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Integrated ecological modelling for evidence-based determination of water management interventions in urbanized river basins: Case study in the Cuenca River basin (Ecuador)

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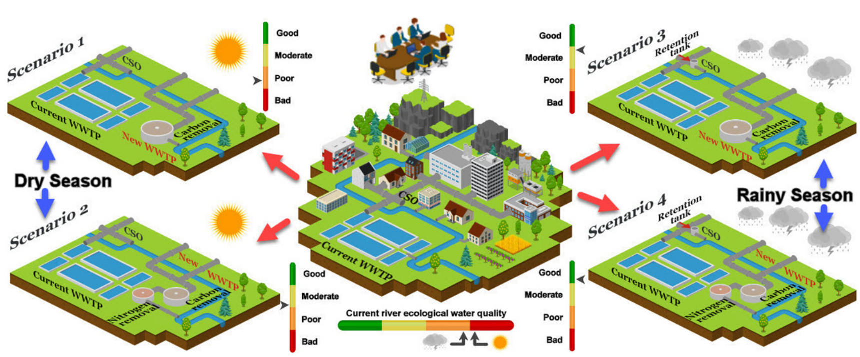
HIGHLIGHTS

- An integrated urban wastewater system model was linked with ecological models
- Four scenarios were analyzed to improve the river water quality
- The benefits of the inclusion of a new WWTP will be most significant in dry season
- Retention tanks before discharges from CSOs play an important role in rainy season
- The integrated model was an essential tool to support decisions in the Cuenca Basin

GRAPHICAL ABSTRACT



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ABSTRACT

The growth of urbanization worldwide has contributed to the deterioration of the ecological status of water bodies. Efforts at improving the ecological status have been made either in isolated form or by means of integrated measures by stakeholders, but in many cases, these measures have not been evaluated to determine their benefit. In this study, we implemented a scenario analysis to restore the ecological water quality in the Cuenca River and its tributaries, which are located in the southern Andes of Ecuador. For this analysis, an integrated ecological model (IEM) was developed. The IEM linked an urban wastewater system (IUWS) model, which gave satisfactory results in its calibration and validation processes, with ecological models. The IUWS is a mechanistic model that incorporated the river water quality model, a wastewater treatment plant (WWTP) with activated sludge technology, and discharges from the sewage system. The ecological status of the waterways was evaluated with the Andean Biotic Index (ABI), which was predicted using generalized linear models (GLMs). The GLMs were calculated with physicochemical results from the IUWS model. Four scenarios that would enhance the current ecological water quality were analyzed. In these scenarios, the inclusion of a new WWTP with carbon, and with carbon and nitrogen removal as well as the addition of retention tanks before the discharges of combined sewer overflows (CSOs) were assessed. The new WWTP with carbon and nitrogen removal would bring about a better

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restoration of the ecological water quality due to better nitrogen removal. The retention tanks would help to enhance the ecological status of the rivers during rainy seasons. The integrated model implemented in this study was shown to be an essential tool to support decisions in the Cuenca River basin management.

1. Introduction

The growth of urbanized areas around the world has increased the pressure on their nearby aquatic ecosystems, which includes rivers, estuaries, and lakes. Generally, these pressures originate in the management of the wastewater and runoff during rainfall. Thus, point discharges from outfalls and wastewater treatment plants (WWTPs) or combined sewer overflows (CSOs), and surface water outfalls (SWOs) during rain events disturb the ecological equilibrium of the receiving water body. This disturbance affects both the chemical composition of the water and the biotic structure (Hvitved-Jacobsen, 1982; Mulliss et al., 1997) and contributes to the deterioration of the morphology of the waterways such as river beds, flow changes, and bank structures (Walsh et al., 2005).

To date, stakeholders have implemented isolated or integrated measures to enhance the water quality of the streams. These actions have been addressed to comply with national or local regulations that in most cases control the physicochemical composition of the water, but not the ecological status of the receiving stream (Holguin-Gonzalez et al., 2013b). However, the optimization of resources invested in improving water quality requires planning tools. These tools are used to simulate an individual process, which when are run separately do not achieve good results in the restoration of the ecosystem (Kraft, 2011). In the case of municipal water management, a good planning tool for this optimization is the implementation of an integrated urban water system (IUWS) model that includes WWTPs, sewage networks and receiving water. The IUWS model combines cost-efficient solutions, the analysis of possible synergies and the optimization of the wastewater system performance (Benedetti et al., 2013). The final outputs of an IUWS model are the concentration of organic pollutants and some hydrological variables in rivers such as water flows. This integrated model has demonstrated great potential in the scenario analysis to improve the river water quality. Basins located in South Africa (Deksissa et al., 2004), Luxemburg (Solvi et al., 2005), Italy (Benedetti et al., 2007) and Colombia (Holguin-Gonzalez et al., 2013b) are examples of the applications of the IUWS models in river management.

The ecological quality of freshwater bodies is affected by both physicochemical changes and hydro-morphological variation produced by natural and anthropogenic stressors (Burneo and Gunkel, 2003; De Pauw et al., 2006). To preserve the aquatic ecosystems, regions such as the European Union (Griffiths, 2002) and countries such as the United States (USEPA - U.S. Environmental Protection Agency, 2011) have introduced the concept of ecological status in river management into their policies. This status includes an integrated assessment of the biological, hydro-morphological and physicochemical elements of the water quality (Griffiths, 2002). In this context, ecological models have been developed as tools that can support environmental management and policy development. These types of models have provided an understanding of the influence of different pressures such as the impact of wastewater and CSOs, in addition to supporting the wastewater treatment selection or predicting the ecological water quality (Goethals and Forio, 2018). As such, the integration of hydro-morphological aspects with physicochemical water quality has been applied as a tool in basin management to predict potential restoration of ecological water quality (Tomsic et al., 2007; Mouton et al., 2009). Basins with urban areas such as the Cauca River in Colombia (Holguin-Gonzalez et al., 2013b) and the Drava River in Croatia (Holguin-Gonzalez et al., 2014) are examples of the application of IUWS models to predict the final ecological status of rivers.

Since 1984, the Water Supply and Wastewater Management Municipal Company ETAPA-EP has been working on improving the water quality status of the Cuenca River and its tributaries. Thus, the combined sewage network and sewage interceptors have been expanded in the uppermost part of the urban area, some suburban neighborhoods and closed towns. Similarly, in 1999, the first facility to treat urban wastewater with pond technology began. This was located downstream from Cuenca in the Ucubamba area. This effort has also included biological water quality assessment as well as the use of a mathematical modelling approach to predict final physicochemical water quality; results that have supported the water management implementation (ETAPA-EP, 2007). Holguin-Gonzalez et al. (2013a) evaluated the contribution of the Ucubamba wastewater treatment plant (U-WWTP) to the ecological water quality of the Cuenca River during the dry season. This assessment was executed by means of the linkage of the physicochemical water quality results provided by the QUAL2K model with ecological models. Although the water quality has improved with the establishment of the facilities as described above, biological water quality varies between deficient and bad in most of the stretches of the urban area of the Cuenca River and its tributaries during the dry season. This quality changes from moderate to good during the rainy season (Jerves-Cobo et al., 2018b). The leading causes that affect the ecological status of the Cuenca River system are management problems of the wastewater and runoff such as leakage from the sewage system, a suburban population that discharge directly into streams, industrial discharges, an overloaded U-WWTP, CSOs discharges, SWOs and disperse pollution from livestock areas (Greeley and Hansen and ACSAM, 2017; Jerves-Cobo et al., 2017; Jerves-Cobo et al., 2018b).

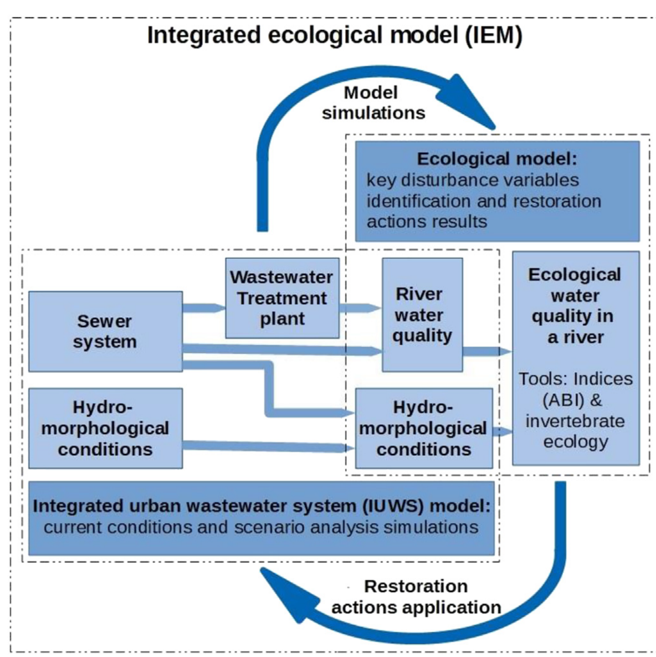

Fig. 1. Framework of the proposed integrated ecological modelling for scenario analysis in river restoration.

The framework and objectives of this research are presented in Fig. 1. We aimed to develop and to validate an integrated ecological model (IEM) for the analysis of possible scenarios to be applied to river restoration and management. This IEM assesses the variation in the ecological water quality produced simultaneously by physicochemical pollution and hydro-morphological pressures. In this regard, the IEM incorporated four models: (1) a river model used to predict the physicochemical water quality in watercourses; (2) an activated-sludge wastewater treatment plant model; (3) a sewer model that generates combined sewer overflows; and (4) ecological models to assess the ecological river water quality that was developed in earlier research by Jerves-Cobo et al. (2020). This framework was applied to analyze potential measures and their impact on the restoration of the Cuenca River that is located in the southern Andes of Ecuador.

2. Materials and methods

2.1. Study area

The study area corresponds to the urban and suburban areas of the Cuenca River basin, which is situated in the southern Province of Azuay in the Andes of Ecuador. The Cuenca River is an Andean mountain stream that is part of the Paute Upper Basin. The latter is one of the tributaries of the Santiago River, which is an affluent of the Amazon River. The Cuenca River is located downstream from the city of Cuenca (Fig. 2) and its four main tributaries: the Tarqui, Yanuncay, Tomebamba and Machangara Rivers that run upstream and through the urban areas of Cuenca. Of the four sub-catchments, only the Machangara Basin is regulated all year by the presence of two hydropower dams: the Labrado and the Chanlud, which are located upstream from the city of Cuenca (Jerves-Cobo et al., 2018a).

The study area is 223 km^2, representing 13% of the Cuenca River basin, of which 16% is an urban area. This urban area had approximately 382,000 inhabitants in 2017 (SENPLADES, 2016). The City of Cuenca contributes about 4.4% of the national GDP, and has a poverty level <5%. Its economy is supported by two main activities: the manufacturing industry (18%) and the construction industry (17%) (ODS-Territorio-Ecuador, 2017). The majority of these industries, about 145, are located in the northeast area of the City of Cuenca, in an industrial park close to the Machangara River (CGA, 2007). Two natural reserves are also located upstream from the Cuenca River basin: Cajas National Park and the Machangara-Tomebamba protected forest. Both are water sources for the Tomebamba, Yanuncay and Machangara Rivers.

The mean altitude of the study area is 2655 m a.s.l., while its average rainfall between 1977 and 2011 was approximately 879 mm per year, and the yearly average temperature was 16.3 °C (Aereopuerto-Mariscal-Lamar, 2012). There are two seasons during the year: the rainy season, which starts from the middle of February until the beginning of July and from the second half of September until the first two weeks of November, while the rest of the year constitutes the dry season. The average flow of the Cuenca River that was measured between 1990 and 2012, prior to its discharge into the Paute River was 28 m^3·s^{-1} (Cordero Domínguez, 2013). Similarly, between 2014 and 2017, the 5th percentile of the annual flow happened during rainy

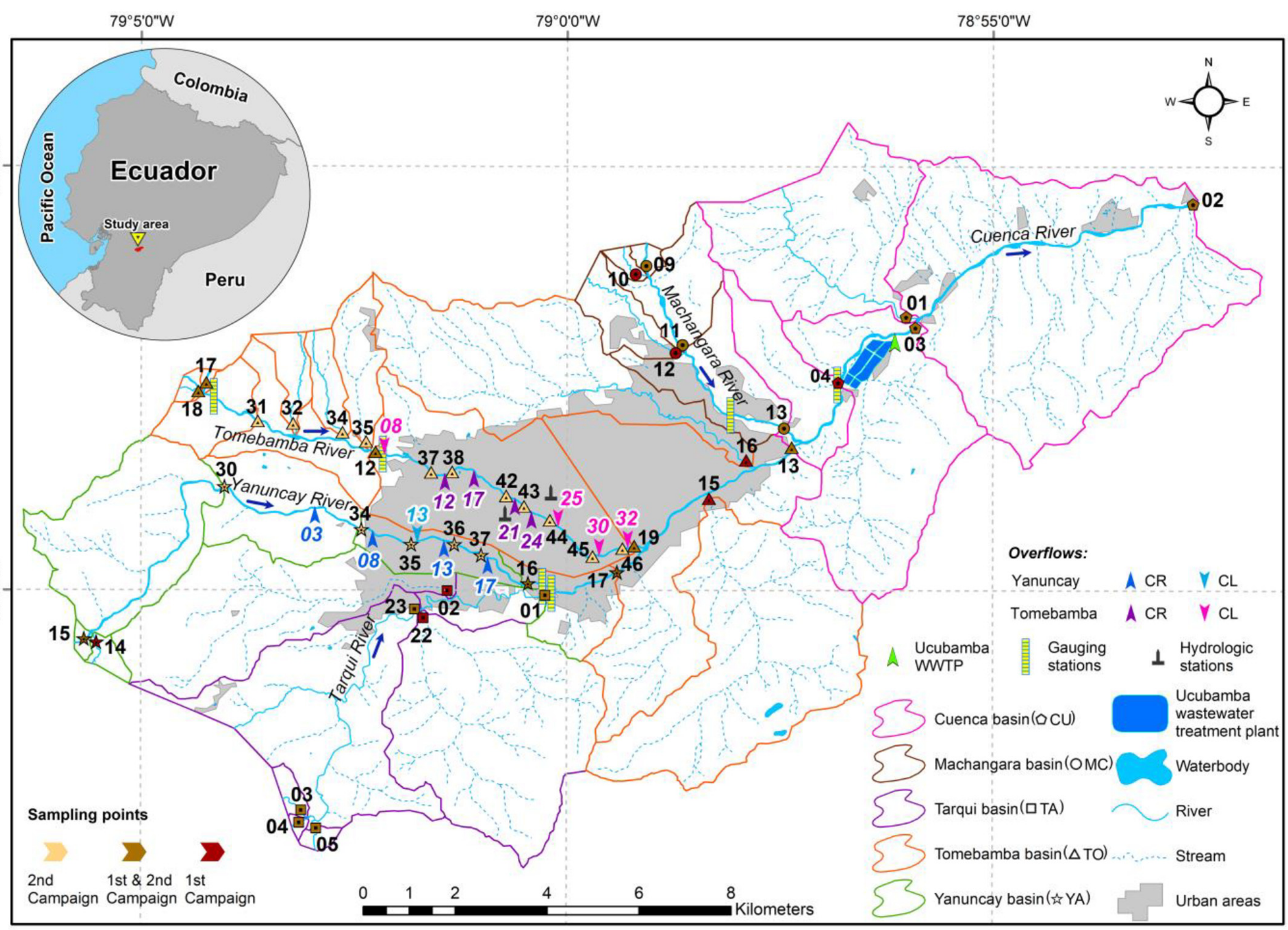

Fig. 2. Map of the studied area, including sampling sites – adapted from Jerves-Cobo et al. (2018b). The large gray area represents the city of Cuenca.

season – March to Jun – was around 71 $m^3.s^{-1}$, while the 95th annual flow percentile was about 6 $m^3 \cdot s^{-1}$, which occurred in dry season between August and September (ETAPA-EP, 2018).

The Water Supply and Wastewater Management Municipal Company, ETAPA-EP, has been working since 1984 to improve the water quality in the rivers that pass through the city of Cuenca. Accordingly, projects have been implemented in Cuenca and its surrounding areas, such as the construction of the U-WWTP, the expansion of sewer interceptors and the enlargement of the combined sewer network (ETAPA-EP, 2007). The U-WWTP treats both municipal and industrial wastewater so that its daily-average effluent meets the Ecuadorian standards (MAE-Ecuador, 2015). However, water quality in rivers is being affected by different contributing sources of pollution that include point discharges from sewer networks and industries, storm water overflows from the sewer network and excess flow from the by-pass located before the U-WWTP as well as surface water outfall (SWO) (Jerves-Cobo et al., 2018b). Adding to the degradation of the river is the diffuse fluxes of organic pollution from extensive livestock, mainly in the Tarqui Basin (Beltrán et al., 2013).

2.2. Data collection

In total, 43 sampling sites located in the city of Cuenca and nearby areas were considered for this research. From these sites, 27 were sampled during the dry season in July 2015, while 35 sites were sampled during the rainy season in March 2016 (Fig. 2). Of these, during both seasons 19 sites were sampled. At each sampling site, we recorded physicochemical and hydro-morphological conditions and collected biological (macroinvertebrates) samples. In particular, 28 physicochemical, hydraulic and microbiological variables were measured. Table A1 presents the overview of the chemical data per and during both seasons. Additionally, we compiled the following information at each sampling site: elevation, land use, river morphology, substrate characteristics, macrophytes, shading in the rivers and flow variation. The kick-sampling procedure as described by Gabriels et al. (2010) was applied to take the samples of benthic macroinvertebrates from the river and its tributaries. In total, 43 taxa of macroinvertebrates were identified up to family level. For a detailed overview of the sampling campaigns and description of the locations, the authors refer to Jerves-Cobo et al. (2018b).

At each sampling site, the Andean Biotic Index (ABI) (Encalada et al., 2011; Ríos-Touma et al., 2014) was calculated to assess the water quality. The ABI was used as it was shown to be more suitable for the Andes in Ecuador above 2400 m a.s.l. (Jerves-Cobo et al., 2018b). Fig. A1 shows the ABI score of each sampling site in both seasons that was recorded by Jerves-Cobo et al. (2018b).

The physicochemical information of the Cuenca River was complemented with nine samples taken before and after the discharge of the U-WWTP during 2017 (Espinoza Berrezueta and Zumba López, 2018).This information was included because no significant changes were registered in population, sewage networks, livestock areas or yearly rainfall in comparison to the previous years of 2015 and 2016.

2.3. Integrated river water quality model: Building, validation and implementation

In the simulation of the integrated river water quality system, an open model structure simulator called WEST® (Vanhooren et al., 2003) was used. WEST® has mainly been applied to the modelling and simulation of wastewater treatment plants, although this software also works with river water quality models and sewer models that are based on a cascade of continuously stirred tank reactors in series (CSTRS) (Deksissa et al., 2004). The structure of WEST® is flexible enough to allow for the inclusion of a relevant process for the study or exclusion of a less critical process. Thus, in the first instance, we developed the river water quality model for the Tomebamba and Cuenca Rivers, so when this model was calibrated during the dry and rainy seasons (for details, see Section 2.3.1), we included the elements shown in Fig. 3 for the scenario analysis. In the analysis, we simulated a new wastewater treatment plant during the dry season (Fig. 3) as well as the new WWTP and tanks to control overflows from the combined sewer network from the city of Cuenca during rainfall events (Fig. 3). Finally, to understand the ecological water quality in the stretches of the Tomebamba and Cuenca Rivers, the information obtained from the WEST® simulations was entered into the equations that link the physicochemical and morphological variables with the ecological water quality that was achieved in the ecological model.

2.3.1. River water quality model

For modelling the river, we used the River Water Quality Model No.1 (RWQM1) (Reichert et al., 2001) implemented in the WEST® software. The RWQM1 is based on the concept of CSTRs to represent, the transport of pollutants along rivers. The river model was constructed with 20 CSTRs in series that simulates 27.1 km of the Tomebamba and Cuenca Rivers from Cuenca and its surrounding areas. Correspondingly, their urban tributaries such as the Tarqui (0.3 Km) and Machangara (0.5 Km) Rivers. These were also simulated with one CSTR each, while the Yanuncay River (3.2 Km) was simulated with two CSTRs. Moreover, the small streams, the Amarillo River, El Tejar, Saucay, El Valle, Milchichig and Sinincay Brooks were included in the IEM as point discharges (Fig. 3).

The concentration of pollutants measured in the two sampling campaigns of 2015 and 2016 (Section 2.2) was mostly computed as model state variables. These included dissolved oxygen (S_{O2}), ammonium-nitrogen (S_{NH}), nitrite-nitrogen (S_{NO2}), nitrate-nitrogen (S_{NO3}) and phosphate (S_{PO}). However, the fractions of model components concerning the COD were derived from the measurements, according to Eqs. (1) and (2) (Reichert et al., 2001). These derived compounds were readily biodegradable soluble COD (S_S), inert soluble COD (S_I), particulate inert COD (X_I), particulate organic matter (X_S), heterotrophic biomass (X_H), first stage nitrifying bacteria (X_{N1}), second stage nitrifying bacteria (X_{N2}) and algae and macrophytes (X_{ALG}). The values of the fractional compounds of COD are shown in Table A2 and were taken from previous model applications (Benedetti, 2006; Solvi, 2006).

$$COD_{Total} = COD_{soluble} + COD_{particulate} \tag{1}$$

$$COD_{Total} = (S_S + S_I) + (X_I + X_S + X_H + X_{N1} + X_{N2} + X_{ALG}) \tag{2}$$

The soluble COD represented an average of 78% of the total COD; a value that was the result of the relationship between volatile dissolved solids and total volatile solids obtained from all measurements sampled during the dry and rainy seasons. Chlorophyll-a was transformed into algae and macrophytes using a factor recommended by Jorgensen et al. (1991). The relationship between the COD and the five-day biological oxygen demand (BOD_5) in the rivers showed a wide range of variation, changing from around three in polluted sites to almost 20 in reference sites. For that reason, the variation of this relationship (COD/BOD_5) was calibrated with a logarithmic regression (Fig. A2) that had an R^2 of 0.62. This relationship was applied to calculate the BOD_5 in each stretch of the Tomebamba and Cuenca Rivers.

The Tomebamba and Cuenca Rivers model was calibrated in two steps: the first step was the process of hydraulic calibration and the second step was the physicochemical calibration. The hydraulic calibration was performed by modifying the Manning roughness coefficient for the riverbed. This coefficient was chosen according to the range of values recommended by Te Chow (1959). Thus, the values of flow and water depth predicted by the model were compared with the measured values. For the hydraulic calibration, information from the flow of six gauging stations located in the study area (Fig. 2) was used (ETAPA-

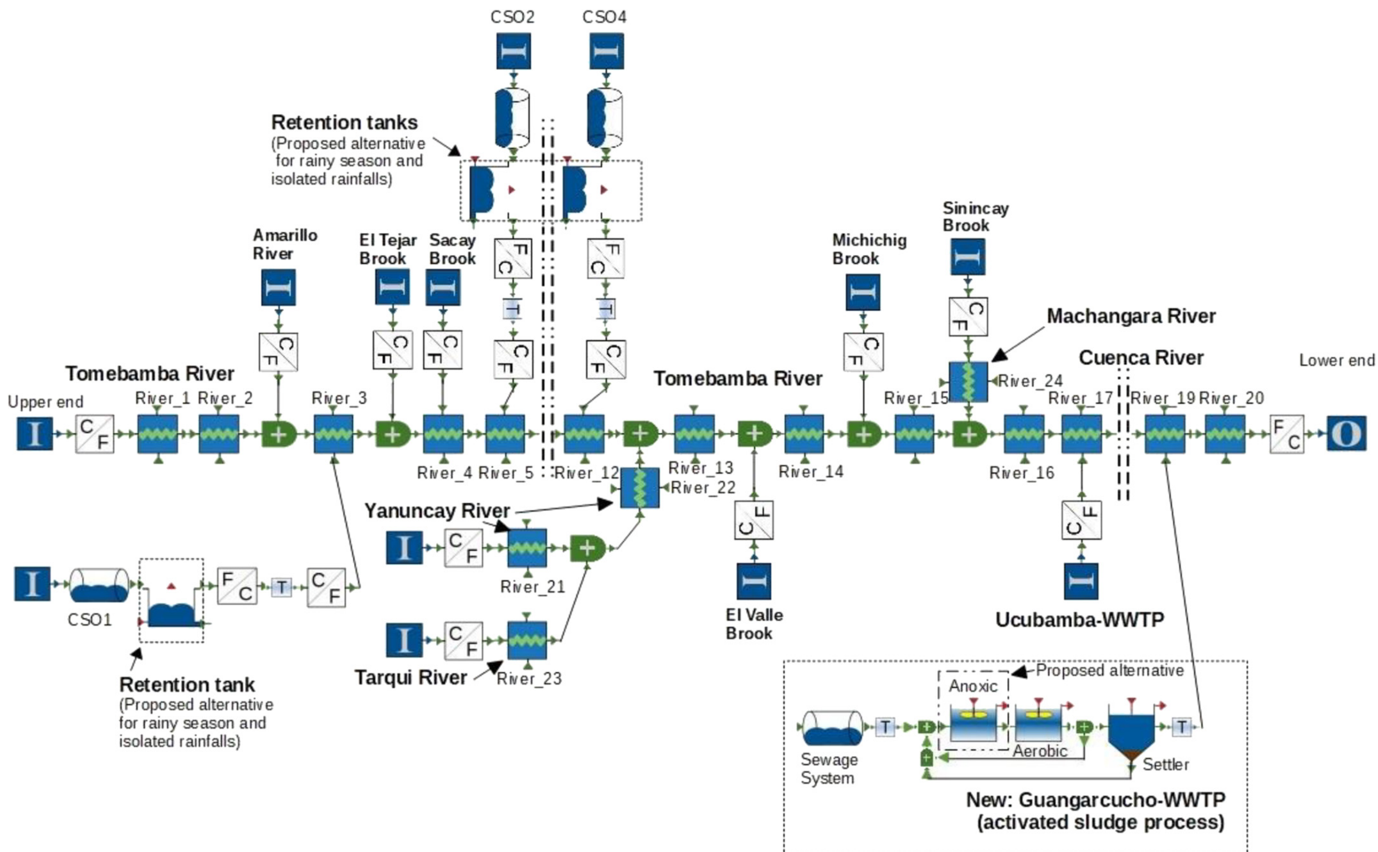

Fig. 3. Scheme of the water quality model of the actual conditions and proposed scenarios (inside dotted rectangles) in WEST®: during dry and rainy seasons.

EP, 2018). In other river stretches, we used the flow and the cross-river sections obtained in the two sampling campaigns of 2015 and 2016 as noted in Section 2.2. Additionally, the flow information and the cross-river section were completed with measurements achieved in four sites on nine occasions during 2013. This information was included in the calibration because the morphology of the rivers has not experienced significant changes from 2013 to 2017. The slope and the cross section from site To12 to To19 in the Tomebamba River (Fig. 2) were obtained from the existing topography (PROMAS-UCuenca, 2005). For the other stretches of the Tomebamba, Yanuncay, Tarqui and Machangara Rivers, the slope was calculated using the height of each site.

To calibrate and to assess the uncertainty of the river water quality model, we used the Monte Carlo analysis, in which each set of parameters that were part of the RWQM1 and that could influence the different variables (DO, COD, BOD_5, NH_4, NO_2, NO_3 and PO_4) was evaluated. The results of this analysis also generated confidence bands for the model, using the concepts of the Generalized Likelihood Uncertainty Estimation (GLUE) methodology (Beven and Binley, 1992). For this calibration, a local sensitivity analysis was first conducted (Saltelli et al., 2004). This analysis was performed in three different stretches located upstream from the Tomebamba River in the urban area. For each stretch, the five parameters selected that influenced each of the variables were those whose mean central relative sensitivity (MeanCRS) had the highest absolute values. If the three analyzed stretches had different parameters per variable, in total a maximum of 15 parameters per variable was selected from the local sensitivity analysis. Next, those chosen parameters per variable underwent a global sensitivity analysis, which was also applied to the three upstream stretches. From the latter analysis, the five parameters whose standardized regression coefficient (SRC) had the highest absolute value were selected for the scenario analysis. Thereafter, in the scenario analysis, these five parameters were evaluated from approximately 1000 possible combinations uniformly distributed in specific ranges, choosing the values of the parameters that gave the best goodness-of-fit for each variable (Table A3). For this selection, in the first place, we applied a visual inspection to evaluate the quality of the model. Finally, the minimum root mean square error (RMSE) of each variable and the set of variables in the analysis was calculated in addition to the square of the Pearson's Product Moment Correlation Coefficient (R^2) and the values of chi-squared (χ^2) (Mac Berthouex and Brown, 2002). We calibrated the following physicochemical variables: DO, COD, BOD_5, ammonium, nitrite, nitrate, phosphate, average water depth and flow velocity. The validation was accomplished in three stretches located downstream in the Cuenca River. The calibration and validation were done during both the dry and rainy seasons.

2.3.2. Wastewater treatment plants

The U-WWTP (Fig. 2), which has been in operation since 1999, processes an average flow of 1.8 $m^3 \cdot s^{-1}$. The U-WWTP is divided into two identical flow lines, each comprising a line of an aerobic pond, a facultative lagoon and a maturation pond (Alvarado et al., 2012). The discharge from the U-WWTP was also included in the integrated model (Fig. 3) to understand its impact on the river water quality, which involved the average flow and concentration of pollutants during the dry and rainy seasons of 2015 and 2016; values that were calculated from the weekly records provided by ETAPA-EP.

A new wastewater treatment plant will be built in the Guangarcucho (G-WWTP) region, located downstream from Cuenca. This new G-WWTP will increase the current capacity of the wastewater treatment, and will include the flow from the suburban areas that will be connected into the urban sewage system. The G-WWTP was designed for carbon removal only with activated sludge technology and it was included in the integrated model in WEST® (Fig. 3). The steady-state simulation of the G-WWTP was performed on the basis of the flow

($1.2\ m^3 \cdot s^{-1}$) and pollutant concentrations, as well as kinetic parameters provided by ETAPA-EP, which were used in its design (Greeley and Hansen and ACSAM, 2017). For the dynamic simulation of the G-WWTP, we calculated the hourly and daily variation of the flow and concentration of pollutants on the basis of the information obtained in the existing U-WWTP (Durazno, 2013). As a model scenario, the G-WWTP was upgraded (Fig. 3) in order to include ammonium removal. The processes of the G-WWTP were simulated with the Activated Sludge Models No. 1 (ASM1) that includes carbon oxidation in aerobic and anoxic conditions as well as nitrification and denitrification (Henze et al., 2000). The upgraded G-WWTP was analyzed in both steady and dynamic states with the same flow and charges used for its design.

2.3.3. Discharges from the combined sewer overflow

The integrated model included four discharges points from the CSOs, which represents 40 existing points along the banks of the Tomebamba River. The variation of the pollution concentration and flow in the discharges of the CSOs were calculated with information collected in three overflows located along the Tomebamba River, during nine rainfall events in the rainy seasons of 2017 and 2018 (Montalvo et al., 2018). Similarly, we calculated the flow of the CSOs according to their contribution area, the surface runoff coefficient obtained for the city of Cuenca by Rubio et al. (2017) and the precipitation from rainfall events. The information on these rainfalls was obtained from one hydrological station located in the contribution area of the three CSOs and verified with the nearest hydrological station (Fig. 2). Three simulations were obtained from rainfall events with their variation on flow and pollutants in the discharges of the CSOs.

In the scenario analysis, we included four retention tanks before the discharges of the CSOs into the Tomebamba River. These tanks reduce both the amount of water spilled from the CSOs, and the peak of pollutant concentrations that affect the water quality in the rivers. Furthermore, the retention tanks allow for the storage of the pollutant water from the CSOs during rainfall and conduction to the WWTP after the storms (Marsalek et al., 2014). The volume of the retention tanks was calculated with a runoff from a 10-year return (EPA, 1999), one-hour storm and obtained with the rational method for areas smaller than 162 ha. This method assumes a uniform rainfall distribution over the tributary area during the duration of this event (NYC Environmental Protection, 2012). The intensity-duration-frequency (IDF) curves used to predict the rainfall of a 10-year return period, which is used in the design of the combined sewer system of the City of Cuenca, was obtained from Estrella and Tobar (2013) and Estrella and Tobar (1994). With a runoff from a 10-year, one-hour storm and a detention time of 40 min, which was calculated with the discharged average flow of a CSO measured hydrogram, the designed retention tanks gave an average storage of $20\ m^3$/ha, a value that was in the range of typical runoff storage in Germany (Zabel et al., 2001).

2.3.4. Ecological model

Ecological models were developed to determine association parameters, which could influence the performance of the Andean Biotic Index (ABI) in the study area. The data measured during the two sampling campaigns of 2015 and 2016 was fitted by a generalized linear model (GLM) since this modelling technique can handle the non-linear behavior of the ecosystem (Guisan et al., 2006; Zuur et al., 2009; Forio et al., 2018). Gaussian, Gamma and inverse Gaussian distributions were considered as the most appropriate distributions for a positive and continuous response variable (McCullagh, 1984; Zuur et al., 2009) such as the ABI. The details of the methodology to develop the ecological models can be consulted in earlier research developed by Jerves-Cobo et al. (2020).

The three most reliable ecological models to predict the ABI class per season were chosen by Jerves-Cobo et al. (2020). The best performing GLMs during the dry and rainy season were selected according to their Cohen's Kappa (κ), correctly classified instances (CCI) and pseudo R^2. Thus, to improve the prediction in the dry season, two models were selected: one model was applied in streams with low concentrations of pollutants, while a second model was used in stretches with higher pollution concentrations. For the rainy season, a season-specific model was applied in the Tomebamba and Cuenca Rivers.

2.3.5. Scenario analysis for restoration of the ecological water quality

In order to understand how the new G-WWTP and other actions could improve the ecological water quality in different stretches of the Tomebamba and Cuenca Rivers, it simulated four different scenarios (Fig. 3) with the calibrated RWQM during the dry and rainy seasons, which are summarized in Table 1. The implementation of the G-WWTP implies that the capacity to handle wastewater treatment will be increased. Consequently, the pollution from suburban areas that is currently discharging into streams could then be connected into the sewage system and conducted to the wastewater treatment system. The total population that could be connected to the G-WWTP was taken from Greeley and Hansen and ACSAM (2017). However, this population was distributed along suburban and rural areas where discharges are occurring directly into streams.

The modeled values of organic pollutants and nutrients such as DO, nitrate, nitrite, phosphorous, BOD_5, turbidity and total ammonium nitrogen were compared with the thresholds (Table A4) given by MAE-Ecuador (2015) to control freshwater aquatic ecosystems.

3. Results

3.1. River water quality model

The river water quality model was calibrated and validated during both seasons for hydraulic and chemical variables, displaying reliable predictions. The hydraulic calibration and validation showed a high determination coefficient ($R^2 > 0.7$) for water depth and flow velocity (Table A5). The physicochemical variables presented a different goodness-of-fit in the calibration and validation processes during both seasons. Fig. 4 and Fig. A3 present the graphs of the calibration and validation for the different variables (DO, BOD_5, COD, ammonium, nitrite, nitrate and orthophosphates) along the Tomebamba and Cuenca Rivers during the dry and rainy season, respectively, along with their confidence bands of the 5th and 95th predicted percentiles, obtained from the Monte Carlo analysis. Similarly, Table A5 presents the values of R^2,

Table 1
Scenarios to recover the ecological water quality in the Tomebamba and Cuenca Rivers.

Scenario	Season	Actions
Sc-1	Dry season	Implementation of the new G-WWTP (carbon removal).
Sc-2	Dry season	Implementation of the upgraded G-WWTP (carbon and nutrients removal).
Sc-3	Rainy season	Implementation of the new G-WWTP (carbon removal).
Sc-4	Rainy season	Implementation of the upgraded G-WWTP (carbon and nutrients removal).
Sc-1 to Sc-4	Dry and rainy seasons	Additional actions to be included in Sc-1 to Sc-4: - Reduction in the concentration of pollutants in 80% of small streams and brooks: Amarillo River and El Tejar, Saucay, El Valle, Milchichig and Sinincay Brooks, due to the connection of the direct discharges with the urban sewage system. - Reduction in the concentration of pollutants in 50% of the main effluents: Yanuncay, Tarqui and Machangara Rivers, due to the connection of the direct discharges with the urban sewage system.
Sc-3 & Sc-4	Rainy season	Additional actions to be included in Sc-3 and Sc-4: - Implementation of four retention tanks before CSO discharges.

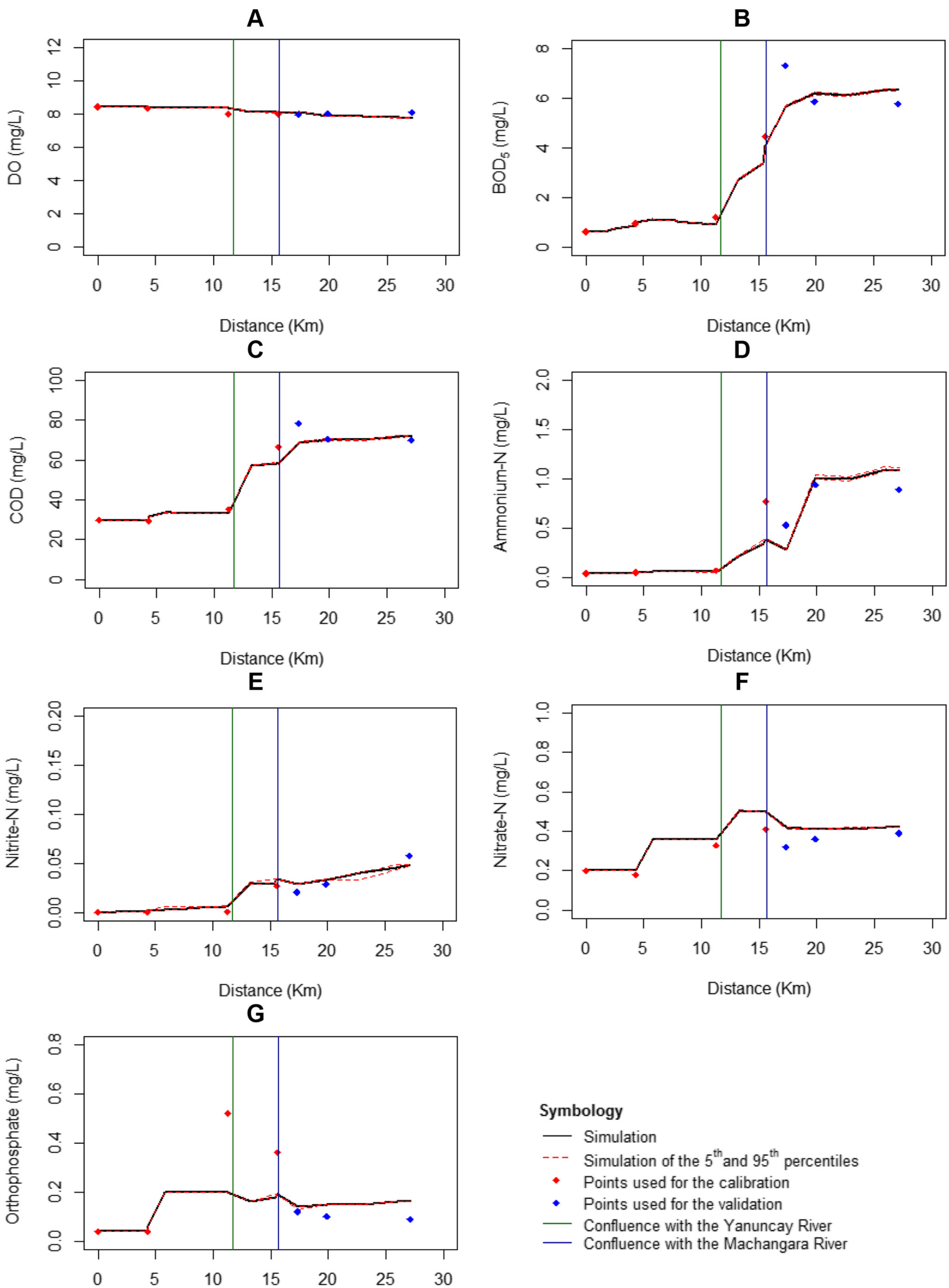

Fig. 4. Calibrated water quality model in the Tomebamba and Cuenca Rivers during the dry season for: (**A**) DO, (**B**) BOD_5, (**C**) COD, (**D**) Ammonium, (**E**) Nitrite, (**F**) Nitrate, (**G**) Orthophosphate.

RMSE and χ^2 obtained with physicochemical variables in both seasons during the calibration and validation processes.

For the calibration during the dry season, the BOD_5, COD, ammonium, nitrite, nitrate and orthophosphate had values higher than 0.7, while the DO showed a lower value ($R^2 < 0.45$). For the validation, nitrate was the only variable with low values ($R^2 < 0.45$), the other variables had values higher than 0.7. Similarly, for the calibration during the rainy season, BOD_5, ammonium, and nitrate had an $R^2 > 0.7$, while nitrite presented a moderate determination coefficient and the DO, COD and orthophosphates had low values ($R^2 < 0.45$). The validation in this season displayed different results than those that were obtained during the dry season. Thus, BOD_5, nitrate and nitrite had values higher than 0.7, while ammonium and orthophosphate indicated moderate values ($0.45 \le R^2 \le 0.7$) and the other variables displayed low determination coefficients.

3.2. Ecological assessment model

The chosen ecological models from Jerves-Cobo et al. (2020) are presented in Table 2. As a result, the ABI class had different predictors per season. Thus, for the dry season, the ABI had three predictors: BOD_5, NH_4 and PO_4, while for the rainy season the six variables that influenced the ABI were: NO_2, NO_3, DO, oxygen saturation, TSol and bank material. The GLMs chosen belong to the Gamma and Gaussian families. When these two chosen GLMs were combined and selectively applied according to their season to the Tomebamba and Cuenca Rivers, the CCI and κ had values higher than 70% and 0.4, respectively (Table 2).

3.3. Integrated ecological model and scenario assessment

Actions to be considered in order to improve biological water quality involve reducing the concentration of organic pollutants in the receiving water body. Consequently, we evaluated the fulfillment of the Ecuadorian regulation to preserve the aquatic ecosystem after the implementation of different scenarios. Thus, in Sc-1 and Sc-2 constructed for dry season and in Sc-3 and Sc-4 developed for rainy season (Table 1), BOD_5, ammonium, nitrite and nitrate would remain under their thresholds (Fig. A4) in the Tomebamba and Cuenca Rivers. Furthermore, with the proposed measurements in Sc-1, Sc-2, Sc-3 and Sc-4, the Tomebamba and Cuenca Rivers will have an important decrease in the concentration of the analyzed pollutants: BOD_5, ammonium, nitrite, nitrate and orthophosphate, than in the current registered concentrations (Fig. A4). However, in Sc-1 and Sc-2 in the Tarqui River, its level of BOD_5 would remain over the regulated thresholds during dry season, despite the fact that this river included a 50% reduction in pollutants.

The scenarios analyzed for the restoration of the ecological water quality have positive impacts in different stretches of the Tomebamba and Cuenca Rivers and their tributaries as can be seen in Fig. 5. Thus, with the implementation of the new G-WWTP, the capacity of the wastewater treatment will be increased, connecting the suburban areas of the city to the urban sewage system, and eliminating the direct sewage discharges into the waterways. With these measures in the dry season, the Tomebamba and Cuenca Rivers (Sc-1) and their streams would ensure the maintenance of the ABI class in the moderate category upstream from the city of Cuenca until the discharge of the current U-WWTP. From this point forward, the ABI class would remain in the deficient range. However, Sc-2 shows that by removing nutrients in the new G-WWTP, the ABI class would remain deficient, but with values closer to the moderate class. During the rainy season, in which the Tomebamba and Cuenca Rivers and their tributaries would have better water quality than during the dry season, the measures included in the Sc-3 such as retention tanks before the CSOs, would also show improvements in the biological water quality. The good ecological water quality of the Tomebamba River would continue until its confluence with the Machangara River. Additionally, the good water quality would be

Table 2
The best-performing season-specific models and their selective combination (Selection from Jerves-Cobo et al. (2020)).

Explanatory variables	Regression parameters	Season-specific models			Selective combination of season-specific models	
		Dry-season		Rainy-season	Combination of the rainy season + dry season models:	
		Gamma models:		Gaussian model:		
		mD1.3fcv2a3	mD8_2.3fcv1a2	mR8.gaussian	mR8.gaussian +mD1.3fcv1a3	mR8.gaussian +mD8.3fcv1a2
		Coefficient[a]	Coefficient[a]	Coefficient[a]		
	A	1.3E-02	1.3E-02	−725.5		
Nitrate	B1		3.5E-02	85.3		
Nitrite	B2			−1100.7		
Ammonium	B3	−4.2E-02	6.2E-03			
BOD_5	B4	5.2E-03	1.2 E-03			
DO	B5			−42.8		
Oxygen saturation	B6			10.9		
Orthophosphate	B8	1.4E-01				
Total solids	B10			−0.1		
Bank material	B18			10.6		
Training subset (2/3)						
AIC:		140.6	133.86	314.0		
Pseudo R^2:		0.7	0.7	0.6		
CCI:		72.2%	47.1%	68.6%		
κ:		0.6	0.3	0.5		
Validation subset (1/3)						
Pseudo R^2:		0.4	0.1	0.3		
CCI:		62.5%	22.2%	72.7%		
κ:		0.4	0.2	0.6		
Applied to the Tomebamba and Cuenca Rivers datasets						
CCI		57.1%	28.6	85.7%	76.2%	66.7%
κ:		0.4	0.1	0.7	0.7	0.5

[a] The coefficient is multiplied by its variable respectively within the GLM equation.

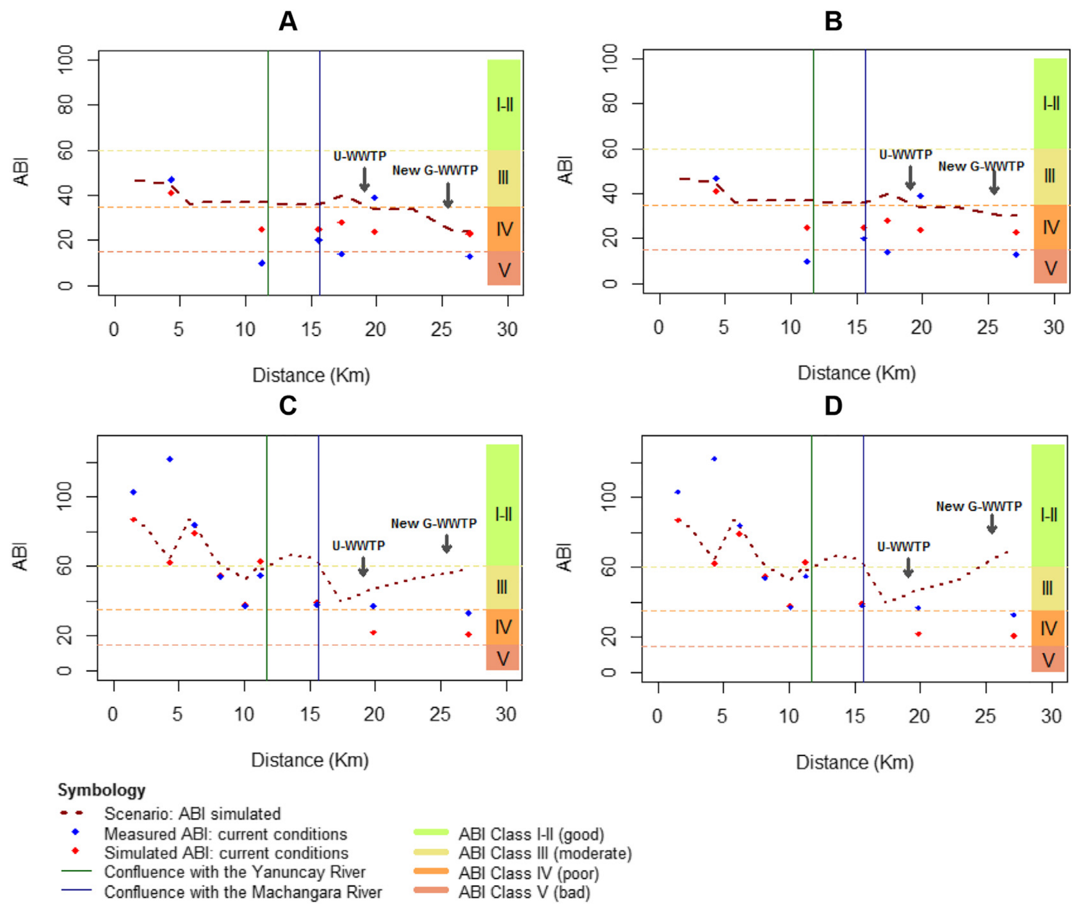

Fig. 5. Scenario analysis concerning restoration needs and expected outcomes: (A) scenario 1 (Sc-1) – dry season, (B) scenario 2 (Sc-2) – dry season, (C) Sc-3 (rainy season) and (D) Sc-4 (rainy season). The difference between Sc-1 and Sc-2 and between Sc-3 and Sc-4, is the technology used in the WWTP-G, thus Sc-1 and SC-3 are only with carbon removal, while for Sc-2 and Sc-4 they are with carbon and nutrients removal.

restored in the last three kilometers of the Cuenca River in the Sc-4. The upgrade to the new G-WWTP to include carbon and nutrients removal would positively influence the ABI class during both the dry and rainy seasons.

The retention tanks before combined sewage overflows (CSOs - Sc-3 and Sc-4) were tested during three rain events, from which one had a runoff from the 10-year, one-hour storm. These tanks showed effectiveness in diminishing pollutants discharged into the Tomebamba River by at least 50%, with the possibility of reduction up to 100% as long as the polluted storage water is conducted to the WWTPs following precipitation.

The analysis of possible technologies to be applied to the new G-WWTP demonstrated that the carbon and nutrient removal technology (Sc-2 and Sc-4) would contribute more effectively to the restoration of the Cuenca River's water quality than with the carbon removal technology (Sc-1 and Sc-3). Thus, for example, under steady-state conditions, the upgraded technology (carbon and nutrients removal) would be more efficient in approximately 70% of the removal of ammonium and approximately 60% in the elimination of nitrate, total nitrogen and total Kjeldahl nitrogen (TKN). In order to obtain the nitrogen species removal, the new G-WWTP must include an anoxic zone in its biological reactors reducing its flow capacity. An analysis of the solids retention

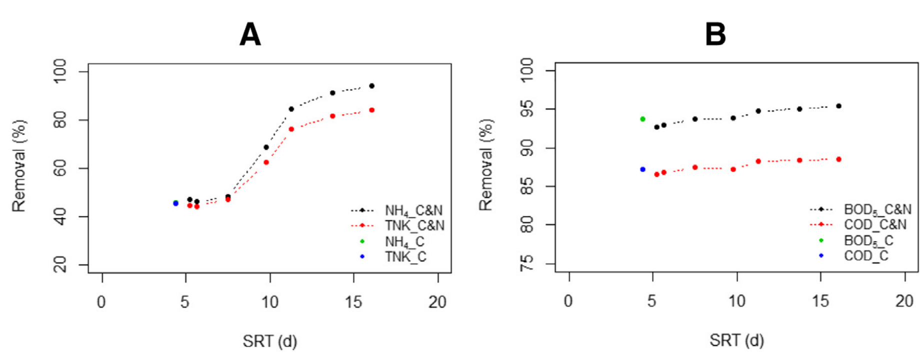

Fig. 6. Analysis of removal for the new G-WWTP according of solids retention time (SRT) with the technologies of carbon (C) and carbon and nitrogen elimination (C&N) for: (**A**) nitrogen family, and (**B**) carbon family.

time (SRT) and hydraulic retention time (HRT) in comparison with the nitrogen and carbon families' removal was applied to the new G-WWTP in Fig. 6 and Fig. A5, respectively. In these graphs, it can be noted that with an SRT of 11 days and an HRT of 0.45 days, increased removal of ammonium and TKN was obtained, while higher values of the SRT or HRT did not influence the elimination of BOD_5 and COD.

4. Discussion

4.1. River water quality model, performance, uncertainty and validation

The primary goal for the calibration of the river water quality model was to have a base with enough accuracy that reflects the behavior of the rivers and a model that can be used as a planning tool to simulate scenario analysis to improve current water quality conditions. The model was calibrated under steady-state analysis during the dry and rainy seasons. This kind of calibration was accomplished with the data, which was taken in both sampling campaigns of 2015 and 2016, but did not consider the variation of the water quality at the sampled sites during a specific day or week. During the calibration process, the parameters set for RWQM1 were adjusted to minimize the difference between measurements and model predictions. Thus, the variables such as COD, BOD_5, NH_4, NO_3 and PO_4 presented acceptable goodness-of-fit with the default values of parameters set on the RWQM1 (Reichert et al., 2001). Only significant parameters were adjusted for this calibration: the re-aeration coefficient (Kla_{base}). This parameter is part of the re-aeration rate and was used to calibrate DO concentration during both seasons resulting in different values for each season. The re-aeration rate is also related to stream velocity, temperature and water depth; variables that registered a relevant change between seasons and likely influenced the different value of the Kla_{base} obtained in the calibration of the DO. The BOD_5 and COD presented acceptable accuracy in both calibrations developed for the dry and rainy seasons. However, the constant relation between these variables ($BOD_5/COD = 0.35$), included in the RWQM1, was changed for each river stretch according to the registered values. This ratio variation range between the COD/BOD_5 in the rivers could have been due to the difference in carbon sources, such as municipal versus industrial pollution in urban areas, or in the rural areas, organic matter from the moorland areas with a decreased biodegradability.

The range of errors probably for both the calibration and validation depend on the variables. Thus, according to the R^2 values obtained in the calibration processes, the DO was low for the dry and rainy seasons, but its RMSE value was very low during both seasons. Similarly, the R^2 of COD during the rainy season was in the low range in the calibration and validation processes, but its RMSE was lower than other variables that presented a higher R^2. Orthophosphates also demonstrated a low value of R^2 and the highest RMSE in its calibration process given during the rainy season. The difference of R^2 between variables and seasons could also be influenced by the limited data available. Furthermore, according to the GLUE technique, the results of the model calibration indicated a good prediction capacity, showing that variables were mainly in the range of the 95% confidence bands. Although the R^2 could likely be increased, changing the values of other parameters could make the model over-parameterized due to the limited dataset and given adequate goodness-of-fit with a different set of calibration parameters (Brun et al., 2001). For future research in which a dynamic calibration may be developed, it is recommended that various sampling sites should be sampled simultaneously with continuous hourly data along a defined period of time. This new information could help to improve the values of the calibration obtained in this research.

The new G-WWTP, which will be constructed in the near future, was not calibrated, because it was included in the integrated model with the same parameters used in its design. This new G-WWTP was designed with the ASM1 using the software GPS-X (Greeley and Hansen and ACSAM, 2017). It is recommended to calibrate the ASM1 used in the integrated model with information that will be collected from the new G-WWTP, when this enters into production. This calibration would allow for the analysis of possible scenarios to improve the performance of the G-WWTP and supply information to the integrated model constructed in this study. Similarly, the operation of retention tanks was analyzed with data from three of nine rainfall events collected, from which one had a return period of 10-year. However, for future analysis, in which a different return period and rainy duration could be included, it is suggested to calibrate the flow caused by the rainfall events in conjunction with their registered pollution.

4.2. Ecological model, performance, uncertainty and validation

To predict the biological water quality (ABI) obtained under the four scenarios proposed, we needed to link the information resulting from the RWQM1 given by WEST®, with three season-specific-ecological models developed by Jerves-Cobo et al. (2020). The implementation of two ecological models in dry season along the Tomebamba and Cuenca Rivers revealed better accuracy than with the application of only one model. The first model had better precision in its application with a low concentration of pollutants — the Tomebamba River until its confluence with the Yanuncay — while the second model performed better in the others stretches and streams which registered higher pollution. The chosen season-specific models were combined and selectively applied according to their season to the Tomebamba and Cuenca Rivers, the accuracy of the combined model was good according to Goethals (2005), with a κ and CCI higher than 0.4 and close or higher than 70%, respectively. The variables of season-specific models changed according to their concentration measured per season; thus, a weaker correlation between biotic indices and organic pollutants was obtained during the rainy season (Jacobsen, 1998; Jacobsen and Encalada, 1998; Jerves-Cobo et al., 2020). The models applied in this study have demonstrated that they have relevant ecological patterns. However, the results obtained with ecological models applied in this study will give a good prediction as long as the range of the variables is the same as those used in the construction of the models (McCullagh and Nelders, 1989).

The DO, organic pollutant expressed as BOD_5 and nutrients, which are variables presented in the chosen ecological models, are widely acknowledged to have influence on the presence/absence of different families of macroinvertebrates (Jacobsen et al., 2003; De Pauw et al., 2006). Total solids (TSol) and bank material, were two variables that were part of the ecological model, but that were not predicted by the RWQM1. The concentration of the TSol was calculated with a simple mass balance, in which the possible sedimentation in the bed of the rivers, the increment by erosion and the loss for reactions were omitted. For the bank material, the classification registered in the survey that was applied to collect the information (Jerves-Cobo et al., 2018b) was included in the application of the ecological model. This classification included bedrock, boulder, cobble, pebble, gravel, sand, silt and clay.

4.3. Integrated ecological modelling and scenario analysis

Two kinds of techniques can be handled in the integration of the models for river management: the integration and the combination approaches (Lam et al., 2004). The integration approach was used to join the river model (RWQM1) with the WWTPs (ASM1) and with the sewage systems (Reichert et al., 2001). This integration allowed for the prediction of the water quality compounds, the flow velocity and the depth into receiving rivers. As such, this approach has been addressed with acceptable accuracy in several studies (Deksissa et al., 2004; Benedetti et al., 2007). Similarly, the combination approach was applied to link the results from the RWQM1 with ecological models obtained in the dry and rainy seasons. This combination was implemented to simulate both the current conditions and to analyze scenarios to restore the environmental state of the Cuenca River and its tributaries.

The implementation of different scenarios for the restoration of the water quality in the Cuenca River system demonstrated their effectiveness in the analysis presented in this research. Thus, the implementation of the new G-WWTP will bring improvements to the biological water quality in Cuenca's River system, collecting wastewater from households located in the suburban area of the city of Cuenca. Additionally, with the new G-WWTP, the current U-WWTP will not overload allowing for improvement to the quality of its effluent. However, with the measures analyzed during the dry season, it was possible to obtain a moderate water quality in the Tomebamba River until its confluence with the Yanuncay River and in the Cuenca River from the U-WWTP onwards. The water quality of the Yanuncay River is affected by the Tarqui River, which also has an impact on the Tomebamba River. In the same way, the Machangara influences the water quality conditions of the Cuenca River. The primary cause of degradation of the Tarqui River is due to the diffuse fluxes of organic pollution from an extensive livestock area (Beltrán et al., 2013). For the restoration of the biological water quality in livestock areas located close to the Tarqui River or other rivers in the Cuenca River basin, a future study could analyze measures to be applied. However, measures such as the implementation of buffer strips in each bank of the Tarqui River and its tributaries, as well as other watercourses, could be applied to reduce the organic and nutrients pollution from livestock areas that reach the streams and affect their water quality (Mouton et al., 2009). Regarding the Machangara River, point discharges from industries must be controlled to enhance the water quality in this tributary and subsequently in the Cuenca River.

The technologies of carbon removal or carbon and nutrients removal that could be implemented in the new G-WWTP will comply with the regulation for discharges to water bodies for nutrients, BOD_5, and COD (MAE-Ecuador, 2015). However, the water quality in the Cuenca River during dry season would not improve to moderate, due to the effluent from the current U-WWTP, which has a concentration of ammonium that varied between 10 and 20 $mg.L^{-1}$ and its BOD_5 is around 30 $mg.L^{-1}$ (ETAPA-EP, 2017). The implementation of a new G-WWTP with carbon and nitrogen removal technology (Sc-2) rather than with only carbon removal technology (Sc-1) would elevate the biological water quality, but its class would remain deficient in the last analyzed stretch of the Cuenca River during the dry season. This improvement in the biological water quality obtained with a WWTP with carbon and nitrogen removal technology (Sc-2) will allow a lower concentration of the ammonium and TKN to enter the Cuenca River. Regarding the activated sludge with extended aeration plants, a technology used in the design of the new WWTP (Greeley and Hansen and ACSAM, 2017), McCarty and Brodersen (1962) indicated that the major problem with this technology is the lack of an appropriate nitrification process, which causes poor ammonium removal. Moreover, if the new G-WWTP could operate in anoxic and aerated conditions to remove ammonium, its limitation would be the SRT as well as the HRT. These factors must be increased to guarantee the nitrogen removal, which would diminish the flow to be processed, an aspect that could likely be managed only within the first few years of plant operation. The new G-WWTP could also be enhanced with the implementation of an anaerobic zone for improving biological phosphorous removal. This analysis of the G-WWTP could be done under the Activated Sludge Model No. 2d (ASM2d) (Henze et al., 2000). Although orthophosphates, a part of total phosphorous, is a component of the ecological model, this pollutant was not included in our study. This was because, with the removal of orthophosphate achieved by the new G-WWTP under Sc-1 and Sc-2, the concentration of orthophosphates in the Cuenca River was increased in 0.01 $mg\ PO_4 \cdot L^{-1}$ by their discharge of the G-WWTP (Fig. A4). Furthermore, if the removal of orthophosphate in the G-WWTP could be increased to 90%, the effluent of this WWTP would not change the concentration of this pollutant in the Cuenca River. Consequently, the GLM used to calculate the ABI class in dry season was not sensitive to a slight variation of orthophosphates.

The scenarios analyzed to improve the biological water quality during the rainy season (Sc-3 and Sc-4) showed that the upgrade of the new G-WWTP from carbon removal to carbon and nitrogen removal, would have an influence in the improvement of the ecological water quality in the last studied stretch of the Cuenca River. In fact, in Sc-4, the ABI would rise to good, although its value would not significantly change in comparison with the ABI obtained in Sc-3. This was likely due to lower concentrations of pollutants as a consequence of both a higher dilution addressed by a higher flow in the river that reduced the effects of the discharge from the WWTP as well as the higher velocity of the water that generated a washing out of the organic material accumulated in the substrate (Jacobsen, 1998; Burneo and Gunkel, 2003).

For rainy season, an analysis of the addition of the retention tanks (Sc-3 and Sc-4) before the discharges of the CSOs was included. Thus, from the CSOs, at the beginning of a rainfall event, high peak concentrations of dissolved compounds are generally discharged into rivers. As a result, the aquatic ecosystem of the receiving water is affected by a short-term and delayed impact in the DO depletion, non-ionized ammonium and shear stress as well as extend periods of anoxic sediment in the rivers bed (Hvitved-Jacobsen, 1982; Borchardt and Sperling, 1997; Reichert et al., 2001). The retention tanks as shown in Section 3.3 would undoubtedly increase the biological water quality and diminish the impact of the first flush as well as the amount of dissolved pollutants discharged directly to the rivers. In addition, these structures would help to reduce the quantity of sediment that is discharged into the rivers. According to the EPA (1999), retention tanks must capture at least 85% of the wet weather volume. In 2017 – the year in which the CSO discharges were measured – registered 9% more rain than the average from years 1977 to 2011 and 3% and 17.5% >2016 and 2015, respectively. With the designed retention tanks, which were the lowest storage sized recommended in Germany, during four rainfalls registered between March and May of 2017, the capture volume was lower than the recommended threshold – a minimum detained volume of 50%. However, the total capture volume during this aforementioned period complied with the recommendation. Consequently, to improve the water quality in the urban area of the Cuenca River system, the inclusion of retention tanks before the discharges of CSOs is recommended. It is also suggested to test the proposed retention tanks during stronger rainfall events than were analyzed in this study and to estimate the capture volume during an entire year.

Linking the results from the IUWS model with the ecological models was done in order to obtain an integrated ecological model (IEM). With this IEM, the improvement to the ecological water quality of the Cuenca River and its tributaries can be understood under the analyzed scenarios. Both the IUWS model and the ecological models present uncertainty. This uncertainty increased when both models were linked. However, this new uncertainty was not calculated, which implies that the accuracy of the IEM model will be lower than the IUWS model and the ecological models. Despite this, the IEM model, provided an understanding of the positive impact of those measures that could enhance the ecological water quality of the study area. The results obtained in this study will enable stakeholders such as ETAPA-EP, to gain insight on how the proposed measures are interrelated with the ecological status of the urban and suburban areas of Cuenca River and its tributaries. Further analysis to enhance the water quality especially in the branches of the Cuenca River located upstream from the city, can be developed. In this analysis, the DPSIR (drivers, pressures, state, impact and response) framework could be adopted, a concept that was used in this manuscript and included in this section, such as recommendations to be applied into livestock areas. Comparable applications to analyze restoration scenarios, in which a river water-quality model was linked with ecological models, have been developed with good performance by Mouton et al. (2009) and Holguin-Gonzalez et al. (2013b).The IEM could be replicated in other basins around the world. However, to develop an ecological model is recommend that the sample size be higher than 50 samples per season to obtain an accuracy close to the maximum, and the minimum quantity be at least 10 to obtain an acceptable accuracy (Stockwell and Peterson, 2002). Accuracy can change based on the chosen technique employed to develop the ecological models. With regard to the IUWM models, in this research, good accuracy was obtained using a minimum dataset of 27 samples developed during the dry season. However, to improve the accuracy, it is recommended that a dynamic data set be taken as was done in seven sites in Crocodile River basin located in South Africa (Deksissa et al., 2004). Additionally, another important factor for the accuracy of the IUWM model is the number of the tanks in series that is used to develop the model.

5. Conclusion

The four scenarios applied to restore the ecological water quality in the Cuenca River system, which was measured with the ABI, demonstrated that the all proposed measures would help to enhance the water quality in the urbanized area of this river system. With the implementation of the new G-WWTP, the benefits in the improvement of the ABI would be most significant during the dry season, with either carbon removal or carbon and nitrogen removal technologies. However, the carbon and nitrogen removal technology would bring a higher restoration of the ecological status of the Cuenca River due to the increased nitrogen removal during both seasons. The retention tanks before the discharges of CSOs would also improve the ecological water quality during rainy seasons. The scenarios analyzed in this research would enable stakeholders to gain insight prior to implementing measures to improve the water quality conditions of the Cuenca River system. Finally, similar applications with the construction of an integrated ecological model could be replicated in other basins to analyze the impact of different measures on their ecological water quality.

Declaration of competing interests

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.

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Appendix A

Table A1
Summary of the physical, chemical and microbiological data collected in the study area based on 43 samples in 2015 and 2016 (Adapted from Jerves-Cobo et al. (2018b)). The mean and median values per seasons are also presented.

Parameter	Units	Both season						Dry season		Rainy season	
		Mean value		Standard deviation	Min value	Max value	Median value	Mean	Median	Mean	Median
Temperature	°C	13.9	±	2.0	10.4	19.8	13.9	13.2	12.6	14.4	14.3
Specific conductivity	$\mu S \cdot cm^{-1}$	135.7	±	194.4	29.0	1396.5	90.9	177.6	90.9	103.4	91.0
pH		7.5	±	0.4	6.6	8.8	7.6	7.7	7.6	7.5	7.5
Turbidity	NTU	27.3	±	33.4	0.8	187.0	11.8	30.5	12.0	24.8	11.4
Chlorophyll-a	$\mu g \cdot L^{-1}$	9.7	±	7.3	2.4	29.3	6.9	9.7	6.9	–	
Dissolved oxygen (DO)	$mg \cdot L^{-1}$	7.5	±	1.3	0.7	8.5	7.7	7.3	8.0	7.6	7.7
Oxygen saturation (OS)	%	97.0	±	15.1	9.6	104.4	100.7	92.3	100.4	100.6	101.1
Five-day biological oxygen demand (BOD_5)	$mg \cdot L^{-1}$	11.6	±	49.9	0.8	384.0	2.4	22.2	3.3	3.4	2.2
Chemical oxygen demand (COD)	$mg \cdot L^{-1}$	98.2	±	195.8	7.9	1036.8	53.8	98.2	53.8	–	
True color (color)	HU	58.9	±	55.0	12.0	293.0	40.5	57.2	58.0	60.3	38.0
Alkalinity	$mg \cdot L^{-1}$ $CaCO_3$	47.0	±	39.6	16.2	209.4	35.9	57.8	38.4	38.7	32.6
Phenolphthalein	$mg \cdot L^{-1}$ $CaCO_3$	0.8	±	3.2	BDL	17.4	0.0	0.6	0.0	1.0	0.0
Total hardness	$mg \cdot L^{-1}$ $CaCO_3$	55.7	±	55.6	16.6	421.2	45.2	60.8	41.6	51.8	46.8
Ca^{++}	$mg \cdot L^{-1}$	16.9	±	16.2	3.8	119.8	13.8	17.5	13.2	16.5	14.7
Mg^{++}	$mg \cdot L^{-1}$	1.5	±	2.2	BDL	9.4	0.5	0.0	0.0	2.6	1.9
Chloride	$mg \cdot L^{-1}$	10.1	±	13.6	3.2	95.3	6.1	13.3	6.5	7.7	5.9
Orthophosphate	$mg \cdot L^{-1}$	0.2	±	0.4	BDL	2.2	0.0	0.3	0.1	0.1	0.0
Total phosphorus	$mg \cdot L^{-1}$	1.2	±	1.3	BDL	5.4	0.6	0.9	0.4	1.4	1.1
Ammonium-N	$mg \cdot L^{-1}$	1.0	±	4.0	BDL	26.4	0.1	2.1	0.2	0.2	0.1
Nitrate-N	$mg \cdot L^{-1}$	0.3	±	0.2	BDL	1.7	0.3	0.3	0.2	0.3	0.3
Nitrite-N[a]	$\mu g \cdot L^{-1}$	28	±	54	BDL	365	12	37	12.6	20	11.4
Total solids	$mg \cdot L^{-1}$	155.1	±	144.9	29.0	998.0	105.5	176.4	104.0	138.7	107.0
Total coliforms	MPN.100 mL^{-1}	4.1E+05	±	3.5E + 01	1.4E + 03	4.3E + 10	1.9E + 05	8.4E + 05	9.2E + 05	2.3E + 05	1.7E + 05
	CFU.100 mL^{-1}	7.5E + 04	±	1.7E + 01	3.9E + 02	8.4E + 09	5.6E + 04	1.2E + 05	1.0E + 05	5.4E + 04	4.5E + 04
Fecal coliforms	MPN.100 mL^{-1}	1.3E + 05	±	3.1E + 01	4.9E + 02	9.2E + 09	8.6E + 04	2.0E + 05	1.7E + 05	9.4E + 04	7.0E + 04
	CFU.100 mL^{-1}	3.1E + 04	±	1.7E + 01	1.1E + 02	4.9E + 08	2.2E + 04	3.6E + 04	4.7E + 04	2.9E + 04	2.0E + 04
Mean stream width	M	13.8	±	8.9	0.9	30.5	13.2	12.6	10.3	14.6	14.2
Mean depth	M	0.6	±	0.4	0.1	1.6	0.6	0.5	0.4	0.6	0.6
Flow velocity	$m \cdot s^{-1}$	1.1	±	0.5	0.2	2.0	1.1	1.0	1.0	1.2	1.2
Flow[b]	$m^3 \cdot s^{-1}$	13.4	±	16.7	0.0	86.7	9.9	14.6	4.5	12.2	10.9

Descriptive statistics of physicochemical and microbiological variables are given as mean values ± standard deviations, minimums and maximums.
NTU = Nephelometric turbidity units; HU = Hazen units; MPN = Most probable number; CFU = Colony-forming unit.
BDL = Below Detection Limit.
[a] Nitrite-N is expressed as $\mu g \cdot L^{-1}$. For its determination the following APHA 4500-NO_2 colorimetric method with a detection value of 2 $\mu g \cdot L^{-1}$ was used.

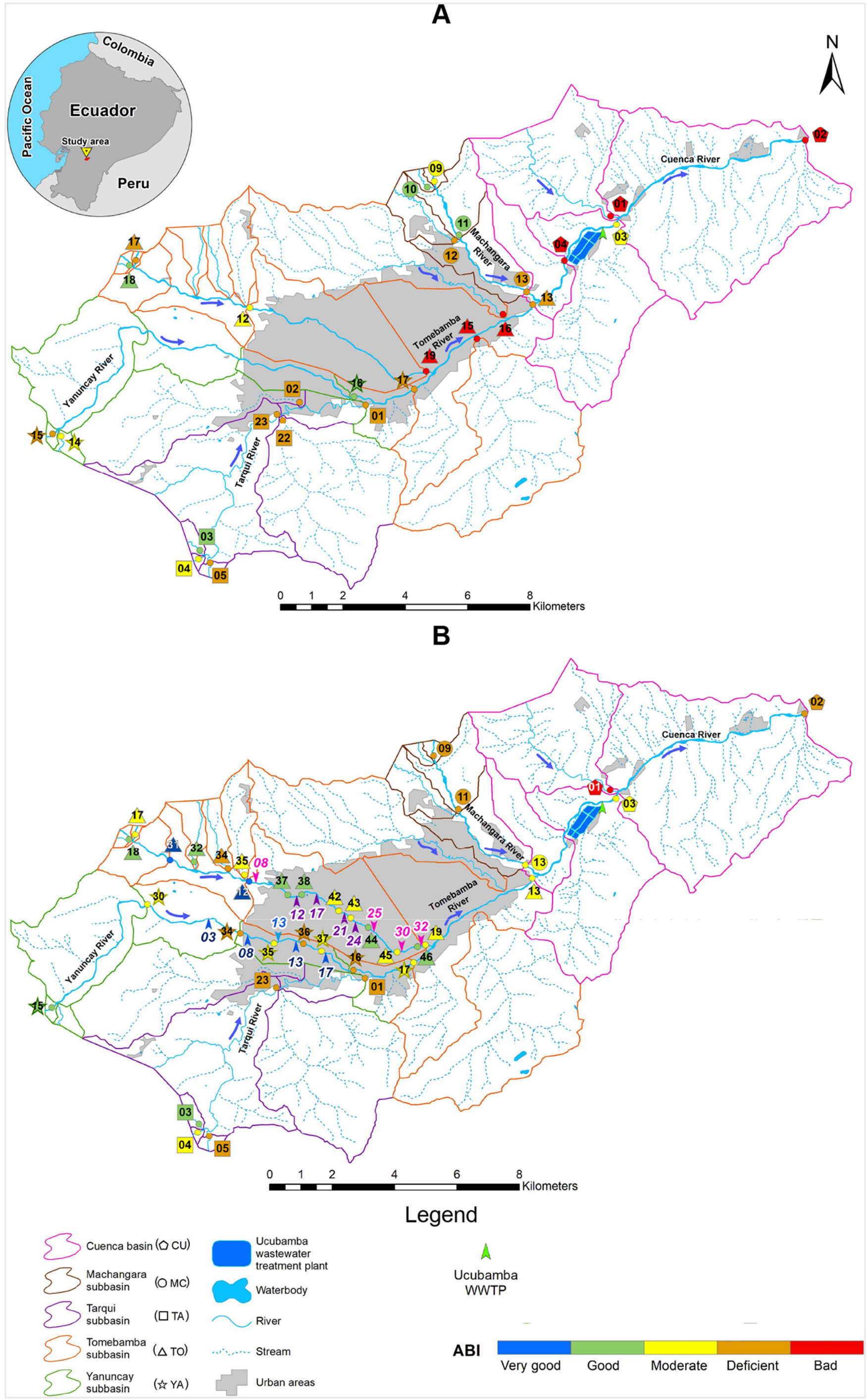

Fig. A1. Sampling sites location with their Andean Biotic Index (ABI) in both seasons – adapted from Jerves-Cobo et al. (2020): **(A)** dry season, (**B**) rainy season.

Table A2
Fractions of the COD as used in the parametrization of the river water quality model (RWQM). Values taken from Solvi (2006).

	Measured variables	RWQM1 variable	Units	Fraction	Value
COD soluble	Readily biodegradable soluble COD	SS	$gCOD\cdot m^{-3}$	0.6	
	Fraction soluble of COD that is inert	SI	$gCOD\cdot m^{-3}$	0.4	
COD particulate	Fraction particulate of COD that is organic material	XS[a]	$gCOD\cdot m^{-3}$	0.65	
	Fraction particulate of COD that is inert	XI[a]	$gCOD\cdot m^{-3}$	~0.35	
	Chlorophyll-a	X_{ALG}	$gCOD\cdot m^{-3}$	0.4167	
	Heterotrophic biomass	XH	$gCOD\cdot m^{-3}$		2
	First stage nitrifying bacteria	XN_1	$gCOD\cdot m^{-3}$		0.4
	Second stage nitrifying bacteria	XN_2	$gCOD\cdot m^{-3}$		0.2

[a] $X_I + X_S = COD_{Total} - COD_{soluble} - X_{ALG} - X_H - X_{N1} - X_{N2}$

Table A3
Default and new parameters values considered in the calibration of the river water quality model.

Parameter		Units	Default	Adjusted value
Dry season: Base value for kla	kla_{base}	d^{-1}	1.0	0.10
Rainy season: Base value for kla	kla_{base}	d^{-1}	1.0	1.55

Table A4
Ecuadorian regulations to control dissolved oxygen, organic pollutants and nutrients in freshwater aquatic ecosystems.

Parameter	Units	Value	Comment
Dissolved Oxygen	% of saturation	>80%	
Nitrate (NO_3)	mg N·L^{-1}	13	
Nitrite (NO_2)	mg N·L^{-1}	0.2	
nitrogen- total phosphorous		A ratio of 15: 1	The value was taken for waters of aesthetic use
Five-day biological oxygen demand (BOD_5)	mg·L^{-1}	2 – 6	Aquatic ecosystems with a moderate impact
Total ammonium (NH_4)	mg N·L^{-1}		Concentrations are regulated concerning the pH and temperature

Temperature (°C)	pH 6.5	7.0	7.5	8.0	8.5	9.0
10	32.4	10.30	3.26	1.04	0.34	0.12
15	22.0	6.98	2.22	0.72	0.24	0.09
20	15.2	4.82	1.54	0.50	0.17	0.07

Table A5
Water quality model evaluation with different indices: R^2, RMSE and χ^2.

Variable	Dry season						Rainy season					
	Calibration			Validation			Calibration			Validation		
	R^2	RMSE	χ^2	R^2	RMSE	χ^2	R^2	RMSE	χ^2	R^2	RMSE	χ^2
Water depth[a]	0.78	0.15	0.95	0.87	0.13	0.90	0.78	0.15	0.95	0.87	0.13	0.90
Flow velocity[a]	0.86	0.26	0.86	0.74	0.11	0.90	0.86	0.26	0.86	0.74	0.11	0.90
DO	0.25	0.03	0.99	0.89	0.03	0.99	0.19	0.03	0.99	0.05	0.05	0.97
BOD5	0.99	0.15	0.95	0.98	0.06	0.80	0.86	0.34	0.75	0.98	0.17	0.86
COD	0.99	0.09	0.54	0.75	0.02	0.54	0.01	0.27	0.01	0.24	0.14	0.21
Ammonium	0.99	0.31	0.90	0.96	0.14	0.92	0.76	0.24	0.99	0.66	0.71	0.84
Nitrite	0.99	7.01	0.97	0.99	0.12	0.99	0.48	0.97	0.99	0.95	0.46	0.99
Nitrate	0.98	0.16	0.99	0.02	0.08	0.98	0.99	0.08	0.99	0.93	0.26	0.98
Orthophosphates	0.93	0.50	0.87	0.83	0.48	0.97	0.40	2.81	0.79	0.67	9.17	0.19

[a] The completed dataset was applied for the calibration and validation of water depth and flow velocity. Namely, these variables were not calibrated and validated per season.

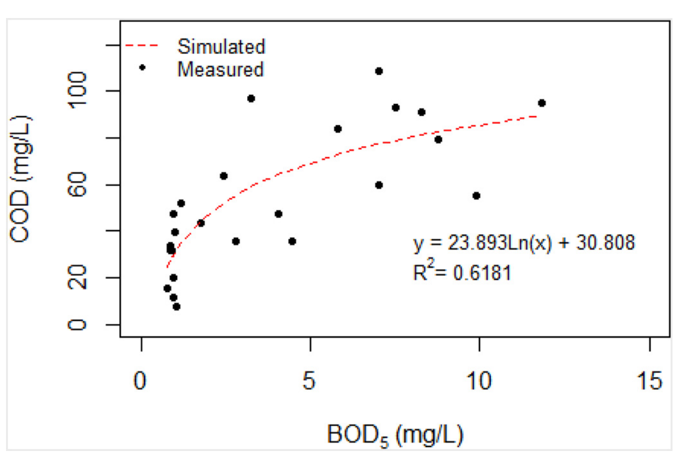

Fig. A2. Adjustment of the relationship between COD and BOD_5 to a logarithmic regression.

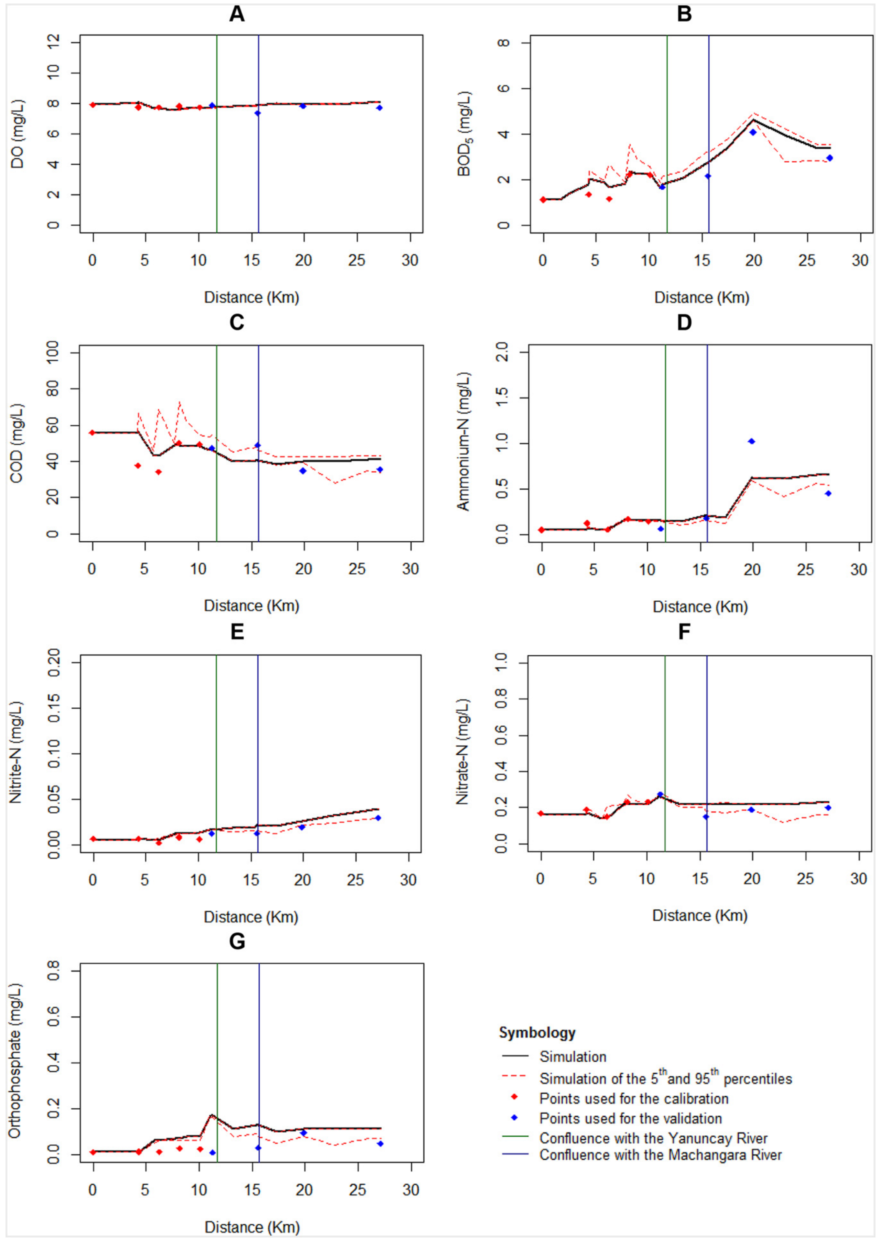

Fig. A3. Calibrated water quality model in the Tomebamba and Cuenca Rivers during the rainy season for: (**A**) DO, (**B**) BOD_5, (**C**) COD, (**D**) Ammonium, (**E**) Nitrite, (**F**) Nitrate, and (**G**) Orthophosphate.

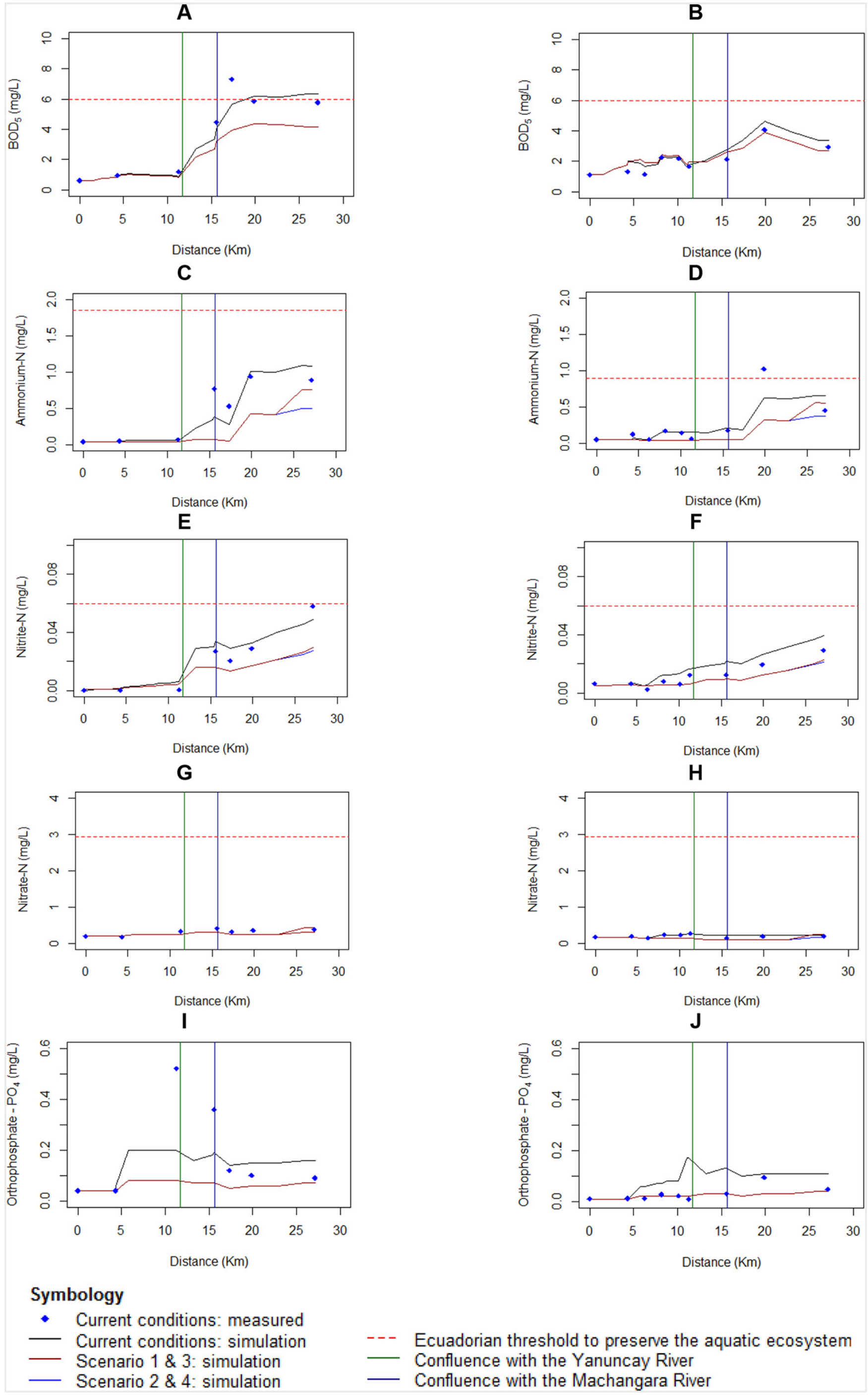

Fig. A4. Simulated concentrations along the Tomebemba and Cuenca Rivers. For dry season - Scenario 1 and 2: (**A**) BOD_5, (**C**) ammonium, (**E**) nitrite, (**G**) nitrate and (**I**) orthophosphate. For rainy season - Scenario 3 and 4: (**B**) BOD_5, (**D**) ammonium, (**F**) nitrite, (**H**) nitrate and (**J**) orthophosphate.

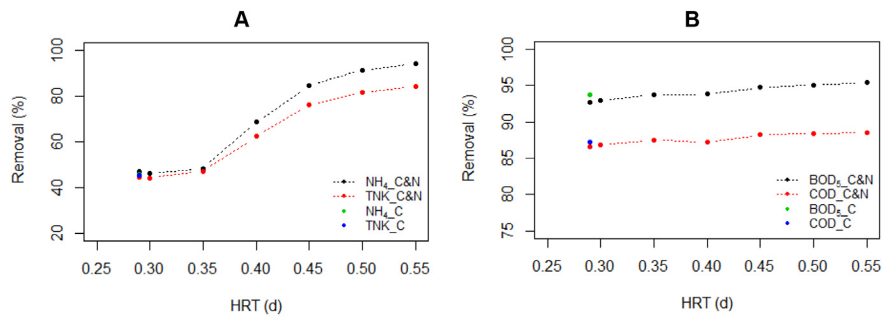

Fig. A5. Analysis of removal for the new G-WWTP according of hydraulic retention time (HRT) with the technologies of carbon (C) and carbon and nitrogen elimination (C&N) for: (**A**) nitrogen family, and (**B**) carbon family.

[b] The average flow calculated from all sampling sites during both the dry and the rainy season could present an erroneous interpretation. For a better understanding, please also check the median of the flow in both seasons.

References

Aereopuerto-Mariscal-Lamar, 2012. Información meteorológica aereopuerto Mariscal Lamar Cuenca. Dirección de aviación civil del Ecuador, Quito - Ecuador.

Alvarado, A., Vedantam, S., Goethals, P., Nopens, I., 2012. A compartmental model to describe hydraulics in a full-scale waste stabilization pond. Water Res. 46, 521–530.

Beltrán, A., Mendieta, P., Vanegas, J., 2013. Calidad del agua y contaminantes en el río Tarqui. Revista Galileo - Universidad de Cuenca 23, 115–121.

Benedetti, L., 2006. Probabilistic Design and Upgrade of Wastewater Treatment Plants in the EU Water Framework Directive Context. Ghent University.

Benedetti, L., Meirlaen, J., Sforzi, F., Facchi, A., Gandolfi, C., Vanrolleghem, P., 2007. Dynamic integrated water quality modelling: a case study of the Lambro River, northern Italy. Water SA 33, 627–632.

Benedetti, L., Langeveld, J., Comeau, A., Corominas, L., Daigger, G., Martin, C., Mikkelsen, P.S., Vezzaro, L., Weijers, S., Vanrolleghem, P.A., 2013. Modelling and monitoring of integrated urban wastewater systems: review on status and perspectives. Water Sci. Technol. 68, 1203–1215.

Beven, K., Binley, A., 1992. The future of distributed models: model calibration and uncertainty prediction. Hydrol. Process. 6, 279–298.

Borchardt, D., Sperling, F., 1997. Urban stormwater discharges: ecological effects on receiving waters and consequences for technical measures. Water Sci. Technol. 36, 173–178.

Brun, R., Reichert, P., Künsch, H.R., 2001. Practical identifiability analysis of large environmental simulation models. Water Resour. Res. 37, 1015–1030.

Burneo, P.C., Gunkel, G., 2003. Ecology of a high Andean stream, Rio Itambi, Otavalo, Ecuador. Limnologica-Ecology and Management of Inland Waters 33, 29–43.

CGA, 2007. Plan Eneregetico de Cuenca 2007–2017.

Cordero Domínguez, I.R., 2013. Evaluación de la gestión territorial de la cuenca del río Paute, estrategias y líneas de acción para superarlas. Master Thesis. Universidad de Cuenca, Cuenca - Ecuador.

De Pauw, N., Gabriels, W., Goethals, P.L., 2006. River monitoring and assessment methods based on macroinvertebrates. Biological Monitoring of Rivers. John Wiley and Son, Ltd, Chichester, UK, pp. 113–134.

Deksissa, T., Meirlaen, J., Ashton, P.J., Vanrolleghem, P.A., 2004. Simplifying dynamic river water quality modelling: a case study of inorganic nitrogen dynamics in the Crocodile River (South Africa). Water Air Soil Pollut. 155, 303–320.

Durazno, G., 2013. Variacion de flujo y de cargas contaminantes en el afluente y cargas de salida de la planta de aguas residuales de Ucubamba - año 2012. Empresa Municipal de Telecomunicaciones, Agua Potable, Alcantarillado y Saneamiento, Cuenca Ecuador.

Encalada, A.C., Sant, M.R., Prat i Fornells, N., 2011. Protocolo simplificado y guía de evaluación de la calidad ecológica de ríos andinos (CERA-S). Proyecto FUCARA, Quito - Ecuador.

EPA, 1999. Combined Sewer Overflows–Guidance for Monitoring and Modeling. Environmental Protection Agency.

Espinoza Berrezueta, D.A., Zumba López, T.C., 2018. Estudio del impacto ambiental en la calidad del agua del río Cuenca producido por la descarga del efluente de la planta de tratamiento de aguas residuales de Ucubamba. Universidad de Cuenca, Cuenca Ecuador.

Estrella, R., Tobar, V., 1994. Actualización de los estudios hidrológicos. Tomo II: Hidrología, Planes Maestros de Agua Potable y Alcantarillado del Área Metropolitana de la Ciudad de Cuenca, ETAPA, Cuenca, Ecuador.

Estrella, R., Tobar, V., 2013. Hidrologia y Climatologia - Formulación del plan de manejo integral de la subcuenca del río Machangara. ACOTECNIC Cia. Ltda. - Consejo de Cuenca del Rio Machangara, Cuenca - Ecuador.

ETAPA-EP, 2007. Evolución de la calidad del agua en los ríos que atraviesan la ciudad de Cuenca. ETAPA-EP, Direccion de Manejo Ambiental.

ETAPA-EP, 2017. Weekly Records of the Water Quality of the Ucumaba Wastewater Treatment Plant of the Years 2015 and 2016. ETAPA-EP, Ecuador.

ETAPA-EP, 2018. Cuenca Basin Gauging Station Network - Records 2014–2017.

Forio, M.A.E., Goethals, P.L., Lock, K., Asio, V., Bande, M., Thas, O., 2018. Model-based analysis of the relationship between macroinvertebrate traits and environmental river conditions. Environ. Model. Softw. 106, 57–67.

Gabriels, W., Lock, K., De Pauw, N., Goethals, P.L., 2010. Multimetric macroinvertebrate index Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). Limnologica-Ecology and Management of Inland Waters 40, 199–207.

Goethals, P., 2005. Data Driven Development of Predictive Ecological Models for Benthic Macroinvertebrates in Rivers. Ghent University, Ghent - Belgium.

Goethals, P., Forio, M., 2018. Advances in Ecological Water System Modeling: Integration and Leanification as a Basis for Application in Environmental Management. Multidisciplinary Digital Publishing Institute.

Greeley & Hansen, ACSAM, 2017. Diseños definitivos de la planta de tratamiento de aguas residuales de Guangarcucho. ETAPA-EP, Cuenca - Ecuador.

Griffiths, M., 2002. The European water framework directive: an approach to integrated river basin management. European Water Management Online 5, 1–14.

Guisan, A., Lehmann, A., Ferrier, S., Austin, M., OVERTON, J., Aspinall, R., Hastie, T., 2006. Making better biogeographical predictions of species' distributions. J. Appl. Ecol. 43, 386–392.

Henze, M., Gujer, W., Mino, T., Van Loosdrecht, M., 2000. Activated Sludge Models ASM1, ASM2, ASM2d and ASM3. IWA publishing.

Holguin-Gonzalez, J.E., Boets, P., Alvarado, A., Cisneros, F., Carrasco, M.C., Wyseure, G., Nopens, I., Goethals, P.L.M., 2013a. Integrating hydraulic, physicochemical and ecological models to assess the effectiveness of water quality management strategies for the river Cuenca in Ecuador. Ecol. Model. 254, 1–14.

Holguin-Gonzalez, J.E., Everaert, G., Boets, P., Galvis, A., Goethals, P.L.M., 2013b. Development and application of an integrated ecological modelling framework to analyze the impact of wastewater discharges on the ecological water quality of rivers. Environ. Model. Softw. 48, 27–36.

Holguin-Gonzalez, J.E., Boets, P., Everaert, G., Pauwels, I.S., Lock, K., Gobeyn, S., Benedetti, L., Amerlinck, Y., Nopens, I., Goethals, P.L.M., 2014. Development and assessment of an integrated ecological modelling framework to assess the effect of investments in wastewater treatment on water quality. Water Sci. Technol. 70, 1798–1807.

Hvitved-Jacobsen, T., 1982. The impact of combined sewer overflows on the dissolved oxygen concentration of a river. Water Res. 16, 1099–1105.

Jacobsen, D., 1998. The effect of organic pollution on the macroinvertebrate fauna of Ecuadorian highland streams. Arch. Hydrobiol. 143, 179–195.

Jacobsen, D., Encalada, A., 1998. The macroinvertebrate fauna of Ecuadorian highland streams in the wet and dry season. Arch. Hydrobiol. 142, 53–70.

Jacobsen, D., Rostgaard, S., Vásconez, J.J., 2003. Are macroinvertebrates in high altitude streams affected by oxygen deficiency? Freshw. Biol. 48, 2025–2032.

Jerves-Cobo, R., Everaert, G., Iñiguez-Vela, X., Córdova-Vela, G., Díaz-Granda, C., Cisneros, F., Nopens, I., Goethals, P.L., 2017. A methodology to model environmental preferences of EPT taxa in the Machangara River basin (Ecuador). Water 9, 195.

Jerves-Cobo, R., Córdova-Vela, G., Iñiguez-Vela, X., Díaz-Granda, C., Van Echelpoel, W., Cisneros, F., Nopens, I., Goethals, P., 2018a. Model-based analysis of the potential of macroinvertebrates as indicators for microbial pathogens in Rivers. Water 10, 375.

Jerves-Cobo, R., Lock, K., Van Butsel, J., Pauta, G., Cisneros, F., Nopens, I., Goethals, P.L.M., 2018b. Biological impact assessment of sewage outfalls in the urbanized area of the Cuenca River basin (Ecuador) in two different seasons. Limnologica 71, 8–28.

Jerves-Cobo, R., Forio, M.A.E., Lock, K., Van Butsel, J., Pauta, G., Cisneros, F., Nopens, I., Goethals, P.L.M., 2020. Biological water quality in tropical rivers during dry and rainy seasons: a model-based analysis. Ecol. Indic. 108, 105769.

Jorgensen, S.E., Nielsen, S.N., Jorgensen, L.A., 1991. Handbook of Ecological Parameters and Ecotoxicology. Elsevier.

Kraft, D., 2011. Model integration: Application in ecology and for management. Modelling complex ecological dynamics. Springer, pp. 301–320.

Lam, D., Leon, L., Hamilton, S., Crookshank, N., Bonin, D., Swayne, D., 2004. Multi-model integration in a decision support system: a technical user interface approach for watershed and lake management scenarios. Environ. Model. Softw. 19, 317–324.

Mac Berthouex, P., Brown, L.C., 2002. Statistics for Environmental Engineers. Lewis publishers, Boca Raton, USA.

MAE-Ecuador, 2015. In: Ministerio-del-Ambiente (Ed.), TULAS - Texto Unificado de Legislación Secundaria. Registro Oficial, Quito - Ecuador.

Marsalek, J., Karamouz, M., Cisneros, B.J., Malmquist, P.-A., Goldenfum, J.A., Chocat, B., 2014. Urban Water Cycle Processes and Interactions: Urban Water Series-UNESCO-IHP. CRC Press.

McCarty, P.L., Brodersen, C., 1962. Theory of extended aeration activated sludge. J. Water Pollut. Control Fed. 1095–1103.

McCullagh, P., 1984. Generalized linear models. Eur. J. Oper. Res. 16, 285–292.

McCullagh, P., Nelders, J.N., 1989. Generalized Linear Models. Chapman & Hall/CRC, Boca Raton, FL.

Montalvo, C., Jerves, R., Domínguez, L., 2018. Determinación de cargas de contaminación en aliviaderos de la red de alcantarillado combinado de la ciudad de Cuenca - Ecuador. XXXVI Congreso Interamericano de Ingeniería Sanitaria y Ambiental. Asociación Interamericana de Ingeniería Sanitaria y Ambiental – AIDIS, Guayaquil - Ecuador, p. 9.

Mouton, A.M., Van Der Most, H., Jeuken, A., Goethals, P.L., De Pauw, N., 2009. Evaluation of river basin restoration options by the application of the water framework directive explorer in the Zwalm River basin (Flanders, Belgium). River Res. Appl. 25, 82–97.

Mulliss, R., Revitt, D., Shutes, R., 1997. The impacts of discharges from two combined sewer overflows on the water quality of an urban watercourse. Water Sci. Technol. 36, 195–199.

NYC Environmental Protection. 2012. Criteria for Detention Facility Design. Page 17 *in* B. O. W. S. Operations, editor., (NY).

ODS-Territorio-Ecuador, 2017. ODS 8: Trabajo Decente y Crecimiento Economico - Azuay. Quito - Ecuador.

PROMAS-UCuenca, 2005. Estudio fluvio-morfológico del río Tomebamba. Cuenca, Ecuador.

Reichert, P., Borchardt, D., Henze, M., Rauch, W., Shanahan, P., Somlyódy, L., Vanrolleghem, P., 2001. River Water Quality Model No. 1. IWA Scientific and Technical Report No. 12. 2001. IWA Publishing, London, UK, p. 131 ISBN 1-900222-82.

Ríos-Touma, B., Acosta, R., Prat, N., 2014. The Andean biotic index (ABI): revised tolerance to pollution values for macroinvertebrate families and index performance evaluation. Rev. Biol. Trop. 62, 249–273.

Rubio, R., Mora, D., Jerves, R., Arias-Hidalgo, M., 2017. Estimación de las pérdidas de la precipitación en una cuenca hidrográfica urbana de la Ciudad de Cuenca. Page 4 First International Conference on Integrated Water Quality Management. Universidad de Cuenca, Cuenca - Ecuador.

Saltelli, A., Tarantola, S., Campolongo, F., Ratto, M., 2004. Sensitivity Analysis in Practice: A Guide to Assessing Scientific Models. Chichester, England.

SENPLADES, 2016. Proyecciones referenciales de población a nivel de distritos de planificación: 2010–2020. Secretaría Nacional de Planificación y Desarrollo Ecuador, Subsecretaría de Información, Dirección de Normas y Metodología, Quito - Ecuador.

Solvi, A., Benedetti, L., Gillé, S., Schosseler, P., Weidenhoupt, A., Vanrolleghem, P., 2005. Integrated urban catchment modelling for a sewer-treatment-river system. 10th International Conference on Urban Drainage.

Solvi, A.-M., 2006. Modelling the Sewer-Treatment-Urban River System in View of the EU Water Framework Directive. PhD thesis. Ghent University, Belgium.

Stockwell, D.R., Peterson, A.T., 2002. Effects of sample size on accuracy of species distribution models. Ecol. Model. 148, 1–13.

Te Chow, V., 1959. Open-Channel Hydraulics. McGraw-Hill New York.

Tomsic, C.A., Granata, T.C., Murphy, R.P., Livchak, C.J., 2007. Using a coupled eco-hydrodynamic model to predict habitat for target species following dam removal. Ecol. Eng. 30, 215–230.

USEPA - U.S. Environmental Protection Agency, 2011. A Primer on Using Biological Assessments to Support Water Quality Management. Washington, DC.

Vanhooren, H., Meirlaen, J., Amerlinck, Y., Claeys, F., Vangheluwe, H., Vanrolleghem, P.A., 2003. WEST: modelling biological wastewater treatment. J. Hydroinf. 5, 27–50.

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., Morgan, R.P., 2005. The urban stream syndrome: current knowledge and the search for a cure. J. N. Am. Benthol. Soc. 24, 706–723.

Zabel, T., Milne, I., Mckay, G., 2001. Approaches adopted by the European Union and selected member states for the control of urban pollution. Urban Water 3, 25–32.

Zuur, A., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer Science & Business Media.