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Modelling climate change impacts on nutrients and primary production in coastal waters

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Abstract

The anthropogenic increase of atmospheric greenhouse gases (GHG) is causing changes in Earth's climate. Coastal waterbodies such as estuaries, bays and lagoons be among those most affected by the ongoing changes on climate. Because of their position at the land-sea interface, they are subjected to the combined changes in the physico-chemical processes of atmosphere, upstream land and coastal waters. Particularly, climate change is expected to alter phytoplankton communities by changing their climate and environmental drivers, thus exacerbating the symptoms of eutrophication events, such as hypoxia, harmful algal blooms (HAB) and loss of habitat. A better understanding of the links between climate-related drivers and phytoplankton is therefore necessary for predicting climate change impacts on aquatic ecosystems. Here we present the case study of the Zero river basin in Italy, one of the main contributors of freshwater and nutrients loadings to the salt-marsh Palude di Cona, a waterbody belonging to the lagoon of Venice. To predict the effects of climate change on nutrient loadings and their effects on the phytoplankton community of the receiving waterbody, we applied an integrated modelling approach made of an ensemble of GCM-RCM climate projections, the hydrological model SWAT and the ecological model AQUATOX. Climate scenarios point out an increase of precipitations in the winter period and a decrease in the summer months, while temperature shows a significant increase over the whole year. Water discharge and nutrient load simulated by SWAT show a tendency to increase in the winter period, and a reduction during the summer months. AQUATOX predicted changes in the concentration of nutrients in the salt-marsh Palude di Cona, and variations in the biomass and species of the phytoplankton community.

Keywords: climate change, nutrient pollution, phytoplankton, eutrophication, integrated modelling approach

1. Introduction

Anthropogenic emissions of greenhouse gases are the main cause of the current Earth's energy imbalance, which is causing the warming of the climate system (von Schuckmann *et al.*, 2016). Global mean temperatures are expected to rise by 0.3 to 4.8 °C by the end of the 21^{st} century (IPCC, 2013), and the water cycle to alter as a result of changes in global atmospheric moisture (Levang and Schmitt, 2015).

Coastal ecosystems, together with the ecological and socio-economic services they provide, could be among those most affected by the ongoing changes on climate (Harley *et al.*, 2006; IPCC, 2014). Coastal waterbodies such as estuaries, bays and lagoons, are transitional systems located at the interface between land and sea and will be subjected to the combined changes taking place in the atmosphere, oceans, and land surface (Raimonet and Cloern, 2016).

Climate change is projected to have substantial effects on the abundance and composition of coastal phytoplankton communities (Winder and Sommer, 2012; Harding *et al.*, 2015), and to increase the frequency and abundance of eutrophication events and related symptoms such as hypoxia, harmful algal blooms (HAB), unsightly scums and loss of habitat (Lloret, Marín and Marín-Guirao, 2008; O'Neil *et al.*, 2012; Paerl and Paul, 2012; Xia *et al.*, 2016). Climate change is expected to have important consequences on phytoplankton because of the potential effects on those factors governing its dynamics. Examples of factors include water temperature, precipitation, wind, solar radiation, water acidity, tides and nutrient availability (Cloern, 1996). Warmer water temperatures affect the physiology and ecology of phytoplankton (Lassen *et al.*, 2010; Hunter-Cevera *et al.*, 2016; Weisse, Gröschl and Bergkemper, 2016), and the thermal stratification and vertical mixing of waters (Cloern et al., 2005). Variations in precipitation patterns and intensity can alter the delivery of freshwaters and loads of nutrients and sediment (Hagy *et al.*, 2004; Moss *et al.*, 2011), with consequences on salinity and nutrient regimes, and water retention times (Dimberg and Bryhn, 2014). Increased CO_2 concentrations will lower the pH of coastal waters, with consequences on phytoplankton calcification and community structure (Beaugrand et al., 2012). Moreover, climate change will increasingly intensify the overwhelming disturbances that already affect coastal areas (Rabalais et al., 2009). Moreover, nutrient loads are expected to increase in the coming decades due to population growth, increased use of inorganic fertilizers and manure, increased fossil fuel burning (associated emission of NO_x), and expansion of sea-based activities such as aquaculture (Burkholder *et al.*, 2007; Bouwman, Beusen and Billen, 2009; Verdegem, 2013). Consequently, the multiple and concomitant effect generated by will influence and be influenced by the effects of climate change, leading to large and detrimental changes on most coastal ecosystems (Rabalais et al., 2009).

Given the large number of factors and complex interactions regulating phytoplankton dynamics, predicting its responses to climate change can be a difficult task. Phytoplankton is at the base of every aquatic food web, and changes in its dynamics and composition may have relevant repercussions on the higher trophic level of ecosystems as well (Hernandez-Farinas *et al.*, 2014; Schloss *et al.*, 2014).

The integration of climate scenarios and mechanistic environmental models can become a valuable tool for the investigation and prediction of phytoplankton dynamics under climate change conditions. In the last decades, the adoption of mechanistic models have become a popular tool for assessing the impacts of climate change on hydrologic and abiotic components of aquatic systems (Vohland et al., 2014). The majority of studies have analysed the impacts of climate change on single environmental aspects such as watershed hydrology and water availability (Leta *et al.*, 2016; Amin *et al.*, 2017; Trinh *et al.*, 2017), loadings of nutrients (Huttunen *et al.*, 2015) and sediments (Bussi et al. 2016; Samaras & Koutitas 2014), and water quality (Wilby *et al.*, 2006). The number of studies that attempted to integrate climate scenarios and process-based models for assessing the impacts of climate change on aquatic ecosystems is growing (Mooij *et al.*, 2007; Taner, Carleton and Wellman, 2011; Glibert *et al.*, 2014; Guse *et al.*, 2015; Trolle *et al.*, 2015) but still limited. Therefore, the further development of integrated modeling approaches can result in a useful contribution to increase the availability of management tools for ecological conservation and adaptation policies of sensitive coastal areas.

This work presents an integrated modelling approach for assessing potential long-term effects of climate change on the productivity and community structure of coastal phytoplankton at a local scale. The approach can investigate the consecutive impacts of climate change along the land-water continuum, from climate-related impacts on stream flow and nutrient loadings, to direct influence of temperature changes on coastal waters. This approach consists of climate scenarios and tools able to provide climate data suitable for impact assessment studies, and two separate environmental models used to depict the physico-chemical and biological characteristics of a watershed and of the receiving waters of a coastal environment. This study is a further attempt to integrate climate scenarios and tools with environmental models for assessing the responses of aquatic ecosystems to climate change. The main objective of the study is to illustrate the modelling approach, demonstrate its applicability through a local case study, and to discuss potential, limitations and areas of improvement.

2. Material and methods

2.1. Selected tools

The developed approach integrates the climate tool CLIME (Villani *et al.*, 2015) for the treatment of climate projections, the hydrological model SWAT (Arnold *et al.*, 1998) and the ecologic model AQUATOX (Park, Clough and Wellman, 2008). The single components, listed in Table 1, are presented in this Section, and their integration and parameterization for the local case study is then described in Section 2.3.

Table 1 – Description of models and tools. The columns Input and Output focus on the connection between the components and include selected climate-related input and outputs of a model.

Component	Model/Tool name	Type of model/tool	Input	Output
Climate	CLIME (Villani *et al.*, 2015)	GIS Tool for climate features	Daily time series of climate variables (Precipitation, P, Temperature, T) from GCM/RCM scenarios	Bias-corrected daily time series of climate variables (P, T)
River basin	SWAT (Arnold et al. 1998)	Eco-hydrological model	Bias-corrected daily time series of climate variables;	Monthly time series of water discharge and nutrient loads
Coastal waters	AQUATOX (Park et al. 2008)	Ecologic model for aquatic environments	Monthly time series of water discharge, nutrient loadings, water temperature	Monthly time series of phytoplankton abundance (chlorphyll-a) and composition

2.1.1. The GIS tool CLIME

In this study, GCM/RCM-based climate scenarios for temperature and precipitation were adopted to simulate future climate conditions. A vast number of climate scenarios obtained from GCM/RCM nested simulations is available to researchers for impact assessment studies (Wilcke and Bärring, 2016). RCMs can increase the spatial resolution of GCMs but, in most cases, biases that prevent an appropriate reproduction of the observed climate conditions can persist (Muerth *et al.*, 2013). Thus, the application of a bias correction method is often recommended (Teutschbein and Seibert, 2012). CLIME is a GIS tool developed for climate data analysis and treatment. In this study, it was adopted to analyse and compare climate observations with climate scenarios, and to apply a bias-correction method for reducing the biases of future climate projections. For this study, the linear scaling method (Lenderink, Buishand and van Deursen, 2007) was selected.

2.1.2. The SWAT model

SWAT is a process-based eco-hydrologic model developed to investigate the impacts of land management practices and climate on water, sediment and agricultural chemical yields in catchment basins over long periods of time. It was selected to simulate the climate-related pressures originating at the watershed level. To simulate watershed processes, SWAT requires information about weather, soil properties, topography, vegetation, and land management practices. The hydrologic response units (HRUs) are the smallest computational units in SWAT, with unique land use, soil and slope, and provide an efficient way of representing the spatial heterogeneity of a watershed (Kalcic, Chaubey and Frankenberger, 2015). SWAT is widely used in climate change impact assessment studies (Cousino, Becker and Zmijewski, 2015; Kim *et al.*, 2016; Sellami *et al.*, 2016), and it was selected for the following reasons: (i) suitability of the model for representation of hydrologic and nutrient cycling processes over long periods of time (i.e. 100 years); (ii) possibility to couple SWAT with other environmental models (e.g. hydraulic, water quality, ecologic models); (iii) capacity of the model to extensively represent land management practices; (iv) possibility to apply to model even to ungauged watershed or with a low availability of observed data; and (v) strong technical support from the SWAT community.

2.1.3. The AQUATOX model

AQUATOX is an aquatic ecological risk assessment model used to evaluate direct and indirect impacts of different stressors such as water temperature, nutrients, sediments, and toxic chemicals. AQUATOX It is a mechanistic model that computes the most important chemical and biological processes at a daily time step of the simulation period within a volume of water. It can be applied to different aquatic environments such as rivers, lakes, ponds, reservoirs and estuaries. The model was selected to assess the effects of water temperature and nutrients loadings on the productivity and community structure of phytoplankton. Phytoplankton biomass in AQUATOX is modeled as a function of nutrient loadings, water retention time, photosynthesis, respiration, excretion, mortality, predation, sinking and sloughing. AQUATOX has been applied to simulation of nutrient-related phytoplankton dynamics (Carleton, Park and Clough, 2009) and climate change impacts on aquatic ecosystems (Taner, Carleton and Wellman, 2011) and it was selected for the ability to (i) assess impacts on the ecosystem from multiple stressors (i.e. water temperature, water discharge, nutrient loadings) for long intervals of time, (ii) select different species of phytoplankton (i.e. diatoms, dinoflagellates, and cyanobacteria), and (iii) implement the output of SWAT (i.e. water discharge, nutrient loadings).

2.2. *Study area*

The Zero river basin (ZRB) and the waters of the salt-marsh Palude di Cona (PdC) belong to the land-water continuum of the lagoon of Venice, Italy (Fig. 1).

2.2.1. *Zero river basin*

The ZRB has a surface area of 140 km^2 and an elevation range from 1 to 110 m. Agriculture is the dominant land use in the Zero catchment (72%) with corn, soy and wheat as dominant crops (ARPAV, 2009). Urban and industrial areas (24%), and semi-natural and forested areas (4 %) cover the remaining surface. The north-western part is characterised by a significant presence of livestock farms, with a density of 5 to 10 farms per km^2 (ARPAV, 2001). Agricultural activities provide the greatest contribution of chemical inputs, specifically, synthetic fertilizers and organic fertilizers in the form of manure and urea. The area features a Mediterranean climate with unique characteristics typical of more Continental climates (Guerzoni and Tagliapietra, 2006). Annual average temperature is 14 °C, with January and December being the coldest months (around 4° C), and July and August the warmest (around 25° C). The average annual rainfall is 1000 mm, with peaks in spring and autumns and minimums in winter and summer periods. Different external factors influence the hydrology and nutrient loads of the Zero river (Essenfelder, Giove and Giupponi, 2016). In particular, the numerous hydraulic infrastructures that regulate the water flow necessary to satisfy the different water needs in the area and the groundwater contributions coming from the external aquifer of the Venetian high plains (Servizio Acque Interne, 2008).

2.2.2. *Palude di Cona*

PdC is a shallow water body surrounded by salt-marshes and located in the upper-north basin of the lagoon of Venice. It is 4 km long, 0.9 km to 1.7 km wide, with a mean depth of about 80 cm. It is surrounded and crisscrossed by navigation channels affecting its hydrology. PdC, as any area of the Venice lagoon, is influenced by the effects of the tide, which exposes the bottom surface of the area during low tide events. Temperatures are, on average, 0.5 – 1°C warmer than those of the Zero river basin and annual precipitations are, on average, 200 to 300 mm lower. Solar radiation, important factor influencing photosynthetic processes in primary producers, reaches peaks of 25 MJ/m^2 in the summer months and lowest of 5 MJ/m^2 in the winter time.

The trophic state of a transitional environment such as PdC is the result of multiple variables such as the loadings and concentrations of nutrients, bathymetry, water retention time (water exchanges between sea and lagoon), climate conditions and biological processes (Cloern, 2001). Considering seasonal variability, the trophic state follows the classic cycle of an aquatic ecosystem in a temperate climate (Facca, 2011). In the winter period, primary production is low and the dynamics of nutrients, which are present in higher concentrations, are mainly influenced by loading and transport phenomena. In the spring time, solar radiation triggers the first phytoplankton blooms, which can be further stimulated or inhibited by the availability or lack of nutrients. Nutrient concentrations show minimum values in the summer period, when phytoplanktonic blooms reach their peak. Phytoplankton composition in the Lagoon of Venice is dominated by diatoms and flagellates (Facca, Sfriso and Ghetti, 2004). In shallow areas on the landward side of the Lagoon such as PdC, the water temperature in winter often get close to freezing point, and phytoplankton biomass is particularly scarce. In contrast, in summer phytoplankton thermophile species find the most favorable environmental conditions to their metabolism, with temperatures of between 25 and 30 °C, triggering exponential growth (Guerzoni and Tagliapietra, 2006).

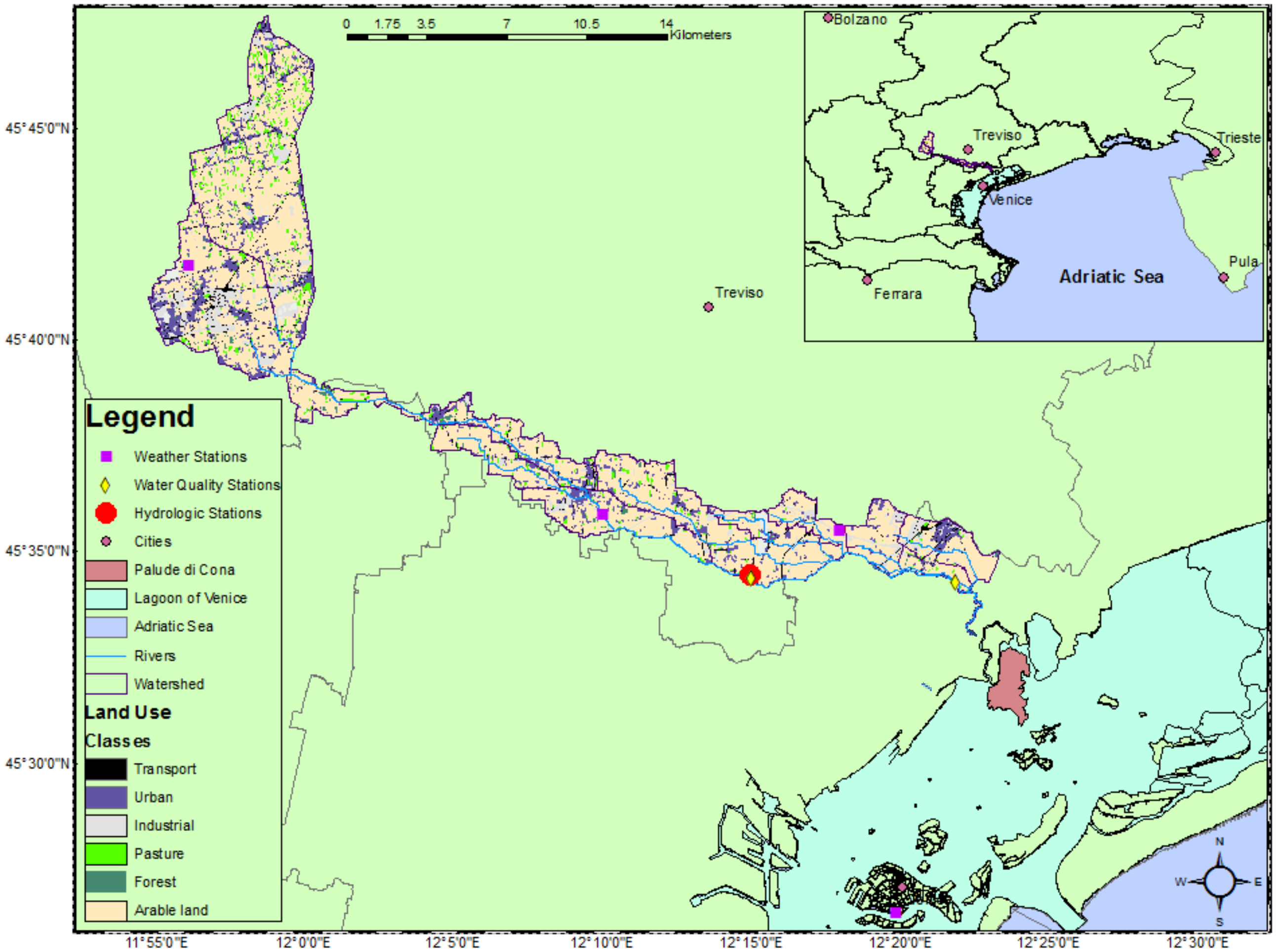


Fig. 1 – Land use in the catchment area of the Zero river, including available hydrologic and weather stations.

2.3. Integration and model parameterization

The integrated modelling approach developed in this study incorporated climate scenarios, a GIS climate tool, and watershed and ecosystem models, as shown in Fig. 2. In this section, the integration of tools and model, and parameterization of models is described in detail.

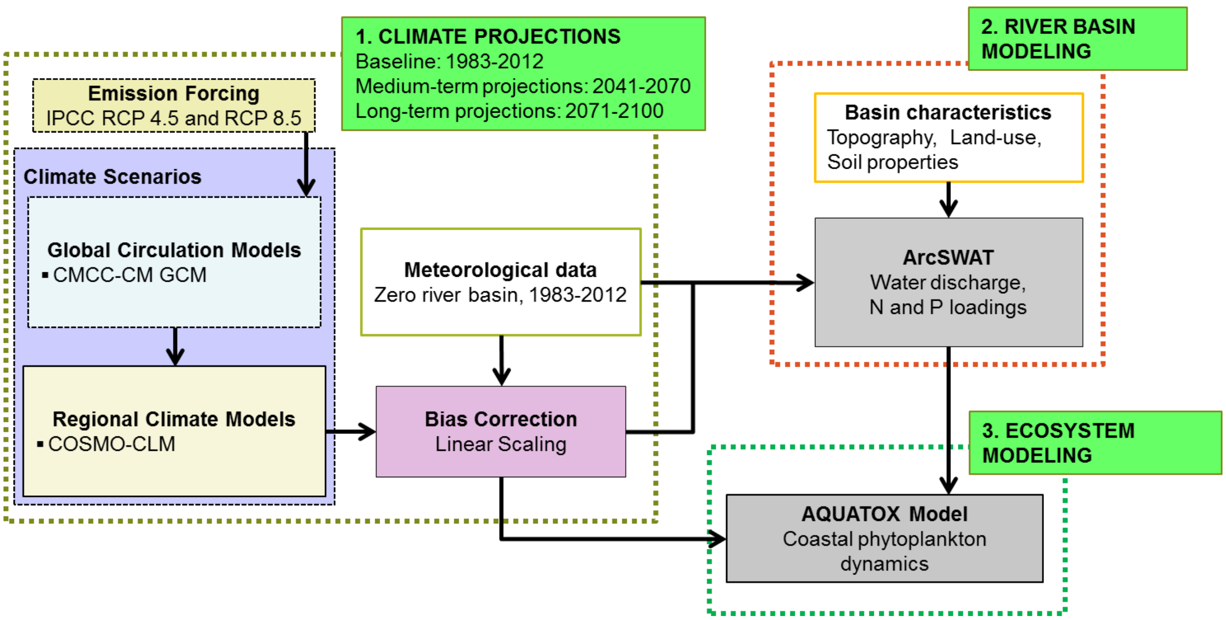


Fig. 2 – The Integrated modelling approach applied in the study.

2.3.1. Climate projections

In this study, only changes in precipitation and temperature were considered. The projected changes in precipitation and temperature were obtained from the climate scenarios generated by the coupling of the GCM CMCC-CM (Scoccimarro et al. 2011) with the RCM COSMO-CLM (Cattaneo et al. 2012) under the RCP4.5 and RCP8.5 concentration scenarios. The selected scenarios have a horizontal resolution of 0.0715° (8 km) and were used to compare the 30-year control period 1983-2012 with future climate simulations representing the mid-term period 2041-2070 and the long-term period 2071-2100. The GIS tool CLIME compared the spatial grid of the observed values (i.e. weather stations) with the spatial grid of the scenario for the control period, and applied the linear-scaling bias-correction method. The obtained delta factors for each month were then applied to the future periods, providing bias-corrected daily values of temperature and precipitation used as input for SWAT and AQUATOX.

2.3.2. River basin modeling of the ZRB

The hydrology and water quality conditions of the Zero river were modeled with SWAT using the following data: daily time series of meteorological data (i.e. precipitation, temperature, wind, solar radiation, relative humidity) from 3 weather stations (i.e. Castelfranco Veneto, Zero Branco, Mogliano; Fig. 1) for the period 2004-2012; 5x5m digital elevation model (Regione Veneto, 2013); a 100x100m land-use map (ARPAV, 2009); 500x500m soil map with soil geomorphological and textural characteristics (ARPAV, 2001); agricultural management practices for the selected cultivations (corn, soy, winter wheat); daily time series of streamflow from one gauging station for the period 2004-2012; daily time series of inorganic nitrogen concentrations (i.e. nitrate and ammonia) for the period 2007-2012. The lack of continuous, high frequency time series of phosphorus concentration values did not allow the calibration and validation of phosphorous loads. However, even though it can be seen as a potential source of uncertainty, the comparison of modeled values with mean annual data for the year 1999 from a previous study (Collavini *et al.*, 2005) allowed to provide a qualitative assessment of the annual mean and seasonal dynamics.

Due to the lack of continuous data, the influence of groundwater recharge of the bordering watersheds was modelled with an additional constant contribution. For the same reason, the influence of point-source pollution (WWTP, Industrial discharges) on nutrient loads was simulated with an additional constant contribution. Values were obtained from previous studies (Salvetti *et al.*, 2008) and ARPAV and Regione Veneto (2007, 2009). A detailed description of model parameterization is provided in the supplementary material A-I.

The SWAT model of the ZRB was run for the period 2004-2012, including a warm-up period of three years. The calibration period was set from 2007 to 2009 and the validation period from 2010 to 2012. The SWAT model outputs relative to water discharge (Q, m^3/s), Nitrogen from nitrate ($N\text{-}NO_3^-$, tons) and ammonium ($N\text{-}NH_4^+$, tons), and phosphorus from orthophosphate ($P\text{-}PO_4^{3-}$, tons) loads were provided as monthly values and were used as input for the AQUATOX model.

2.3.3. Ecosystem modeling of Palude di Cona

A model application with AQUATOX was used to represent phytoplankton dynamics and composition in PdC. The area has been simulated as an enclosed waterbody solely influenced by the freshwater discharge coming from the Zero river basin. In order to avoid introducing further complexity, the effect of tide has been neglected. This should be considered as a potential source of uncertainty. The following parameterization data were used: the modeled area is about 4 km long (L), 1.3 km wide (W), with a mean and maximum depth respectively of 0.8m (H) and 3.2m (H_{max}). Accordingly, the volume (V) of the system was set constant at 4.16 x 10^6 m^3 (V = L x W x H); Monthly means of water discharge (m^3 s^{-1}) and nutrient loads (kg $month^{-1}$) were obtained from the SWAT simulation of the Zero river basin; monthly values of water temperature were obtained from the monitoring network SAMANET (Ferrari, Badetti and Ciavatta, 2004) for the years 2007-2011; the annual average and range of wind speed (m s^{-1}) and light intensity at the surface (Ly day^{-1}) were determined through meteorological data for the period 2007-2011; time-average constant values were assume for pH (7.9); chemical parameters such as inorganic nitrogen and phosphorus concentrations (mg L^{-1}), Dissolved oxygen (DO, mg L^{-1}) and salinity were obtained from MELa3 monitoring campaign (MAV and CVN, 2002) and SAMANET monitoring network (Ferrari, Badetti and Ciavatta, 2004); total suspended solids (TSS, mg L^{-1}) were set equal to 8 mg L^{-1}. This value is based on average values of calm conditions (without the effect of wind and waves) in the lagoon of Venice, found in Thetis (2006).

The modeled ecosystem is composed of detritus, phytoplankton, and zooplankton. Detritus is subdivided into suspended particulate refractory detritus, suspended particulate labile detritus, sediment refractory detritus, sediment labile detritus and dissolved detritus. As no site-specific information was available, no parameterization was performed. Nine phytoplankton compartments, 7 Diatoms (D) and 1 Cyanobacteria (CB), were added to represent the possible evolutions of phytoplankton biomass and composition in present and future conditions (Table 2). In this study, the default organisms in the AQUATOX library (release 3.1) having the most similar characteristics to the species of the lagoon of Venice (Facca, Sfriso and Ghetti, 2004) were selected. *Navicula* and *Cyclotella* spp. are species commonly found in the waters of the lagoon of Venice. *Fragilaria* sp. was selected as a species that can be found in transitional water ecosystems. A species of Cyanobacteria was implemented to observe if future environmental conditions would favor blooms of cyanobacteria. The species *Microcystis* sp. was selected from the AQUATOX database as a species that is fairly salt-tolerant and able to produce microcystin toxins that can be stable and persistent in transitional ecosystems (Gibble and Kudela, 2014).

Table 2 – Phytoplankton compartments added to the system.

N.	Species	Optimal T (°C)	N Half-sat K	P Half-sat K
D1	*Navicula ssp.*	15	0.01	0.002
D2	*Cyclotella nana*	20	0.011	0.017
D3	*Cyclotella nana* (High nutrient waters)	20	0.117	0.055
D4	*Fragilaria spp.* (low nutrient waters)	26	0.0154	0.001
D5	*Cyclotella nana* (warm waters)	25	0.011	0.017
D6	Cyclotella *nana* (extremely warm waters)	30	0.011	0.017
D7	*Cyclotella spp.* (high nutrient and warm waters)	25	0.117	0.055
D8	*Fragilaria spp.* (high nutrient and cold waters)	8	0.117	0.05
CB1	*Microcystis spp*.	30	0.4	0.03

A more detailed description of phytoplankton species is provided in the supplementary material A-II. The AQUATOX model was run for the period 2005-2011, including a warm-up period of two years. Model performance was evaluated by comparing modeled results to data collected by the real-time water quality monitoring system SAMANET (Ferrari, Badetti and Ciavatta, 2004) for the time period 2007-2011. The relative bias (rB) versus variance's test (F) was applied to compare simulation results with SAMANET's data, for a sub-set of biotic and abiotic variables.

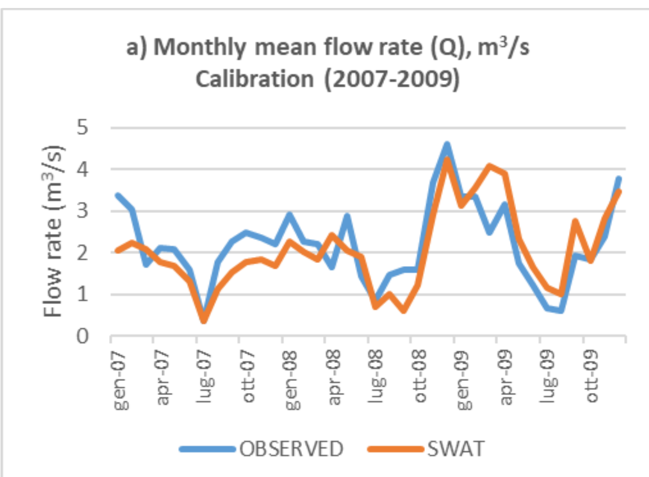
2.3.4. Modelling Assumption and caveats in future scenarios

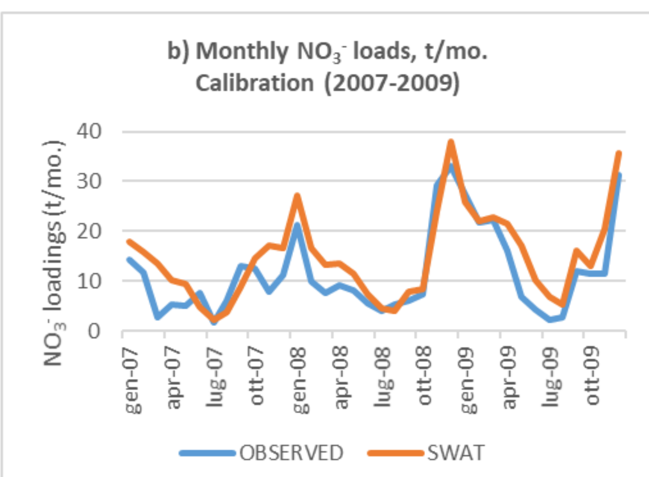
To simulate future physico-chemical and ecologic conditions of the area of study, a number of assumptions and caveats were taken. First, the remaining meteorological parameters (i.e. wind speed, solar radiation, relative humidity) were kept constant. Statistical parameters of monitored data for the calibration and validation periods (2007-2012) were used in the SWAT Weather Generator to generate daily data in according with the temperature and precipitation. In AQUATOX, computed mean and range of wind speed and solar radiation were kept constant for both mid-term and long-term periods. Second, land-use, agricultural practices and other anthropogenic emissions (i.e. WWTP, industrial discharges) in the Zero river basin remained unchanged. Third, the effects of sea level rise and human infrastructures such as MOSE project at the inlets of the lagoon of Venice have been neglected. Finally, given the absence of high-resolution water temperature projections for the lagoon of Venice for RCP4.5 and RCP8.5, projected water temperature was computed by using a linear regression between monitored air temperature and water temperature (R^2=0.99).

3. Results

Calibration of the SWAT Model

Monthly calibration was possible for water discharge, nitrate and ammonium loads. Calibration for a monthly time step for the 2007-2009 period produced “satisfactory” results for flow rate (NSE=0.58, R2=0.63), nitrate (NSE=0.60, R2=0.80) and ammonium (NSE=0.51, R2=0.59).





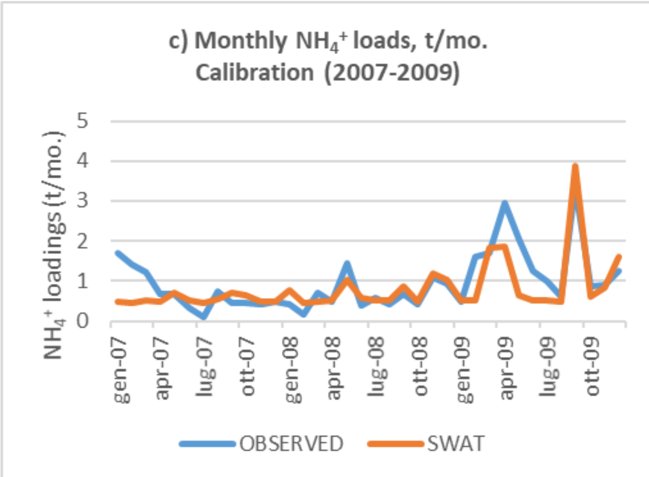
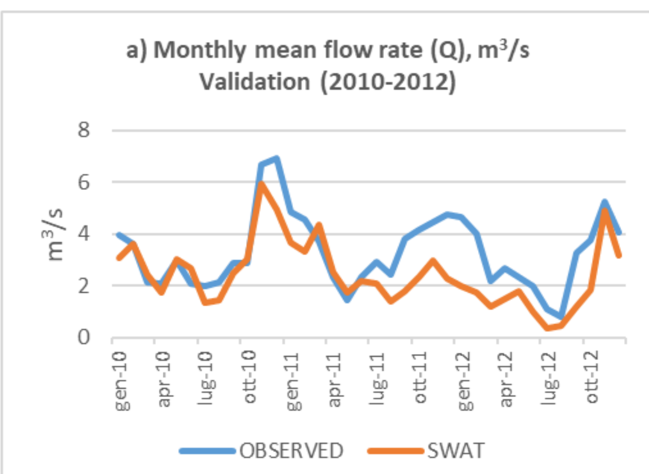
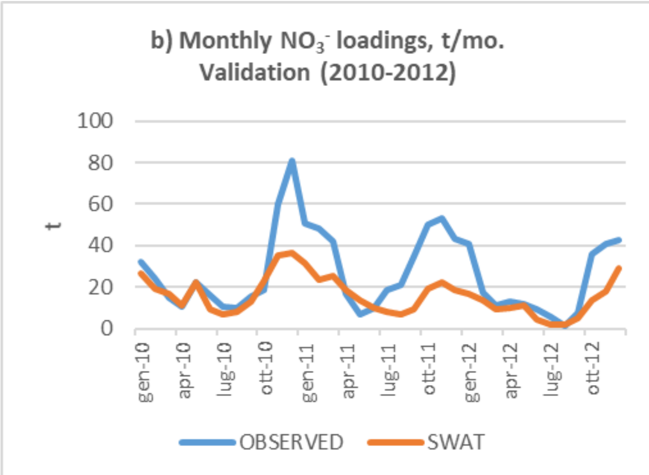


Fig. 3 – Calibration results of the SWAT model: a) water discharge; b) N-NO_3^-; c) N-NH_4^+.

Validation was performed for the 2010-2012 period and resulted in lower NSE for flow rate (NSE=0.20, R2=0.61) and nitrate (NSE=0.25, R2=0.65), related to an extreme precipitation event in October-November 2010 and an underestimation of flow rate during the 2011-2012 autumn-winter, characterised by very low precipitations (Fig. 4a, Fig. 4b). The low performance (NSE=-0.10, R2=0.25) of ammonium during validation period (Fig. 4c) are also attributed to underestimated flow rate during the 2011-2012 autumn-winter period.





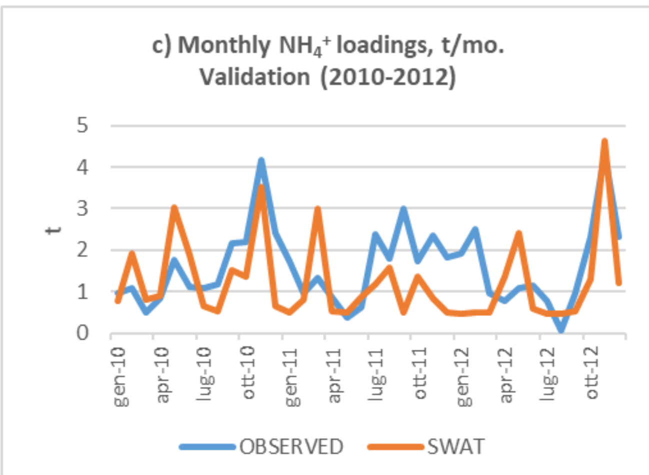


Fig. 4 - Validation results of the SWAT model: a) water discharge; b) N-NO_3^-; c) N-NH_4^+.

Regarding phosphorus loads, the only way to evaluate modelled values was to compare the annual means of the model with those of previous studies in the same areas, even though for different time periods (Zuliani *et al.*, 2005; Salvetti *et al.*, 2008). Given the long-term purposes of the study and the different aspect that characterize the complexity of the Zero river basin (Zaggia *et al.*, 2004; Essenfelder, Giove and Giupponi, 2016), we considered the results acceptable for the validity of the model.

Performance evaluation of the AQUATOX Model

Wind speed, solar radiation, DO, DIN and DIP concentrations, and Chl-a concentrations were used to evaluate the performance of the AQUATOX model of PdC. Relative bias (rB) versus variance test (F) were used to evaluate the performance of the model. Table 3 and **Error! Reference source not found.** summarizes the outcomes of the overlap test through the computed values of relative bias (rb) and F-test (F).

Table 3 – Values of relative bias and F-test for the considered parameters.

Parameter	Mean AQUATOX	Mean Observations	St. Dev. Observations	Variance AQUATOX	Variance Observation	rb	F
Sol. Rad	327.00	334.65	211.06	40325.78	44545.23	-0.03	0.91
Wind	1.38	1.69	0.83	1.67	0.78	-0.38	2.14
DO	8.56	8.16	2.5	2.58	6.25	0.16	0.17
DIN	0.96	0.86	0.69	0.20	0.48	0.14	0.18
DIP	0.03	0.02	0.013	0.00011	0.00017	0.76	0.44
DIN:DIP	30.07	45.35	22.3	303.37	528.9	-0.66	0.34
Chl-a (2007-2011)	3.44	3.5	5.58	35.7	31.09	-0.01	1.32

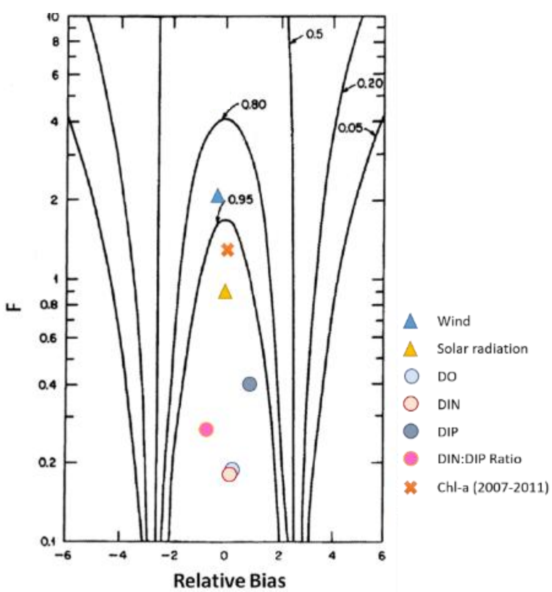


Fig. 5 - Overlap between modeled data and observed data, based on relative bias (rB) and variance (F). Isopleths indicate the probability that the predicted and observed distributions are the same, assuming normality.

In general, it is possible to observe that the majority of modeled variables are in good agreement with the observed data. Discrepancies between modeled and observed values are observed for dissolved inorganic phosphorus and DIN:DIP ratio. The model slightly overestimate phosphorus concentrations over the year. This justifies the higher value of rb (0.76) in the overlap test. A plausible explanation may be the difficulty of AQUATOX in modeling the complex dynamics of nutrient between sediment and water column, specifically the removal of phosphate, as explained in Zirino (2016). Consequently, the modeled DIN:DIP ratio is slightly underestimated, as demonstrated by the rb value (-0.66). However, the seasonal fluctuations are preserved.

Watershed responses to climate change

Changes in temperature and precipitation, as shown in Fig. 6 and Fig. 7, yielded substantial variations in freshwater discharge and nutrient loads in the mid-term (2041-2070) and long-term (2071-2100) periods. Water discharge projections do not show any change in the annual average, with 30-year yearly mean stable at 2 $m^3 s^{-1}$. However, an increase in the late autumn-winter flow, and a marked decrease in the months of July and August for both RCP 4.5 and RCP 8.5 scenarios can be observed (Fig. 8). These results are in agreement with climate projections, which indicate an increase of precipitation in winter and a marked reduction in summer, coupled with an increase in summer evapotranspiration due to the higher temperatures.

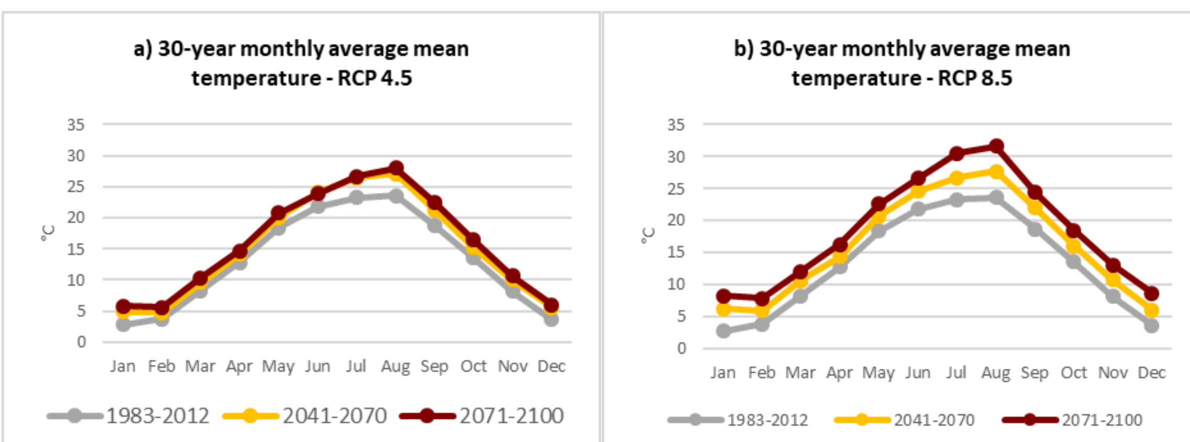


Fig. 6 – 30-year monthly average of mean temperature in the control period (1983-2012) and the mid-term (2041-2070) and long-term (2071-2100) future projections for RCP 4.5 (a) and RCP 8.5 (b).

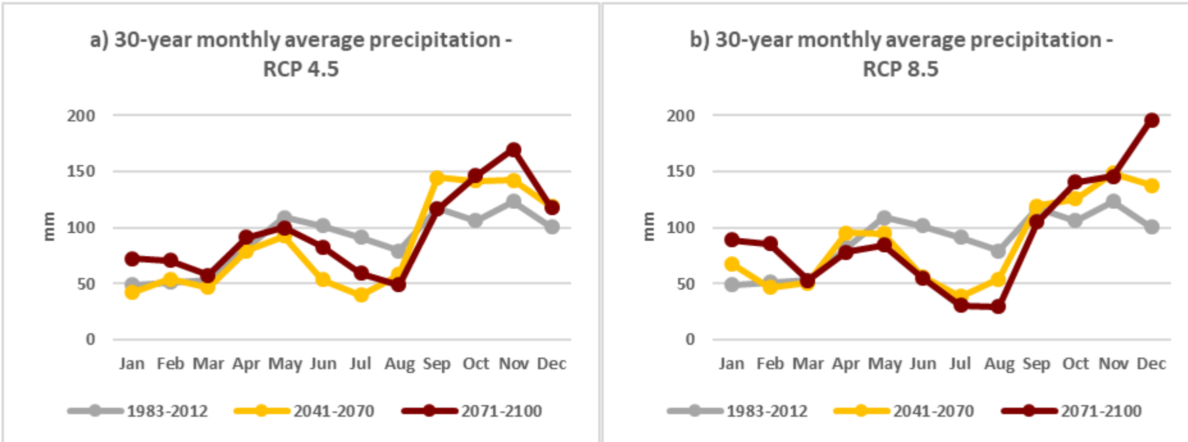


Fig. 7 - 30-year monthly average of mean temperature in the control period (1983-2012) and the mid-term (2041-2070) and long-term (2071-2100) future projections for the RCP 4.5 (a) and RCP 8.5 (b)

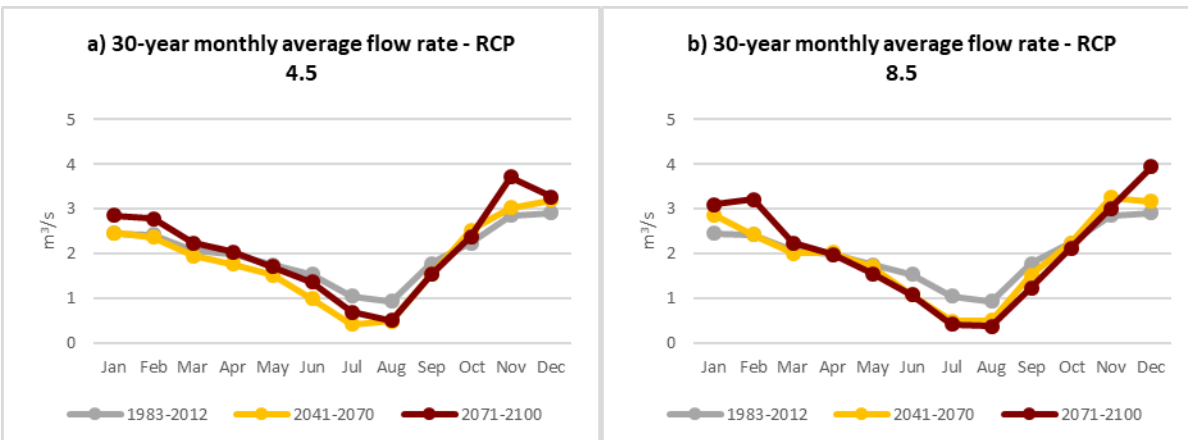


Fig. 8 – Flow rate differences in the 30-year monthly average of the control period (1983-2012) and the mid-term (2041-2070) and long-term (2071-2100) future projections for the RCP 4.5 (a) and RCP 8.5 (b).

The described changes in climate and water flow consequently affected the loads of nutrients. Projections of NO_3^- loads for both RCP4.5 and RCP 8.5 show an increase in the average yearly loadings over the 21st century, with values that increase of up to 5% by the end of the century (Fig. 9). Projections show an increase in winter consequently with projected increased precipitations. NO_3^- loads in summer are influenced by a reduction in precipitation and, therefore, in the water discharge.

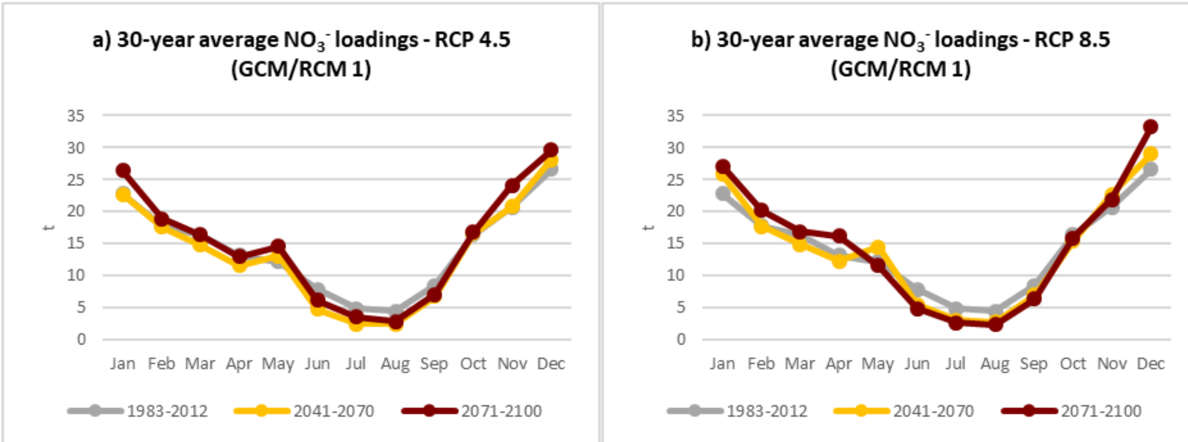


Fig. 9 – Nitrate loadings differences in the 30-year monthly average of the control period (1983-2012), mid-term (2041-2070) and long-term (2071-2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

NH_4^+ loads indicate an increase in the spring and autumn-winter periods, and a decrease in the summer period. For both RCP4.5 and RCP8.5 higher loads of NH_4^+ were observed in the mid-term period (Fig. 10). This is probably caused by the effect of temperatures on nitrogen dynamics, as nitrogen mineralization, nitrification and volatilization processes are influenced by temperature and available water, and reach their optimal values within a range of temperature and humidity in the soil.

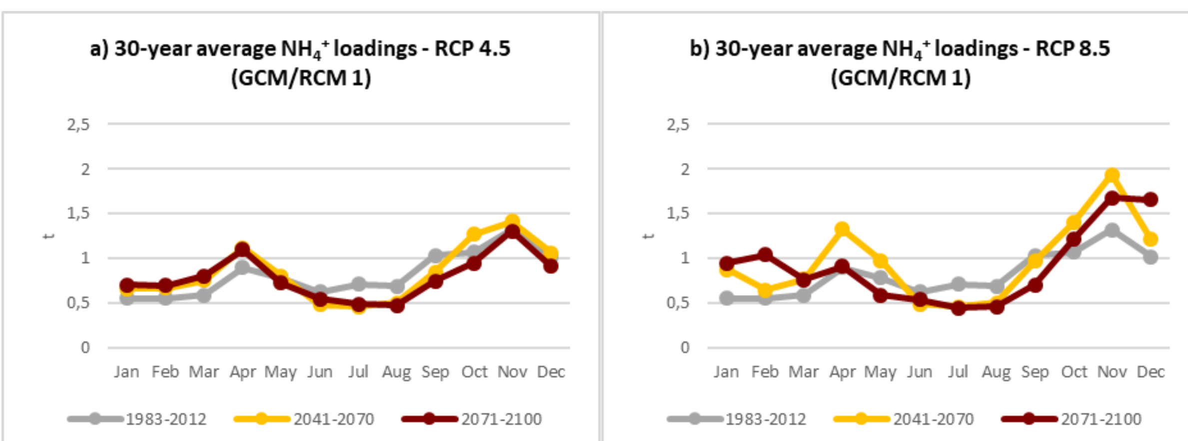


Fig. 10 - Ammonium loading differences in the 30-year monthly average of the control period (1983-2012), mid-term (2041-2070) and long-term (2071-2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

Changes in phosphorus were also observed (Fig. 11). Results indicate marked changes in the magnitude of the winter loads in both RCP4.5 and RCP8.5, probably caused by an enrichment of the topsoil in inorganic phosphorus due to accelerated remineralization, in conjunction with increased leaching and erosion processes caused by increasing precipitations in the autumn-winter period (Jennings, 2009; Pierson et al., 2010). Moreover, drier conditions in the summer might exacerbate the erosion of soil in the autumn season, and consequently increase the runoff of sediments and adsorbed mineral forms of phosphorus.

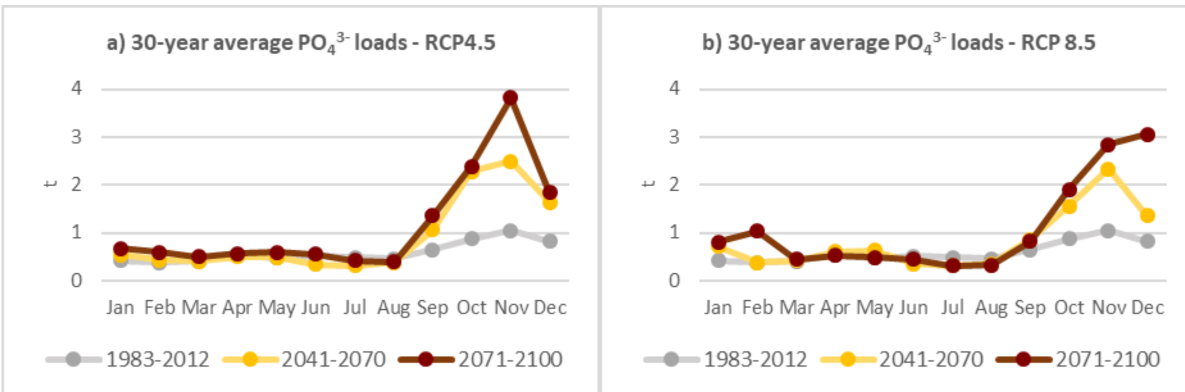


Fig. 11 – Inorganic phosphorus loading differences in the 30-year monthly average of the control period (1983-2012), mid-term (2041-2070) and long-term (2071-2100) future projections for scenarios RCP 4.5 (a) and RCP 8.5 (b).

3.1. Physico-chemical responses of the coastal ecosystem

Water quality variables (DO, DIN, DIP, DIN:DIP ratio) related to the ecosystem of PdC are evaluated in this section. Simulated results indicate that dissolved Oxygen (DO) concentrations features a decrease in both RCP4.5 and RCP8.5. As expected, DO decrease in summer, from 7 mg L^{-1} to 4.5 mg L^{-1}, is more marked than in winter (Fig. 12).

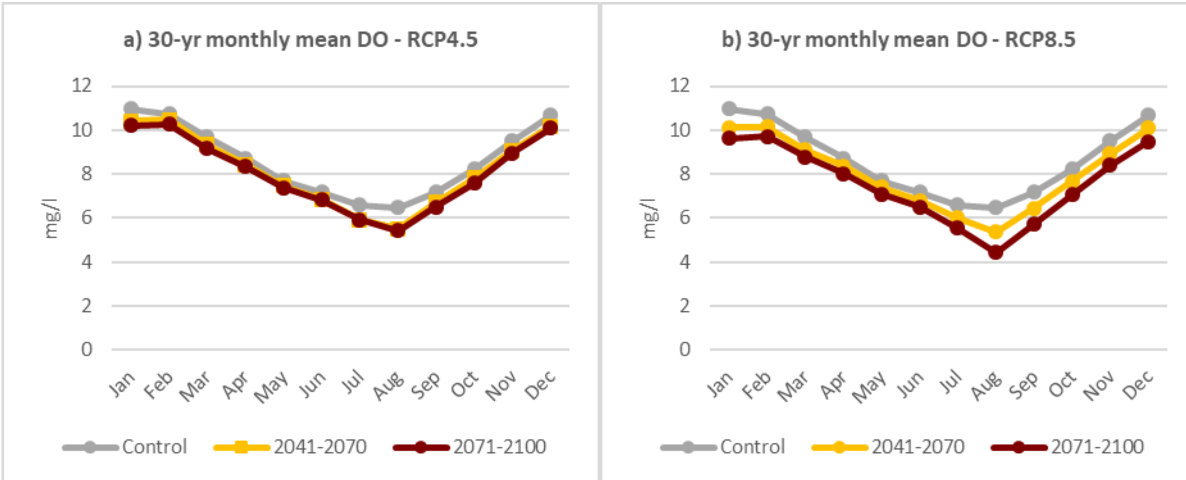


Fig. 12 - Differences in the 30-year DO monthly mean between the control period (1983-2012), and the mid-term (2041-2070) and long-term (2071-2100) future projections for RCP4.5 (a) and RCP8.5 (b).

Future projections of DIN concentrations in water don't show substantial changes from the control period (Fig. 13). Both RCPs indicate an increase in DIN concentration due to an increase of NH_4^+ as temperature increase. In the remaining months DIN features a general stability in its concentration levels.

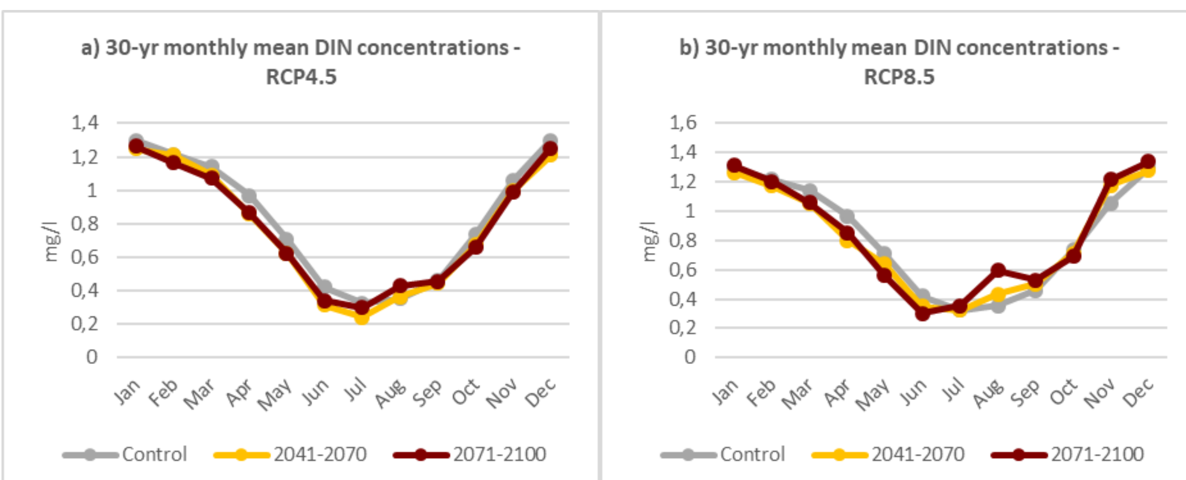


Fig. 13 – Differences in the 30-year DIN monthly mean between the control period (1983-2012), and the mid-term (2041-2070) and long-term (2071-2100) future projections for RCP4.5 (a) and RCP8.5 (b).

DIP concentrations reflect the changes in phosphorus loadings from the ZRB. It is possible to observe a substantial increase of phosphorus concentrations in the winter period, while summer concentrations remain unchanged (Fig. 14).

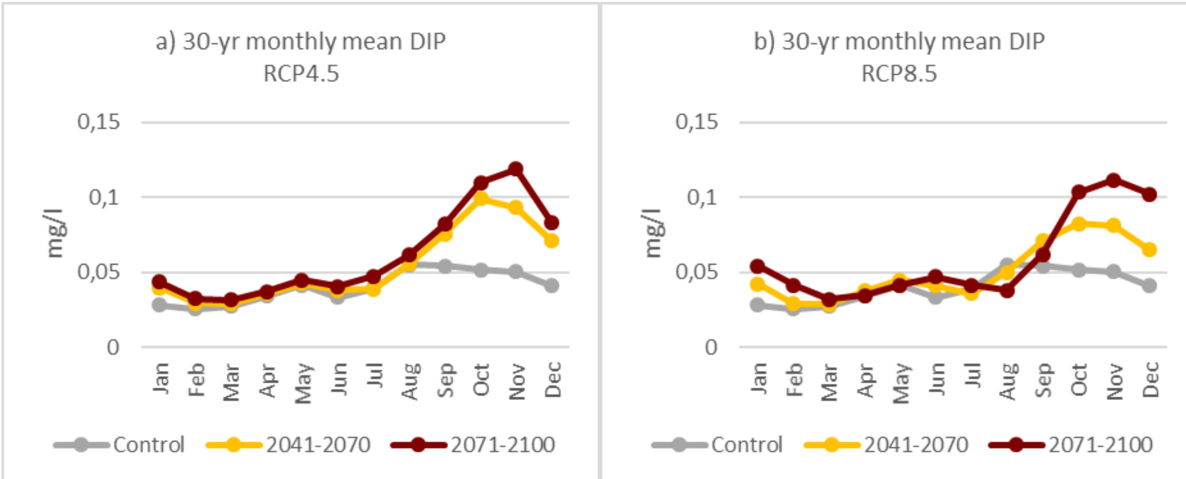


Fig. 14 - Differences in the 30-year DIN monthly average concentrations between the control period (1983-2012), and mid-term and long-term projections for RCP4.5 (a) and RCP8.5 (b).

Future projection of DIN:DIP ratio indicates marked changes in autumn and winter (Fig. 15), due to the marked changes in phosphorus concentrations. Differently, summer months don't show substantial changes. The higher availability of phosphorus in the winter reduce noticeably the DIN:DIP ratio. Given the overestimation of phosphorus in the model and the greatest changes in the DIN:DIP ration happening in the winter season, results do not suggest relevant effects on the composition of phytoplankton in PdC. However, the events in which nitrogen become the limiting nutrient may become more frequent than current times.

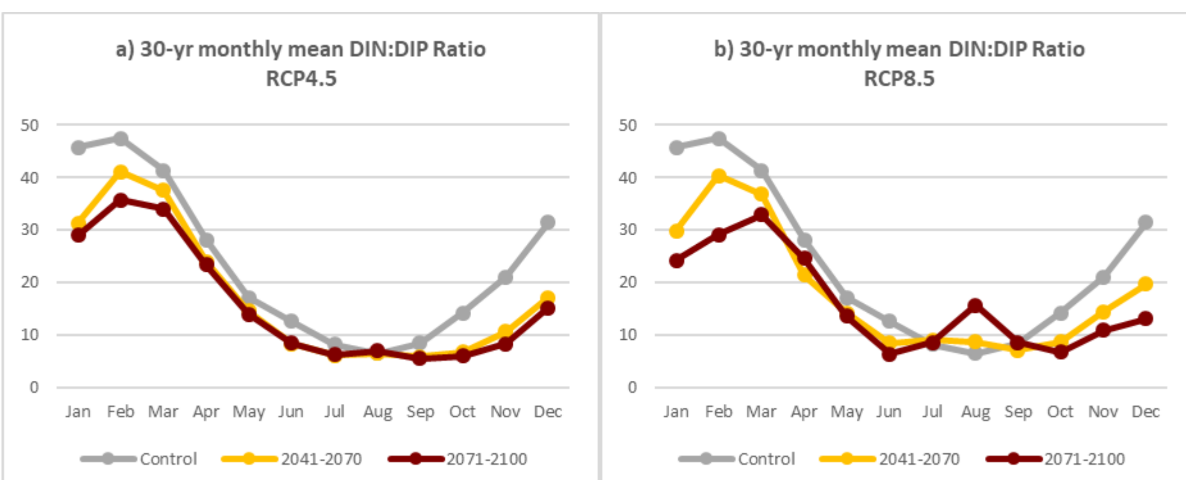


Fig. 15 – Differences in the 30-year DIN:DIP monthly mean between the control period (1983-2012), and the mid-term and long-term projections for RCP4.5 (a) and RCP8.5 (b).

3.2. Phytoplankton responses of the coastal ecosystem

30-year averages of Chl-a concentrations were observed (Fig. 16**Error! Reference source not found.**). Total phytoplankton biomasses increase in both scenarios, particularly in the long-term period of RCP8.5. The RCP4.5 scenario does not indicate marked changes, where only an increase in the summer months is observed. RCP8.5 indicates more marked differences both in concentration and seasonality. Yearly average concentrations rise from 66.44 μg L^{-1} (control period) to 67.7 μg L^{-1} (2041-2070) and 89.48 μg L^{-1} (2071-2100). It is also observable an evident shift in the peak of Chl-a, from June to August. It is important to consider the fact that Chl-a values are representative of the phytoplankton composition in the system and they cannot model adaptation or addition of new, more tolerant species.

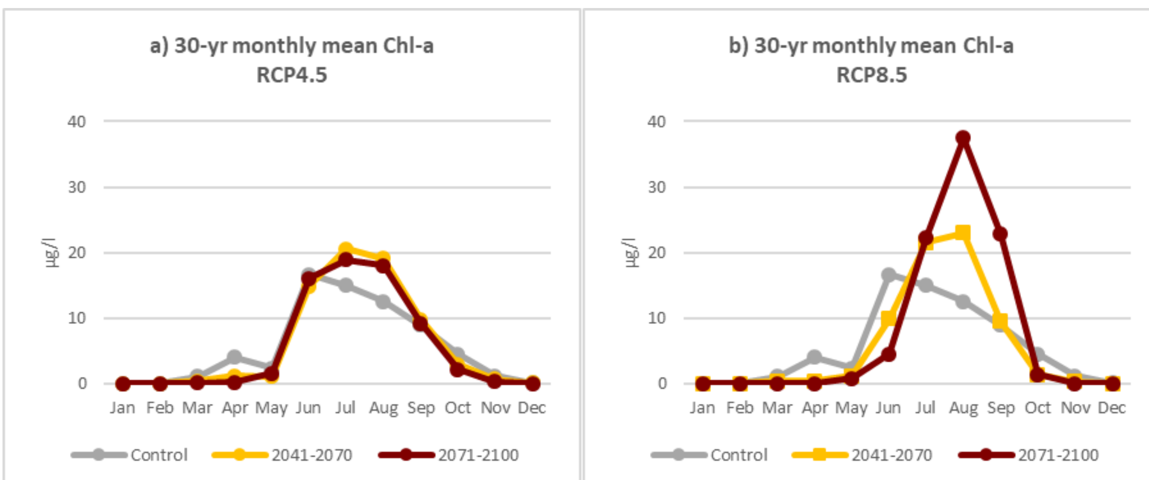


Fig. 16 - Differences in the 30-year monthly average Chl-a concentrations between the control period (1983-2012), and mid-term (a) and long-term (b) projections (2071-2100) of RCP4.5 and RCP8.5.

Marked differences are also observable in the composition of phytoplankton (Fig. 17 & Fig. 18). In the control period, phytoplankton is mainly composed of D1, D2, D5, and D8 (representative of the spring bloom in PdC). The major changes happen in the summer months, where the abundances of Cyanobacteria (CB1) and diatoms adapted to warmer temperatures (D6) increase noticeably. Another observation is that Navicula (D1), a common diatom in PdC and other areas of the lagoon of Venice, tends to disappear in every future scenario. These results illustrate how different climate condition will promote the growth of species which are more resistant to warm temperatures, and inhibit the growth of the species that currently dominate the composition of phytoplankton. Substantial stability in the DIN:DIP ratio in the blooming period does not promote growth of phytoplankton with different nutrient ratios, such as D3, D6 and D7. Thus, results indicate that major changes in the phytoplankton community will be caused by higher temperatures, while changes in nutrient loadings might be a secondary driver of change.

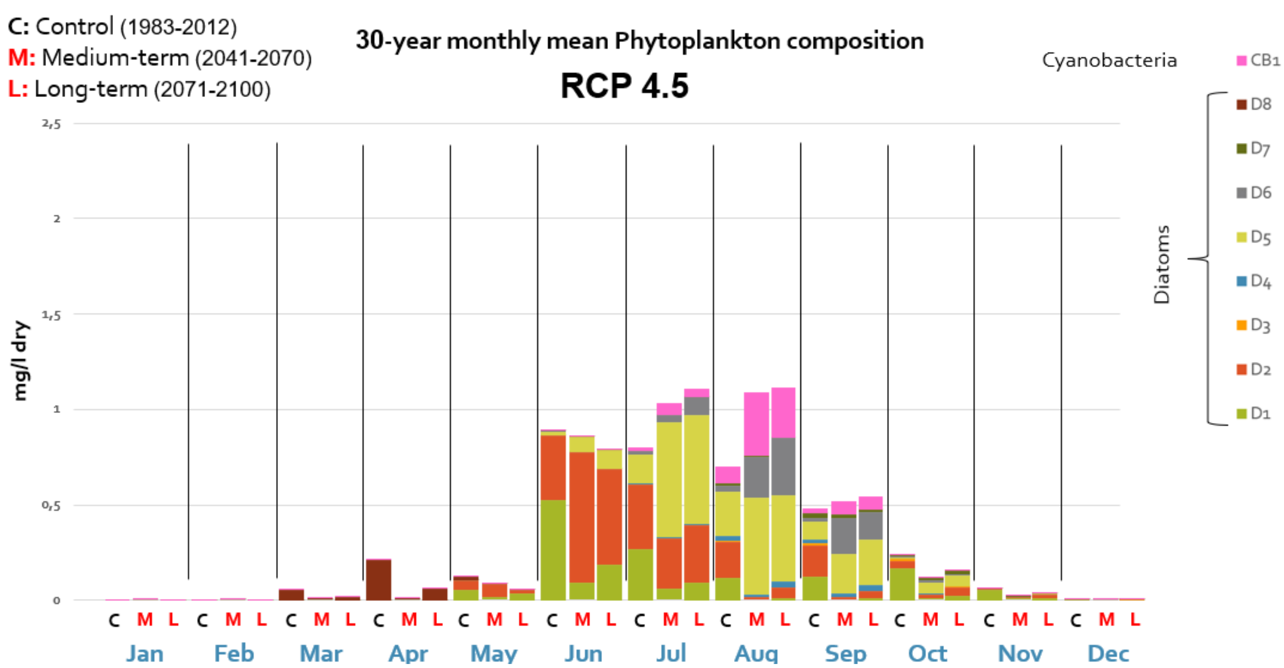


Fig. 17 – Comparison of abundance of different species of phytoplankton between control period and mid- (2041-2070) and long-term (2071-2100) periods for RCP4.5.

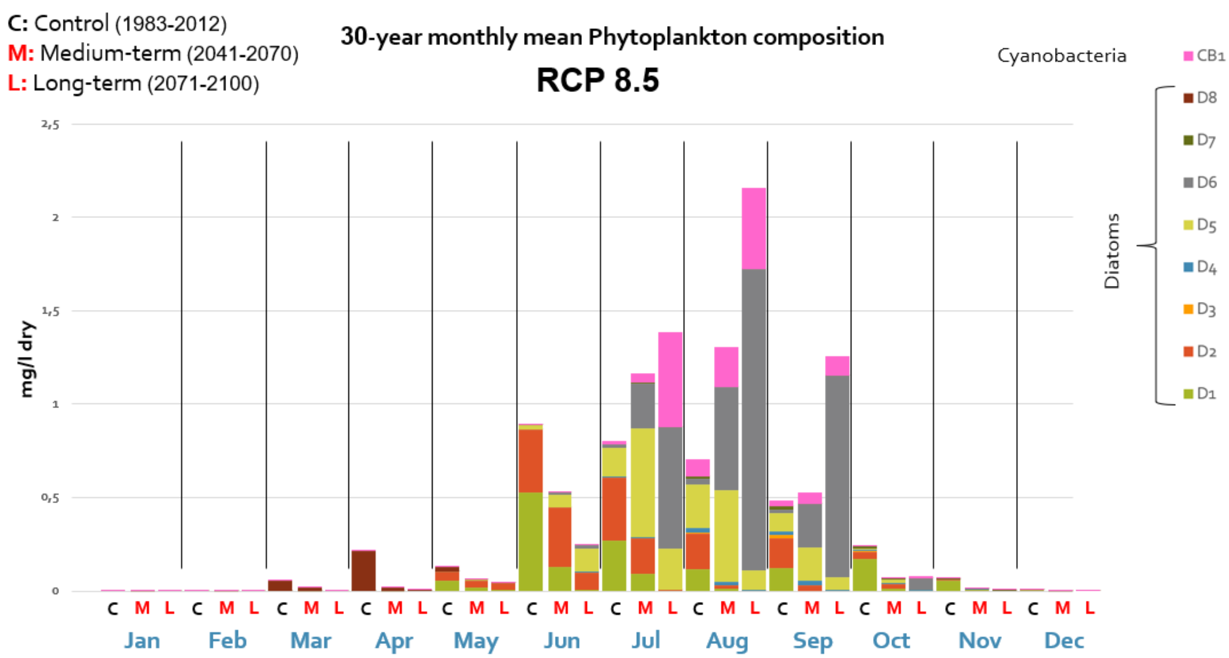


Fig. 18 - Comparison of abundance of different species of phytoplankton between control period and mid- (2041-2070) and long-term (2071-2100) periods for RCP8.5

4. Discussion and conclusion

Results indicate that climate change will exacerbate current hydrological conditions of the Zero river basin, thus affecting nutrient loads. Freshwater discharge will increase in winter and decrease in summer, and so will do nutrient loads, especially P, to the lagoon of Venice. The ecological impact of modified loads and increased temperatures is a projected increase in eutrophication events in the summer, with peaks of phytoplanktonic biomass occurring at greater magnitude. Moreover, phytoplankton composition will change considerably. Results indicate that species adapted to warmer waters will dominate and replace current species, and that will generate greater blooms. Moreover, the increased frequency of nitrogen limited conditions, together with a greater availability of phosphorus, may favor the growth of nitrogen-fixing cyanobacteria.

The implemented integrated modelling approaches necessitated some assumptions and simplifications that added uncertainty to the study. First, meteorological drivers such as wind, relative humidity and solar radiation were not changed from baseline scenarios, due to the uncertainty of projected values of these variables. As wind speed has an important role in the hydrodynamics of the lagoon and on the biological processes of phytoplankton this can be considered a potential area of improvement. Second, here only results for one GCM/RCM scenario are presented but, considering that the obtained results are highly dependent on the assumptions of the selected GCM/RCM combination, others GCM/RCM should be applied to the integrated methodology in order to assess the uncertainty due to climate change projections, and this could be part of future research activities. Third, land-use, agricultural activities and nutrient loadings from other anthropogenic sources (e.g. WWTPs) were kept as constant. Moreover, global events such as the projected P shortage anticipated in coming decades, and the potential reduction in fertilizers were not considered (Glibert *et al.*, 2014). The integration of projections on land-use and other human influences on the environment could add further value to the modelling approach. Finally, the effect of tide, sea level rise and hydraulic infrastructures (e.g. Mose Project) were neglected in this study but could have important consequence on the hydrodynamics, and therefore on the ecosystems of the lagoon in the coming decades.

The study demonstrates the utility of integrating climate scenarios and environmental models to project impacts of multiple stressors on aquatic ecosystems, defining future climate conditions and predicting the impact on abiotic and biotic components of the ecosystem. Specifically, SWAT and AQUATOX offer valuable tools to project the impacts of climate change on watersheds and on aquatic ecosystems. Both models have been applied in assessing climate change independently; however, to our knowledge, they have not previously been coupled together for this purpose. Moreover, this work is the first application of AQUATOX to a waterbody of the lagoon of Venice. In conclusion, to further improve this integrated modelling approach and use it for planning/management purposes, potential improvements include: (1) adoption of more GCM/RCM scenarios to assess the uncertainty coming from climate projections and the sensitivity of ecological parameters to climate drivers; (2) incorporating additional atmospheric drivers (i.e. wind speed, solar radiation, relative humidity); (3) implementing a hydraulic model able to simulate the specific hydrodynamics of the lagoon; (4), integrating land-use change scenarios and other models capable of simulating and predicting changes in land-water interactions; and, finally, (5) collecting more data to implement a rigorous calibration and validation of hydrological, water quality and ecological parameters. All these aspects may provide a mean for improving the impact assessment of climate change on aquatic ecosystems in the future.

Acknowledgments

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