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Improving the ecohydrological and economic efficiency of Small Hydropower Plants with water diversion

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ABSTRACT

Water exploitation for energy production from Small Hydropower Plant (SHP) is increasing despite human pressure on freshwater already being very intense in several countries. Preserving natural rivers thus requires deeper understanding of the global (i.e., ecological and economic) efficiency of flow-diversion practice. In this work, we show that the global efficiency of SHP river intakes can be improved by non-proportional flow-redistribution policies. This innovative dynamic water allocation defines the fraction of water released to the river as a nonlinear function of river runoff. Three swiss SHP case studies are considered to systematically test the global performance of such policies, under both present and future hydroclimatic regimes. The environmental efficiency is plotted versus the economic efficiency showing that efficient solutions align along a (Pareto) frontier, which is entirely formed by non-proportional policies. On the contrary, other commonly used distribution policies generally lie below the Pareto frontier. This confirms the existence of better policies based on non-proportional redistribution, which should be considered in relation to implementation and operational costs. Our results recommend abandoning static (e.g., constant-minimal-flow) policies in favour of non-proportional dynamic ones towards a more sustainable use of the water resource, also considering changing hydroclimatic scenarios.

1. Introduction

Small Hydropower Plants (SHP) are a class of low-capacity (typically lower than 10 MW) energy production power plants often based on either flow diversion from water intakes or run-of-the-river water use concepts. Whenever there is water diversion from the river, and depending on the operational policy, a residual flow is generally released downstream the intake. In part driven by the fear of a Fukushima scenario and in view of limiting carbon emissions from fossil fuel power generation, energy production is turning to renewable sources. Among others, SHP installations are growing although the installed global (i.e., all power plant types) hydropower potential in some countries already exceeds 70% of the feasible potential (e.g., USA and Switzerland, see Fig. 1). Some other country, e.g the United Kingdoms, currently uses less than 60% of its potential. Indeed, due to both economic reasons and limitations of technology, sites with lower hydraulic heads or power outputs were not considered as suitable for energy production in the past. This offers some interesting development opportunities for the future provided that environmentally friendly solutions are adopted for further exploitation of freshwater resources. In this work we show how the global (i.e. economic and environmental) performance of flow-diversion practice for feeding SHPs can be improved by engineering a new class of dynamic residual flow policies, and will show this on three real SHP case studies.

We focus on SHPs without significant storage capacity, which withdraw water from an intake installed at a specific river transect, and return it downstream below the power house (Fig. 2). Among SHPs, the latter is the scheme with the highest environmental impact in terms of affected riverine corridor length. In the majority of the cases, SHPs also apply residual flow policies set to constant minimal amounts (minimum flow release, henceforth referred to as MFR). Politically simple to define, MFR policies have no specific ecological basis, and their extensive use systematically affected first the morphology and then the ecosystem of river corridors (Moyle and Mount, 2007; Poff et al., 2007). As today's society acknowledges the value of ecosystem services under resource

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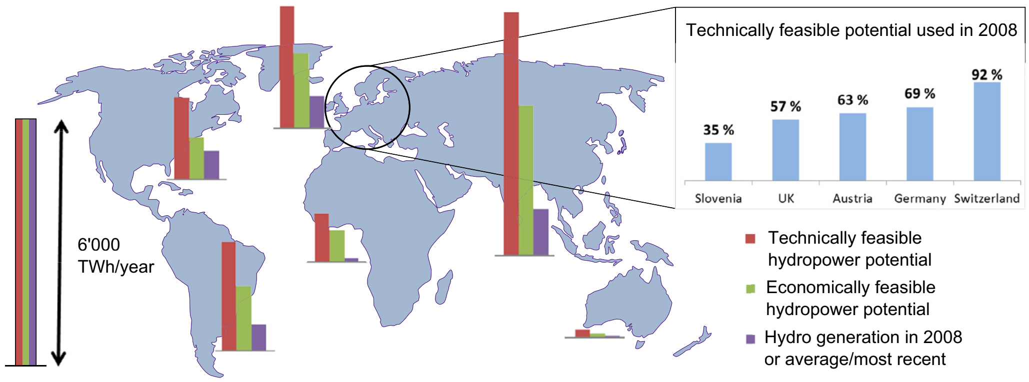

Fig. 1. Worldwide consumption of hydropower energy potentials. A detailed view of selected European countries is also provided. Up-to-date (2016) installed vs potential SHP power capacities for Africa (580 vs 12198 MW), Americas (7864 vs 44161 MW), Asia (7231 vs 120588 MW), Europe (18685 vs 32943 MW), Oceania (447 vs 1206 MW) are available in detail from UNIDO (2016).

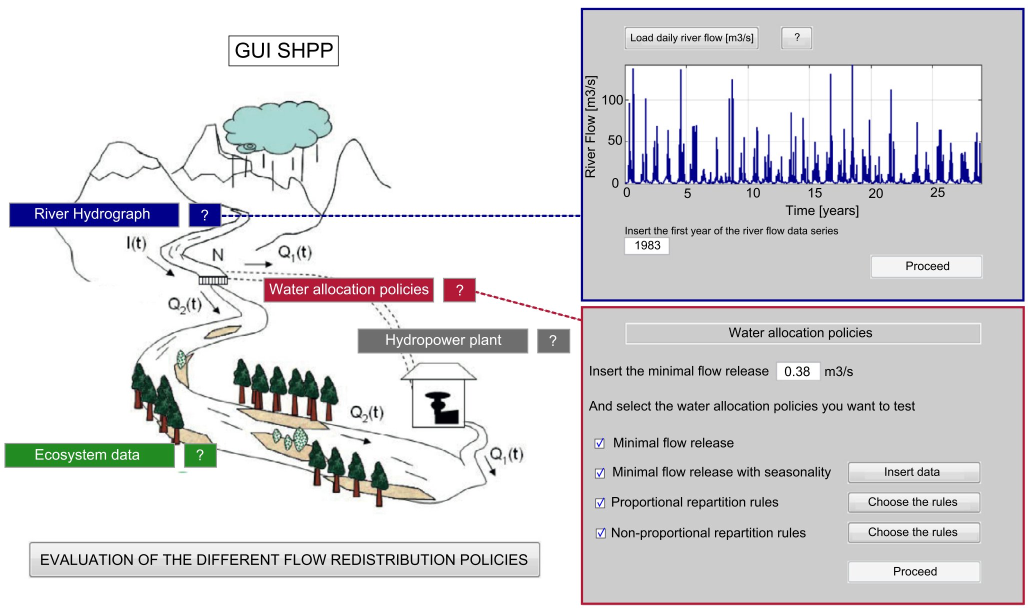

Fig. 2. SHP schematics and the corresponding river reach affected by reduced water variability. The two panels on the right show the Graphical User Interface (GUI) developed to perform the numerical simulations. In the top panel the user enters the natural hydrograph used as an input for the model. On the bottom panel, the different water allocation policies simulated by the model can be selected.

exploitation (Arthington et al., 2006), the classic MFR policy is not sustainable anymore (Poff et al., 2010). Hence, dynamic environmental flow releases mimicking the natural flow regime variability have recently been suggested as preferable (e.g. Basso and Botter, 2012; Perona et al., 2013) in order to cope with the ecosystem resilience to perturbations and reduce the risk of critical transitions to different statistical equilibrium states (Scheffer, 2009; Scheffer et al., 2012). Such dynamic redistribution practices (called “proportional” from now on) consist of the release of a certain percentage of the total flow to the environment (e.g., 20%, 30%) while exploiting the remaining fraction up to the plant nominal capacity. Although innovative and beneficial for the environment compared to minimal-flow, proportional policies suffer from the fact that the percentage of redistribution is, by definition, independent of the incoming flow carried by the river.

In order to find more efficient redistribution rules, non-proportional policies have been proposed (Gorla and Perona, 2013; Perona et al., 2013) and their global efficiency preliminary investigated by Gorla (2014) and Razurel et al. (2016). In contrast to proportional policies, the fraction of water released to the environment is defined by a non-linear function which depends on the value of the incoming flow . The conceptual basis of non-proportional redistribution is the paradigm of sustainable development, which recognizes the right of applying limited human pressure to the environment (Arthington et al., 2006). Hence, the more flexible the redistribution rule is, the more efficient the use of water by the riverine ecosystem will be. In this paper we extend the work of Razurel et al. (2016) by first improving the description of the ecohydrological indicators; second, we numerically simulate hundreds of thousands of non-proportional policies and show that Pareto efficient redistribution rules (i.e., the Pareto frontier) are indeed made by non-proportional policies; third, we perform a sensitivity analysis on the weight used to compute the ecohydrological indicator. We show the results for three Swiss case studies also under the effect of changing hydroclimatic scenarios. Potentially, these policies may be successfully applied to any river intake structures, which are primarily used to intercept and divert water from the main stream to serve, as either a storage reservoir or directly for a human use.

2. Methodology and data description

2.1. Non-proportional water allocation policies

The problem of defining the optimal water allocation for dammed systems (Castelletti et al., 2007; Niayifar and Perona, 2017; Soncini-Sessa et al., 1999) clearly simplifies for water intakes with negligible storage capacity. With reference to Fig. 2, let us assume that the fraction $Q_1(t)$ of the total incoming flow $I(t)$ at the intake is delivered to the power house. By virtue of the conservation law, the difference

$$Q_2(t) = I(t) - Q_1(t) \tag{1}$$

will be allocated to the riparian ecosystem. The environmental utility for using that water has been shown to be indirectly evaluated by the human use benefit function (Perona et al., 2013). The optimal water allocation can be identified by evaluating which redistribution rule maximizes the global (i.e., economic and environmental) benefits obtained by assigning $Q_1(t)$ to the power house and $Q_2(t)$ to the environment over a reference time frame (Gorla and Perona, 2013).

With the purpose of systematically exploring a large number of water allocation policies representing both proportional and non-proportional redistribution rules, Razurel et al. (2016) introduced a class of nonlinear functions (Gorla, 2014) by modifying the Fermi–Dirac distribution well known in quantum physics (Lifshitz and Landau, 1984). Other ways could have been used to define the non-proportional allocation function but this one has been chosen because it comprises many reasonable redistributions in a simple mathematical function, which is also parsimonious in the number of involved parameters. Thus, the fraction of water that is released to the environment is defined by the following equation:

$$f(x) = \left[1 - M - \frac{Y}{\exp[a(x-b)] + c}\right](j-i) + i \tag{2}$$

with $M = \frac{A}{A-1}$, $Y = (1-M)[\exp(-ab)+c]$ and $A = \frac{\exp(-ab)+c}{\exp[a(1-b)]+c}$. This function allows the generation of water allocation policies by varying only few parameters (i, j, a, b), as hereafter described. The parameters i and j are used to set the bound of the Fermi function. The parameter i ranges within [0;1] and represents the fraction of water left in the river at the beginning of the competition ($I = I_{min}$). The parameter j ranges also within [0;1] and correspond to the fraction of the incoming flow rate left in the river at the end of the competition ($I = I_{max}$). Non-proportional allocation starts for an incoming flow rate $I_{min} = Q_{mfr} + Q_{mec}$, where Q_{mfr} represents the minimal flow release and Q_{mec} is the minimum flow required to activate the turbines; below I_{min}, all the water goes to the environment. Initially, a fraction i of the dimensionless flow $x = \frac{I - I_{min}}{I_{max} - I_{min}}$ above 0 (for $I = I_{min}$) is allocated to the environment as

$$Q_2 = f(x) \cdot (I - I_{min}) + Q_{mfr}, \tag{3}$$

the minimal flow requirement being thus always guaranteed. The competition ends at an incoming flow rate $I_{max} = \frac{Q_N - Q_{mec}}{1-j} + Q_{mfr} + Q_{mec}$, when the nominal power of the turbine is reached at $Q_1 = Q_N$. Therefore, for $I_{min} < Q < I_{max}$ the water is dynamically allocated between the environment and the hydropower plant, depending on the value of the incoming flow I. At the end of the competition, $j < 1$ is the fraction of x left to the environment (see also Razurel et al., 2016 for details). Beyond I_{max}, river discharge exceeding Q_N is allocated to the environment spilling.

When $i = j$ the model generates proportional repartition rules. In this particular case, the quantity of water Q_2 allocated to the river is a fixed percentage (e.g., 10%, 20%) of the water inflow I in addition to the minimal flow requirement. The parameter a allows a variation of the smoothness of the transition between the environmental water allocation i relative to low flows and j relative to high flows (see Fig. 3). In the limit of a very large a, one obtains a steep-like transition. Conversely, a small a yields a linear interpolation between i and j. By varying the parameter b, one introduces a change of concavity and controls the position of the inflection point. If the change of concavity is outside the interval [I_{min}, I_{max}], one obtains either a convex or a concave function. Finally, the parameter c gives the overall shape of the curve. Gray curves in Fig. 3 show a representative sample of feasible non-proportional water repartition rules given by Eq. (2). These were obtained from 36 combinations of a and b, while fixing i and j. Pink curves correspond to the same 36 combinations of a and b, but are obtained by inverting i and j.

2.2. Ecohydrological indicators

River rehabilitation often relies on restoring a more natural flow regime (Bartholow, 2010; Petts, 2009), which suggests that optimal flow releases should be dynamic and show a variability similar to that of the natural flow regime (Poff et al., 1997). We propose to evaluate the environmental performance of the dynamic releases by building a dimensionless synthetic ecohydrological indicator. In particular, this joins the assessment provided by the Indicators of Hydrologic Alteration proposed by Richter et al. (1996) with an evaluation of the habitat availability for fish (Fig. 4). Other indicators like the hydro-morphological index of diversity (HMID) developed by Gostner et al. (2013a) exist, and have already been applied to real case studies (Gostner et al., 2013b). Their choice is a valid alternative, which depend, however, on river morphological complexity and general data availability.

The 32 Indicators of Hydrologic Alteration (IHA) proposed by Richter et al. (1996) are an effective attempt to quantify the variability of the natural flow dynamics and deviations from it for altered flow regimes. Coherently with this idea we use the IHAs to minimize the "hydrologic distance" (in terms of *Rate of non Attainment (RnA)* and *Coefficient of Variation (CV)*) between natural conditions and the flow regime resulting from every regulation policy, as detailed in Gorla and Perona (2013). We recall here that the *RnA* is defined as the fraction of simulated years in which each IHA falls outside a range defined from the natural flow regime (for each IHA).

From $RnA(k)$ and $CV(k)$ we compute the indicators $Hyd1_{sim}$ and $Hyd2_{sim}$ by first intra- and subsequently inter-groups of arithmetic means of the IHA (see Gorla and Perona, 2013; Razurel et al., 2016 for details),

$$Hyd1_{sim} = 1 - E\left[(RnA_{sim}(k) - RnA_{nat}(k))^2\right], \tag{4}$$

$$Hyd2_{sim} = 1 - E\left[(CV_{sim}(k) - CV_{nat}(k))^2\right], \tag{5}$$

where k refers to each of the 32 IHA.

In addition to hydrologic alteration, habitat availability also plays an important role in species protection. This can be assessed by modelling habitat preference curves generally obtained from river surveys and hydraulic measurements (Bloesch et al., 2005; Maddock, 1999; Milhous et al., 1984a). In the three projects considered in this work, surveys were made on the river reaches impacted by reduced flow with PHABSIM (Physical Habitat Simulation) (Milhous et al., 1984b). Fishing being the main ecosystem of interest in our case, Weighted Usable Areas (WUA) curves were computed for one dominant fish species, the *brown trout*, discriminating between *juveniles* and *adults* (EcoControl, 2011; 2012; 2013). This method was chosen according to the available data, mainly the hydrograph. Fig. 4(b) shows a qualitative example of the prefer-

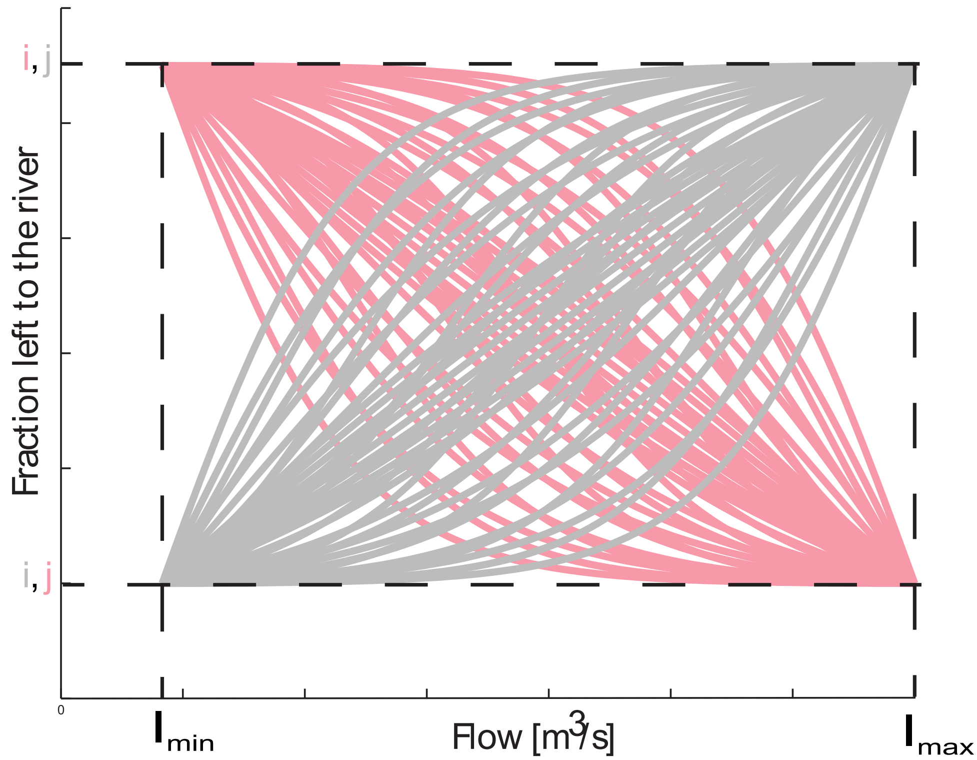

Fig. 3. Example of non-proportional repartition rules obtained with the modified Fermi function (Eq. (2)). The gray curves show an example of 36 non-proportional functions obtained for different combination of the parameters *a* and *b* while *i* and *j* are fixed ($i < j$).The pink curves correspond to the same combinations of *a* and *b* but parameters *i* and *j* are inverted ($i > j$). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

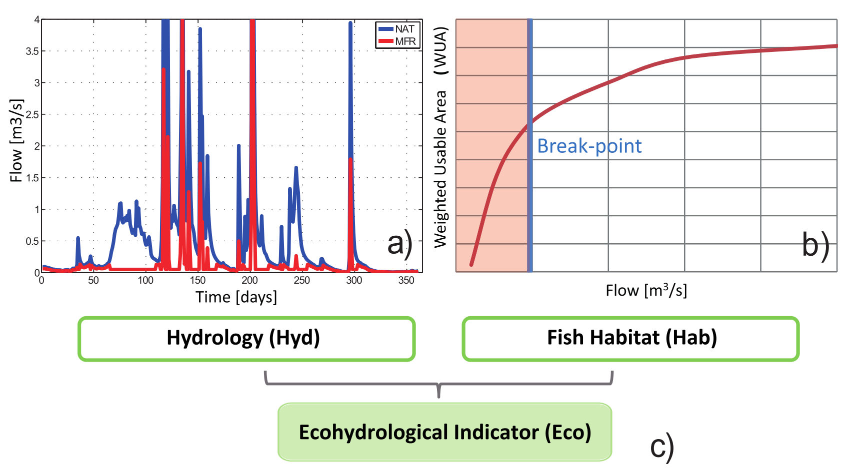

Fig. 4. a) Hydrologic differences between the natural flow and environmental releases generated by a classic minimal flow requirement approach (data from the Buseno case study). b) Sketch of the common shape of a Weighted Usable Area (WUA) curve, computed on the basis of surveying and PHABSIM simulations. The break-point generally corresponds to a remarkable change in the slope of the curve. c) Generation of the dimensionless and synthetic ecohydrological indicator *Eco* from hydrologic (*Hyd*) and fish-habitat (*Hab*) information. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

ence curve resulting from PHABSIM method. A common practice to define static threshold, like Q_{mfr}, is to define a breaking point, intended as significant changes of the WUA curve slope, and to consider it as the limit above which a further increase in environmental flow is marginally low. As this method represents a static concept, we improve and extend its use for evaluating dynamic flow releases. We assume that fish stress due to inadequate combination of substrate, water depth and speed, is more relevant when prolonged in time (Payne, 2003). We use the original WUA curves reproducing empirical data and the breaking points recommended in the official project reports in order to identify the threshold (blue line in Fig. 4(b)). Eventually, we quantify the number of consecutive days the environmental release is below the threshold and use this as a proxy for fish habitat conditions.

$Hab1_{sim}$ and $Hab2_{sim}$ thus represent the maximal number of consecutive days, computed over the whole simulation time, characterized by flows under the critical thresholds identified by break-points, for juveniles and adults, respectively. Such thresholds were fixed equal to 1.2 m^3/s for young fish and 0.73 m^3/s for adults in Buseno, 0.50 m^3/s for both categories in Cauco, and 0.55 m^3/s for young fish in Ponte Brolla, where impacts on adults were considered as negligible (EcoControl, 2011; 2012; 2013).

Table 1
List and parameters of the three case studies considered in this work.

Location	Catchment [km^2]	Head [m]	Turbine type	Q_N [m^3/s]	Q_{mfr_1} [m^3/s]	Q_{mfr_2} [m^3/s]	Power [kW]	Energy Production [GWh]
Buseno	120	66.5	Cross-flow	4.5	0.38	0.60	2340	8.8
Cauco	89	49.9	Cross-flow	3.5	0.315	0.60	1390	5.0
Ponte Brolla	592	39.5	2 x Francis	12	0.55	0.86	1900	13.9

We then aggregate $Hyd1_{sim}$ and $Hyd2_{sim}$ into two hydrological sub-indicators, E_1 and E_2, bounded between 0 and 1 as

$$E_1 = 1 - \frac{Hyd1_{sim} - Hyd1_{min}}{Hyd1_{max} - Hyd1_{min}}; E_2 = 1 - \frac{Hyd2_{sim} - Hyd2_{min}}{Hyd2_{max} - Hyd2_{min}}. \quad (6)$$

The indicators with subscript *min* and *max* correspond to the scenarios having the minimal and maximal impact on the river, respectively; in this work they correspond to the natural flow regime (no-impact) and to the minimal flow requirement policy.

Similarly, we aggregate $Hab1_{sim}$ and $Hab2_{sim}$ into two fish habitat availability sub-indicators, E_3 and E_4,

$$E_3 = 1 - \frac{Hab1_{sim} - Hab1_{min}}{Hab1_{max} - Hab1_{min}}; E_4 = 1 - \frac{Hab2_{sim} - Hab2_{min}}{Hab2_{max} - Hab2_{min}}. \quad (7)$$

The hydrological indicator *Hyd* is calculated by doing the weighted geometric average of the sub-indicators E_1 and E_2,

$$Hyd = e^{w_1 \cdot \ln E_1 + w_2 \cdot \ln E_2}, \quad (8)$$

where w_1 and $w_2 = 1 - w_1$ are the weighting factors of E_1 and E_2. The exponential form is used here as a convenient way of representing the weighted geometrical mean.

The fish habitat indicator *Hab* is calculated by doing the weighted geometric average of the sub-indicators E_3 and E_4,

$$Hab = e^{w_3 \cdot \ln E_3 + w_4 \cdot \ln E_4}, \quad (9)$$

where w_3 and $w_4 = 1 - w_3$ are the weighting factors of E_3 and E_4.

The indicators *Hyd* and *Hab* are finally aggregated to calculate the dimensionless synthetic ecohydrological indicator *Eco*,

$$Eco = e^{w_5 \cdot \ln Hyd + w_6 \cdot lnHab}, \quad (10)$$

where w_5 and $w_6 = 1 - w_5$ are the weighting factors of *Hyd* and *Hab*.

Weights should be defined case-by-case, on the basis of expert's opinion and considering the status of the specific riparian ecosystem. In this work we chose not to express preferences and weighted all the indicators as equally important in all numerical simulations (Richter et al., 1997; 1996). However, in order to explore how weighting impact the results, we performed a sensitivity analysis for the weighting factor w_5.

2.3. Case studies

We chose three small hydropower case studies (henceforth denominated Buseno, Cauco, and Ponte Brolla) located in Southern Switzerland, whose details are reported in Table 1. For the three case studies we compared the effects of the following sub-classes of water allocation policies: (i) scenarios MFR_1 and MFR_2, representing traditional minimal flow requirement policies with one or two thresholds (the second one is introduced to increase the minimal flow value from April 1st to September 30th), respectively Q_{mfr_1} and Q_{mfr_2} defined in Table 1; (ii) dynamic flow releases, proportional to $I(t)$ (fixed percentages going from 10% to 50% with a step of 5%); (iii) dynamic flow releases, non-proportional to $I(t)$ (flow-dependent, variable percentages as previously described). In particular, the non-proportional water allocation policies were obtained by varying i and j from 0.02 to 0.70 with 0.01 increment, a from 2 to 8 with step equal to 2, b from 0 to 1 with step 1/8, and considering c constant and equal to 1, for a total of 168912 considered alternatives. The minimal flow requirement Q_{mfr_1} was enforced by law and was therefore always guaranteed for each simulated scenarios.

We used 29 years of streamflow data measured by the Swiss Federal Office for the Environment as natural inflows $I(t)$ to evaluate scenarios in the period 1983–2011. For Cauco and Ponte Brolla, power plant locations along the river are not the same as the locations from which the historic flow series have been obtained. We therefore transposed streamflows measured at Buseno (https://www.hydrodaten.admin.ch/fr/2474.html) and Bignasco (https://www.hydrodaten.admin.ch/fr/2475.html) gauging stations using a surface ratio by rescaling them to the respective catchment areas (Brutsaert, 2005; Dingman and Dingman, 1994). The dependence of hydropower production B_1 on river discharge Q_1 was approximated by a 2nd degree polynomial equation $B_1 = m \cdot Q_1^2 + p \cdot Q_1 + q$, with m, p, and q depending on each plant turbine and associated to a fitting law showing a fitting correlation coefficient R^2 larger than 0.9 (see Gorla, 2014 for details).

2.4. Climate change impact on streamflow

The effect of climatic changes on water availability for the the periods 2020-49 and 2070-99 has been obtained by considering the emission RCP 6.0 scenario (Flato et al., 2013), which has been extensively applied to project future climate in several alpine regions of Switzerland. In brief, this scenario foresees by the end of the century a mean global increase of Earth surface temperature of about 2.8°C during summer, with a possible range of +1.7 to +4.5°C in Alpine Swiss Cantons. The expected winter temperature variations are approximately 2°C smaller. The projected precipitation regime is even more uncertain given the present inherent stochasticity of the phenomenon (Brönnimann et al., 2014). Overall, streamflows are expected to increase in magnitude in the period 2020–2049 due to the melting and shrinking of alpine glaciers. This scenario will progressively move to a nivo-pluvial flow regime in the period 2070–2099 characterized by higher flows during late winter, early spring time. Those changes are shown in Fig. 5. A recent report (Job et al., 2011) describes the evolution of the Gornera basin (located in Southern Switzerland near the considered catchments) in response to such changes and to stored ice and snow in the basin. We considered this scenario as representative for the three basins chosen and based on that we generated time series of daily streamflow expected for the periods 2020–2049 and 2070–2099 for each each basin (e.g. see Gorla, 2014).

2.5. Development of a Graphical User Interface and numerical simulations

A Graphical User Interface (GUI) (Fig. 2) has been developed using the software Matlab to facilitate the data treatment and the selection of the optimal water allocation functions among the different scenarios (non-proportional, proportional and MFRs repartition rules). For each scenario, the energy production and the ecohydrological indicators were computed based on the generated flows As a result, the efficiency graph, showing the mean annual energy produced during the analyzed period versus the ecohydrological

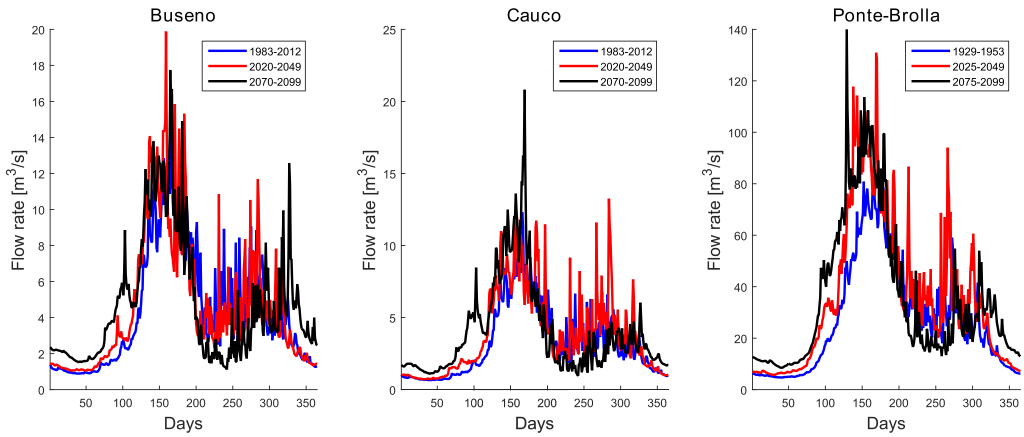

Fig. 5. Changes in the mean annual hydrograph for medium and long term under the considered climate scenario RCP 6.0 (Flato et al., 2013) for the three different case studies: Buseno, Cauco and Ponte Brolla.

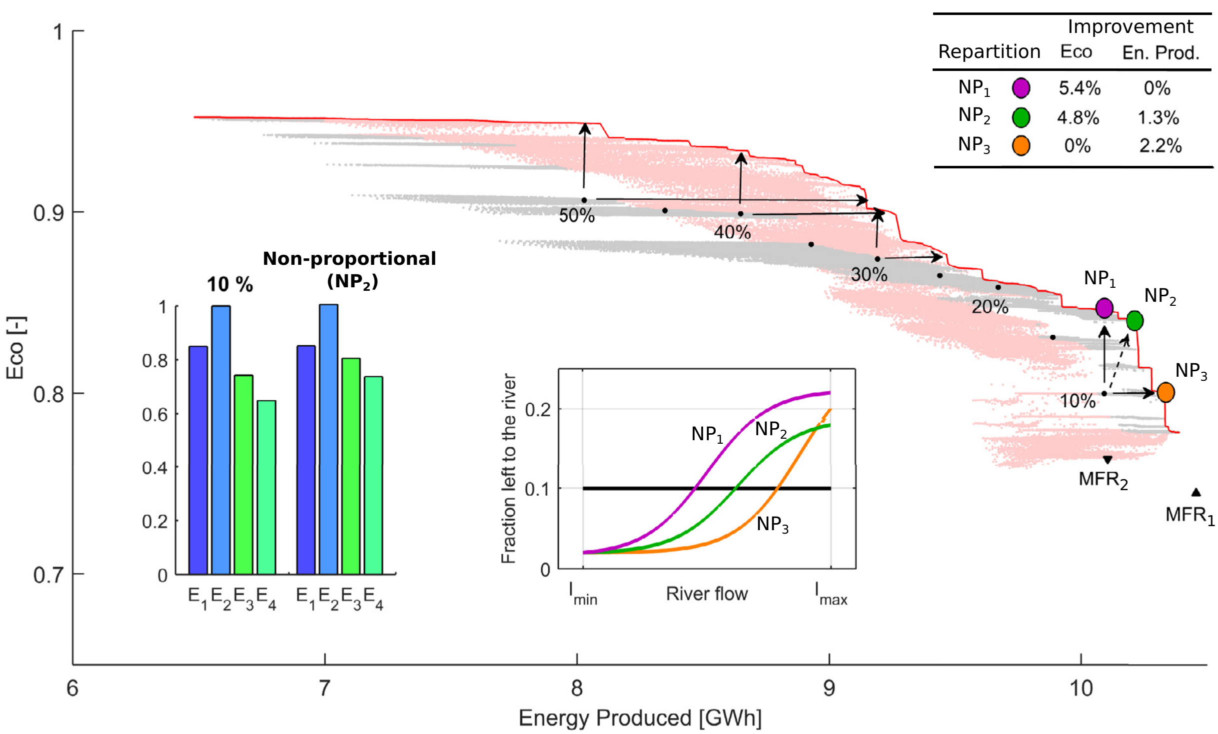

Fig. 6. Pareto frontier (red line) and alternatives repartition rules simulated from the 29-years hydrograph (1983–2011) for the Buseno case study. In black are MFR and proportional allocation policies; grey and pink points correspond to non-proportional policies (a subset of these is shown in Fig. 3). The black arrows indicate the improvement in term of ecohydrological indicator (vertical ones) and energy produced (horizontal ones) by switching from proportional to non proportional alternatives. The histograms show an example of sub-indicators performances of a proportional (10%) and a non-proportional alternative (green point on the Pareto frontier). The colored curves in the central panel represent the Fermi functions obtained for the three effcient non proportional alternatives to the 10% policy. In the table, the percentages of improvement in ecohydrological indicator and energy production of the non-proportional alternatives NP_1, NP_2 and NP_3 with respect to the 10% proportional rule are shown.(For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

indicator, was plotted. The Pareto front, representing the ensemble of optimal water allocation scenarios, was identified and enhanced with a red line in the efficiency plot. More details are provided in Appendix A.

3. Results

3.1. Efficiency plot and selection of optimal scenarios

Fig. 6 shows the performances of Buseno hydropower plant in terms of efficiency plot for all the 168912 water repartition rules obtained from Eq. (2). Each gray and pink point of the efficiency plot corresponds to a non-proportional repartition policy, and can thus be compared to more classic scenarios, e.g. based on minimal flow requirement and proportional water allocation policies.

As expected, scenario MFR_1 has the highest hydropower production and the lowest environmental performance. The scenario MFR_2 in Buseno, in which the minimal release is increased from April 1st to September 30th to a second fixed threshold, shows a reduction of hydropower production by 3.4% and an increase of ecohydrological indicators by 2.5% with respect to the performances of MFR_1. This scenario may be improved by applying proportional repartition rules. Among these, the one that leaves 10%

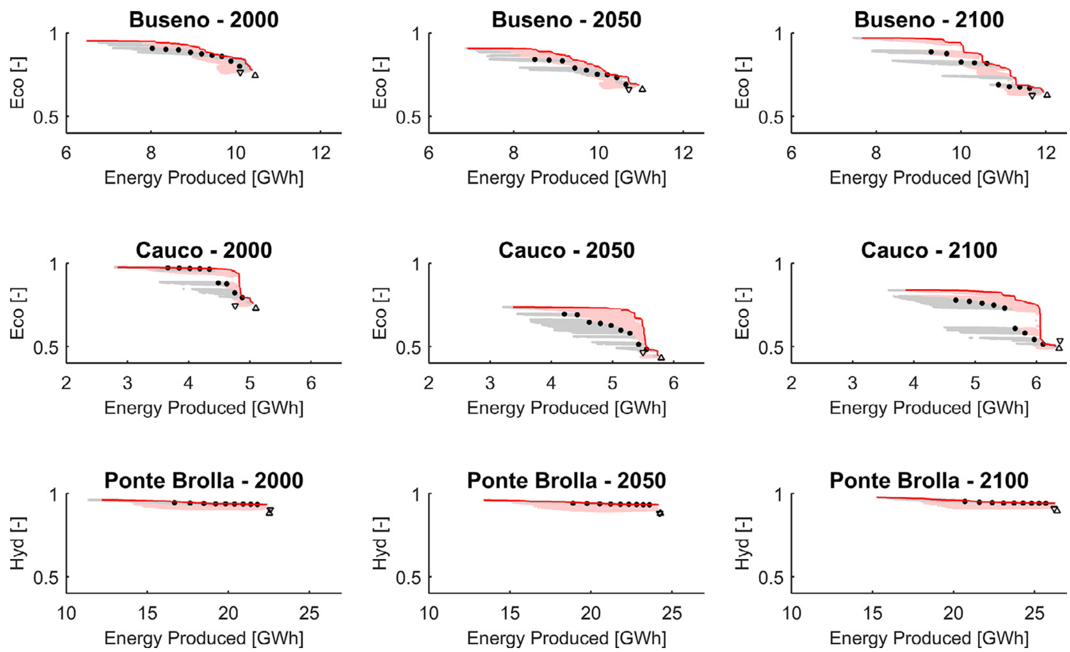

Fig. 7. Overview of the alternatives simulated, and the relative Pareto frontiers, for the three case studies under the three considered climatic scenarios (RCP 6.0). Equal weights were assigned for ecohydrological indicators. Colours and symbols are the same of Fig. 6.

of the incoming flow to the environment preserves the energy production of scenario MFR_2, while increasing the ecohydrological benefits by 4.7%.

However, the benefits obtained with the 10% proportional rule, can still be improved by moving vertically or horizontally toward the Pareto frontier, enhancing the ecohydrological indicators and the energy produced, respectively. A notable result is that the Pareto frontier is entirely composed by non-proportional repartition rules (henceforth referred to as “efficient”). It is worth recalling here that, at the Pareto frontier, it is not possible to improve a scenario by making an indicator better without making another one worse. For this power plant, changing a proportional repartition rule with an efficient one (i.e., that lies on the Pareto frontier) causes a 5% hydropower production average improvement and a 3% improvement for the ecohydrological indicators. These percentages were obtained, with reference to Fig. 6, by moving vertically and horizontally from proportional alternatives towards points located on the Pareto frontier.

Similar results are obtained for Cauco power plant, but not for the one in Ponte Brolla, as shown in the left-hand side panels of Fig. 7. For the latter case, proportional repartition rules perform already well and the ecohydrological indicator resulting from the simulated alternatives is already high, thus making the improvement almost negligible, (the potential improvement of using efficient non-proportional distribution to replace proportional distribution is between 0.0% and 0.1%). This is mainly due to the fact that, in Ponte Brolla, habitat thresholds (the blue line shown in Fig. 4(b)) turned out to be lower than Q_{mfr} because of the particular canyoning morphology of the regulated reach, where a minimal flow release also guarantees fish survival. Consequently, among the indicators, mainly the hydrologic one (i.e., *Hyd*) concurred to the definition of the global ecohydrological indicator *Eco*. This result is consistent with that shown by the sensitivity analysis performed while changing the weights used to build the ecohydrological indicator (shown ahead). That is, results similar to Ponte Brolla power plant can be obtained for both Cauco and Buseno in the limit of non considering the fish habitat availability. A backwards control on sub-indicators and Fermi's functions (see e.g. subplots in Fig. 6) should also be done case-by-case on the basis of experts opinions in order to check the soundness of interesting alternatives.

3.2. Climate change scenarios

Our study shows that a general increase in hydropower production is foreseen for the periods 2020–2049 and 2070–2099 for all the three basins (Fig. 7). This right shift toward higher energy production of the efficiency plot can be explained by an increase of streamflow from 2020 to 2049 and a seasonal temporal shift of water availability in the period 2070–2099, as predicted by climate models (Fig. 5). While the aftermath of glacier melting in 2020–2049 is obvious as far as energy production is concerned, the effects of higher winter and spring precipitation expected in the second three decades requires an explanation. The latter regime sees a flattening of the current river hydrograph with a strong reduction of the summer maximum. As a consequence of such redistribution of water availability during the year, the number of days when turbines can be activated will increase, as the flow necessary for the turbine to operate, Q_{mec}, will be reached more often. The impact of climate change on the number of possible operation hours at Q_N per year is more uncertain, especially if no storage is available.

The ecological effects of regulation under climate change are complex and must be analyzed case-by-case. While an exception can be made for Ponte Brolla, where river morphology always guarantees good habitat availability (even under low-flow MFR scenario), both Buseno and Cauco will see a worsening of both the proportional and constant flow release policies with respect to non-proportional ones. Table 2 presents the average improvements obtained by moving from proportional to efficient non-proportional repartitions located on the Pareto frontier, for the three case studies and the three time periods. The results show that gains can be obtained through the use of optimal allocation rules for the three case studies. For Buseno, the potential gain in ecohydrological indicator goes from 1.8% for the period 1983–2012 to 4.6% for the period 2020–2049. The foreseen amelioration of the energy production is around 2% for the three considered periods. The most important results concerning the ecohydrological indicator are those obtained for Cauco. Indeed, the foreseen amelio-

Table 2
Quantification of the averaged improvements for the alternatives shown in Fig. 7. They were obtained by replacing proportional repartition rules with efficient non-proportional ones, improving one indicator at a time.

Foreseen amelioration of non-proportional policies						
Case study	1983–2012		2020–2049		2070–2099	
	Eco	HP	Eco	HP	Eco	HP
Buseno	3.1%	2.4%	4.6%	2.2%	1.8%	1.9%
Cauco	8.6%	1.0%	19.8%	1.0%	22.8%	0.8%
Ponte Brolla	0.1%	2.3%	0.1%	2.6%	0.0%	0.3%

ration of the ecohydrological indicator goes from 8.6% for the period 1983–2012 to 22.8% for the period 2070–2099. However, the potential gain in energy production is around 1%, which is lower than the two other case studies on average. Ponte Brolla shows the lowest gain in ecohydrological indicator, less than 1%, but the improvement of the energy production for the periods 1983–2012 and 2020–2049 are close to Buseno. These scenarios are valid assuming that even though the morphology of single river banks is dynamic, average fish habitat conditions in a river reach will not change over the considered time horizon.

4. Discussion

4.1. Role of ecohydrological indicator and sensitivity analysis

Fig. 8 shows the results of the sensitivity analysis performed for the three case studies: (a) Buseno, (b) Cauco and (c) Ponte Brolla. For each of the three plots, the two weighting factors w_1 and w_3 were set to 0.5 while the third factor w_5 was progressively increased from 0 to 1 with a step of 0.001. Thus the only parameter that was changed is the weighting of the hydrological indicator *Hyd* and the fish habitat indicator *Hab* to compute the final ecohydrological indicator *Eco*. For each combination of factors, a new efficiency plot is computed. The corresponding average amelioration in both ecohydrological indicator and energy production when replacing proportional rules by non-proportional ones were thus calculated and shown on the Y-axis of the plot.

Notably, the sensitivity analysis shows some different results depending on the case study. As far as Buseno (Fig. 8(a)) is concerned, the average improvement of the ecohydrological indicator (red curve) with respect to proportional policies is decreasing when the weighting of the hydrological indicator is bigger than the habitat one, i.e. more weight is given to the hydrological indicator. The gain of energy production (blue curve) starts decreasing when w_5 is above 0.6. This shows that giving a superior weight to the hydrological indicator leads to a reduction in the power production gain. For Cauco (Fig. 8(b)), the same tendency is observed for the environmental gain. However, the variation of the power production as a function of the weighting factor w_5 shows some fluctuations. In contrast to Buseno, no clear tendency is observed. The results for Ponte Brolla (Fig. 8(c)) are different and the improvements of the power production and the ecohydrological indicator are constant, independently of the value of w_5. This is explained by the fact that for this specific case, the minimal flow release MFR is always greater than the value of the threshold defined to calculate the fish habitat indicator. Thus, the indicator *Hab* is always set to the constant maximum value. The order of magnitude of the power production gain is comparable to the other stations but the environmental gain is lower.

The absolute value of the ecohydrological indicator has to be interpreted carefully since there is no other previous study applying the same methodology to combine the hydrological and fish habitat suitability indicators. The indicator has been built to evaluate how far from the natural series each scenario is, a value of 1 corresponding to the natural condition. Thus, we are more interested in the comparison of the different allocation scenarios and the results we are showing are more focused on the relative gain that may be obtained by using non-proportional policies. We show a method to choose the optimal distribution functions by comparing all the possible distribution methods. The sub-indicators have been chosen according to the available data, being mainly the natural hydrograph and the characteristics of the power plant, but may be improved if more data are available. The allocation rules we are presenting in the paper (non-proportional) have not been implemented yet so there are no empirical data available that allows a comparison between the pre-impact and post-impact systems.

4.2. General considerations and recommendations

Managing water resources to their maximal extent in Alpine countries will necessarily force people to be aware that each unit of energy is generated at some expense of the ecology of the riverine ecosystem. As a consequence, all the feasible measures to improve in efficiency should be taken into consideration together with implementation costs. Some costs are very much country dependent and this aspect is not addressed in this work, being beyond the scope of the work. However, the implementation costs for generating dynamic flow releases are worth a few comments.

This work showed that gains in hydropower production and ecohydrological indicator could be made on average by replacing proportional water allocation policies (today's best practice though not yet widespread) with non-proportional ones located on the Pareto frontier (Table 2). Improving both criteria, such increments must be considered as actual win-win solutions. These results are based on testing non-proportional redistribution rules on only three homogeneous SHP case studies limited to the Swiss environment and its socio-economic context. We showed that the potential improvement lies in the wider range of non-proportional repartition rules, with respect to traditional policies. Moreover, Fig. 6 demonstrates how classic minimal flow requirement approaches (MFR_1 and MFR_2) can be improved, mainly in term of ecohydrological benefit, by applying non-proportional policies even more than by applying proportional ones (both dynamic). Considering the environment as an independent water user (Perona et al., 2013), with specific needs and features, is thus the key to obtaining efficient environmental flow releases. Such rules will generally result in being non-proportional and flow-dependent. In fact, while the efficiency curve of a turbine does not change throughout the year, the environmental use of water follows seasonal trends. This could easily be added in the model and weighted case-by-case when specific ecological information is available. Increasing the number of case studies would statistically strengthen the results and suggest more general rules to understand which power plants can actually be improved in global performances. This can be challenging to show, particularly because data are often not easily available.

In this work, we decided to express the economical indicator as the Energy Production in GWh. This study focuses on Small hydropower plants without storage, hence, this suggests that the optimal strategy would be to always turbine the water diverted according to the chosen allocation rule. However, a further improvement would consist in considering the variability of the electricity market price. This could be made by changing the dimensionless variable x of the Fermi function (Eq. (3)) so it does not depend only on the flow rate but also on the market price. Thereby, the value of the produced hydropower production would be optimised (Pereira-Cardenal et al., 2016).

Energy provision from renewable sources is a sign of human being responsibility, which however requires a strong harmonisation

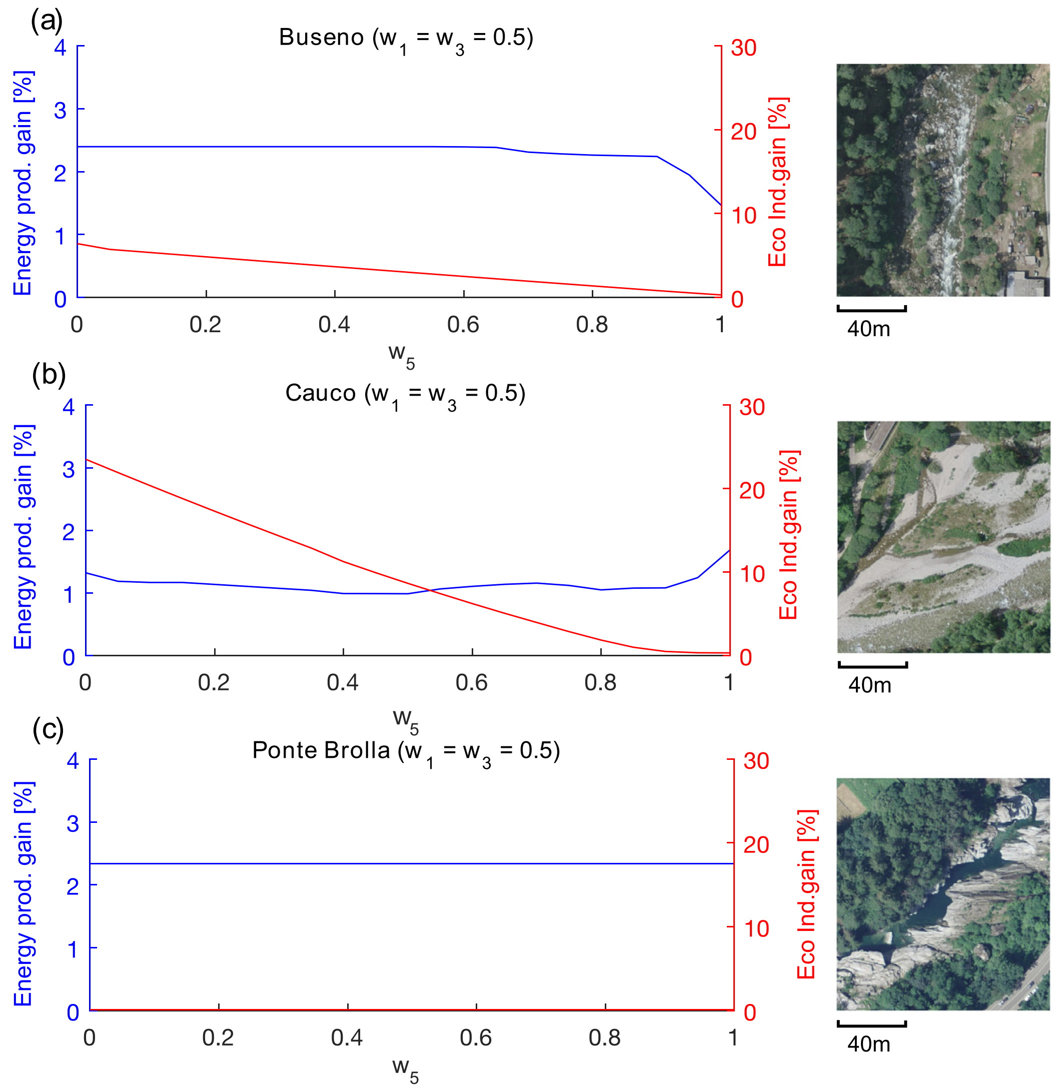

Fig. 8. Sensitivity analysis showing the gain in power production (blue curve) and ecohydrological indicator (red curve) with respect to proportional policies and obtained by changing the sub-indicator weighting factors w_1, w_3 and w_5 as described in Section 2.2. Pictures of the river reach morphologies corresponding to the three case studies are also shown.(For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

among social, economic and political parts. The question of how to implement non-proportional flow release rules has not been addressed in this work. However, our present research started to address this problem, particularly looking at suitable hydraulic infrastructures that may generate Fermi function redistribution rules at zero energy costs (Bernhard and Perona, 2017). This is highly desirable in order to pursue innovation not only from an intelligent technological infrastructure point of view, but also from a sustainable one.

5. Conclusions

This work shows a simple and innovative numerical approach for defining sustainable and efficient environmental flow releases in river reaches of SHP without storage. The method has been tested on real data and constraints, and could be adopted as a prompt answer to the actual need to conciliate environmental protection and growth of hydropower production. A convenient class of functions, developed by Gorla (2014) and Razurel et al. (2016), was here comprehensively tested as a practical tool for exploring a representative sample of dynamic flow releases. Such functions provide a direct link between the practice of comparing different environmental flow policies, in particular those using fixed percentages of the incoming flows (proportional) and those with variable splits between diverted and released flows (non-proportional). The Pareto frontier is obtained from the simulated alternatives for each case study and it shows that non-proportional rules are generally more efficient than traditional ones, both proportional and static. It was shown that when applying efficient non-proportional repartition rules for regulating the run of the river hydropower plants, ameliorations in hydropower and ecohydrological performances can be attained, with respect to proportional policies. Although the three case studies are located in Switzerland the results vary from one case to another, leading to the conclusion that they depend on the river morphology. Indeed, the canyoning morphology in the case of Ponte Brolla implies that the MFR value is always higher than the threshold given by the WUA curve, which results in a maximum value for the fish habitat suitability indicator. For Cauco, the foreseen amelioration for the ecohydrological indicator is the most important, it goes from 8.6% for the period 1983–2012 to 22.8% for the period 2070–2099 but the gain in energy production is the lowest (around 1%) in comparison to the two other case studies. Buseno and Ponte Brolla show some similar potential gains in energy production (around 2%) but for the latter the ecohydrological improvement is almost irrelevant (between 0.0% and 0.1%).

Author contribution

Lorenzo Gorla and Pierre Razurel contributed equally to this work.

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Appendix A

Graphical User Interface (GUI) (Fig. 2) has been developed using the software Matlab to facilitate the data treatment and the selection of the optimal water allocation functions among the different scenarios (non-proportional, proportional and MFRs repartition rules). This tool takes the natural river hydrograph and the hydropower plant features (efficiency function, design flow, etc) as inputs. The desired water allocation policies as well as the ecological threshold can also be set. The user-friendly architecture of the GUI (freely available to any user that wants to reservedly test the performances of his own cases)[1] makes the model particularly suitable for stakeholder planning, for water managers operations or for academic purposes.

Numerical simulations were performed in order to model the different allocation functions. The natural daily flow, $I(t)$, was redistributed between the hydropower plant and the river by simulating Eqs. (1)–(3) according to the selected Fermi function and for the entire time series of $I(t)$. For each scenario, the energy production and the ecohydrological indicators were computed based on the generated flows Q_1 and Q_2, respectively. The same procedure was repeated for the whole set of selected Fermi function parameters as well as for the proportional and MFRs repartition rules. As a result, the efficiency graph, showing the mean annual energy produced during the analyzed period versus the ecohydrological indicator, was plotted. The Pareto front, representing the ensemble of optimal water allocation scenarios, was identified and enhanced with a red line in the efficiency plot.

The simulations to asses the impact of the climate change have been performed in the same way for the three case studies (i.e., Buseno, Cauco and Ponte Brolla). The time series of daily streamflow for the three different time periods (i.e., 2000, 2050 and 2100) have been generated from the current natural data series by applying the trend of the RCP 6.0 scenario described in the previous Section 2.4.

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[1] Free download from http://www.sccer-soe.ch/research/hydropower/task2.4/ or by simply contacting the authors (PR, PP).

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