Designing Eco-Friendly Water Intake Portfolios in a Tropical Andean Stream Network

D. González-Zeas, D. Rosero-López, T. Walter, A. Flecker, P. Lloret, B. de Bièvre, T. Condom, R. Osorio, O. Dangles

AGU100 ADVANCING EARTH AND SPACE SCIENCE

Water Resources Research

RESEARCH ARTICLE
10.1029/2019WR025190

Designing Eco-Friendly Water Intake Portfolios in a Tropical Andean Stream Network

D. González-Zeas[1], D. Rosero-López[2], T. Walter[2], A. Flecker[3], P. Lloret[4], B. De Bièvre[5], T. Condom[6], R. Osorio[4], and O. Dangles[1,3]

[1]Institut de Recherche pour le Développement, Centre d'Ecologie Fonctionnelle et Evolutive, UMR 5175, CNRS, Université de Montpellier-Université Paul-Valéry Montpellier-EPHE-IRD, Montpellier, France. [2]Department of Biological and Environmental Engineering, Cornell University, Ithaca, NY, USA, [3]Department of Ecology and Evolutionary Biology, Cornell University, Ithaca, NY, USA, [4]Empresa Pública Metropolitana de Agua Potable y Saneamiento, Quito, Ecuador, [5]Fondo para la Protección del Agua, Quito, Ecuador, [6]Université Grenoble Alpes, CNRS, IRD, Institut des Geosciences de l'Environnement (IGE), Grenoble, France

Key Points:
- We develop a methodological framework to perform trade-off analyses between water withdrawal and ecological impacts at the basin scale
- Through a Pareto frontier analysis, the operation of 11 water intake structures used to provide potable water to Quito City is optimized
- Application of new water management practices are necessary to reach a more dynamic operation strategy to balance human and ecological needs

Correspondence to:
D. González-Zeas and O. Dangles,
duniapgz@yahoo.com.mx;
olivier.dangles@ird.fr

Citation:
González-Zeas, D., Rosero-López, D., Walter, T., Flecker, A., Lloret, P., De Bièvre, B., et al. (2019). Designing eco-friendly water intake portfolios in a tropical Andean stream network. *Water Resources Research*, *55*, 6946–6967. https://doi.org/10.1029/2019WR025190

Received 18 MAR 2019
Accepted 27 JUL 2019
Accepted article online 1 AUG 2019
Published online 19 AUG 2019

Abstract In view of the rapid proliferation of water infrastructures worldwide, balancing human and ecosystem needs for water resources is a critical environmental challenge of global significance. While there is abundant literature on the environmental impacts of individual water infrastructures, little attention has been paid to their cumulative effects in river networks, which may have basin-to-global impacts on freshwater ecology. Here we developed a methodological framework based on Pareto frontier analysis for optimizing trade-offs between water withdrawal and ecological indicators. We applied this framework to a mountainous Ecuadorian headwater river network that is part of a continental water transfer for supply and demand management to optimize ecological conditions and the operation of 11 water intake structures used to provide potable water to the city of Quito. We found that the current water intake configuration has an important effect on the total length of fifth-order stream sections (65% reduction compared to premanaged condition) and isolates 70.9% of the headwater stream length. The Pareto frontier analysis identified water intake portfolios (i.e., different combinations of intake sites) that decreased ecological impacts by 7.8% points (pp) and 13.0 pp for connectivity and stream order change, respectively, while meeting Quito's water demands. Additional portfolios accounting for monthly variability in water demand and resources further decrease the ecological impact up to 9.6 pp in connectivity and 13.4 pp in stream order. These eco-friendly portfolios suggest that adaptive management at basin level may help optimize water withdrawal to fulfill urban demands while preserving ecological integrity.

1. Introduction

Worldwide, the trend of increasing urbanization, along with associated energy needs, requires increasing demand for potable water and hydropower (Palmer, 2014). The combination of increased human needs (e.g., agriculture, urban and industrial water supply, or energy production) and migration (toward large cities) causes rapidly changing water use portfolio, which may conflict with variation in water availability related to human perturbations and climate change (Liu et al., 2008). Moreover, many regions of the world are becoming increasingly prone to drought or, at least, declining precipitation coupled with rising water demand, which could threaten the ecological integrity and ecosystem services of freshwater ecosystems across the globe (Hartmann et al., 2013). In the endeavor to manage water to meet human necessities, freshwater biota have largely been neglected. Yet, healthy freshwater ecosystems provide crucial goods and services for society (Grizzetti et al., 2016). Consequently, human appropriation of freshwater flows must be better managed if we hope to sustain these benefits and freshwater biodiversity (Vörösmarty et al., 2010). In view of the rapid proliferation of dams and other water infrastructures, particularly in biodiverse regions such as the Andean Amazon (Finer & Jenkins, 2012;Latrubesse et al., 2017), it is, more than ever, a critical environmental challenge of global significance to balance human and ecosystem needs for water resources.

While there is an abundant literature on the environmental impacts of individual water infrastructures from temperate regions (e.g., Wood et al., 2008), our knowledge is relatively slim for rapidly developing regions where watersheds are heavily managed. In particular, little attention has been paid to the cumulative effects of complex water infrastructure networks (such as intakes obtaining water from rivers to supply drinking

10.1029/2019WR025190

water systems), which may have basin-to-global impacts on vital ecosystem services that are dependent on naturally functioning hydrological regimes (Kibler & Tullos, 2013). Assessing cumulative effects in rapidly developing regions that possess some of the most biodiverse river systems is especially challenged by few data on hydrology, ecology, and biodiversity. Moreover, cumulative effects may be manifested dozens of kilometers downstream of water control structures, and such spatially distant impacts are rarely considered. Also, it is increasingly recognized that biodiversity and ecosystem services are spatially structured. For example, one of the major effects of flow alteration on aquatic biota is related to the modification of the connectivity within and between waterscapes with impacts on species distribution, local endemism, and metacommunity structure (Cauvy-Fraunié et al., 2014). Consequently, our understanding of spatial patterns in environmental variables urgently needs to be matched with spatial patterns in human impact at a watershed scale (Jager et al., 2015).

The rapid expansion of water infrastructure and the deficit of baseline information have created an urgency to develop innovative frameworks for decision makers to evaluate thresholds and trade-offs between ecosystem services and human water use under different water infrastructure development and climate change scenarios (Brown et al., 2015). While there have been previous attempts to propose frameworks of ecologically sustainable water management (Richter et al., 2003), there remain major bottlenecks that impede concrete recommendations in terms of flow to preserve the need: (1) to deal with the scarcity of hydrological and meteorological data that characterize most tropical watersheds, (2) to integrate the spatial configuration of both environmental variables (biodiversity, ecosystem health) and cumulative effects of water infrastructures in river networks, and (3) to frame water management conflicts into socio-ecological scenarios (e.g., increase in water demand) and propose portfolios (i.e., different combinations of intake sites) usable for policy makers. For these two last bottlenecks, multi-objective optimization approaches have been used in previous studies (Hof & Bevers, 1998; Reed et al., 2013; Salazar et al., 2016) in order to find Pareto optimal solutions that could define the optimal trade-offs between human needs and environmental protection (e.g., Kuby et al., 2005; Null et al., 2014). Optimized portfolios can give way to an increased environmental flow regime, with limited impacts on water uptake.

The overall objective of this study is to develop a methodological framework for building ecologically sustainable water management scenarios, in which human needs are met designing water intake configurations in a manner that can sustain the ecological integrity of freshwater at a basin scale. The four main steps of this framework are to (1) evaluate a water supply system at the basin scale under premanaged and managed scenarios, (2) model the water system by defining and quantifying relevant indicators of water use and ecological health, (3) design water intake portfolios that optimize both water use and ecological indicators on a Pareto frontier, and (4) suggest possible improvements for water management planning based on optimization criteria. To implement and test our model, we focused on the Andean region of South America, specifically headwaters of the Amazonian Napo basin in Ecuador. Located at the eastern side of the continental water divide, these waters are used for provisioning Quito, Ecuador's capital city, with drinking water. Due to the extremely complex topography, and high altitude of Quito at 2,850 m asl, the system consists of a high number of intakes in small headwater streams. This area of the Ecuadorian Andes is characterized by relatively low rainfall, but with an extreme heterogeneity of spatial and temporal distribution of precipitation (Hofstede et al., 2003). Our rationale for focusing on this region is that it provides all the ingredients to develop a general decision framework for examining socio-ecological trade-offs and thresholds of water infrastructure development. Moreover, it is a biodiversity hot spot where high densities of water infrastructure are being planned for waterscapes that provide important ecosystem services such as water purification, carbon storage, and climatic refuges for biodiversity (Jacobsen & Dangles, 2017).

2. Material and Methods

2.1. Methodological Framework

We developed a methodological framework for optimizing water use for urban supply and ecological health that integrates four main components (data, system operation, modeling with indicators and scenarios, and optimization) to perform trade-off analyses between water yields and ecological impacts. The procedure is summarized in Figure 1 and includes the following steps:

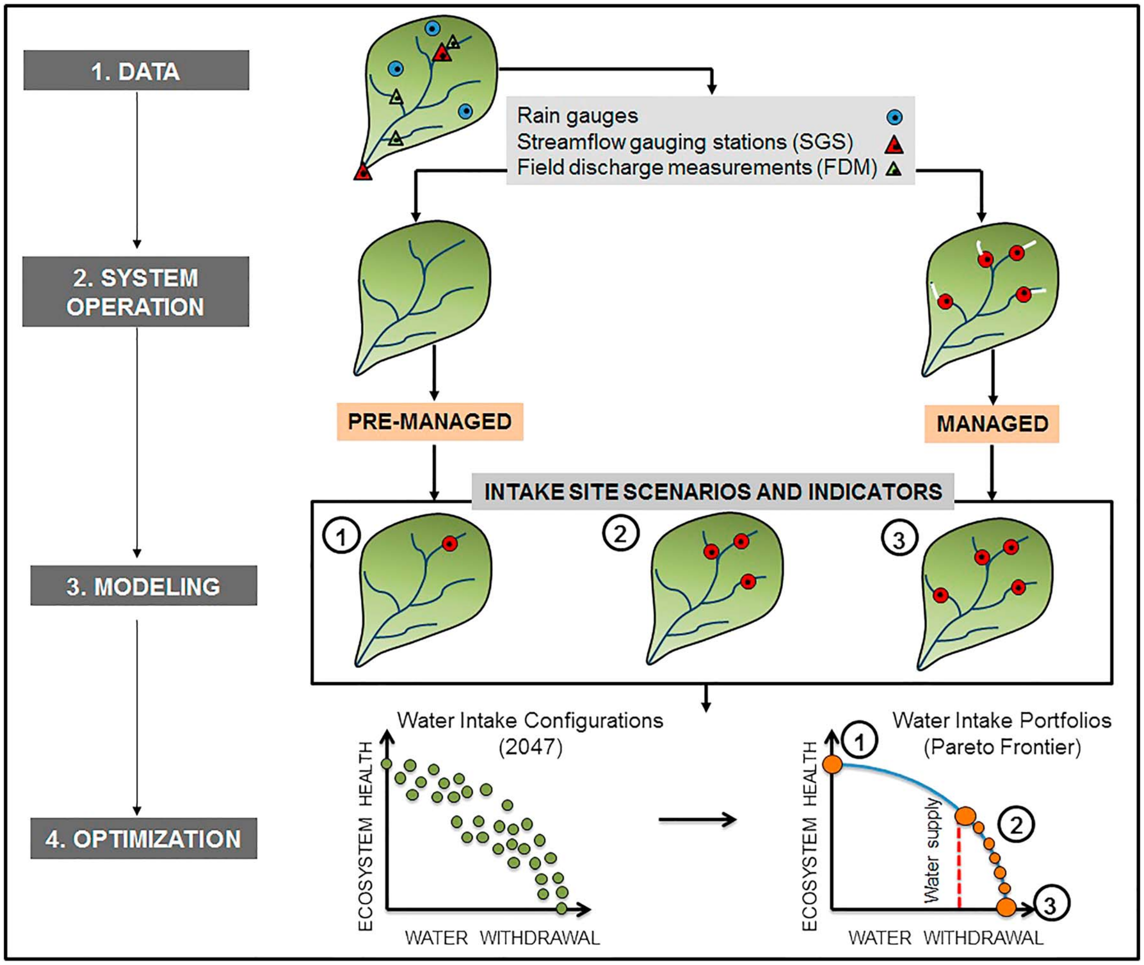

Figure 1. Schematic of the methodological approach. Red dots indicate water intake structures, and white segments of the river represent the loss of connectivity because of water intake placement. Based on physiographic and climatic data, a cluster analysis allows to regionalize the study basin to obtain daily flow duration curves at every water intake structure (red dots). Green dots represent the different water intake configurations (2,047), and orange dots show the ensemble of multiple optimum water intake configurations of the Pareto frontier under a scenario of increasing water pressure (from 1 to 3). Water supply specifies a restriction of the minimum water withdrawal (historical water supply data) from the system from which the Pareto frontier is defined. Sets of optimal water intake configurations represent eco-friendly portfolios for water management.

1. collection and processing of available data (both physiographical and hydroclimatic) in the study basin, as well as water withdrawal data (water supply);
2. diagnosis of the water supply system operation under the current situation (period 2008–2016);
3. modeling of the system through the definition of indicators of water withdrawal (i.e., available discharge) and ecosystem health (i.e., stream order and connectivity), and the definition of input scenarios of increases in water withdrawal pressure, from low (1) to high (3), (see Figure 1); and
4. Pareto frontier analysis to identify optimized portfolios of the number and placement of water intakes (see 2 in Figure 1). The optimization aims to satisfy water needs (water supply) within the context of minimizing possible impact on ecosystem health. It can also be useful to develop adaptive management programs of water resources to better balance human and ecological needs.

2.2. Data Use in This Study

2.2.1. Study Area

The study was conducted in the Chalpi basin located southeast of Quito within the Napo province of Ecuador (centroid of the basin situated at 823417 east, 9966968 north, WGS84 UTM 17S). The basin is approximately of 95 km^2 with elevations ranging from 2,880 to 4,200 m asl. A digital elevation model

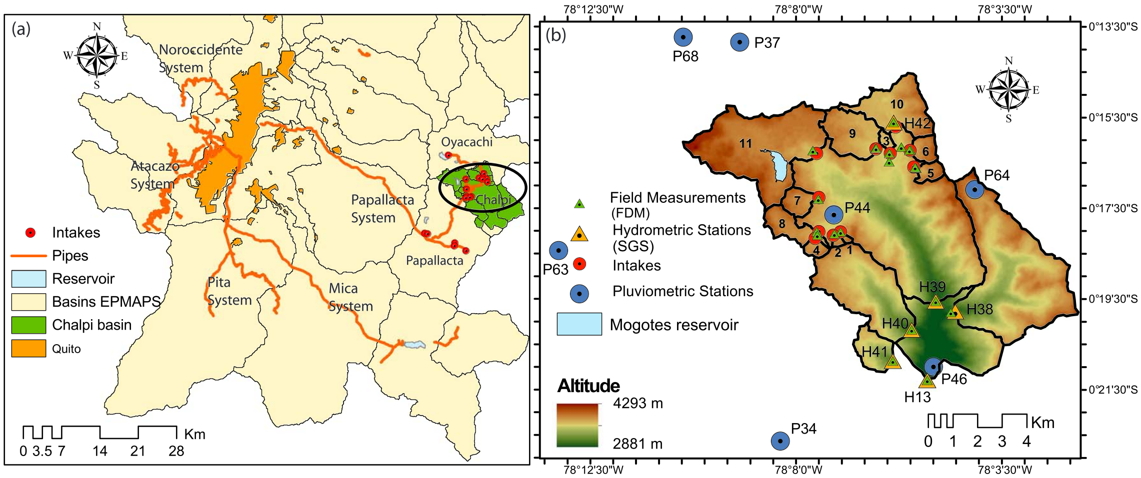

Figure 2. (a) Location of the 11 water intakes in operation (red dots in the black oval) in the upper part of the Chalpi basin of the Papallacta system. (b) Studied sub-basins and location of rain gauges, six SGSs as well as FDM data upstream of water intakes. Sub-basins codes are defined in Table 1. SGSs = streamflow gauging stations.

(DEM) was obtained from National Aeronautics and Space Administration's Advanced Spaceborne Thermal Emission and Reflection Radiometer and has a resolution of 30 m. The Chalpi River drains to the Quijos River, a tributary of the Napo River. The study basin is one of the main sources of water supply to the Papallacta System, which provides potable water to about 30% of the 2.6 million people in the city of Quito (Figure 2a). Predominant land cover in the Chalpi basin is páramo, composed of herbaceous (*Calamagrostis* sp.) high-altitude prairies with scattered patches of *Polylepis* forests laying on thick, highly humic soils. Páramo soils have many properties in common with upland peat soils with high organic C content and high water retention (Nanzyo et al., 1993). Páramos are well known for their large water surplus and sustained base flow throughout the year, feeding rivers descending to both the Amazon basin and coastal regions (Buytaert et al., 2006). Additionally, streamflow of the Ecuadorian Andes primarily consists of subsurface flow with the yearly amount of streamflow tightly controlled by the annual rainfall depth (Crespo et al., 2011). Many smaller lakes and local depressions exist in the páramo, but they are disconnected from the drainage network collecting overland flow from slopes. This allows for infiltration and recharge of deep ground water storage reservoirs (Price & Waddington, 2000).

Table 1
Design and Regulated Flows of the Water Intakes According to the Manual of Operation Operation and Maintenance of the Papallacta System (EPMAPS, Confidential Document)

Code	Intake	Design flow (m^3/s)	Regulated flow (operation) (m^3/s)
1	Vikingos 2	0.15	0.05
2	Vikingos 3	0.15	0.05
3	Quillugsha 3	0.15	0.05
4	Vikingos	0.15	0.05
5	Glaciar	0.15	0.11
6	Gonzalito	0.15	0.11
7	Venados	0.15	0.11
8	Guaytaloma	1.00	0.20
9	Quillugsha 1 y 2	0.15	0.05
10	Chalpi Norte	1.00	0.45
11	Mogotes	1.40	0.70

Note. The design flow represents the maximum flow that can be withdrawn by the structure, while the regulated flow represents the mean water flow actually extracted from the river to meet water needs.

In our study, we considered 16 sub-basins (see the sub-basin code in Figure 2b and Table 1), which were defined from the 11 operating intake sites (codes from no. 1 to no. 10) and an additional sub-basin (no. 11) corresponding to a water intake, Mogotes, that has an upstream regulation structure. We also had six streamflow gauging stations (SGSs): one of them (H42) was located upstream of water intake no. 10, and five additional sites with available streamflow gauging data (codes H13, H38, H39, H40, and H41; Figure 2b).

2.2.2. Data Collection

Precipitation and stream discharge data were provided by seven rain gauges and six SGSs, respectively (Figure 2b). Rain gauges had a 5-min temporal resolution. P34, P37, and P44 provided information from 2004 to 2016; P63, P64, and P68 had information from 2012 to 2016; and P46 recorded information from 2007 to 2016. SGSs H38, H39, H40, and H41 (see Figure 2b) recorded information from 2011 to 2015, while the H42 and H13 stations registered information from 2011 to 2016 and 2003 to 2016, respectively. All discharge series had a time resolution of

10 min. We are aware that the short data time series for some stations can generate uncertainty in simulations with regards to interannual variability (wet and dry periods). Unfortunately, the scarcity of streamflow data is a common problem, especially in tropical regions (e.g., Auerbach et al., 2016; Dile & Srinivasan, 2014; Singh et al., 2001; Yu et al., 2002). In addition to the six SGSs, field discharge measurements (FDMs) made several times a year (on average once a month using current meters) were available upstream of the 11 water intakes in operation (See Appendix A and Table A2). Otherwise, the operation department of Quito's water utility (EPMAPS in spanish) provided monthly water withdrawal data (water withdrawn to meet urban water demands) of the 11 water intakes in operation over the period 2008–2016 (See Table B1). FDM and water withdrawal data of the Mogotes intake used in the study take into account water stored in Mogotes reservoir. Taking into consideration all available discharge information in the basin, as well as the water withdrawn, we used a common period of complete monthly time series from 2008 to 2016. Although the length of the analysis period is short (9 years), it captures the modes of variability associated with the El Niño–Southern Oscillation phenomenon (Erazo et al., 2018; Takahashi et al., 2011; see Figure A1 and Appendix A). Furthermore, according to Segura et al. (2019), in the transition Andean zone (from 15°S to 1°N) the El Niño and La Niña events are not the main factors influencing interannual precipitation variability.

2.3. System Operation

The study basin is one of the water sources contributing to the Papallacta System, together with the Oyacachi and Papallacta basins. The Papallacta system consists of a continental water transfer, where the water withdrawal by the different intakes structures of the system converge to a single water pipe in order to cover 30% of Quito's drinking water demand through the Bellavista and Paluguillo water treatment plants. In this case, the water management considers each water intake individually and all are jointly operated to supply a single urban water demand. The Papallacta system provided 2.54 m^3/s (average of the period 2008–2016) with a supply reliability of 99.85% (EPMAPS, personal information, 2019). This system, together with other existing systems—Pita, Mica, Atacazo, and Noroccidente—(see Figure 2a), covers the entire demand of the drinking water for Quito. Our study is focused on the Chalpi basin where Quito's water utility (EPMAPS) has built different structures in the upper part of the basin (11 water intakes and the Mogotes reservoir, see Figure 2b). Based on historical data of water withdrawn from the system (period 2008–2016), the Chalpi basin provided an average of 0.93 m^3/s to the Papallacta system (see Table B1). The remaining 1.61 m^3/s was covered by hydraulic structures located in Oyacachi and Papallacta basins (see Figure 2a) whose analysis will be incorporated in futures studies. The operation rules of the Papallacta system permit withdrawing water from the river limited by the regulated flow (see Table 1). During months in which streamflow is greater than the regulated flow, the surplus overflows the weir. Moreover, the current system operation does not release an *ecological flow* downstream of the water intakes and many times the water withdrawal is equal to the available flow in the river. The operation of the system also prioritizes that Mogotes' reservoir is at suitable levels of operation.

2.4. Data Analyses and Modeling

2.4.1. Regionalization Method for the Construction of Monthly Streamflow Series

There is a large number of methodologies proposed for streamflow predictions in poorly gauged basins. Here we have focused on using available observed discharge records rather than developing another new method of flow prediction in ungauged basins.

Estimation of monthly streamflow for all water intake study basins is of practical interest for sustainable and optimal water management. We recognize that substantial advances in flow predictions in poorly gauged and ungauged catchments have been made (Alipour & Kibbler, 2018; Kuzmin et al., 2019; Nepal et al., 2017). For example, methods based on regionalization techniques (e.g., Kult et al., 2014; Pumo et al., 2016; Tegegne & Young-Oh, 2018) are important tools, yet their application may be limited in areas with low density of climatic/hydrologic data. An additional alternative in such cases is relying on existing data, including regional/global data of climatic variables (e.g., Tuo et al., 2016). However, precipitation is very heterogeneous in this mountainous watershed, with high spatial-temporal variability, so using regional data sets in a rainfall-runoff model would likely lead to large uncertainties at the basin scale. Since direct measurements of SGSs are available for only six sites in the study basin, the value of a few short FDMs can support predictions in an ungauged basin (e.g., Pool et al., 2017; Viviroli & Seibert, 2015). More precisely, we adopted a commonly used regionalization approach to evaluate catchment similarity in order to transpose the monthly distribution from the gauged to ungauged catchments, and then, the monthly response at ungauged water intake basins was calculated based on the FDM data (see Table A2) as an index of streamflow. The procedure is detailed in Appendix B.

2.4.2. Goodness of Fit Indicators

Based on Legates and McCabe (1999) and Fekete and Vörösmarty (2004), we used four quantitative indicators of goodness of fit for comparing the FDM and SGS flow duration curves (FDCs) to corroborate the value of FDM data for constructing monthly streamflow in the ungauged water intake basins: the mean relative error ($\overline{E}_s$), the standard deviation of the error ($\sigma_{E,s}$), the index of agreement (d, Willmott, 1981), and the Nash-Sutcliffe indicator (NS, Nash & Sutcliffe, 1970).

The mean relative error and its standard deviation were computed as follows:

$$E_{s,j} = \frac{Qs_{s,j} - Qobs_{s,j}}{Qobs_{s,j}} \tag{1}$$

$$\overline{E_s} = \frac{1}{N_d} \sum_{j=1}^{Nd} E_{s,j} \tag{2}$$

$$\sigma_{E,s} = \sqrt{\frac{1}{N_d - 1} \sum_{j=1}^{Nd} \left(E_{s,j} - \overline{E_s}\right)^2} \tag{3}$$

where $E_{s,j}$ is the relative error, $Qs_{s,j}$ and $Qobs_{s,j}$ are the FDM and SFS daily data for site s and duration j, and Nd refers to the total number of durations j considered in the comparison.

The index of agreement (d) was used as a measure of the mean relative error with respect to the potential error (Krause et al., 2005; Moriasi et al., 2007). This dimensionless indicator, which varies between 0 for poor adjustment and 1 for perfect adjustment, is expressed by the following equation:

$$d = 1 - \frac{\sum_{j=1}^{Nd} \left(Qs_{s,j} - Qobs_{s,j}\right)^2}{\sum_{j=1}^{Nd} \left(\left|Qs_{s,j} - \overline{Qobs_s}\right| + \left|Qobs_{s,j} - \overline{Qobs_s}\right|\right)^2} \tag{4}$$

Finally, the NS dimensionless indicator represents the relative improvement of the FDM data with respect to the SGS data. The values of NS range from minus infinity (poor fit) to 1 (perfect fit) and are expressed as follows:

$$NS = 1 - \frac{\sum_{j=1}^{Nd} \left(Qs_{s,j} - Qobs_{s,j}\right)^2}{\sum_{j=1}^{Nd} \left(Qobs_{s,j} - \overline{Qobs_s}\right)^2} \tag{5}$$

where $\overline{\mathbf{Qobs}_s}$ is the mean SGS daily data of the common period of study for site s.

2.5. Portfolio Building

2.5.1. Intake Site Number and Placement

Currently, the 11 intake sites located in the Chalpi basin are in operation to supply Quito with potable water. However, there is no existing water management plan taking into account the dynamic nature of water demand and supply, or the trade-off between water needs and ecological impacts. To address this issue, we analyzed the trade-off between water supply for urban demand versus ecological impacts under different scenarios of water withdrawal pressure, by varying the number and placement of operating intakes. Water pressure scenario ranged from a single operating water intake at the lowest flow site (lower water withdrawal pressure) to the 11 operating intakes (higher withdrawal pressure). The following equation gives the total number of potential portfolios.

$$C(n_{\text{sites}}, \text{int}) = \frac{n_{\text{sites}}!}{(n_{\text{sites}} - \text{int})! * \text{int}!} \tag{6}$$

where int corresponds to the number of intake sites to be considered in the generation of the configurations and n_{sites} is the number of potential intake site placements. Taking into consideration that int is variable from 1 to the total number of built intakes (*total*), hence the total number of generated configurations can be expressed as follows:

$$\boldsymbol{C}_{\mathbf{total}} = \sum \mathrm{int} = \sum_{\mathbf{int=1}}^{\mathbf{total}} \boldsymbol{C}_{\boldsymbol{i}}(\boldsymbol{n}_{\mathbf{sites}}, \mathrm{int}) \quad (7)$$

For our study basin, this represents 2,047 different configurations.

2.6. Optimization

2.6.1. Indicators

For each configuration, we used the sum of mean flows at each operating intake site as an indicator of total water withdrawal. The mean annual water withdrawal and the mean monthly water withdrawal were used to cover the range of management needs of operation within our study system. With regards to the ecological health of the hydrosystem, we used two types of indicators: (1) the change in stream orders downstream of the water intake sites (e.g., reduction from second order to first order due to a complete withdrawal of the water), associated with two ecological indicators as a measure of resilience or disturbance of the ecosystem due to placement of water intakes, and (2) stream connectivity within the basin, that is, the total length of streams not disconnected from the lower parts of the basin. Both of them were calculated using Geographical Information System based on Strahler's system of ordering streams for determining the change in stream order, and the total of stream length of each sub-basin as a measure of stream connectivity. These two indicators are complementary as they assess potential ecological impacts downstream and upstream of a given water intake site, respectively. Previous studies have used these stream health metrics to characterize disturbances associated with human activities (Feio et al., 2010; Fryirs, 2013; Strecker & Brittain, 2017). Connectivity has been addressed in different ways through analyses in numerous fields of study including hydrology (Bracken & Croke, 2007; Freeman et al., 2007; Pringle, 2003). Similarly, changes in stream order have been considered for evaluating the physical modification of the river and a corresponding degradation of habitats and loss of natural functions, causing a variety of ecological impacts (Hughes et al., 2011; Kwang-Seuk et al., 2010; Oaks et al., 2005). We further refined *stream order* metrics by quantifying how stream order affects two key components of community structure and ecosystem function of high-altitude streams: benthic macroinvertebrate assemblages and the breakdown of organic matter (Dangles et al., 2011). There is a rich literature on the use of benthic macroinvertebrates (mainly insect larvae and crustaceans) to detect the ecological effects of human disturbance (including flow alteration) on the integrity of freshwater systems (e.g., see Rosero-López et al., 2019; Vannote et al., 1980). Also, because headwater streams receive a large proportion of allochthonous input from the riparian vegetation (mainly grasses), leaf breakdown plays a pivotal functional role and is a key process for assessing the possible impact of flow alteration on stream ecosystem (Martínez et al., 2016; Mendoza-Lera et al., 2012). For these reasons, we used published data collected in our study basin (Dangles et al., 2011; Rosero-López et al., 2019) to plot the relationship of the regional diversity of macroinvertebrates at the basin scale (gamma diversity) and the decomposition rate of grass as a function of stream order (see Figure A2 and Appendix A). These two ecological indicators were then implemented in the optimization process.

2.6.2. Trade-Off Analysis and Optimization

Trade-off analysis between water withdrawal and ecosystem health indicators was performed for the 2,047 water intake configurations. The main objectives of our analysis focused on optimizing both water withdrawal (to meet water needs for urban supply) and ecological indicators. A multi-objective optimization technique was used to define the Pareto-optimal (PO) solutions (Deb, 2002; Lautenbach et al., 2014; Peñuela & Granada, 2007). The concept of dominance was applied for the search of the portfolio of configurations that were dominant, also known as the *no-dominated set*. The remaining configurations were part of the dominated set (see Cibin & Chaubey, 2015; Kougias & Theodossiou, 2013; Lautenbach et al., 2013 for further details on multiobjective optimizations). The Pareto front was obtained when the set of dominant configurations was determined across the entire objective space (2,047 configurations). Pareto frontier analyses were performed in MATLAB using the bisection method (Cordero et al., 2011; Deb, 2004). Our code generated a recursive division of the entire space of solutions until it converged to the optimal solutions.

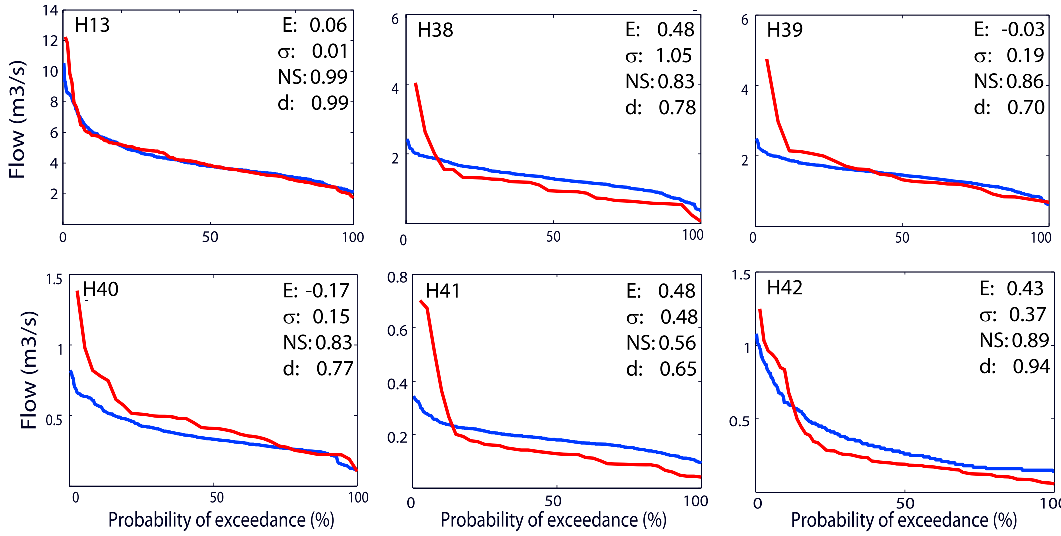

Figure 3. Comparison between field discharge measurements (FDMs, red lines) and the six streamflow gauging stations (SGSs, blue lines) FDCs of daily streamflow data. FDCs = flow duration curves.

Assuming that the 11 water intakes in operation take all the water from the river at each site, the current system configuration has a maximum withdrawing capacity of 1.29 m^3/s. Nevertheless, the water withdrawal necessary for satisfying urban demands of Quito from the Chalpi basin was 0.93 m^3/s during the period 2008–2016 (see Table B1 for details of mean and coefficient of variation of the discharge and water supply data of the analysis period). Consequently, the range between maximum water withdrawal and maximum water needs was taken into account to define PO configurations. Considering that the Pareto frontier provides a portfolio of multiple optimum configurations where each solution is *best*, the optimum configuration was reached by comparing the different solutions of the Pareto frontier and defining a posteriori the configuration with the maximum value of connectivity and maximum value of the maintenance in stream order. In addition, our analysis defined a maximum water withdrawal of each water intake based on the existing intake capacity to derive the water (design flow, see Table 1). Consequently, flows exceeding this capacity cannot be diverted, and they are not considered in the optimization process. We further explored whether water intake configuration could be optimized on a monthly basis by taking into account the seasonality in water discharge and water supply. For this, we used historical data of water withdrawal provided by the EPMAPS of Quito (see Table B1) and water discharge obtained by constructed monthly streamflow series.

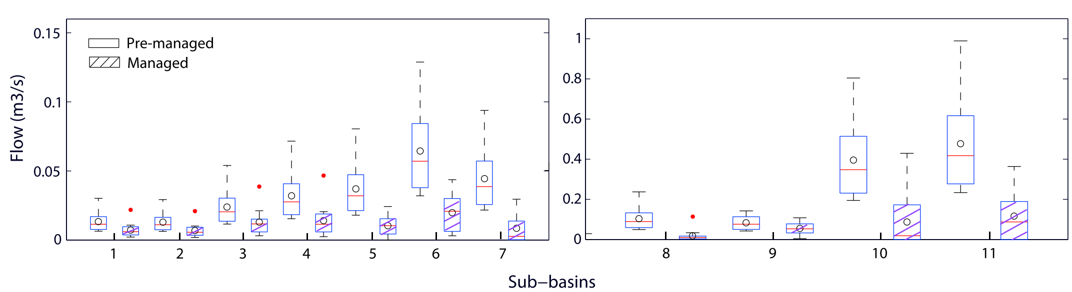

Figure 4. Comparison between premanaged and managed monthly flow. The box plot shows the interannual variation of the flow. The lines extend up to 1.5 times the interquartile range above and below the box. The box extends from the 25th to the 75th percentiles. The line within the box indicates the median flow. The open black circle shows the mean annual flow (average flow of the period 2008–2016); the red points outside the box indicate the outliers for the 12 months of data. Basin codes refer to Table 1.

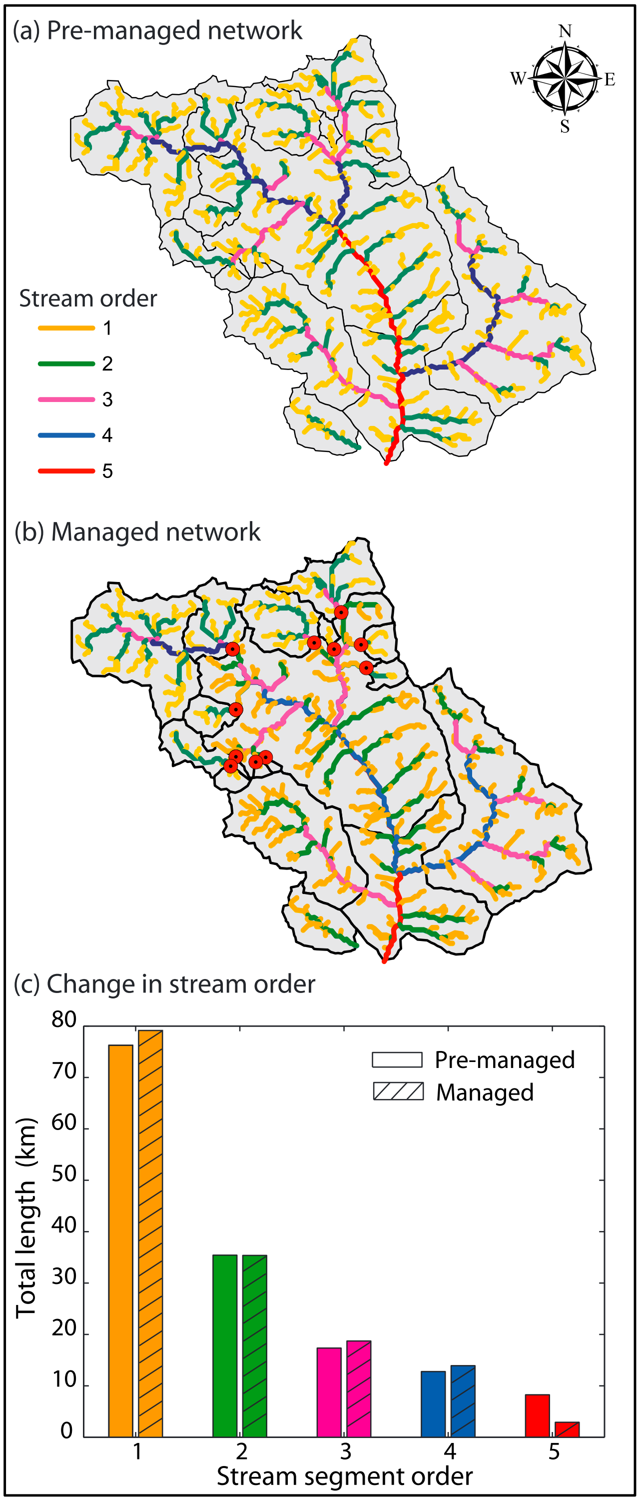

Figure 5. Distribution of stream-segment orders in the Chalpi network under (a) premanaged conditions, (b) managed conditions (with the assumption of a complete water withdrawal at the 11 water intakes in operation), and (c) change in stream segment length for different order categories. The major impact is the disappearance of the fifth-order stream length.

3. Results

3.1. Constructed Monthly Streamflow Series

Based on physiographic and climatic characteristics (Table A1), the sub-basin cluster analysis revealed one homogeneous region in the Chalpi basin. We found good overall agreement between FDM and SGS FDCs in all the sub-basins (Figure 3). Poor fits occurred mostly for extreme flow values (probability of exceedance <20%), in particular low flows for which discharge measurements could produce large errors. Despite these limitations, the goodness of fit values confirmed the good performance of FDM FDC (Figure 3). Mean relative error and mean standard deviations were 0.24 and 0.45 m^3/s, respectively, while mean index of agreement and mean NS coefficient were 0.79 and 0.77, respectively. Consequently, mean FDM data were used to calculate monthly discharges as a streamflow metrics for our subsequent analyses.

3.2. Current Hydrological and Ecological Impacts of Water Intakes

Ongoing water resource management in the Chalpi basin has caused an alteration of the natural flow regime downstream of the 11 operating water intakes (Figure 4). Overall, changes in mean annual water discharge vary between 33% and 82%. Some water intake sites (7, 8, 10, and 11) experienced the highest water withdrawal pressure with values of 81% (0.036 m^3/s), 82% (0.082 m^3/s), 78% (0.307 m^3/s), and 76% (0.361 m^3/s), respectively. In terms of annual variability, some water intakes (7, 10, and 11) withdrew the maximum water in the driest months (August to January).

Impacts of water intake structures on ecosystem health were assessed based on changes in connectivity and stream order measurements compared to premanaged conditions (Figure 5a). Water withdrawal has resulted in significant changes in the distribution of stream order segments in the Chalpi network (Figure 5b). The major impact is a 65% (5.3 km) reduction of the total length of fifth-order stream sections when compared to premanaged conditions (Figure 5c). In contrast, stream sections of orders 1, 3, and 4 show an increase of 4% (2.9 km), 8% (1.4 km), and 9% (1.1 km) in length, respectively, while the length of second-order sections remained unchanged (Figure 5c). These changes in lengths of stream order suppose a reduction from 9.74 to 7.98, and from 0.0051 to 0.0041 of the regional diversity of benthic invertebrates and decomposition rate of organic matter (*Calamagrostis* sp.), respectively. In terms of connectivity, we estimated that water intakes isolate a total 60.1 km of headwater streams.

3.3. Optimal Trade-Off Between Water Withdrawal and Ecological Indicators

The current water intake configuration provides a water withdrawal capacity of 1.29 m^3/s and generates a 32% decrease in stream connectivity and 18% change in stream order when compared to premanagement conditions (with an important impact on the length of fifth-order streams). From the different configurations (2,047), only 496 fulfill the annual water withdrawal for the current urban demand. The optimization analysis of water withdrawal versus connectivity results in the definition of a Pareto frontier with a portfolio of 47 optimal water intake configurations (Figure 6a). The best configuration (higher connectivity) includes five

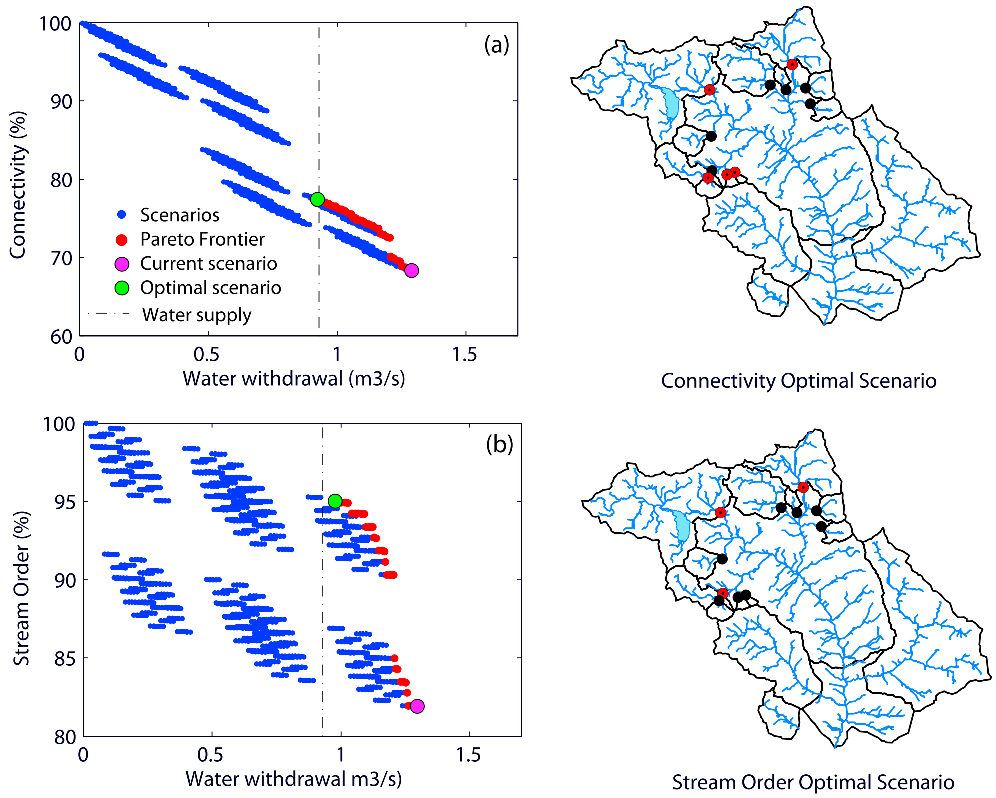

Figure 6. Trade-off between water withdrawal and ecosystem health metrics (a, connectivity; b, stream order) as represented by a Pareto-optimal frontier and associated portfolio maps of water intake configurations (red dots are the intakes that should withdraw the water according to the optimal portfolio, and black dots are the intakes not in the optimal scenario). The dotted line indicates the water withdrawal required for the water supply. Given that we used historical data of water withdrawal of the system (period 2008–2016), the Pareto frontier was defined once the water needs are met.

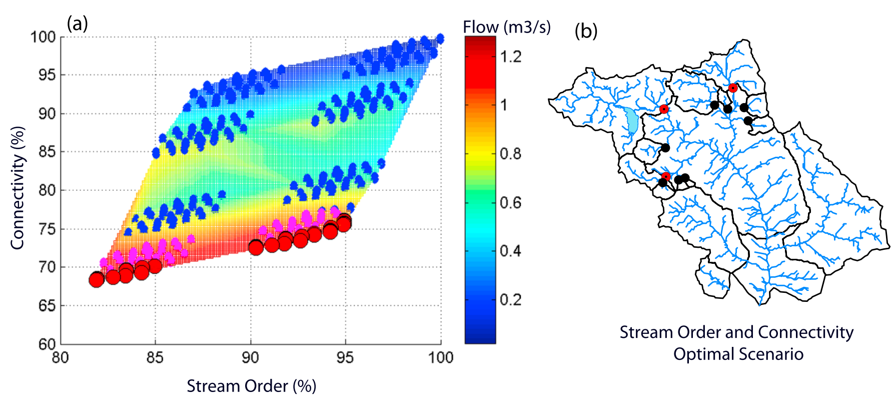

Figure 7. (a) Multi-objective optimization among water withdrawal, connectivity, and stream order. Blue points show the 2,047 different configurations of the system, magenta points represents portfolios that met water needs (>0.93 m^3/s), and red circles show optimal water intake configurations of the Pareto frontier. (b) Associated portfolio map of water intake configuration (red dots are the intakes that should withdraw the water according to the optimal portfolio, and black dots are the intakes not in the optimal portfolio).

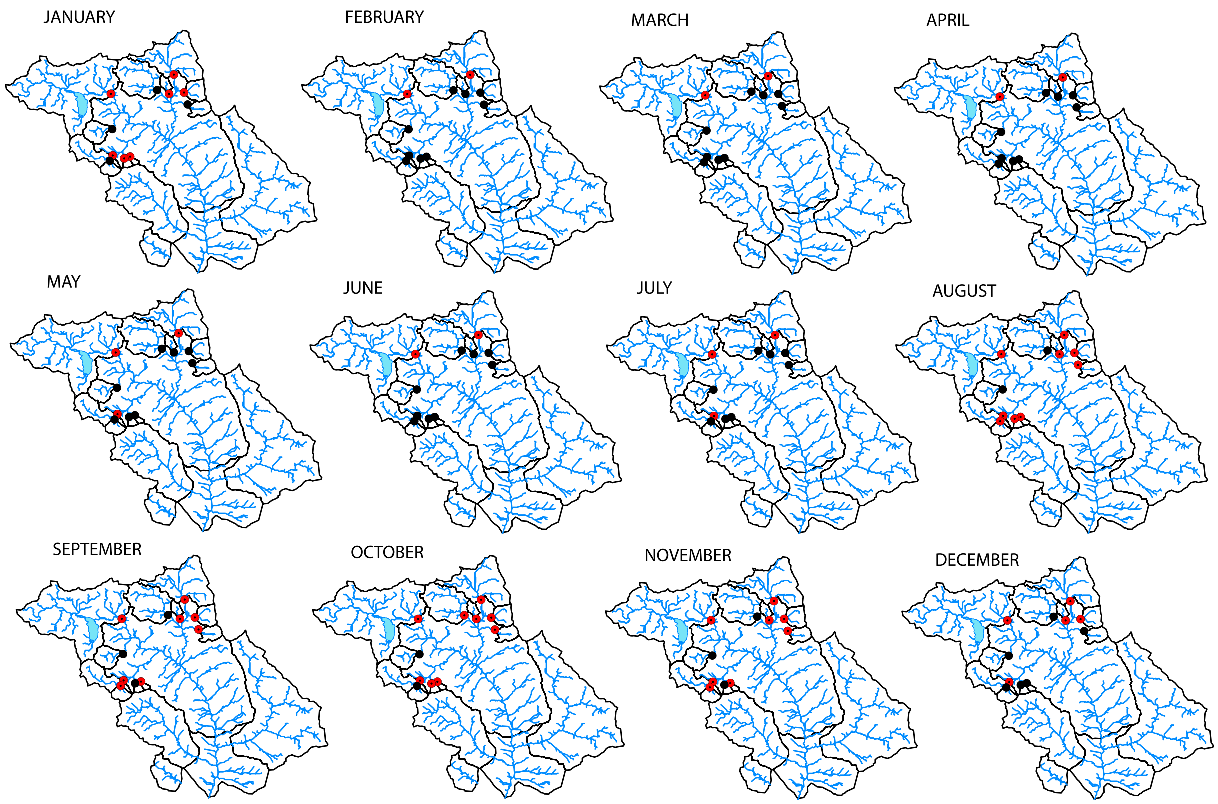

Figure 8. Portfolio maps of the Chalpi basin showing the monthly optimal water intake structure configuration based on a trade-off analysis between water withdrawal, stream order, and stream network connectivity (red dots are the intakes that should withdraw the water according to the optimal portfolio, and black dots are the intakes not in the optimal portfolio).

water intakes with a capacity to supply a water demand of 0.93 m^3/s and a 23% decrease in stream connectivity. The optimization between water withdrawal and maintenance in stream orders generates 32 optimal water intake configurations (Figure 6b). The optimal configuration (minimum changes in stream segment orders) encompassed three water intakes with water withdrawal of 0.98 m^3/s and a change in stream order of 5% relative to premanagement conditions.

Figure 7 highlights the optimization among water withdrawal, connectivity, and stream order. There were 496 scenarios that fulfill mean annual water supply, 32 of which were optimal. The optimal configuration (maximum water withdrawal, maximum connectivity, and maximum maintenance of stream order) includes three water intakes with a 24% decrease in connectivity, 5% decrease in stream order, and a total water withdrawal of 0.98 m^3/s.

Our historical data of water discharge and water withdrawal for supply (sum of 11 water intakes in operation, see Table B1) revealed clear temporal interannual variability with the maximum supply in the wettest months. For this reason, it was possible to develop a multi-objective monthly optimization of needs versus ecological indicators (connectivity and stream order) accounting for the dynamic nature of the water supply system (see Figure 8). When compared to the current water intake configuration, a portfolio of monthly configurations would limit ecological impacts by up to 9.6 percent points (pp) decrease in stream connectivity and up to 13.4 pp decrease in stream order change, respectively, while still meeting the urban water demand. The drier months, August to January, require the greatest number of water intakes (from 5 to 9) to fulfill water needs while the wetter month of June only needs two water intakes.

Additionally, changes in monthly and annual streamflow (see Figures A3a and A3b, respectively) were considered in the optimization process (10% increase and 10% decrease in streamflow) in order to evaluate the optimization results of water intake portfolios for the historical water supply. Overall, results highlight an increase in the number of intakes with the 10% decrease scenario and reductions or little change in the number of intakes with increased monthly and annual streamflow (see Figure A3c and A3d, respectively). However, in some months (April and June) where the difference between streamflow and water supply is higher, the number of intakes for the 10% increase scenario is greater with respect to the other scenarios. In this case, the optimal solution includes a greater number of intakes with smaller contributing area, which therefore generates lower ecological impacts. Finally, optimal solutions for connectivity (see Figures A3e and A3f for monthly and annual results, respectively) and maintenance of stream order (see Figure A3g and A3h for monthly and annual results, respectively) highlight in general the increase in connectivity and increase in the maintenance of stream order with the 10% increase scenario and the opposite for the 10% reduction scenario.

4. Discussion

4.1. Model Limitations

Data scarcity constitutes a serious problem to modeling water resources in tropical regions (e.g., Ibrahim et al., 2015), in particular in mountainous areas. Located near the continental divide of the Pacific and Amazon drainages, our study basin is characterized by complex hydrological behavior where even observed precipitation is underestimated with the current rain gauge network (González-Zeas et al., 2019). Given the large uncertainty in observations, it is therefore challenging to quantify the spatiotemporal variability of streamflow on a daily scale, especially by applying a rainfall-runoff model. Our model used the sum of mean monthly and annual flows at each operating intake as a flow metric of the total water withdrawal; therefore, accounting for daily observations would likely provide different model outcomes, especially at extreme flows. Hence, the temporal scale of work (monthly) as well as the short length of the data record used may affect the results of our optimization process by uncertainty.

However, our model remains an important contribution to enhance water resource management in the region and the reliability of its predictions could be further improved as new hydrological data become available. Beyond its application, our methodological framework is a contribution of potentially broad interest for other regions, as it addresses novel concepts in the study of environmental flows by explicitly integrating the spatial configuration of both environmental variables and the cumulative effects of infrastructures in the watershed (Anderson et al., 2018; Arthington et al., 2018; Wang et al., 2018).

4.2. Regionalization Approach for Monthly Response at Ungauged Sub-Basin

Thanks to the water utility's interest in monitoring flow upstream of water intakes, we were able to develop a simple regional model to construct monthly streamflow data (period 2008–2016). In particular, our FDM FDCs demonstrate a good fit for the general pattern of a daily flow regime in the SGS of our study sub-basins, which is consistent with other studies applied at a basin scale (Flores-López et al., 2016; Masih et al., 2010). However, our approach and results still have some limitations. First, some hydrological processes are difficult to generalize in an empirical way. For example, high flows were difficult to estimate with the proposed method, given that, the monthly temporal scale and the range of available hydrological information (period 2008–2016) prevented the representation of more extreme conditions occurring at lower frequencies. These results support the idea that the length of available flow time series strongly influences model accuracy for floods, as reported in other studies (e.g., Castellarin et al., 2001; Smakhtin et al., 1997).

Second, the applicability of our empirical model may be restricted to relatively small watersheds with a relatively homogeneous land cover. Moreover, because the utility of regionalization process to assess the impact of change in water yield and catchment regulation is limited (Crespo et al., 2011), more complex hydrological models would be needed to assess the impact of land use change on headwater basin hydrology (e.g., Buytaert et al., 2004; Buytaert & Beven, 2009). However, building mechanistic models requires understanding the mechanisms controlling water balance in high-altitude páramo hydrosystems. Several studies have tested hydrological models of different complexity for representing the hydrology of páramo ecosystems by using a diverse array of parameters (e.g., Buytaert & Beven, 2011; Célleri et al., 2010). Ochoa-Toachi et al. (2016) report that the hydrological response of Andean catchments is strongly related to their soil conditions, highlighting that saturation excess flow is a dominant process in the páramo ecosystem. Therefore, infiltration excess overland flow is nearly absent in páramo ecosystems (Buytaert et al., 2008) and surface or near-surface runoff can be an important contributor of river discharge due to the many saturated areas in the páramo.

4.3. Pareto Frontier Solutions

We intended to design eco-friendly water intake portfolios by exploring optimal solutions of the Pareto frontier for trade-off scenarios between water withdrawal pressure and ecological indicators. Other studies have also developed an optimized multicriteria approach to identify the best performing designs within multireservoir systems (e.g., Goor et al., 2010; Herman et al., 2014), dam portfolios for optimizing both energy production and low sediment trapping (Schmitt, 2016), or optimization algorithms for sustainable land use from global to subglobal scales (Seppelt et al., 2013). However, to our knowledge, none of them have jointly addressed hydrological and ecological impacts. Our optimal solutions of the Pareto front highlighted that the size of a managed sub-basin generally prefigured the subsequent impact of water intakes on ecological indicators. For example, a single water intake (no. 1 in Table 1) with a contributing catchment of 0.26 km^2 caused 0.15 km of disconnected rivers without changes in stream order, while another intake (no. 11 in Table 1) with a contributing catchment of 13.28 km^2 generated a total of 32 km of disconnected rivers and a change in stream order from 4 to 1. Also, the large sub-basins had considerable impact on ecological indicators. As a consequence, they were most likely excluded from water portfolios in the optimization process, as long as the sum of water withdrawal in the smaller sub-basins was able to satisfy the water supply.

These results are mainly due to the fact that we assumed that each intake site withdrew the maximum diversion capacity of the water intakes and not only the regulated flow established in the current system operation. Given that environmental flows have not been established in the Ecuador water legislation, it was the easiest way to manage water withdrawal as a first management approach. However, as environmental flows are a key feature of environmental water management (Acreman & Dunbar, 2004; Richter & Thomas, 2007), further refinements of our optimization approach may account for variable amounts of water withdrawn by each intake. Moreover, while our optimization objective functions were well defined (i.e., we maximized both water withdrawal and ecosystem health), formulation constraints were established only for water withdrawal. Indeed, we did not consider any constraints for connectivity and change in stream order as we lacked information related to the ecological thresholds acceptable for these indicators. Even so, our annual optimal configurations revealed a substantial minimization of ecological impacts through a 7.8% increase in connectivity and 13% decrease in stream order change when compared to the actual management plan. Overall, our study confirms that multi-objective optimization algorithms are promising tools for planning more balanced human and ecosystem needs in watersheds exposed to rapid infrastructure building (e.g., Hurford et al., 2013; Yin et al., 2010).

In addition, PO portfolios result in a minimal ecological impact for a given water production capacity equal or higher than the required urban demand. That also implies that increasing water withdrawal capacity from any of the PO portfolio requires increasing impact on ecological indicators. PO solution has the advantage of supplying historical water demand with the minimum connectivity loss and change in stream order with respect to the other PO portfolios. However, the PO solution might not be achievable given the current intake portfolio (with 11 water intakes in operation) and existing operational policies. Therefore, without a concerted strategy of system operation based on an adaptive management, the gap between the PO portfolio and the current portfolio could not be used to improve the ecosystem health.

4.4. Adaptive Water Management

By taking into account the monthly variability in water discharge and water supply in the studied basin, we proposed a *dynamic portfolio* of water intake sites, which may represent a first step toward truly adaptive management (AM) of this water supply system. AM is a *systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices* (Millennium Ecosystem Assessment, 2005). As a result, water AM provides a way to adjust to change and uncertainty in water flow provision and demand. Our results highlighted that implementing a month-by-month management of water intake in operation would allow satisfying water needs while reducing ecological impacts by 9.6 and 13.4 pp (connectivity and stream order, respectively) when compared to the present situation (and 7.8 and 13 pp when compared to a yearly portfolio). They further suggest that the current configuration of water intakes in the Chalpi basin is oversized with respect to a portfolio of monthly configurations, considering that during some periods of the year (e.g., high flows in wet season) a relatively small number of intake structures are needed for water supply. Therefore, by operating a dynamic portfolio of water intakes, Quito's Water Company may have a certain volume of water available every month, considering that under current operation, the monthly reservoir capacity used varies between 18% and 48% of the maximum diversion capacity of the Mogotes design water intake. Therefore, an optimization of the Mogotes reservoir operation could be used to plan adaptation measures in the face of climate change, ecological flow consideration, and increasing potable water demand. Rather than making a single, inflexible intake operational framework, the water company could adapt a more dynamic strategy to meet its needs via the monitoring of provisional portfolios and changing conditions, and incremental adjustments in the light of new information (Castelletti et al., 2013; Susskind et al., 2011). Developing dynamic water intake portfolios through Pareto frontier analyses may be a powerful approach for an efficient AM of water resources at the basin level. In this context, the implementation of different strategies of operation based on AM of the water supply system is necessary (for example, reducing water intake withdrew at each location, optimizing reservoir operation, or considering different amounts of environmental flows). This will provide new portfolios with better ecological benefits and solutions. Thus, to balance human and ecological water needs, Quito's water utility could adopt a new water management paradigm in the study area.

4.5. New Water Management Practices

In order to achieve AM of the complex water system in the study area, the following new management practices would be needed: (1) Improvements in the climate and hydrological monitoring network. Indeed, the uncertainty of our model is due mainly to inherent hydrologic variability and a fundamental lack of hydrological information. (2) Appropriate management of the Mogotes reservoir is of utmost importance to meet both ecological and water demand targets, in particular during the dry season (August to January). (3) New rules of intake and reservoir operation must be adopted by the water company, in particular during the dry season when water quality is most critical. This implies new challenges for the water company such as evaluating organizational implications of adaptive water intake operations, defining procedures at local levels and needs and conditions of their application, minimizing risks of errors associated with new procedural deployment, and planning to demonstrate the value of these efforts for the company and their staff.

5. Conclusion

Through an innovative methodological framework based on a Pareto frontier analysis, we demonstrated a simple, yet relevant model to identify eco-friendly portfolios, which allowed optimizing water management at the basin level. In particular, our monthly optimization analysis affords the opportunity for developing adaptive management across the year by optimizing water use and reducing ecological impacts at a fine temporal scale. This study provides valuable information for water managers when planning the operation of a water supply system, proposing new water management practices focused on strengthening the conditions and capacities of decision makers. Currently, we are working with the Quito's Water Company and Water Fund in charge of source water protection to implement optimal portfolio of water intakes in the Chalpi basin. To this end, further refinements may be applied to our model such as (1) scaling the framework to the entire Papallacta system, (2) including future changes in both water demand (e.g., increasing population) and water availability (e.g., effect of climate change), (3) including economical costs of the different water intake portfolios in the optimization process (e.g., Heinz et al., 2007; Higgins et al., 2011; Jenkins et al., 2004), and (4) implementing environmental flow thresholds downstream of water intakes as part of an adaptive system management (e.g., Pahl-Wostl et al., 2013).

Appendix A

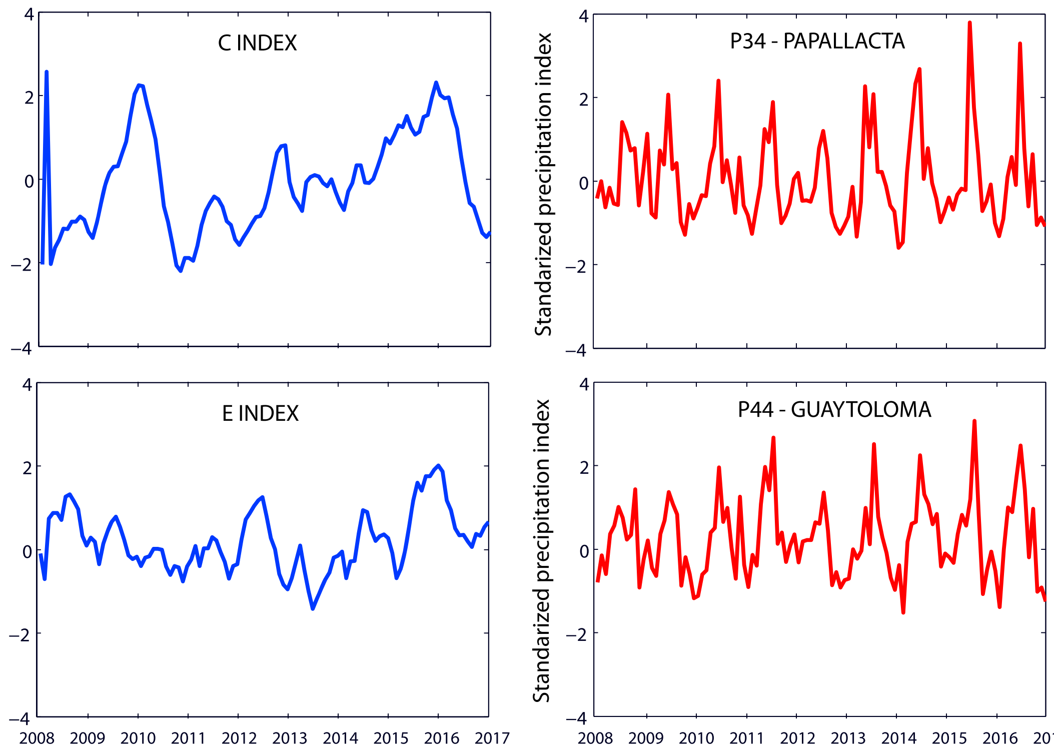

Figure A1. Modes of variability associated with the ENSO phenomenon (blue curves): (1) the increase in the central Pacific (C Index) variability that is associated with stronger cold events, and (2) the reduction in the eastern Pacific (E Index) variability within the cold/moderate warm regime. The standardized precipitation of the two rain gauge stations (P34 and P44) of our study basin in the period of analysis (red curves). ENSO = El Niño–Southern Oscillation.

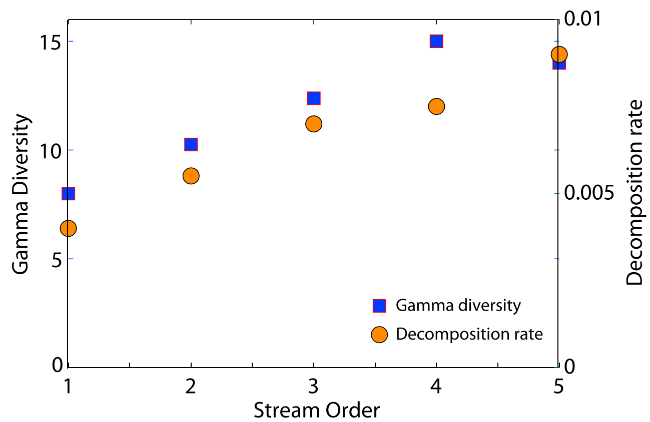

Figure A2. Relationship of the regional diversity of benthic invertebrates and decomposition rate of organic matter (*Calamagrostis* sp.) with increasing stream order in the study basin (see publication of Dangles et al., 2011; Rosero-López et al., 2019, respectively, for details of the methods).

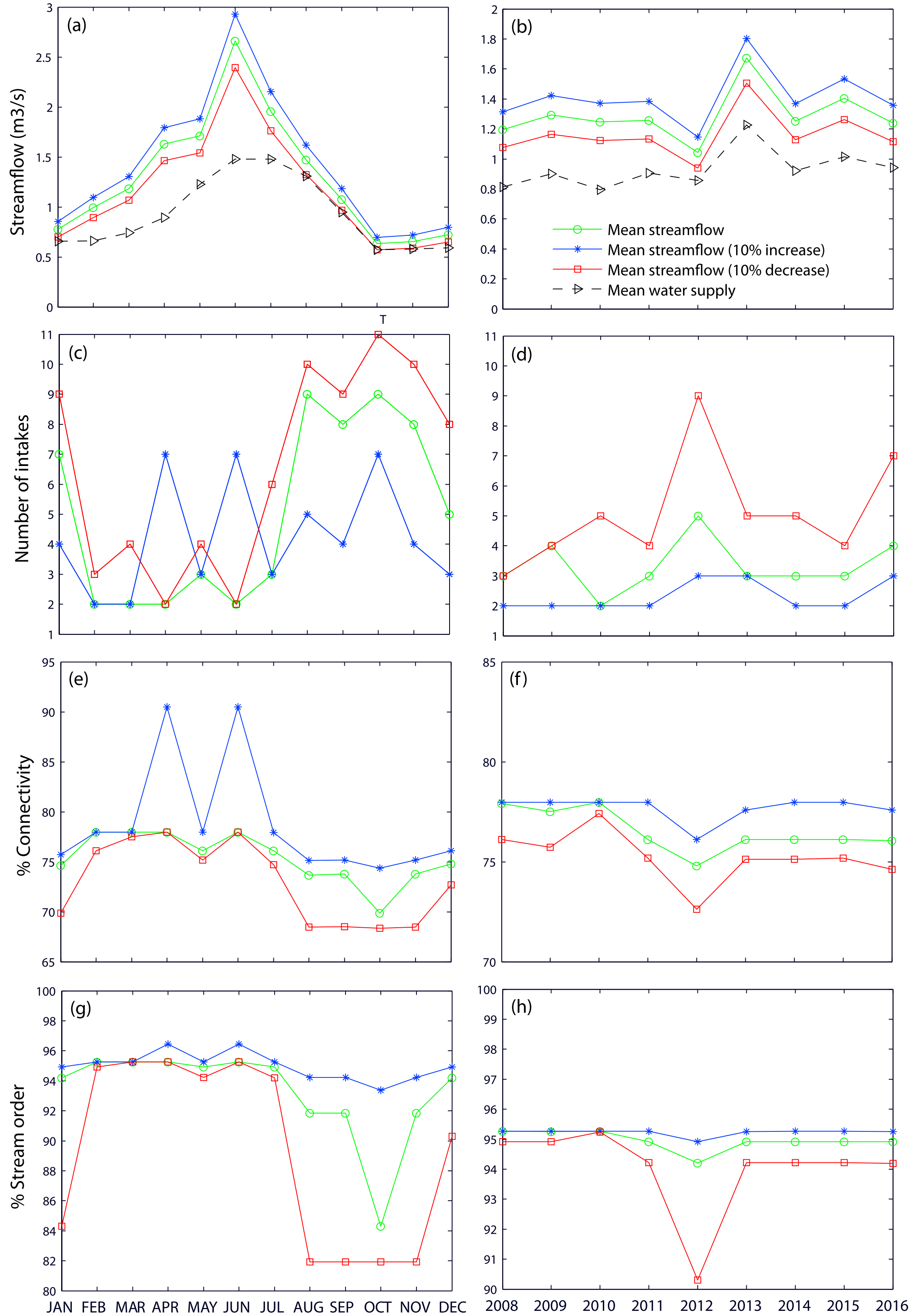

Figure A3. Pareto-optimal (PO) solution for monthly (left panels) and annual (right panels) streamflow scenarios: mean streamflow, 10% increase, and 10% decrease in mean streamflow. (a and b) Comparison between streamflow and water supply. (c and d) Number of intakes of the PO solution. (e and f) Percentage of connectivity of the PO solution. (g and h) Percentage of stream order of the PO solution.

10.1029/2019WR025190

Table A1
Available Data of Field Discharge Measurements in the 16 Sub-Basins

Code of FDM	Location	Water intake	SGS	Period of data	No. years	No. FDM	Mean streamflow (m^3/s)
PN14	Upstream Chalpi Norte	Chalpi Norte	H42	1989–1992, 2000–2016	21	123	0.290
R013	Upstream H13		H13	2000–2016	17	156	3.371
R043	Upstream H38		H38	2011–2015	5	22	1.088
R042	Upstream H39		H39	2011–2014	4	23	1.476
R047	Upstream H40		H40	2011–2015	5	32	0.490
R046	Upstream H41		H41	2011–2015	5	40	0.197
PN11	Upstream Quillugsha 1 and 2	Quillugsha 1 and 2		2001–2016	16	123	0.092
PN13	Upstream Quillugsha 3	Quillugsha 3		2001–2016	16	123	0.022
PN53	Upstream Gonzalito	Gonzalito		2007–2016	10	127	0.067
PN54	Upstream Glaciar	Glaciar		2008–2016	9	37	0.037
PN29	Upstream Mogotes	Mogotes		2000–2016	17	147	0.473
PN30	Upstream Venado	Venado		2001–2016	16	135	0.043
PN09	Upstream Guaytoloma	Guaytoloma		1989–1992, 2001–2016	20	156	0.140
PN64	Upstream Vikingos	Vikingos		2009–2016	8	38	0.032
PN64	Upstream Vikingos 2	Vikingos 2		2009–2016	8	30	0.013
PN64	Upstream Vikingos 3	Vikingos 3		2009–2016	8	29	0.013

Note. FDM = field discharge measurements; SGSs = streamflow gauging stations.

Table A2
Historical Data: Mean and Coefficient of Variation (CV) (Period 2008–2016) of Water Withdrawal for Supply (m^3/s) and Simulated Water Discharge (m^3/s) of the Chalpi Basin (Sum of 11 Water Intakes in Operation)

Description	Jan	Feb	May	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Mean water discharge	0.78	0.99	1.18	1.63	1.71	2.66	1.96	1.47	1.08	0.63	0.65	0.72	1.29
CV water discharge	0.34	0.24	0.16	0.25	0.04	0.27	0.36	0.29	0.30	0.18	0.24	0.27	0.13
Mean water withdrawal (supply)	0.66	0.66	0.74	0.89	1.23	1.48	1.48	1.31	0.95	0.59	0.58	0.59	0.93
CV water withdrawal (supply)	0.37	0.43	0.38	0.30	0.17	0.21	0.31	0.29	0.23	0.18	0.34	0.31	0.14

Note. The monthly variability of water withdrawal for urban supply is the result of the historical operation of the Papallacta system.

Appendix B: Procedure for the Construction of Monthly Streamflow Series

We first classified the 16 sub-basins according to their similarity in both climatic and physiographic characteristics (Morris et al., 2008). For each sub-basin, mean annual precipitation (2008–2016) was obtained using the inverse distance weighted interpolation method applied to rainfall data (see Table A1). We then determined on ArcGis the morphometric parameters of the sub-basins (Palaka & Sankar, 2014; Waikar & Nilawar, 2014) and ran cluster analyses (Euclidean distance) to calculate similarity among groups of sub-basins (Burn, 1989; Unal et al., 2003). Best clusters were identified using the Cophenet index (Ouarda et al., 2006), which measures the degree of consistency and similarity among members of each group. As the value of this indicator is closer to unity, there is greater similarity between the physiographic and climatic characteristics of each group. Second, the suitability of using mean FDM for the construction of the upstream water intake discharges was evaluated by comparing daily flow duration curves (FDCs) obtained from the six SGS with corresponding field data (See Table A2). Third, we extended the SGS time series of the six stations over a common 2008–2016 period, based on the station with the greater length of observed data. Finally, a dimensionless regional monthly flow time series was then generated in each hydrological region, through the standardization of SGS data by an index of streamflow (mean streamflow of the analysis period). The identification of the dimensionless regional model was carried out using cross-validation processes. The product between the monthly regional time series and the mean FDM data provided a simulated monthly time series at each water intake site.

Table B1
Physiographic and Climatic Characteristics of the Sub-Basins of the Study Area

		Vikingos 2	Vikingos 3	Quillugsha 3	Vikingos	Glaciar	Gonzalito	Venados	Guaytaloma	Quillugsha 1 y 2	Chalpi Norte	Mogotes	Encantada	Chalpi A	Chalpi B	Chalpi C	Chalpi grande
Code[a]	Unit	1	2	3	4	5	6	7	8	9	10-H42	11	H38	H39	H40	H41	H13
A	km^2	0.26	0.29	0.36	0.56	0.75	0.95	1.31	1.99	3.62	3.80	13.28	18.53	55.93	9.22	2.72	94.93
P	km	1.46	1.79	2.80	3.04	3.90	3.94	4.85	8.37	8.53	11.18	19.53	22.46	38.17	14.20	7.21	52.32
Zmin	masl	3,884	3,886	3,815	3,850	3,853	3,875	3,872	3,814	3,810	3,829	3,833	3,246	3,239	3,221	3,243	2,881
Zmax	masl	4,117	4,090	3,881	4,082	4,184	4,270	4,120	4,216	4,059	4,006	4,323	4,277	4,284	4,100	3,970	4,284
Zm	masl	4,001	3,988	3,848	3,966	4,019	4,073	3,996	4,015	3,935	3,918	4,072	3,762	3,762	3,661	3,607	3,583
Lp	km	0.64	0.68	1.49	1.04	1.40	1.46	2.19	3.22	3.80	3.53	7.56	8.87	16.01	6.66	3.30	20.27
Nc		1	1	1	2	2	2	2	3	3	3	4	4	5	3	2	5
D1	-	1	1	1	2	3	3	6	8	12	12	39	72	184	29	10	319
Sg	%	48	38	13	34	41	41	34	33	22	25	27	47.97	35.31	42.36	46.01	40
Ap	km	0.40	0.43	0.24	0.54	0.54	0.65	0.60	0.62	0.95	1.07	1.76	2.09	3.49	1.38	0.82	4.68
Scp	%	36.14	30.10	4.43	22.41	23.61	27.08	11.35	12.48	6.55	5.01	6.48	11.63	6.53	13.19	22.00	6.92
K	-	0.81	0.93	1.31	1.14	1.26	1.13	1.19	1.66	1.25	1.61	1.50	1.46	1.43	1.31	1.23	1.50
Lt	km	0.15	0.20	0.79	0.74	1.15	1.84	2.51	3.73	8.45	9.28	32.43	33.34	127.36	18.56	4.96	206.40
Dd	km/km^2	0.57	0.68	2.21	1.33	1.53	1.94	1.92	1.87	2.33	2.44	2.44	1.80	2.28	2.01	1.82	2.17
Ff	-	0.62	0.64	0.16	0.52	0.38	0.45	0.27	0.19	0.25	0.30	0.23	0.24	0.22	0.21	0.25	0.23
Pm	mm	1,950	1,950	1,914	1,860	2,265	2,160	1,868	1,827	1,761	1,813	1,592	2,045	1,857	1,771	1,627	1,869

[a] Area (A), perimeter (P), minimum height (Zmin), maximum height (Zmax), mean height (Zm), length of principal river (Lp), stream order (Nc), number of first-order rivers (D1), mean slope of the basin (Sg), average width of the basin (Ap), mean slope of the river (Scp), gravelious index (K), total length of the rivers (Lt), drainage density (Dd), shape factor (Ff), and precipitation (Pm).

Acknowledgments

This work was part of the project *Développer des solutions pour la gestion durable et adaptative des ressources en eau dans les páramos de la ville de Quito (Équateur)*—CHALPI-FLOW—funded by the Agence Française pour le Développement (AFD), in collaboration with the French Institute for a sustainable Development (IRD, convention 2017000345). The financial support to O. D. by the ECOBIO Department of IRD for his stay at Cornell University is greatly appreciated. The data used are available at https://github.com/duniapgz/CHALPI-FLOW.git. The authors are thankful to Francois Sueur, from Division Eau et Assainissement of the AFD, Nelson Hairston and the Limnology Group in the Department of Ecology and Evolutionary Biology at Cornell University, and the water resources management group of Quito's Water Company.

References

Acreman, M. C., & Dunbar, M. J. (2004). Defining environmental river flow requirements—A review. *Hydrology and Earth System Sciences*, *8*(5), 861–876. https://doi.org/10.5194/hess-8-861-2004

Alipour, M. H., & Kibbler, K. M. (2018). A framework for streamflow prediction in the world's most severely data-limited regions: Test of applicability and performance in a poorly-gauged region of China. *Journal of Hydrology*, *557*, 41–54. https://doi.org/10.1016/j.jhydrol.2017.12.019

Anderson, E. P., Jenkins, C. N., Heilpern, S., Maldonado-Ocampo, J. A., Carvajal-Vallejos, F. M., Encalada, A. C., et al. (2018). Fragmentation of Andes-to-Amazon connectivity by hydropower dams. *Science Advances*, *4*(1), eaao1642. https://doi.org/10.1126/sciadv.aao1642

Arthington, A. H., Bhaduri, A., Bunn, S. E., Jackson, S. E., Tharme, R. E., Tickner, D., et al. (2018). The Brisbane declaration and global action agenda on environmental flows. *Frontiers in Environmental Science*, *6*, 45. https://doi.org/10.3389/fenvs.2018.00045

Auerbach, D., Buchanan, B. P., Alexiades, A. V., Anderson, E. P., Encalada, A. C., Larson, E. I., et al. (2016). Towards catchment classification in data-scarce regions. *Ecohydrology*, *9*(7), 1235–1247. https://doi.org/10.1002/eco.1721

Bracken, L. J., & Croke, J. (2007). The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. *Hydrological Processes*, *21*, 1749–1763.

Brown, C. M., Lund, J. R., Cai, X., Reed, P. M., Zagona, E. A., Ostfeld, A., et al. (2015). The future of water resources systems analysis: Toward a scientific framework for sustainable water management. *Water Resources Research*, *51*, 6110–6124. https://doi.org/10.1002/2015WR017114

Burn, D. H. (1989). Cluster analysis as applied to regional flood frequency. *Journal of Water Resources Planning and Management*, *115*(5), 567–582. https://doi.org/10.1061/(ASCE)0733-9496(1989)115:5(567)

Buytaert, W., & Beven, K. (2009). Regionalization as a learning process. *Water Resources Research*, *45*, W11419. https://doi.org/10.1029/2008WR007359

Buytaert, W., & Beven, K. (2011). Models as multiple working hypotheses: Hydrological simulation of tropical alpine wetlands. *Hydrological Processes*, *25*(11), 1784–1799. https://doi.org/10.1002/hyp.7936

Buytaert, W., Célleri, R., De Bièvre, B., Cisneros, F., Wyseure, G., Deckers, J., & Hofstede, R. (2006). Human impact on the hydrology of the Andean páramos. *Earth-Science Reviews*, *79*(1-2), 53–72. https://doi.org/10.1016/j.earscirev.2006.06.002

Buytaert, W., De Biévre, B., Célleri, R., Cisneros, F., Wyseure, G., & Deckers, S. (2008). Comment on "Human impacts on headwater fluvial systems in the northern and central Andes". (Carol P. Harden. Geomorphology 79, 249-263). *Geomorphology*, *96*(1-2), 239–242. https://doi.org/10.1016/j.geomorph.2007.04.003

Buytaert, W., De Biévre, B., Wyseure, G., & Deckers, J. (2004). The use of the linear reservoir concept to quantify the impact of land use changes on the hydrology of catchments in the Ecuadorian Andes. *Hydrology and Earth System Sciences*, *8*(1), 108–114. https://doi.org/10.5194/hess-8-108-2004

Castellarin, A., Burn, D. H., & Brath, A. (2001). Assessing the effectiveness of hydrological similarity measures for regional flood frequency analysis. *Journal of Hydrology*, *241*(3-4), 270–285. https://doi.org/10.1016/S0022-1694(00)00383-8

Castelletti, A., Pianosi, F., & Restelli, M. A. (2013). A multiobjective reinforcement learning approach to water resources systems operation: Pareto frontier approximation in a single run. *Water Resources Research*, *49*, 3476–3486. https://doi.org/10.1002/wrcr.20295

Cauvy-Fraunié, S., Espinosa, R., Andino, P., Dangles, O., & Jacobsen, D. (2014). Relationships between stream macroinvertebrate communities and new flood-based indices of glacial influence. *Freshwater Biology*, *59*(9), 1916–1925. https://doi.org/10.1111/fwb.12395

Célleri, R., Buytaert, W., De Biévre, B., Tobón, C., Crespo, P., Molina, J., & Feyen, J. (2010). Understanding the hydrology of tropical Andean ecosystems through an Andean Network of Basins. *IAHS-AISH Publication*, *336*, 209–212. https://doi.org/10.13140/2.1.4187.3608

Cibin, R., & Chaubey, I. (2015). A computationally efficient approach for watershed scale spatial optimization. *Environmental Modelling and Software*, *66*, 1–11. https://doi.org/10.1016/j.envsoft.2014.12.014

Cordero, A., Torregrosa, J. R., & Vassileva, M. P. (2011). Three-step methods with optimal eighth-order convergence. *Journal of Computational and Applied Mathematics*, *235*(10), 3189–3194. https://doi.org/10.1016/j.cam.2011.01.004

Crespo, P. J., Feyen, J., Buytaert, W., Bücker, A., Breuer, L., Frede, H. G., & Ramírez, M. (2011). Identifying controls of the rainfall-runoff response of small catchments in the tropical Andes (Ecuador). *Journal of Hydrology*, *407*(1-4), 164–174. https://doi.org/10.1016/j.jhydrol.2011.07.021

Dangles, O., Crespo-Pérez, V., Andino, P., Espinosa, R., Calvez, R., & Jacobsen, D. (2011). Predicting richness effects on ecosystem function in natural communities: Insights from high-elevation streams. *Ecology*, *92*(3), 733–743. https://doi.org/10.1890/10-0329.1

Deb, K. (2002). A Fast Elitist Multiobjective Genetic Algorithm: NSGA-II. *IEEE Transactions on Evolutionary Computation*, *6*(2), 182–197. https://doi.org/10.1109/4235.996017

Deb, K. (2004). *Multi-objective optimization using evolutionary algorithms*. Kanpur, India: Departmental of mechanical Engineering. Indian Institute of technology.

Dile, Y. T., & Srinivasan, R. (2014). Evaluation of CFSR climate data for hydrologic prediction in data-scare watersheds: An application in the Blue Nile river basin. *Journal of the American Water Resources Association*, *50*(5), 1226–1241. https://doi.org/10.1111/jawr.12182

Erazo, B., Bourrel, L., Frappart, F., Chimborazo, O., Labat, D., Dominguez-Granda, L., et al. (2018). Validation of satellite estimates (Tropical Rainfall Measuring Mission, TRMM) for rainfall variability over the Pacific slope and Coast of Ecuador. *Water*, *10*(2), 213. https://doi.org/10.3390/w10020213

Feio, M. J., Alves, T., Boavida, M., Medeiros, A., & Graca, A. S. (2010). Functional indicators of stream health: A river-basin approach. *Freshwater Biology*, *55*(5), 1050–1065. https://doi.org/10.1111/j.1365-2427.2009.02332.x

Fekete, B. M., & Vörösmarty, C. J. (2004). Uncertainties in precipitation and their impacts on runoff estimates. *Journal of Climate*, *17*(2), 294–304. https://doi.org/10.1175/1520-0442(2004)017<0294:UIPATI>2.0.CO;2

Finer, M., & Jenkins, C.N. (2012). Proliferation of hydroelectric dams in the Andean Amazon and implications for Andes-Amazon connectivity. *PLoS ONE*, *7*(4), E35126, e35126, DOI: https://doi.org/10.1371/journal.pone.0035126.

Flores-López, F., Galistsi, S. E., Escobar, M., & Purkey, D. (2016). *Water*, *8*(3), 8. https://doi.org/10.3390/w8030094

Freeman, M. C., Pringle, C. M., & Jackson, C. R. (2007). Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association*, *43*(1), 5–14. https://doi.org/10.1111/j.1752-1688.2007.00002.x

Fryirs, K. (2013). (Dis)Connectivity in catchment sediment cascades: A fresh look at the sediment delivery problem. *Earth Surface Processes and Landforms*, *38*(1), 30–46. https://doi.org/10.1002/esp.3242

González-Zeas, D., Erazo, B., Lloret, P., De Biévre, B., Steinschneider, S., & Dangles, O. (2019). Linking global climate change to local water availability: Limitations and prospects for a tropical mountain watershed. *Science of the Total Environment*, *650*(Pt 2), 2577–2586. https://doi.org/10.1016/j.scitotenv.2018.09.309

Goor, Q., Halleux, C., Mohamed, Y., & Tilmant, A. (2010). Optimal operation of a multipurpose multireservoir system in the Eastern Nile River basin. *Hydrology and Earth System Sciences*, *14*(10), 1895–1908. https://doi.org/10.5194/hess-14-1895-2010

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

Hartmann, D. L., Tank, A. M. K., Rusticucci, M., Alexander, L. V., Brönnimann, S., Charabi, Y. A. R., & Soden, B. J. (2013). Observations: Atmosphere and surface. In *Climate Change 2013 the Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (Chap. 2, 96 pp.). Cambridge: Cambridge University Press.

Heinz, I., Pulido-Velazquez, M., Lund, J. R., & Andreu, J. (2007). Hydro-economic modeling in river basins management: Implications and applications for the European Water Framework Directive. *Water Resources Management*, *21*(7), 1103–1125. https://doi.org/10.1007/s11269-006-9101-8

Herman, J. D., Zeff, H. B., Reed, P. M., & Characklis, G. W. (2014). Beyond optimality: Multistakeholder robustness tradeoffs for regional water portfolio planning under deep uncertainty. *Water Resources Research*, *50*, 7692–7713. https://doi.org/10.1002/2014WR015338

Higgins, A. J., Bryan, B. A., Overton, I. C., Holland, K., Lester, R. E., King, D., et al. (2011). Integrated modelling of cost-effective siting and operation of flow-control infrastructure for river ecosystem conservation. *Water Resources Research*, *47*, W05519. https://doi.org/10.1029/2010WR009919

Hof, J. G., & Bevers, M. (1998). *Spatial optimization for managed ecosystems*. New York: Columbia University Press.

Hofstede, R., Segarra, P., & Mena, P. V. (2003). *Los Páramos del Mundo*. Quito: Global Peatland Initiative/NC-IUCN/EcoCiencia.

Hughes, R. M., Kaufmann, P. R., & Weber, M. H. (2011). National and regional comparisons between Sthraler order and stream size. *Journal of the North American Benthological Society*, *30*(1), 103–121. https://doi.org/10.1899/09-174.1

Hurford, A. P., Huskova, I., Harou, J. J., Wallingford, H. R., Park, H., Lane, B., & Ox, W. (2013). Using many-objective trade-off analysis to help dams promote economic development, protect the poor and enhance ecological health. *Environmental Science & Policy*, *38*, 72–86.

Ibrahim, B., Wisser, D., Barry, B., Fowe, T., & Aduna, A. (2015). Hydrological predictions for small ungauged watersheds in the Sudanian zone of the Volta basin in West Africa. *Journal of Hydrology: Regional Studies*, *4*, 386–397.

Jacobsen, D., & Dangles, O. (2017). *Ecology of high altitude waters* (p. 284). New York, NY: Oxford University Press.

Jager, H. I., Efroymson, R. A., Opperman, J. J., & Kelly, M. R. (2015). Spatial design principles for sustainable hydropower development in river basins. *Renewable and Sustainable Energy Reviews*, *45*, 808–816. https://doi.org/10.1016/j.rser.2015.01.067

Jenkins, M. W., Lund, J. R., Howitt, R. E., Draper, A. J., Msangi, S. M., Tanaka, S. K., et al. (2004). Optimization of California's water supply system: Results and insights. *Journal of Water Resources Planning and Management*, *130*(4), 271–280. https://doi.org/10.1061/(ASCE)0733-9496

Kibler, K. M., & Tullos, D. D. (2013). Cumulative biophysical impact of small and large hydropower development in Un River, China. *Water Resources Research*, *49*, 3104–3118. https://doi.org/10.1002/wrcr.20243

Kougias, I. P., & Theodossiou, N. P. (2013). Application of the Harmony Search optimization algorithm for the solution of the multiple dam system scheduling. *Optimization and Engineering*, *14*(2), 331–344. https://doi.org/10.1007/s11081-011-9183-x

Krause, P., Boyle, D. P., & Bäse, F. (2005). Comparison of different efficiency criteria for hydrological model assessment. *Advances in Geosciences*, *5*, 89–97. https://doi.org/10.5194/adgeo-5-89-2005

Kuby, M. J., Fagan, W. F., ReVelle, C. S., & Graf, W.l. (2005). A multiobjective optimization model for dam removal: An example trading off salmon passage with hydropower and water storage in the Willamette basin. *Advances in Water Resources*, *28*(8), 845–855. https://doi.org/10.1016/j.advwatres.2004.12.015

Kult, J. M., Fry, L. M., Gronewold, A. D., & Choi, W. (2014). Regionalization of hydrologic response in the Great Lakes basin: Consideration of temporal scales of analysis. *Journal of Hydrology*, *519*, 2224–2237. https://doi.org/10.1016/j.jhydrol.2014.09.083

Kuzmin, V., Pivovarova, I., Shemanaev, K., Sokolova, D., Batyrov, A., Tran, N. A., & Dang, D. (2019). Method of prediction of the stream flows in poorly gauged and ungauged basins. *Journal of Ecological Engineering*, *20*, 180–187.

Kwang-Seuk, J., Dong-Gyun, H., Myeoung-Seop, B., Jong-Chul, J., Han-Gook, K., Dong-Kyun, K., & Gea-Jae, J. (2010). Stream modification patterns in a river basin: Field survey and self-organizing map (SOM) application. *Ecological Informatics*, *5*, 293–303.

Latrubesse, E. M., Arima, E. Y., Dunne, T., Park, E., Baker, V. R., d'Horta, F. M., & Ribas, C. C. (2017). Damming the rivers of the Amazon basin. *Nature*, *546*(7658), 363–369. https://doi.org/10.1038/nature22333

Lautenbach, S., Volk, M., Strauch, M., & Whittaker, G. (2013). Optimization-based trade-off analysis of biodiesel crop production for managing an agricultural catchment. *Environmental Modelling and Software*, *48*, 98–112. https://doi.org/10.1016/j.envsoft.2013.06.006

Lautenbach, S., Volk, M., Strauch, M., Whittaker, G., & Seppelt, R. (2014). Management support by optimization-based trade-off analysis—The example of biodiesel crop production. *Geo-Öko*, *35*(1-2), 39–77.

Legates, D. R., & McCabe, G. J. Jr. (1999). Evaluating the use of "goodness of fit" measures in hydrologic and hydroclimatic model validation. *Water Resources Research*, *35*, 233–241.

Liu, Y., Gupta, H., Springer, E., & Wagener, T. (2008). Linking science with environmental decision making: Experiences from an integrated modeling approach to supporting sustainable water resources management. *Environmental Modelling & Software*, *23*, 846–858.

Martínez, A., Larrañaga, A., Basaguren, A., Pérez, J., Casado, C., Casas, J. J., et al. (2016). Climate modulates the magnitude of the effects of flow regulation on leaf-litter decomposition. *Aquatic Sciences*, *79*(3), 507–514. https://doi.org/10.1007/s00027-016-0513-0

Masih, I., Uhlenbrook, S., Maskey, S., & Ahmad, M. D. (2010). Regionalization of a conceptual rainfall-runoff model on similarity of the flow duration curve: A case study from the semiarid Karkheh basin, Iran. *Journal of Hydrology*, *391*(1-2), 188–201. https://doi.org/10.1016/j.jhydrol.2010.07.018

Mendoza-Lera, C., Larrañaga, A., Pérez, J., Descals, E., Martínez, A., Moya, O., et al. (2012). Headwater reservoirs weaken terrestrial-aquatic linkage by slowing leaf-litter processing in downstream regulated reaches. *River Research and Applications*, *28*(1), 13–22. https://doi.org/10.1002/rra.1434

Millennium Ecosystem Assessment (2005). *Ecosystems and human well-being: Synthesis*. Washington, DC: Island Press.

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., & Velth, T. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE*, *50*(3), 885–900. https://doi.org/10.13031/2013.23153

Morris, A. J., Donovan, J. J., & Strager, M. (2008). Geospatial analysis of climatic and geomorphic interactions influencing stream discharge, Appalachian Mountains, USA. *Environmental Modeling and Assessment*, *14*(1), 73–84. https://doi.org/10.1007/s10666-008-9145-7

Nanzyo, M., Shoji, S., & Dahlgren, R. (1993). *Volcanic ash soils: Genesis, properties and utilisation*, *Developments in Soil Science*, (Vol. 21). Amsterdam: Elsevier.

Nash, J. E., & Sutcliffe, J. V. (1970). River flow forecasting through conceptual models, Part I—A discussion of principles. *Journal of Hydrology*, *10*(3), 282–290. https://doi.org/10.1016/0022-1694(70)90255-6

Nepal, S., Flügel, W. A., Krause, P., Fink, M., & Fischer, C. (2017). Assessment of spatial transferability of process-based hydrological model parameters in two neighbouring catchments in the Himalayan region. *Hydrological Processes*, *31*(16), 2812–2826. https://doi.org/10.1002/hyp.11199

Null, S. E., Medellin-Azuara, J., Escriva-Bou, A., Lent, M., & Lund, J. R. (2014). Optimizing the dammed: Water supply losses and fish habitat gains from dam removal in California. *Journal of Environmental Management*, *136*, 121–131. https://doi.org/10.1016/j.jenvman.2014.01.024

Oaks, R. M., Gido, K. B., Falke, J. A., Olden, J. D., & Brock, B. (2005). Modelling of stream fishes in the Great Plains, USA. *Ecology of Freshwater Fish*, *14*(4), 361–374. https://doi.org/10.1111/j.1600-0633.2005.00110.x

Ochoa-Toachi, B. F., Buytaert, W., De Biévre, B., Célleri, R., Crespo, P., Villacís, M., et al. (2016). Impacts of land use on hydrological response of tropical Andean catchments. *Hydrological Processes*, *30*(22), 4074–4089. https://doi.org/10.1002/hyp.10980

Ouarda, T. B. M. J., Cunderlik, J. M., St-Hilairea, A., Barbet, M., Bruneau, P., & Bobeé, B. (2006). Data-based comparison of seasonality-based regional flood frequency methods. *Journal of Hydrology*, *330*(1-2), 329–339. https://doi.org/10.1016/j.jhydrol.2006.03.023

Pahl-Wostl, C., Arthington, A., Bogardi, J., Bunn, S. E., Hoff, H., Lebel, L., et al. (2013). Environmental flows and water governance: Managing sustainable water uses. *Current Opinion in Environmental Sustainability*, *5*, 1–11.

Palaka, R., & Sankar, G. J. (2014). Study of watershed characteristics using Google elevation service. *India Geospatial Digest*, *2*, 18–26.

Palmer, L. (2014). The next water cycle. *Nature Climate Change*, *4*(11), 949–950. https://doi.org/10.1038/nclimate2420

Peñuela, C. A., & Granada, M. (2007). Multi-objective optimization using elitist non-dominated sorting genetic algorithm (NSGA-II). *Scientia et Technica*, *35*, 175–180.

Pool, S., Viviroli, D., & Seibert, J. (2017). Prediction of hydrographs and flow-duration curves in almost ungauged catchments: Which measurements are most informative for model calibration? *Journal of Hydrology*. https://doi.org/10.1016/j.jhydrol.2017.09.037

Price, J. S., & Waddington, J. M. (2000). Advances in Canadian wetland hydrology and biogeochemistry. *Hydrological Processes*, *14*(9), 1579–1589. https://doi.org/10.1002/1099-1085(20000630)14:9<1579::AID-HYP76>3.0.CO;2-#

Pringle, C. (2003). What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes*, *17*(13), 2685–2689.

Pumo, D., Viola, F., & Noto, L. V. (2016). Generation of natural runoff monthly series at ungauged sites using a regional regressive model. *Water*, *8*(5). https://doi.org/10.3390/w8050209

Reed, P. M., Hadka, D., Herman, J. D., Kasprzyk, J. R., & Kollat, J. B. (2013). Evolutionary multiobjective optimization in water resources: The past, present, and future. *Advances in Water Resources*, *51*, 438–456. https://doi.org/10.1016/j.advwatres.2012.01.005

Richter, B. D., Mathews, R., Harrison, D. L., & Wigington, R. (2003). Ecologically sustainable water management. *Managing River Flows for Ecological Integrity*, *13*(1), 206–224.

Richter, B.D., & Thomas, G.A. (2007). Restoring environmental flows by modifying dam operations. *Ecology and Society*, *12*(1): 12. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art12/

Rosero-López, D., Walter, M. T., Flecker, A. S., Lloret, P., De Bièvre, B., González-Zeas, D., et al. (2019). Streamlined eco-engineering approach helps define environmental flows for tropical Andean headwaters. *Freshwater Biology*, *64*(7), 1315–1325. https://doi.org/10.1111/fwb.13307

Salazar, J. Z., Reed, P. M., Herman, J. D., Giuliani, M., & Castelletti, A. (2016). A diagnostic assessment of evolutionary algorithms for multi-objective surface water reservoir control. *Advances in Water Resources*, *92*, 172–185. https://doi.org/10.1016/j.advwatres.2016.04.006

Schmitt, R. (2016). CASCADE—A framework for modeling fluvial sediment connectivity and its application for designing low impact hydropower portfolios. Doctoral Dissertation. Politecnico di Milano, 245p.

Segura, H., Junquas, C., Espinoza, J. C., Vuille, M., Jauregui, Y., Rabatel, A., et al. (2019). New insights into the rainfall variability in the tropical Andes on seasonal and interannual time scales. *Climate Dynamics*, *53*(1-2), 405–426. https://doi.org/10.1007/s00382-018-4590-8

Seppelt, R., Lautenbach, S., & Volk, M. (2013). Identifying trade-offs between ecosystem services, land use, and biodiversity: A plea for combining scenario analysis and optimization on different spatial scales. *Current Opinion in Environment Sustainability*, *5*, 1–6.

Singh, R. D., Mishra, S. K., & Chowdhary, H. (2001). Regional flow-duration models for large number of ungauged Himalayan catchments for planning micro hydro projects. *Journal of Hydrologic Engineering*, *6*(4), 310–316. https://doi.org/10.1061/(ASCE)1084-0699(2001)6:4(310)

Smakhtin, V. Y., Hughes, D. A., & Creuse-Naudine, E. (1997). Regionalization of daily flow characteristics in part of the Eastern Cape, South Africa. *Hydrological Sciences Journal*, *42*(6), 919–936. https://doi.org/10.1080/02626669709492088

Strecker, A. L., & Brittain, J. T. (2017). Increased habitat connectivity homogenizes freshwater communities: Historical and landscape perspectives. *Journal of Applied Ecology*, *54*(5), 1343–1352. https://doi.org/10.1111/1365-2664.12882

Susskind, L., Camacho, A. E., & Schenk, T. (2011). A critical assessment of collaborative adaptive management in practice. *Journal of Applied Ecology*, *49*(1), 47–51. https://doi.org/10.1111/j.1365-2664.2011.02070.x

Takahashi, K., Montecinos, A., Goubanova, K., and, Dewitte, B. (2011). ENSO regimes: Reinterpreting the canonical and Modoki El Niño. *Geophysical Research Letters*, *38*, L10704, doi:https://doi.org/10.1029/2011GL047364.

Tegegne, G., & Young-Oh, K. (2018). Modelling ungauged catchments using the catchment runoff response similarity. *Journal of Hydrology*, *564*, 452–466. https://doi.org/10.1016/j.jhydrol.2018.07.042

Tuo, Y., Duan, Z., Disse, M., & Chiogna, G. (2016). Evaluation of precipitation input for SWAT modeling in Alpine catchment: A case study in the Adige river basin (Italy). *Science of the Total Environment*, *573*, 66–82. https://doi.org/10.1016/j.scitotenv.2016.08.034

Unal, Y., Kindap, T., & Karaca, M. (2003). Redefining the climate zones of Turkey using cluster analysis. *International Journal of Climatology*, *23*(9), 1045–1055. https://doi.org/10.1002/joc.910

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*, *37*(1), 130–137. https://doi.org/10.1139/f80-017

Viviroli, D., & Seibert, J. (2015). Can a regionalized model parameterization be improved with a limited number of runoff measurements? *Journal of Hydrology*, *529*, 49–61. https://doi.org/10.1016/j.jhydrol.2015.07.009

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., et al. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555–561. https://doi.org/10.1038/nature09440

Waikar, M. L., & Nilawar, A. P. (2014). Morphoetric analysis of a drainage basin using Geographical Information System: A case study. *International Journal of Multidisciplinary and Current Research*, *2*, 179–184.

Wang, Y., Zhang, N., Wang, D., Wu, J., & Zhang, X. (2018). Investigating the impacts of cascade hydropower development on the natural flow regime in the Yangtze River, China. *Science of the Total Environment*, *624*, 1187–1194. https://doi.org/10.1016/j.scitotenv.2017.12.212

Willmott, C. J. (1981). On the validation of models. *Physical Geography*, *2*(2), 184–194. https://doi.org/10.1080/02723646.1981.10642213

Wood, P. J., Hannah, D. M., & Sadler, J. P. (2008). *Hydroecology and ecohydrology: Past, present and future*. Chichester, England: Wiley. https://doi.org/10.1002/9780470010198

Yin, X., Yang, Z., Yang, W., Zhao, Y., & Chen, H. (2010). Optimized reservoir operation to balance human and riverine ecosystem needs: Model development, and a case study for the Tanghe reservoir, Tang river basin, China. *Hydrological Processes*, *24*, 461–471.

Yu, P. S., Yang, T. C., & Wang, Y. C. (2002). Uncertainty analysis of regional flow duration curves. *Journal of Water Resources Planning and Management ASCE*, *128*(6), 424–430. https://doi.org/10.1061/(ASCE)0733-9496(2002)128:6(424)