in which furthermore hypolimnetic withdrawal is active and was implemented into the model; (2) a very deep oligomictic mesotrophic lake with a multi-basin morphometry, which challenges the 1D schematisation (Guyennon et al., 2014; Copetti et al., 2020); (3) a deep, potentially hypertrophic lake with permanent chemical stratification.

3. Results and discussion

3.1. Reproduction of observations in the calibration period

The observed and modelled series of volume-weighted T_w, *DO* and *TP* in the epilimnion and hypolimnion and of Chl-*a* in the photic layer are shown for the pilot lakes in Figs. 2–4.

While the accuracy of the hydrodynamic model GOTM in reproducing epilimnetic temperatures (Figs. 2a, 3a and 4a) is comparable to that of the DYRESM and GLM 1D models, it results for all lakes in much more precise hypolimnetic T_w predictions than those generally obtainable with such models (Fenocchi et al., 2017; Bruce et al., 2018; Dresti et al., 2023a), both for shallow (Fig. 2b) and deep (Figs. 3b and 4b) basins. This also holds within the calibration period for chemically stratified Lake Idro and Lake Iseo, even though the assumed constant salinity profiles would affect long-term projections, given the expected increase in the salinity gradient. The first reason for the accurate hypolimnetic T_w reproduction obtained here is that, through the ParSAC calibration, a large set of physical parameters can be considered, tuning each of them much more precisely than possible through manual calibration. Furthermore, the k-ε turbulence closure employed by GOTM gives a more precise representation of hypolimnetic mixing than the energy budget approach of DYRESM and GLM, especially for the extremely low-turbulence environments of deep lakes (Figs. 3b and 4b).

The robust simulation of mixing leads to a solid overall reproduction of the seasonal hypolimnetic *DO* dynamics for shallow lakes (Figs. 2d), including summer anoxia and winter reoxygenation, and of the deep-hypolimnion dynamics of *DO* and nutrients for deep lakes (Figs. 3d, f, 4d and f). The latter are therefore proven to mainly depend on the implemented meteorological variability, which leads to a properly simulated interannual variation of mixing dynamics, and on long-term biochemical processes, such as nitrification, denitrification and mineralisation, whose parameters are calibrated for each lake. The overlooked yearly variable deep insertions of nutrients and oxygen by lake tributaries hence appear to have a significantly smaller influence on hypolimnetic dynamics, likely also due to the decreased intensity and frequency of such phenomena in the last 20 years due to climate change (Dresti et al., 2023b). For Lake Como, this also occurs as the lake deepest point near Argegno where samples are taken lies in a branch in which the hypolimnion is disconnected from the rest of the lake by an underwater ridge, hence separating it from the intrusions of the main inflows River Adda and River Mera (Copetti et al., 2020).

The constant load assumption reveals its weakness for nutrient variables in most of the water volume for shallow lakes (Fig. 2e and f) and in the epilimnion for deep lakes (Figs. 3e and 4e). For these layers, an

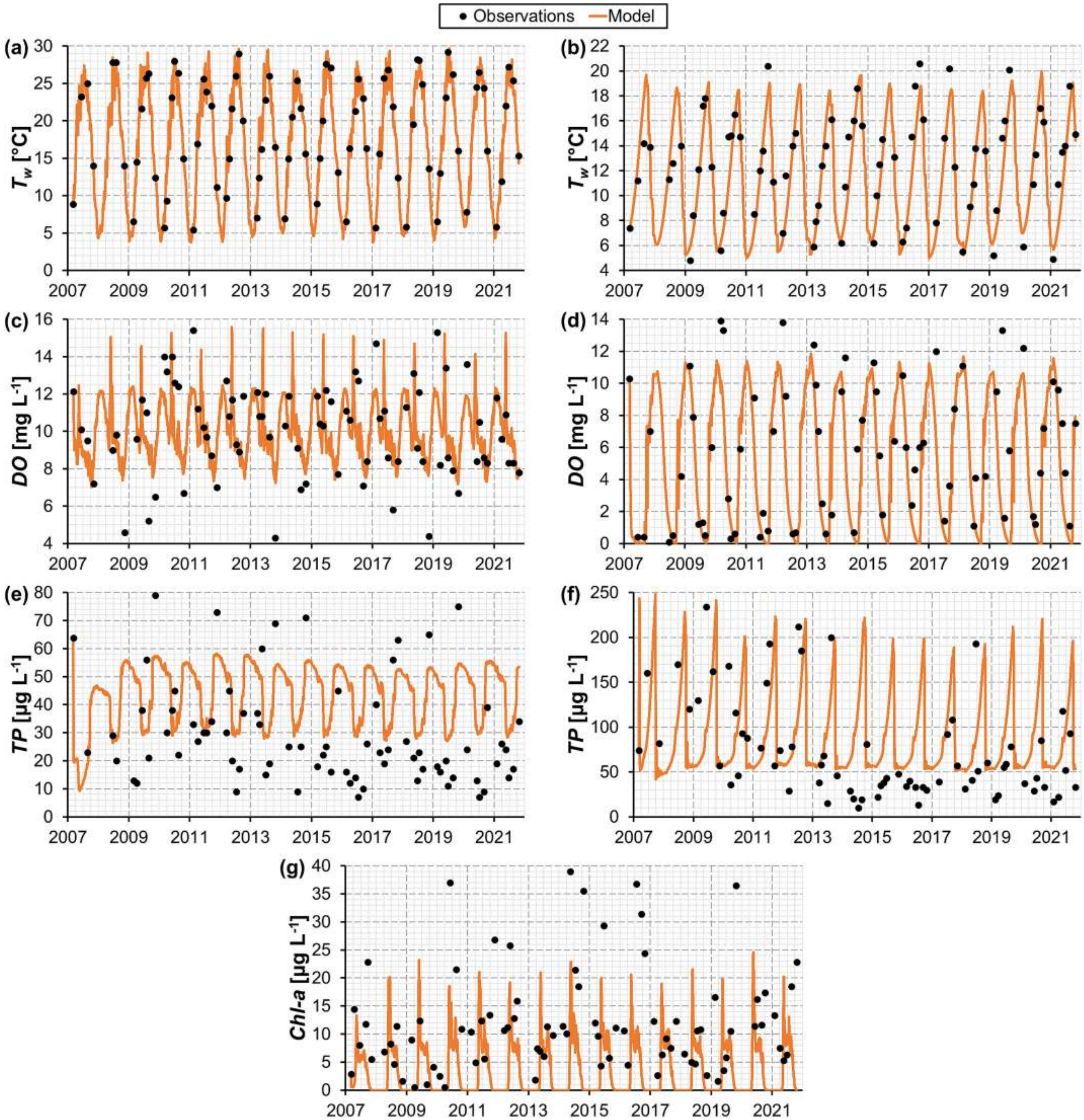

Fig. 2. Observed and modelled volume-weighted water temperatures (T_w), dissolved oxygen (*DO*) and total phosphorus (*TP*) concentrations in the 0–1 m epilimnion (a, c, e, respectively) and in the 8–11 m hypolimnion (b, d, f, respectively) and Chlorophyll-*a* (Chl-*a*) concentrations in the 0–6 m photic layer (g) for Lake Annone East.

averaged behaviour is in fact reproduced by the models, which does not represent the full observed variability. This is also due to the horizontal patchiness of nutrient concentrations over the mixed layer, so that the observations represent the local result at the sampling point, while the 1D model returns a horizontally averaged result (Gal et al., 2009; Fenocchi et al., 2019). The worst agreement with observations is obtained for all lakes for Chl-*a* in the photic layer (Figs. 2g, 3g and 4g). For Chl-*a*, a first additional error source compared to nutrients is the higher parameterisation level of biological variables compared to chemical ones in coupled models (Gal et al., 2009). Furthermore, the use of a single phytoplankton group is a radical approximation in the ecological chain, as the higher trophic levels and the constituent algal groups at finer taxonomic ranks, which are indeed characterised by specific ecological features and seasonality (Reynolds, 2006), are not considered. Nonetheless, the main aim of the implemented generic phytoplankton group in the present simplified model schematisation is to close the NPD cycle over an annual basis, these errors not being crucial for the evaluation of *TP* concentrations at spring mixing. Flaws in nutrient and especially phytoplankton biomass reproduction contribute to the errors in *DO* prediction in the epilimnion, for which a more

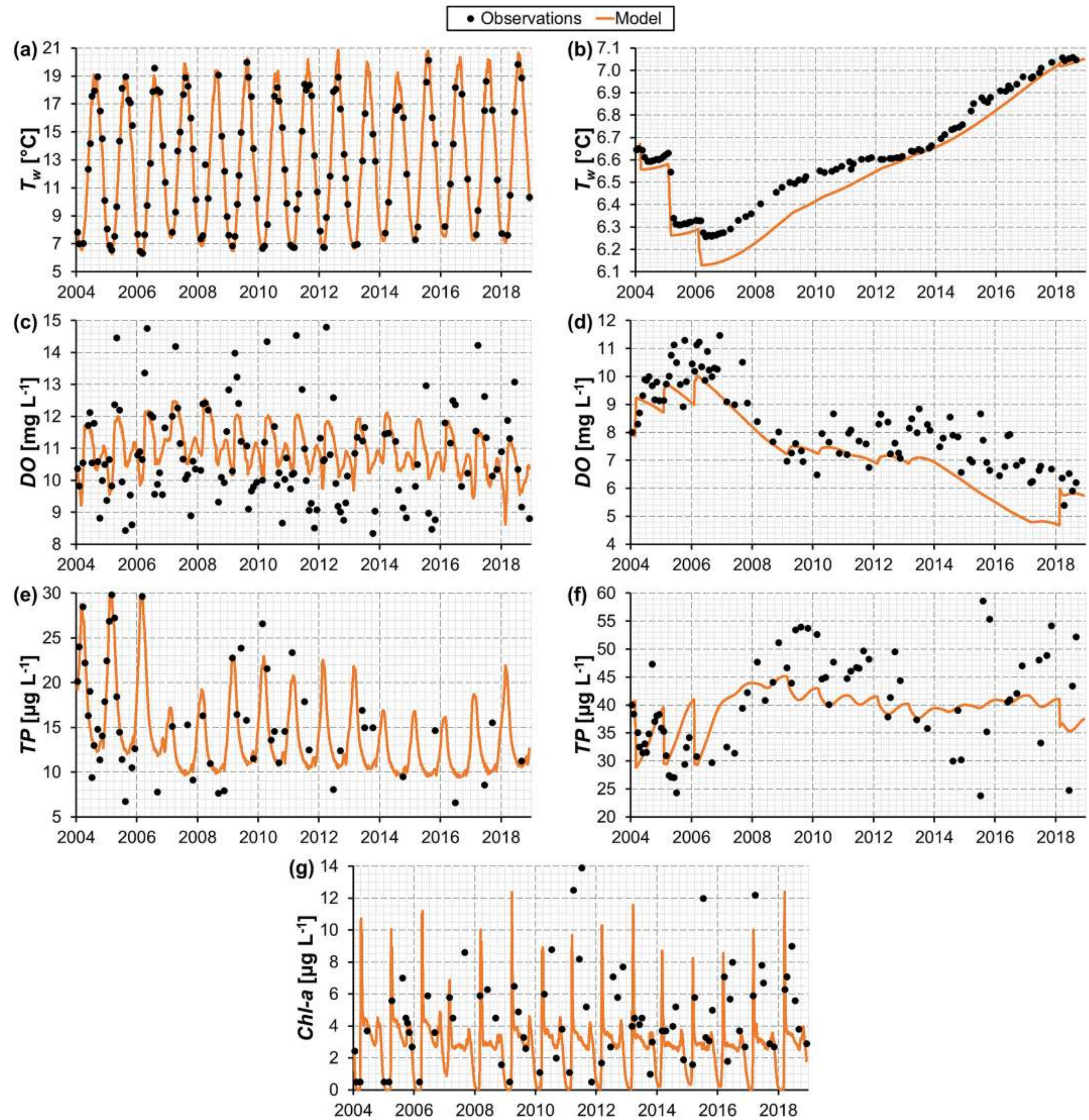

Fig. 3. Observed and modelled volume-weighted water temperatures (T_w), dissolved oxygen (*DO*) and total phosphorus (*TP*) concentrations in the 0–25 m epilimnion (a, c, e, respectively) and in the 200–426 m deep hypolimnion (b, d, f, respectively) and Chlorophyll-*a* (Chl-*a*) concentrations in the 0–20 m photic layer (g) for Lake Como.

smoothed behaviour than the observed one is reproduced, more evident for large lakes (Figs. 3c and 4c) than for small ones (Fig. 2c), likely due to the greater extent of horizontal patchiness over wider areas.

The constant load assumption cannot be employed when definite variations in input loads occur during the simulated period, exceeding the interannual oscillations due to rainfall variability. Among the modelled basins, this happened for Lake Pusiano, in which external loads from the watershed were significantly reduced from 2007 to 2012 circa as part of the PIROGA cooperative restoration project (Copetti et al., 2017). For this lake, the model allowed determining through preliminary simulations that an external mean annual *TP* load of 4.5 t a^{-1} (Table 2) best reproduces presently observed lake concentrations, lower than the available literature estimations obtained prior to the interventions on the watershed.

3.2. Simulation of volume-weighted TP concentrations at spring mixing

The observed and modelled series of annual volume-weighted *TP* concentrations at spring mixing in the calibration period and in the following near-future extension are shown in Fig. 5 for the pilot lakes,

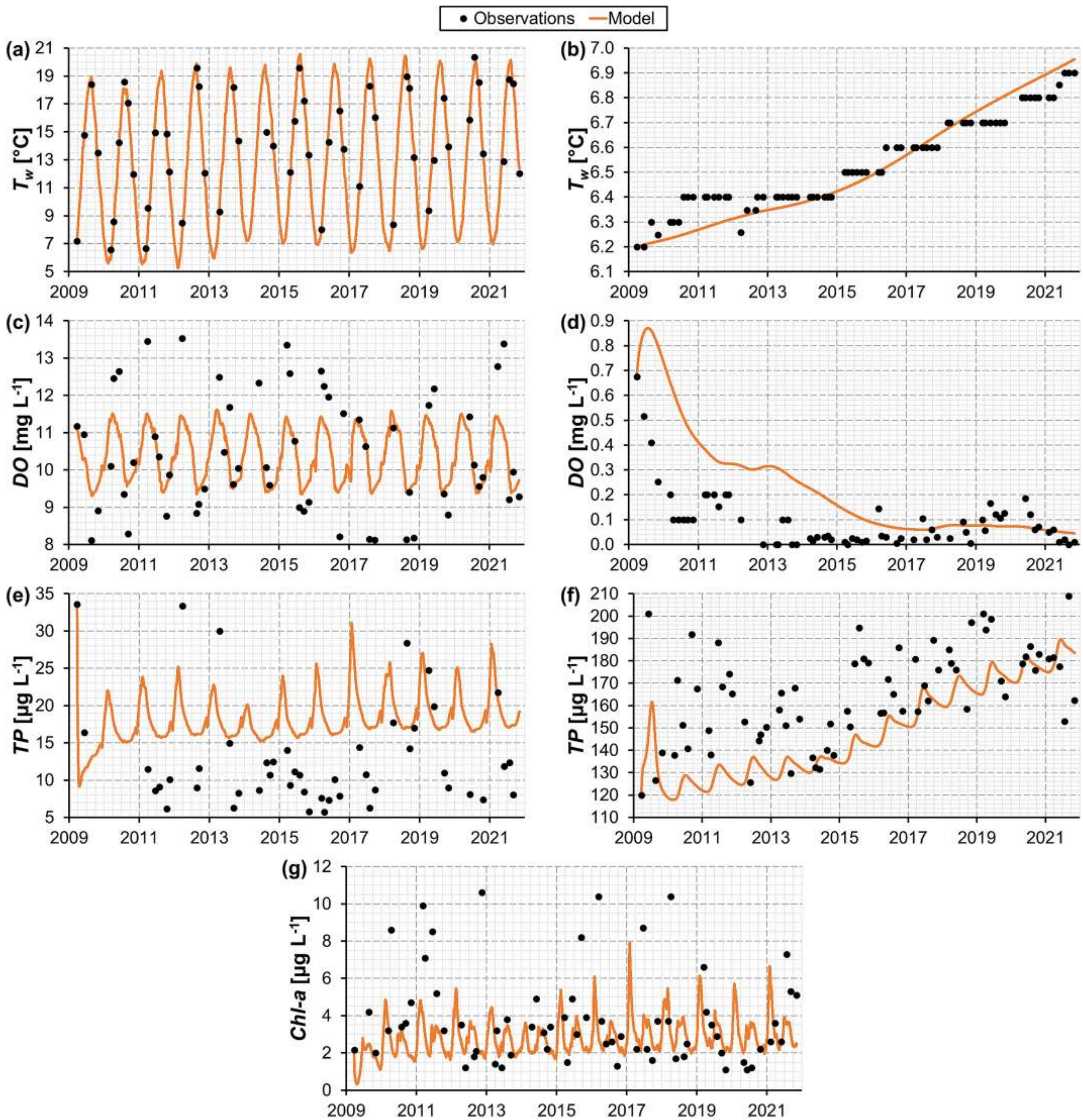

Fig. 4. Observed and modelled volume-weighted water temperatures (T_w; observations have 0.1 °C resolution), dissolved oxygen (*DO*) and total phosphorus (*TP*) concentrations in the 0–25 m epilimnion (a, c, e, respectively) and in the 200–252 m deep hypolimnion (b, d, f, respectively) and Chlorophyll-*a* (Chl-*a*) concentrations in the 0–20 m photic layer (g) for Lake Iseo.

over the entire water column and in the 0–60 m mixolimnion for Lake Iseo.

The implemented simplified modelling schematisation effectively reproduced the observed general trend of this derived variable for all the studied lakes. However, observed values display a higher interannual variability than simulated ones. This is due to: (1) the constant-load approximation; (2) the differences in the time of sampling from the annual day of spring mixing identified in the simulations; (3) the availability for many considered basins, including the displayed Lake Annone East (Fig. 5a) and Lake Iseo (Fig. 5c and d), of data not systematically sampled at the same depths, which influences the volume-weighted value calculations for the observations. For lakes in which there is a significant ongoing contribution by internal load (Lake Alserio, Lake Idro, Lake Iseo, Lake Pusiano), variations in the external input *TP* in the near-future extension period affect the volume-weighted concentrations at spring mixing less than linearly, due to the internal nutrient release from the sediments. For meromictic Lake Idro and Lake Iseo, the agreement between modelled and observed volume-weighted *TP* concentrations at spring mixing over the mixolimnion alone (Fig. 5d) is worse than over the entire water column (Fig. 5c). This is due

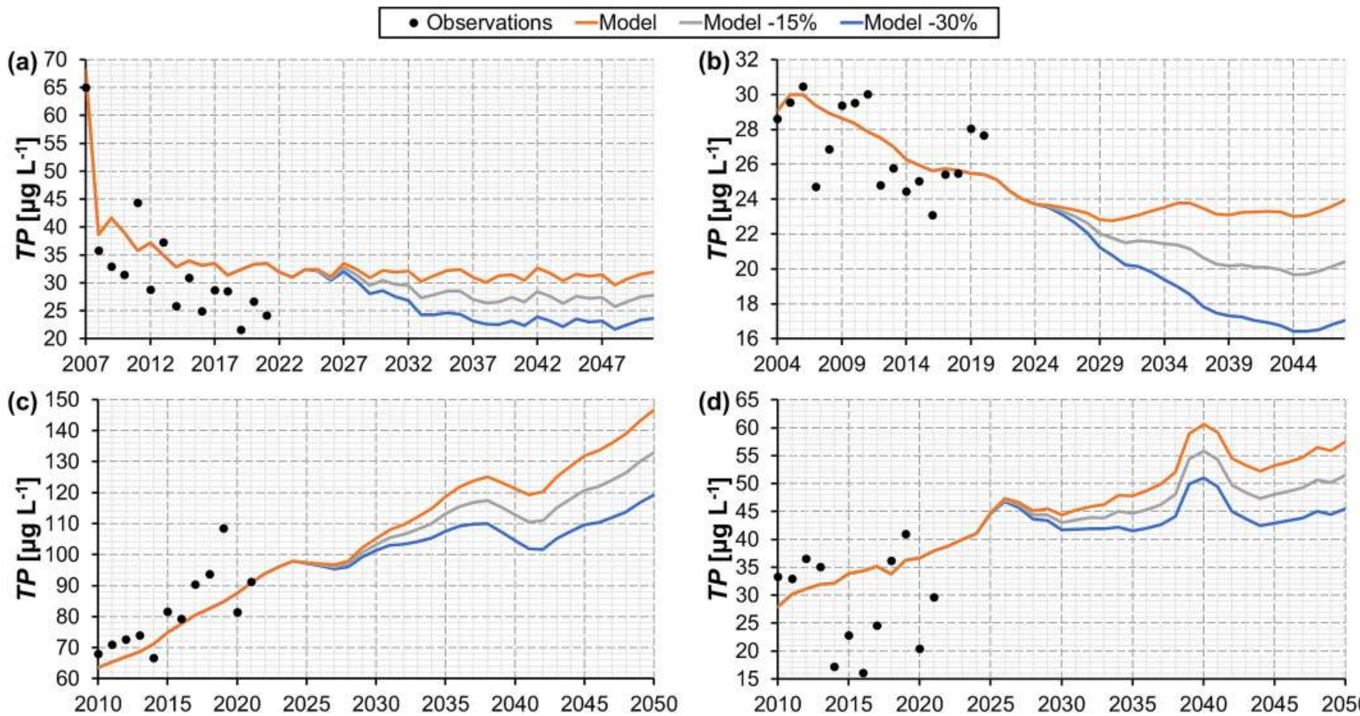

Fig. 5. Observed and modelled (constant nutrient loads and − 15 % and − 30 % external loading variations) volume-weighted total phosphorus (*TP*) concentrations at spring mixing over the water column for Lake Annone (a), Lake Como (b), Lake Iseo (c) and in the 0–60 m mixolimnion for Lake Iseo (d).

to model calibration minimising the global error across all depths, so that a bias may be present over a portion of them, hence also explaining the better agreement of the *TP* concentrations averaged over the whole depth in Figs. 5a – 5c compared to those over partial layers (Figs. 2e, f, 3e, f, 4e and f). Furthermore, the influence of interannual load variability and of horizontal patchiness is far more prominent in the mixolimnion than in deep separated waters.

Focusing on pilot Lake Annone East (Fig. 5a), the simulation reproduces a decrease in volume-weighted *TP* concentrations at spring mixing in the calibration period, even though at a slower rate than in the observations. Such decrease is due to the hypolimnetic withdrawal, active since 2008 and implemented into the model. The model correctly reproduces that, through the withdrawal, the high *TP* concentration of the first simulated year 2007 is never reached again. The lower simulated decrease rate than the observed one may be due to both interventions to reduce external loads having been performed and to the imperfect modelling of nutrient loss from bottom sediments, an accurate calibration of such phenomenon requiring periodic data of nutrient content in the sediments. This would be also proven by the higher-than-observed reproduced sediment release in the latter half of the calibration period (Fig. 2f). For Lake Annone East, both the simulated and the observed volume-weighted *TP* concentrations at spring mixing stabilise by the end of the calibration period, a prompt adaptation to external load reductions occurring in the relative near-future scenarios, given the low inertia reproduced for all considered small shallow lakes.

As regards displayed Lake Como (Fig. 5b), the model well reproduces the slow decrease in volume-weighted *TP* concentrations at spring mixing observed in the calibration period. External nutrient loading to Lake Como has been pretty much stable overall in the last 20 years due to the absence of interventions on the watershed (Fenocchi et al., 2023), the oscillations around the reduction trend in the observations revealing the interannual load variability due to rainfalls. This gradual decrease in *TP* concentrations is therefore to be ascribed to the significant reduction of external loading performed to oppose eutrophication in the 1980s and 1990s, its effects on Lake Como being prolonged by the inertia due to the large lake volume. In the reference near-future scenario with constant loads, stabilisation of *TP* concentrations is obtained by 2030. Instead, in the scenarios with external load reduction the decreasing trend extends after loading stabilisation in 2034, approaching equilibrium by the end of the simulated period. This further proves the relevant inertia of this large lake. Similar results were obtained for Lake Garda, which shares watershed intervention history and morphometrical traits with Lake Como.

Considering the whole water column of pilot Lake Iseo (Fig. 5c), the model properly simulates the observed rapid increase in volume-weighted *TP* concentrations at spring mixing in the calibration period. This upsurge is due to a hypolimnetic accumulation process triggered by the increase of both thermal and chemical stabilities with global warming, leading to hypolimnetic anoxia and meromixis (Ambrosetti and Barbanti, 2005; Rogora et al., 2018), supplemented by the still high external loading and the increasing internal loading (Scibona et al., 2022; Valerio et al., 2022). This trend in *TP* concentrations carries on unaltered up to the end of the simulated period in the reference near-future scenario, milder increase rates being reproduced in the external load reduction ones. The increase process would nevertheless be further intensified by future climate warming, not considered in the present simulations, causing an additional growth of water-column stability. The relevance of hypolimnetic accumulation and internal loading phenomena on the increase of volume-weighted *TP* concentrations at spring mixing is confirmed by considering the mixolimnion alone (Fig. 5d). Within such layer, in fact, the −30 % nutrient variation is reproduced to eventually stabilise *TP* concentrations, whereas an enduring rise occurs in the monimolimnion.

4. Conclusions

The adopted simplified process-based coupled ecological-hydrodynamic modelling approach, employing data from public monitoring and automatic calibration, allowed a deeper understanding of the present and near-future conditions of the studied lakes than allowable through Vollenweider-OECD-type approaches, offering more solid mechanistic bases and an insight into aspects other than volume-weighted total phosphorus (*TP*) concentrations at spring mixing. For such relevant variable, in any case, process-based coupled models allow computing the evolution in time with a much less parametrical approach than traditional steady-state models. All of this translates into valuable information for lake water ecosystem service protection planning. Further enhancement of the lake models implemented in this study would first require more precise quantifications of external loads from the watersheds, including their variability in time.

For an additional water quality improvement of the lakes considered in this study, supplementary external load reduction from their watersheds is central. In fact, except for the very large Lake Como and Lake Garda, for which residual effects of past major interventions against eutrophication are still ongoing due to their relevant inertia, the other lakes have reached a virtual equilibrium, which could yet be compromised by climate warming, not included in the present simulations, causing phenomena such as unexpected phytoplankton blooms and increased nutrient release from sediments. The effects of climate change are already critical for Lake Iseo, for which extensive external load reduction is compelling. Shallow lakes have largely disposed their internal load due to past widespread nutrient pollution, so that they seem to now react faster than in the past. Again, accurate quantification of present external loading levels is crucial, being the first step towards the identification of the further nutrient abatement potential in the watershed. This would lead to a more informed planning of interventions aimed at lake ecosystem service protection, being directed to either reducing nutrient release through policies on human activities or to decreasing the fraction which is delivered to lakes through sewage collection and treatment improvements.

CRediT authorship contribution statement

Andrea Fenocchi: Writing – review & editing, Writing – original draft, Visualization, Supervision, Software, Resources, Methodology, Investigation, Formal analysis, Conceptualization. **Nicolò Pella:** Writing – review & editing, Software, Investigation, Formal analysis. **Diego Copetti:** Writing – review & editing, Writing – original draft, Investigation, Funding acquisition, Conceptualization. **Fabio Buzzi:** Writing – review & editing, Validation, Resources, Data curation. **Daniele Magni:** Writing – review & editing, Funding acquisition, Data curation. **Nico Salmaso:** Writing – review & editing, Validation, Data curation. **Claudia Dresti:** Writing – review & editing, Writing – original draft, Project administration, Investigation, Funding acquisition, Conceptualization.

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.

Acknowledgments

Support by Regione Lombardia inside the “Modellazione idrodinamico-ecologica dei laghi lombardi a supporto dell’aggiornamento del Piano di Tutela delle Acque” research agreement with the University of Pavia and CNR-IRSA is deeply acknowledged. This study was developed within the operative and data sharing agreement of Fondazione Edmund Mach di San Michele all’Adige (FEM) with CNR-IRSA and the University of Pavia for Lake Garda research. The logistic support provided by ARPA Veneto for the limnological survey of Lake Garda is highly appreciated. Dr. M. Rogora of CNR-IRSA is gratefully acknowledged for providing us with the salinity data of Lake Idro and Lake Iseo. We thank all the ARPA Lombardia and FEM staff who collected, analysed and archived field samples over the years.

Data availability

The datasets analysed and generated in this study are available from the Corresponding Author upon reasonable request.

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