Journal of Hydrology 489 (2013) 98–107

Contents lists available at SciVerse ScienceDirect

Journal of Hydrology

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

On quantifying ecologically sustainable flow releases in a diverted river reach

Lorenzo Gorla *, Paolo Perona

Group AHEAD, Institute of Environmental Engineering, EPFL-ENAC, Lausanne, Switzerland

ARTICLE INFO

Article history:
Received 29 October 2012
Received in revised form 20 February 2013
Accepted 24 February 2013
Available online 20 March 2013
This manuscript was handled by Geoff Syme, Editor-in-Chief

Keywords:
Run-of-the-river hydropower plants
Environmental benefits
Water allocation policy
Principle of equal marginal utility
Dynamic flow releases
Hydrological alteration

SUMMARY

Water demand for hydropower production has been increasing along with a growing awareness of the importance of preserving riparian ecosystems and their biodiversity. Some Cantons in Switzerland have begun replacing the inadequate concept of Minimum Flow Requirement (MFR) with a dynamic one, by continuously releasing a percentage of the total inflow.

In this work, we evaluate both ecological and economical benefits of dynamic release policies within a diverted river reach in the Swiss Canton of Graubünden. We compare such policies to another one that generates inflow-dependent variable releases as a result of an economic competition between traditional (e.g., hydropower) and non-traditional (e.g., environment) water uses (Perona et al., 2013). We propose to compute flow statistics from different release polices by using Indicators of Hydrologic Alteration (Richter et al., 1996; Richter et al., 1997). Then, we aggregate the hydrological differences from the natural flow regime as a proxy for assessing environmental benefits and we look at the mean of the ratio of the allocated net flows between environment and hydropower as a suitable engineering parameter to represent their relative value. Eventually all the simulated release policies find an economical significance explained by marginal utility functions.

We show that dynamic redistribution policies can perform better than MFR-like ones. Moreover, by introducing the concept of inflow-dependent water allocation, both economic and ecological indicators can be further improved, without necessarily implying higher installation costs.

This method aims to provide a simple but effective step towards eco-sustainability in the growing market of mini hydropower plants, where MFR-rules are still widespread.

1. Introduction

The worldwide increase of energy demand associated to the need for renewable energy sources is leading to an increase of hydropower production and, in turn, more pressure on riverine ecosystems. During the past decade, the number of installations and studies involving small hydropower plants functioning on a run-of-the-river principle has been growing (Wehse et al., 2011; Niadas and Mentzelopoulos, 2008). This is a result of the limited necessary infrastructure and the suitability of installations in small catchments, the majority of which are still free from impoundment and diversions. In these systems (Fig. 1a) water is diverted from the river mainstream to powerhouses often leaving only a Minimum Flow Release (henceforth referred to as MFR) in the mainstem. Fig. 1b shows an example of a regulation based on a MFR policy: the resulting hydrograph is significantly different from the natural one, as much of the variability characterizing the natural flow regime (e.g., seasonality) is no longer visible. Calanca valley (Fig. 1c), in southern Switzerland, is a good example of societal pressure caused by energy demand, where the planned installation of four new run-of-the-river mini-hydropower plants will affect several river mainstreams less than ten km apart.

When river branches are regulated by simple rules such as the MFR, then biodiversity changes and river homogenization can be expected at medium and long term (Poff et al., 2010; Bartholow, 2010) as a result of perturbing the natural flow regime (Poff et al., 1997). Quantifying in advance specific environmental effects due to a certain degree of flow modification by impoundment or diversions is a challenging problem (Petts, 2009; Bartholow, 2010; Groffman et al., 2006), e.g. due to ecosystem resilience to perturbations (Scheffer et al., 2001). In order to minimize ecological shortcomings, a common sense measure would suggest to maintain the variability of water releases as close as possible to that of the natural flow regime (Richter et al., 1996; Richter et al., 1997; Suen and Eheart, 2006; Richter and Thomas, 2007; Costigan and Daniels, 2012). The study of flow release strategies is thus of deemed importance (Arthington et al., 2006) for both scientific and practical reasons, particularly when conducted in not yet diverted rivers. Since the ecosystem is still in pristine condi-

* Corresponding author. Tel.: +41 (0)21 693 0863.
E-mail address: lorenzo.gorla@epfl.ch (L. Gorla).

0022-1694/$ - see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.jhydrol.2013.02.043

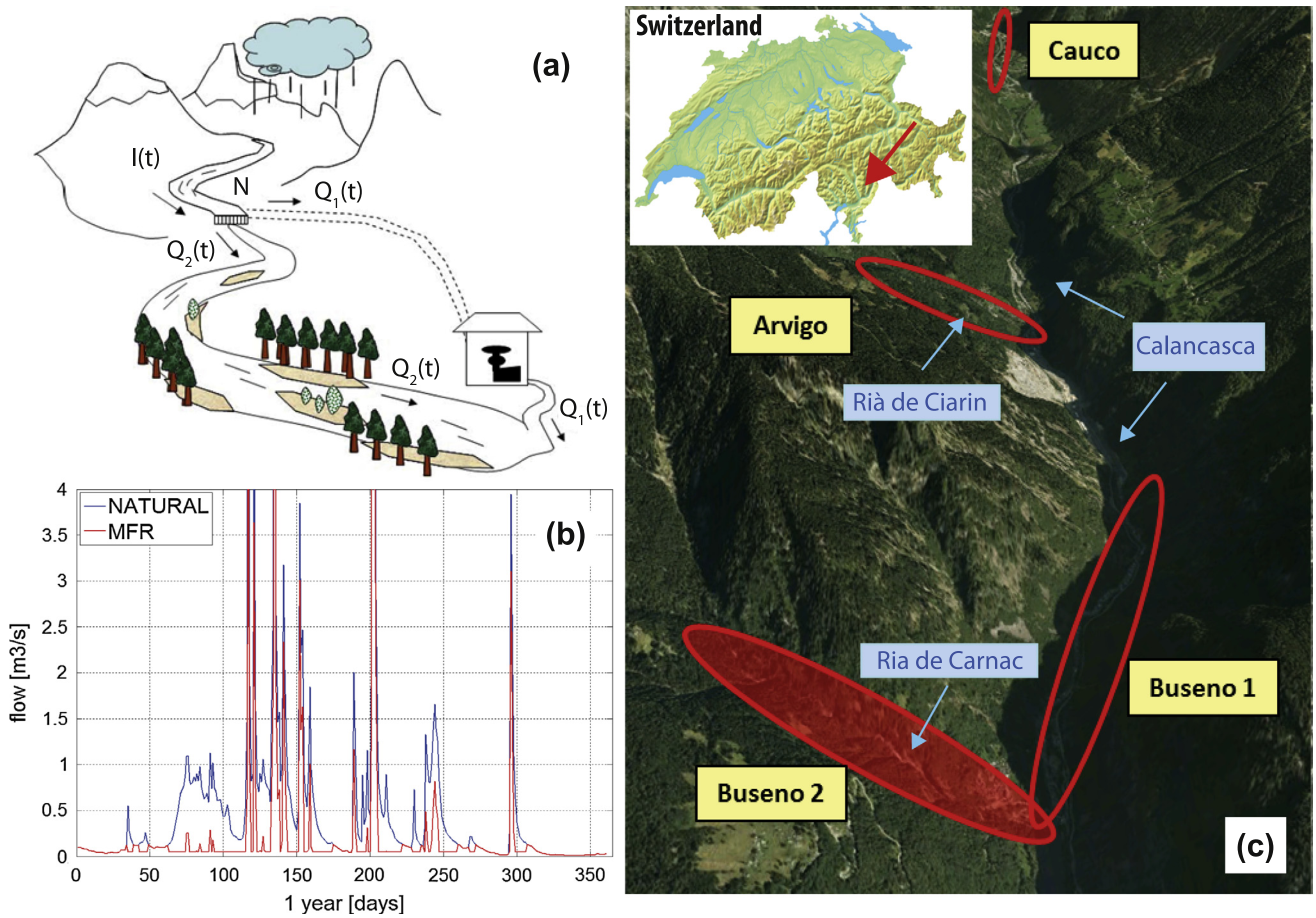

Fig. 1. (a) Sketch of the run-of-the-river system, referring to all the four cited projects, in which an upstream diversion node *N* affects the riparian environment withdrawing a flow Q_1 and thus modifying the natural regime downstream. Eventually, the diverted flow Q_1 used by the exploitation activity will re-join the residual flow Q_2 downstream of the river reach. (b) Effects of a classic MFR regulation on a natural hydrograph: the time series have been taken from Buseno2 project, explored in detail hereafter, but the effect of losing a consistent part of the natural river dynamic can be generalized to all these kinds of run-of-the-river systems. (c) The Calanca Valley in the Swiss Canton of Graubünden is a good example of the spreading rate concerning small hydropower plants: in fewer than 10 km long valley four mini hydropower systems have been projected and they will be completed in the next one or two years. The portions of river affected by water diversion are highlighted in red circles; "'Buseno 2" is the only project considered in this paper for simulations.

tions, ecological impacts can be ascribed to a specific flow alteration, which is a substantial advantage in respect to already altered environments.

In Calanca valley several alternatives to the MFR have been proposed by practitioners, which are mainly based on the common sense of a proportional redistribution at the diversion node. Similar approaches have for instance been applied in Austria and Northern Italy. Those proportional repartition rules generates natural-like variability in the released flow, but they are not flexible: the environment and the hydropower receive indeed their constant fraction of water, which means that their relative importance is inflow independent. This is an implicit strong constraint, which can be removed if the environment and the hydropower were considered as independent users with independent water demand functions. This leads to inflow-dependent water allocation, which possibly improves both economical and ecological efficiency. In the present work we address this issue by using marginal benefit analysis (e.g., Perona et al., 2013) to show similarities and suggest possible ameliorations.

We focus on one of the planned run-of-the-river projects in the Calanca valley (Buseno 2, Fig. 1c) with the goal of simulating and quantifying the economic return and the ecologic sustainability of six flow release scenarios. Two scenarios are based on MFR with and without seasonal threshold, three are empirical dynamic releases based on proportional water partitioning rules, and one results from the Principle of Equal Marginal Utility (PEMU): it has been proposed in an earlier work (Perona et al., 2013) and generates inflow-dependent water allocation.

We consider the riparian ecosystem from a "holistic" point of view (Tharme, 2003; Arthington et al., 2006), aiming to address the flow requirements of different ecosystem components and not just a few species (e.g. fishes). Hence, the performance of release scenarios can be evaluated both economically and ecologically by using hydropower revenues and the Indicators of Hydrologic Alteration (IHA), originally introduced by Richter et al., 1996; Richter et al., 1997. These indicators quantify the difference between regulated and natural flow regimes by computing the "virtual distance" between altered scenarios and natural conditions. Rather than presenting all of the IHA separately, we propose two new synthetic and dimensionless indicators that can be used to comprehensively evaluate the efficiency of water distribution at the node for the global system (hydropower plus riverine environment).

We obtain three notable results: first, we demonstrate that some of the dynamic redistribution policies, which allow for more natural streamflow, are Pareto-dominant over other scenarios based on MFR with seasonal thresholds. Second, we show that dynamic release strategies based on simple proportional repartitions find a plausible economical interpretation in the PEMU model of Perona et al. (2013). Third, we claim that such solutions do not necessarily imply higher costs or advanced management tools to be implemented.

Thus this paper suggests the PEMU as an evolution of nowadays employed water allocation rules, and offer an aggregation of the IHA as a proxy to evaluate them. The next section describes the methodology and the case study; Section 3 presents the results and discuss them; Section 4 is left to conclusions.

2. Data description and methodology

2.1. Carnac river, hydrology and regulation policies

Carnac basin (7.45 km^2, shown in Fig. 1a) is a small inner valley of Val Calanca (Graubünden, Switzerland). The homonymous river receives waters from a stream called *Rià de Ciarin* and the 350 m height difference between the diversion site (1050 m a.s.l) and Buseno village makes it interesting for hydropower exploitation. The catchment has no glaciers and its runoff regime is nival–pluvial. The mini-hydropower plant *Buseno 2* is not active yet and the measured flow-series of *Carnac* river are available from October 2010 to January 2012, with several discontinuities. This is not a sufficient amount of data for evaluating water allocation policies by means of IHA. Recommendations by Richter et al. (1997) indicate at least 20 years of continuous data for the indicators to be significant. For the purpose of this paper, where different policies are compared to each other, a synthetic long enough time series of flow data will be obtained using data from nearby rivers.

We considered the time series of two relatively close rivers, which are constantly monitored by the Federal Office for the Environment (FOEN). These are the *Calanasca* river, whose catchment measures about 120 km^2 and which receives *Carnac*'s waters less than 500 m from Buseno village, and *Rià di Roggiasca*, in another small catchment (about ten km apart from *Calanca* Valley), of 8.06 km^2. For the period October 2010–December 2011, hourly runoff data from the two rivers was transformed in logarithmic scale, standardized with respect to the relative catchment area, and a cross-correlation analysis was performed. Results show that *Carnac-Calancasca* streamflows are correlated at 86%, whereas *Carnac-Roggiasca* ones at 65%. *Calancasca* has a better cross-correlation with *Carnac* but its catchment is one order of magnitude bigger and 1.1% of its surface is constituted by glacier; moreover, an artificial water withdrawal from high in the *Calancasca* valley to a next one has been affecting its flows for about 50 years. This suggests that the high correlation value may reflect spurious components more than the natural catchment dynamics. *Roggiasca* river is not affected by any regulation/withdrawal and its catchment and hydrology are comparable with those of *Carnac* river. Such a correlation and a reconstruction of *Carnac* streamflow are shown in Fig. 2. For the period October 2010–December 2011 *Carnac* river reconstructed data by using *Roggiasca* time series above-said are slightly overestimated (about 13%), which we assume to be acceptable for the purposes of this paper (Mathez, 2011).

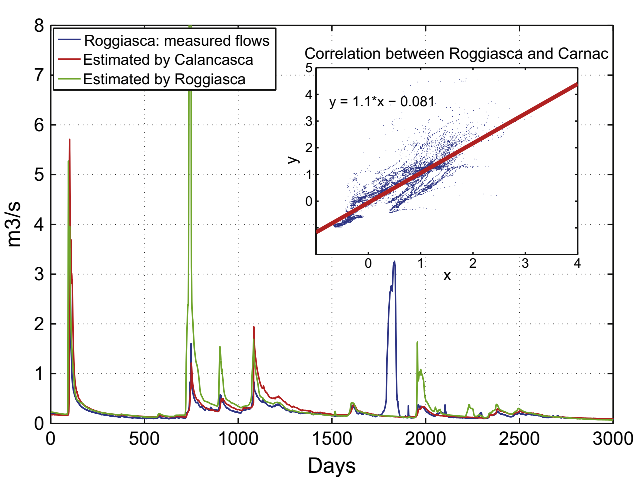

Fig. 2. Hourly flows in Carnac river, directly measured and obtained from Roggiasca and Calancasca. The smaller panel shows the correlation with the chosen signal: *X* represents Roggiasca flows expressed as the logarithm of the product between the hourly measured flow and its catchment surface; *Y* represents Carnac flows, expressed in the same way.

We focus on Buseno 2 project (Fig. 1a), for which the following five release policy scenarios have been proposed by practitioners:

- Scenario 1: to apply a standard MFR of 0.05 m^3/s, which corresponds to the minimum quantity imposed by law in the region, and leads to the modified flow regime shown in Fig. 1b;
- Scenario 2: to apply a two step MFR, which doubles the MFR in Scenario 1 from April 1th to September 30th;
- Scenarios 3–5: to release to the river a fixed percentage of the net inflow, respectively 20%, 25% and 30%, by assuring when possible the minimum of 0.05 m^3/s of Scenario 1. When the inflow is lower than 0.05 m^3/s, it is fully left to the river.

Scenario 6 is obtained by numerically generating the optimal allocation rule at the node based on the model of Perona et al. (2013) and presented in the next section.

2.2. Economical model for optimal water allocation

This model applies to situations like the one sketched in Fig. 1a and considers the environment as a non-traditional water user, which competes for water with an exploitation activity (King et al., 2003). Competition is based on the Principle of Equal Marginal Utility (PEMU) to define an efficient allocation strategy that maximizes the total benefit of the system (environment and hydropower). As a result, the hydropower activity and the riverine ecosystem respectively receive the flow rates Q_1 and Q_2, computed from the flow continuity at the node

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

and the PEMU (Perona et al., 2013),

$$b_1(Q_1(t); r_1) = b_2(Q_2(t); r_2), \tag{2}$$

once the marginal benefits functions for the hydropower $b_1(Q_1)$ and for the environment $b_2(Q_2)$ are known. The vectors of parameters r_1 and r_2 define the curves of each user, in particular b_1 being calculated as shown below. Eq. (2) expresses the concept of PEMU: unconstrained optimal water allocation for the global system (that is for the two users) results from distributing the incoming flow $I(t)$ in such a way that both users assign the same marginal benefit to their parts (Brown et al., 1990; Brown and daniel, 1991; Brown and Duffield, 1995; Alfieri et al., 2006).

The power house adopts a vertical shaft Pelton turbine with four-nozzles and a design flow of 0.84 m^3/s, for a maximum installed power of 2.23 MW. A few points of the efficiency curve of a vertical shaft Pelton turbine with four nozzles have been merged to the efficiencies of the generator and the voltage transformer, and converted into hydropower energy production. Since it is a run-of-river power plant it is not possible to control the production on the basis of the energy price. The annual production and revenues are linearly correlated by the annual mean net price: thus the shape of the hydropower benefit function would not change when expressed in revenues or energy production. The mean price depends on subsidies and market oscillations: To avoid such uncertainties and coherently with the purpose of this paper, we expressed hydropower benefits in terms of energy production [kW h]. The efficiency curve of a Pelton turbine is lower when using a small percentage of the nominal flow, than for higher per-

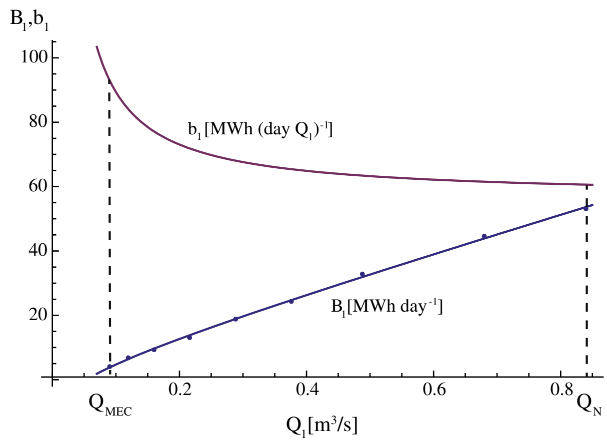

Fig. 3. Hydropower benefit (blue line) and marginal benefit (red line) for the case-study. These curves are the result of a fitting of available efficiency data (turbine, generator and transformer), converted into benefit [MW h] and represented as blue points. Q_{mec} and Q_N represent the lower and the upper boundaries of the working range. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

centages (Anagnostopoulos and Papantonis, 2007; Paish, 2002). It follows that the hydropower benefit $B_1(Q_1)$ for receiving a flow Q_1 is not linear for the case study and can be interpolated by the function

$$B_1(Q_1) = a - bQ_1 + c\ln(Q_1), \tag{3}$$

and the related marginal benefit (Alfieri et al., 2006) is obtained by differentiation

$$b_1(Q_1) = \frac{dB_1}{dQ_1} = -b + c/Q_1. \tag{4}$$

Both functions ($a = 1.28, b = 0.46, c = 0.28$) are shown in Fig. 3.

As far as the environmental marginal benefit function (henceforth referred to as EMBF) for receiving an amount of water Q_2 is concerned, we adopt the linear function

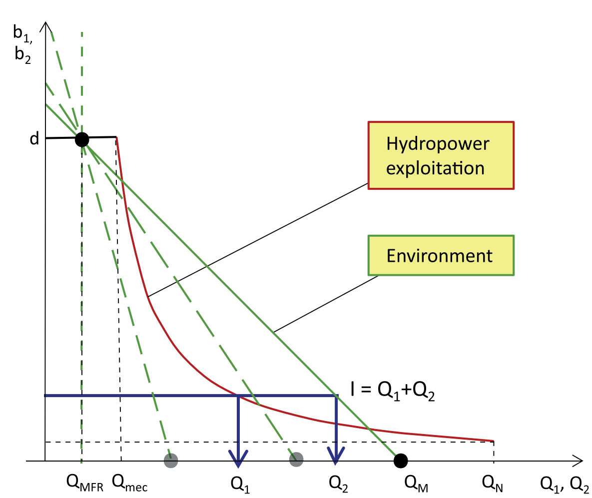

Fig. 4. Graphical application of PEMU: the red line represents the marginal benefit function for the power plant; the EMBF (in blue) is linear and depends on Q_{MFR} and Q_M (black dots). Once Q_{MFR} has been fixed, only Q_M remains as free parameter which determines the slope (see family of green curves). The space of competition between environment and hydropower starts respectively from Q_{MFR} and Q_{mec}; it ends in correspondence of the horizontal dotted line. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

$$b_2(Q_2) = \frac{dB_2}{dQ_2} = \left(d\frac{Q_M}{Q_M - Q_{MFR}}\right)\left(1 - \frac{Q_2}{Q_M}\right), \tag{5}$$

which is represented schematically in Fig. 4. The linear shape has been chosen for the sake of simplicity (Perona et al., 2013) and a broader justification is reported in Section 3.2. The corresponding environmental benefit (or utility) can be obtained by integration and is the 2nd order polynomial[1]

$$B_2(Q_2) = d\left(\frac{Q_2(2Q_M - Q_2)}{2(Q_M - Q_{MFR})}\right), \tag{6}$$

where d is the value of the marginal benefit functions at the beginning of the competition and fixes the economic scale (Fig. 4). Q_{MFR} is the minimal flow, which sets the theoretical lower limit of the competition (let alone some mechanical threshold Q_{mec} imposed on turbine functioning) and below which only the environment receives water (Perona et al., 2013). Q_M is a maximum flow setting the higher limit of the competition that identifies the point of maximum benefit for the environment, i.e. above which the flow rate starts yielding a smaller (even if still positive) benefit.

The discussion about destructive floods is still open (Tharme, 2003; Perona et al., 2009), because negative effects due to direct loss in biomass trigger in turn positive effects associated to renewal and biological transport processes. In our case this argument can be considered as secondary, since high flow rates easily overcome the power plant nominal flow and are not really influenced by adopted policies. Upon fixing Q_{MFR} (often imposed by law), then Q_M is the only degree of freedom left to the economical model. Notice that by changing Q_M, the slope of the EMBF changes as well (Fig. 4), and this affects the water redistribution between the two users. The resulting expressions for the allocated flow to each user are given in the Appendix, and are graphically shown in Fig. 4. Here we recall that the competition for water between the two users starts at the incoming flowrate $I = Q_{MFR} + Q_{mec}$, then the optimal water allocation corresponds to horizontal intercepts of the two marginal functions Q_1 and Q_2, while respecting Eq. (1), until the plant reaches its maximum flow. We therefore define the competition range as the range of natural inflows $I(t)$ between $Q_{MFR} + Q_{mec}$ and the flow leading the turbine to its maximum capacity Q_N (see Appendix).

Numerical simulations of all six scenarios have been run on a daily time step as requested by the implementation of IHA. Over the simulation time T_{sim} and within the competition range we compute the average

$$\Gamma = E[(Q_1(t) - Q_{mec})/(Q_2(t) - Q_{MFR})] \quad 0 \leqslant t \leqslant T_{sim}. \tag{7}$$

For scenarios 1 and 2, defined without competition range, the quantity Γ (Eq. (7)) is therefore undefined (see Section 3.2 for details). For scenarios 3, 4 and 5, Γ is equal to the ratio of the percentages of net water allocated (i.e., Scenario 3 has $\Gamma = 0.8/0.2 = 4$). In the competition range each unit of water left to the environment would correspond to an increment of energy production if diverted. Thus Γ identifies the relative importance between hydropower and environment following each release policy. For the purposes of this work, numerical simulations were run and the value of Q_M fixed ($Q_M = 0.208$ m^3/s, $\Gamma = 4.8$) in order to show that a more efficient solution than the one of Scenario 3 is possible. The term *more efficient* must be intended here as having both economic and ecological improved performances: the power production can in fact be increased by withdrawing a bit more water from the river; the ecological indicators (see hereafter) get benefit from changing the water redistribution policy (details in Section 3.2).

[1] It represents a mean environmental benefit, from the point of view of the other user; however environmental performances are evaluated in this work by ecological indicators (Section 2.3).

Table 1
Summary of the 32 indicators of hydrological alteration originally proposed by Richter in 1996, divided into five groups.

IHA statistics group	Regime charateristics	Hydrological parameters
Group 1: Magnitude of monthly water conditions	Magnitude timing	Mean value for each calendar month (1:12)
Group 2: Magnitude and duration of annual extreme water conditions	Magnitude duration	Annual minima 1-day means (13) Annual maxima 1-day means (14) Annual minima 3-day means (15) Annual maxima 3-day means (16) Annual minima 7-day means (17) Annual maxima 7-day means (18) Annual minima 30-day means (19) Annual maxima 30-day means (20) Annual minima 90-day means (21) Annual maxima 90-day means (22)
Group 3: Timing of annual extreme water conditions	Timing	Julian date of each annual 1-day maximum (23) Julian date of each annual 1-day minimum (24)
Group 4: Frequency and duration of high/low pulses	Frequency duration	No. of high pulses each year (25) No. of low pulses each year (26) Mean durattion of high pulses withineach year (27) Mean duration of low pulses withineach year (28)
Group 5: Rate/frequency of water condition changes	Rates of changes frequency	Means of all positive differences between consecutive daily values (29) Means of all negative differences between consecutive daily values (30) No. rises (31) No. falls (32)

2.3. Ecological indicators

Ecological performances of each policy are not computed by the integral of b_2 (Eq. (6)), but by means of IHA described in Table 1. They were originally proposed to characterize a given flow regime and, by comparison with the natural state, to quantify the degree of alteration after human intervention (Richter et al., 1996; Richter et al., 1997). In the absence of detailed empirical information about flow requirements, or in addition to them, they have also been used as a proxy of the riverine ecological status (Arthington et al., 2006; Petts, 1996; Petts et al., 2006; Petts, 2009). The goal is to safeguard the riparian environment and it can translate into operative terms by conserving as much as possible the natural flow variability which contributed to create and sustain such an environment (Poff et al., 2007; Rosenberg et al., 2000; Petts et al., 2008; Bizzi et al., 2012).

In the Range of Variability Approach (RVA) method introduced by Richter et al. (1997), the *Rate of non Attainment* (hereafter referred to as *RnA*) is defined as the fraction of post-regulation years in which each indicator falls outside the range defined by plus/minus one standard deviation around the mean. Notice that values outside the boundaries are not undesirable, but are important features of the flow regime. As an example, Fig. 5 shows this parameter computed for an exemplary natural flow ($RnA = 0.26$) and a classic MFR policy ($RnA = 0.10$). Clearly, *RnA* is not a sufficient descriptor because it does not take signal amplitude into account, so two signals can easily have same *RnA*, although looking very different (Richter et al., 1996; Richter et al., 1997). We propose to include this information with the coefficient of variation (*CV*), which is dimensionless and enhances the differences between the two regimes ($CV = 0.06$ and $CV = 0.01$ for the natural and the MFR regime, respectively).

Aiming to achieve flow requirements which can sustain the riparian ecosystem, we propose to look at the minimization of the distance between *RnA* and *CV* of the regulated and the natural flow conditions, expressed as

$$E[RnA_{sim}(i) - RnA_{NAT}(i)]^2, \tag{8}$$

$$E[CV_{sim}(i) - CV_{NAT}(i)]^2, \tag{9}$$

where i refers to each IHA listed in Table 1.

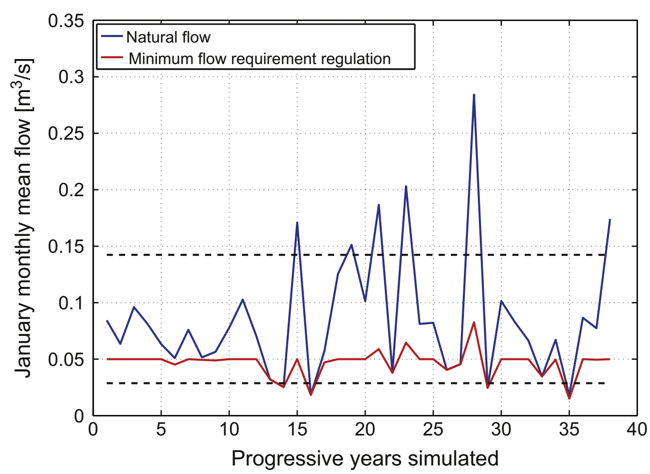

Fig. 5. Comparison for the monthly average flow in January for all years computed for the natural flow regime and a MFR regulation policy (Scenario 1). The blue line represents the monthly mean of the natural regime; the red one refers to Scenario 1 (regulation by classic MFR); dotted lines have been computed as *mean* ± *SD* of the natural signal, unless they overcome extreme values occurred during the simulation time; in this case extremes are used.

Another measure has recently been proposed (Bizzi et al., 2012) based on the minimization of the *virtual distance*, at daily time step, between simulated flow Q_{sim} and natural ones Q_{NAT} through the simulation time T_{sim}, i.e.

$$E[Q_{sim}(t) - Q_{NAT}(t)]^2 \quad 0 \leqslant t \leqslant T_{sim}. \tag{10}$$

Such a formulation, however, takes only volume differences and not benefits associated to variability and timing into account (i.e., effects of reduced flow in June are worse than in January). Even if suitable for complex optimization problems, e.g. involving dynamic programming, it is not used in the present work.

The 32 couples of *RnA* and *CV* relative to each alternative or policy being simulated can be aggregated using weighted averages, first computed inside each group and then across groups (Bizzi et al., 2012). We recommend that weights are defined case by case, with input from experts and stakeholders, so to consider the status of the entire riparian ecosystem and not only a part of it (e.g. fishes, which many technical studies focus on). In this work, we weighted 0.5 months October–March, and 1 months April–September for indicators of Group 1 because during summer months environmental needs for water are generally higher. This is reflected in nowadays popular MFR policies characterized by seasonal thresholds (e.g. scenario 2). Indicators of Group 3 (Julian dates of maximum and minimum annual flow) are less influenced by regulation in run-of-the-river systems (without water storage), and were not considered. All other indicators were averaged with weight 1 inside each group. Finally, values from the four considered groups were again averaged. By doing this for both *RnA* and *CV*, we eventually obtained two synthetic and dimensionless ecological indicators, called hereafter *Eco1* and *Eco2*. Other ecological indicators could be added when available, referring to single species habitat conditions (Milhous et al., 1984; Vismara et al., 2001) or ecosystem state (Scheffer et al., 2001). No specific ecological data are available for the small case study so we did not explore this possibility.

3. Results and discussion

3.1. Comparison between regulation policies

Fig. 6 summarizes all the simulated policies and their relative performances evaluated in terms of economical benefit (energy production) and ecological indicators (Eco1 and Eco2). The best policy is the one generating a maximal energy production and showing a minimal value of the ecological indicators, given that the latter measures the differences of the flow regime from natural conditions.

As expected, Scenario 1 has the greatest hydropower production and the worst environmental performance for both indicators (Fig. 6). Scenario 2, which accounts for an increase of the MFR during summer months, improves environmental performances at the expenses of a strong drop in the energy production compared to Scenario 1. Scenario 3, which is based on proportional repartitioning (80% hydropower, 20% environment) interestingly shows higher energy production and still a remarkable amelioration in terms of environmental indicators (*RnA* in particular) if compared to those of Scenario 2. Scenarios 4 and 5 are then determining a decrease in energy production and in the value of the indicators Eco1 and Eco2 as the proportion of releases to the environment increases. Finally, results from Scenario 6, though quite close, are better than those of Scenario 3, especially as far as the energy production is concerned. Notice, that while both *Eco1* and *Eco2* of Scenario 6 are located between those of scenarios 3 and 4, the energy production does not lie at all between the productions offered by such two scenarios, but it is higher for Scenario 6 (Fig. 6). A parametric analysis to study the sensitivity to changes in Q_M of ecological indexes, energy production and related importance of the users was also made and is shown in (Fig. 7): if from one hand Q_M affects almost linearly energy production, its impact on Eco1 and Eco2 is more complex. In general Eco1 and Eco2 decrease with Q_M but fluctuations are also present, reflecting how each IHA quantifies year by year the simulated flow dynamics. A similar non-linearity is expected when introducing other environmental indicators (Scheffer et al., 2001).

The results afore discussed find also explanation when looking at the actual allocated flows for all scenarios, which are shown in Fig. 8 and in the related inset panel. Scenario 3 (black line) does not imply systematic allocation of more water to the environment if compared to Scenario 2 (dotted red line), but better follows the pulsation of the natural flow regime. Thus, when river flows naturally decrease, Scenario 3 is able to leave less water to the environment, and vice versa. Nevertheless this could be problematic for certain species (e.g. because of reduced availability of fish habitat), when the flow decreases in Summer-time below 0.10 m^3/s (the second threshold of Scenario 2). Scenario 6 is able to mitigate this problem respect Scenario 3 (Fig. 8), although the two policies withdraw, in average, similar volumes ($\Gamma = 4$ and $\Gamma = 4.8$ for scenarios 3 and 6, respectively). We recall here that $\Gamma = 4$ means that the hydropower receives in average 4 times more water than the environment. However, while for Scenario 3 the ratio averaged for computing Γ is constant, this is generally not the case for Scenario 6. Indeed, the ratio $(Q_1(t) - Q_{mec})/(Q_2(t) - Q_{MFR})$ is a variable quantity being $Q_1(t)$ and $Q_2(t)$ not tighten to a fixed ratio but resulting from the optimal allocation imposed by PEMU. Thus, this ratio becomes a inflow dependent quantity, which can range from low values for relatively low inflows, to higher values for higher inflows.

A comparison between the two extreme scenarios 1 and 6 (respectively the classic MFR and the PEMU approach) in terms of four exemplary IHA, is plotted in Fig. 9, together with the value

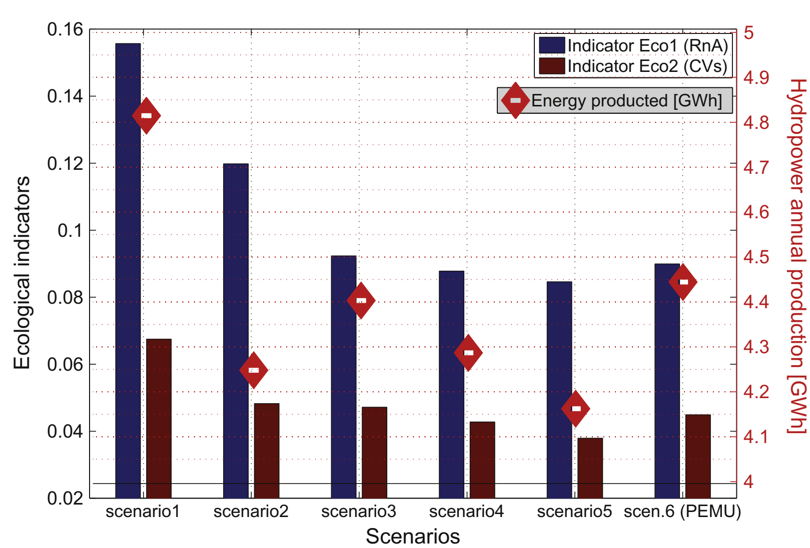

Fig. 6. Results in terms of synthetic ecological indicators (blue and dark-red bars) and hydropower production (red rhombs). The best policy is the one generating a maximal energy production and showing a minimal value of the ecological indicators, given that the latter measures the differences of the flow regime from natural conditions.

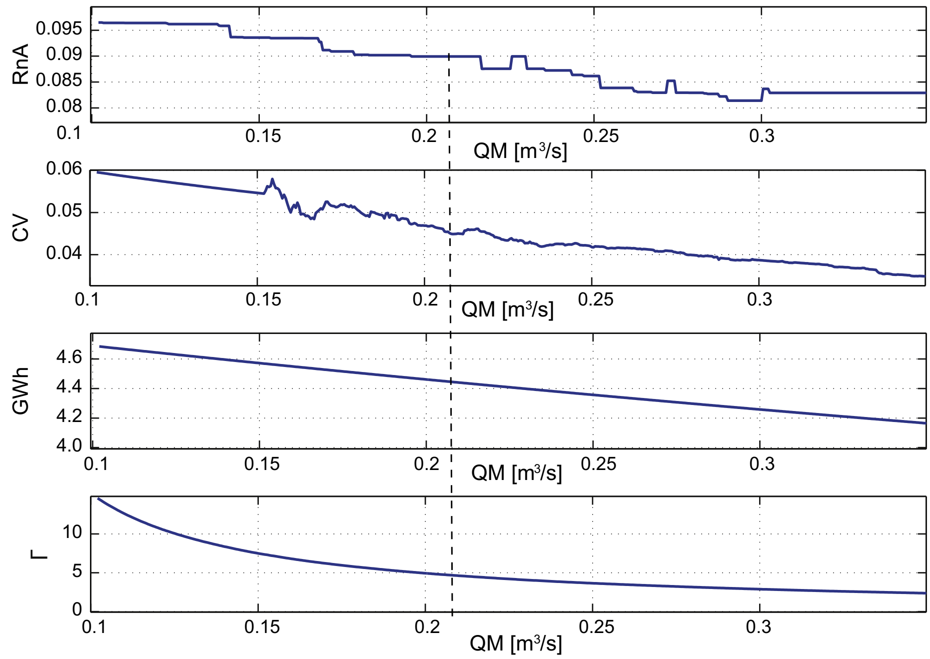

Fig. 7. Parametric analysis for Scenario 6. The free parameter of the linear EMBF, Q_M, is plotted against Eco1, Eco2, energy production and Γ. The dotted black line identifies $Q_M = 0.208\ m^3/s$, the value used to compute Scenario 6.

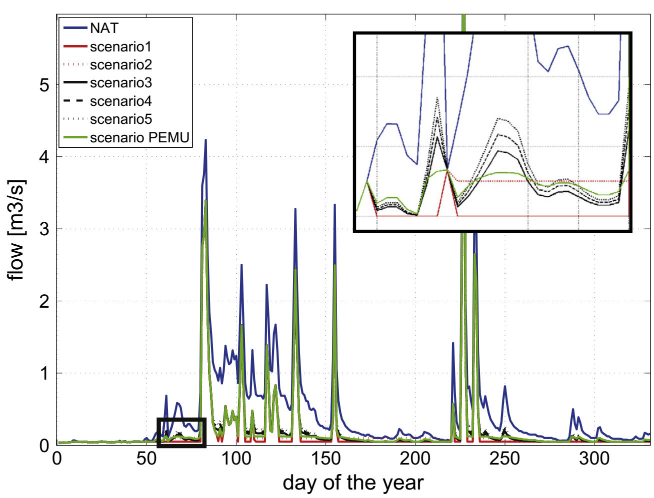

Fig. 8. Hydrographs corresponding to the six alternatives simulated during a representative year, from the January the 1st to December the 31st. Note that the green line leaves to the environment more water than the black line, for relatively low inflows; less water for higher inflows.

that the indicators take for the natural flow regime. Indicators 1 and 17, which describe properties of the flow magnitude (groups 1 and 2, see Table 1) do not show important differences between Scenario 1 and Scenario 6 if compared to the indicators of the natural flow regime (blue curve). This results from the fact that Scenario 6 withdraws, in the competition range, on average more than 80% of the water to the power plant. Hence, amelioration in terms of flow magnitude cannot be expected to exceed 20% with respect to the performances of Scenario 1. The amelioration is more significant when considering groups 4 and 5, which describe flow variability through frequency, duration, rate of change, etc., i.e. all features whose importance is well recognized (Poff and Ward, 1989; Poff et al., 1997; Arthington et al., 2006). In Fig. 9c, the values of indicator 28 for the natural flow regime and for Scenario 6 are practically identical, and very similar are those of indicator 30. If we compared Scenario 2 and 6 in terms of the same plots of Fig. 9, they would show very small differences among the indicators of groups 1 and 2, but still high differences among indicators of groups 4 and 5. Consistent with the resulting hydrographs of Fig. 8, IHA show in fact that the ecological amelioration achievable changing Scenario 1 by Scenario 2 would be just in terms of volumes, the water repartition rule remaining static.

3.2. Significance of all scenarios in terms of PEMU

Considering the 5 Scenarios proposed by practitioners from a PEMU point of view helps to deepen the understanding of the consequences of each policy. In particular, modeling scenarios 3, 4 and 5 (proposed on the basis of nothing but the common sense) is a way to support the idea behind them and a step towards dynamic flow releases. Scenario 1 can be seen in terms of PEMU as the limiting case of a vertical EMBF ($Q_M = Q_{MFR}$, see Fig. 4), which would quantify water demand as a perfect *inelastic good* (Perona et al., 2013), always requiring and receiving a constant flow. This is in conflict with general understanding that ecosystem benefits are not linear in the amount of water (Scheffer et al., 2001), definitely not constant, and so dynamic flow releases are important to sustain biodiversity (Poff et al., 1997; Arthington et al., 2006). Similar situation occurs for Scenario 2, which just shows a seasonal threshold for MFR.

Let us analyse the correspondence between the PEMU method and the simple practice of proportional redistribution adopted by practitioners. Both approaches result in more naturally-shaped hydrographs compared to MFR policies as seen in Fig. 8. With reference to Table 2 Scenario 6 required an assumption concerning the form of the EMBF, in this case chosen linear for the sake of simplicity, and consequently two parameters, Q_{MFR} and Q_M, to mathematically define it. Q_{MFR} is a constraint which is generally imposed by law, whereas we linked the choice of Q_M to the relative importance of the two competitors, that is the parameter Γ. Thus, defining at both political and social level the importance of the environment with respect to the hydropower, means choosing a mean value for Γ. In turn, this means to fix Q_M to a specific value and thus giving an economical meaning to the EMBF in relation

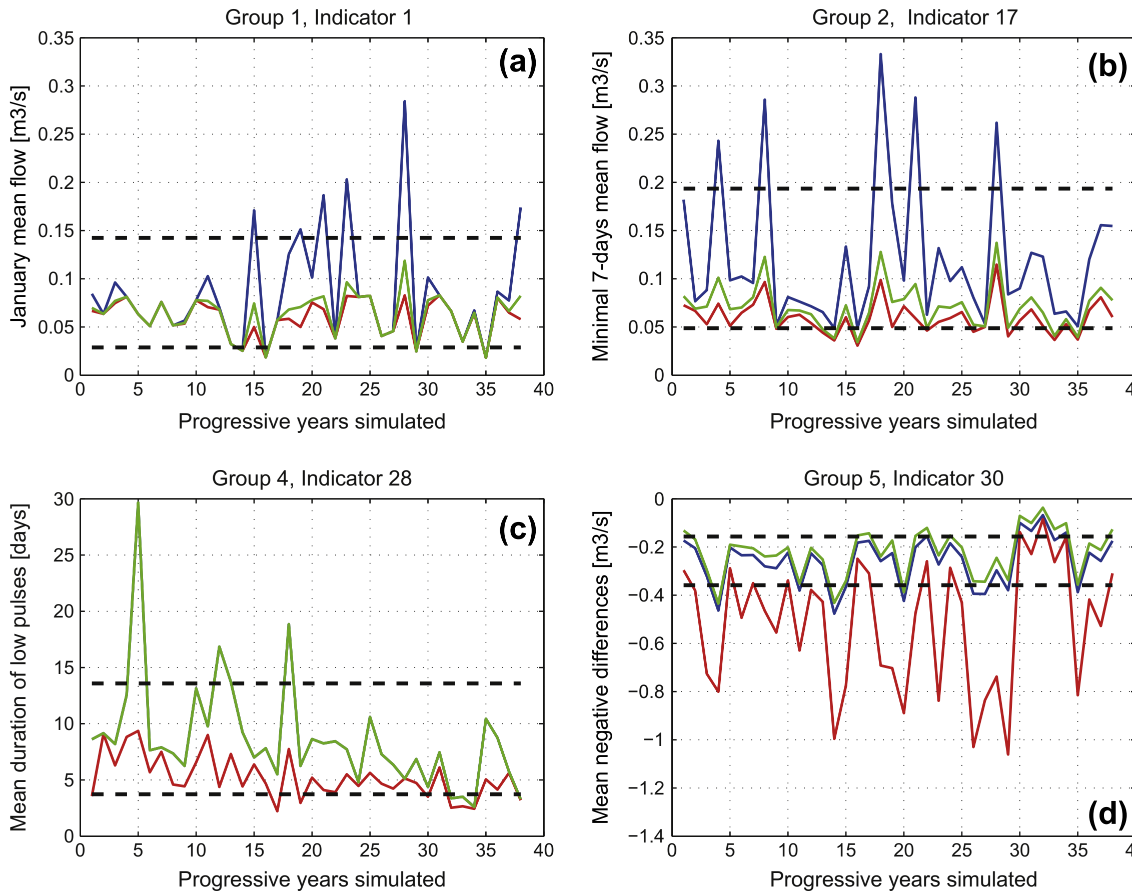

Fig. 9. Comparison between Scenario 1 and Scenario 6: four indicators, belonging to four different groups have been chosen as representative. The blue line corresponds to the natural flow regime, the red one to Scenario 1 and the green one to Scenario 6.

Table 2
Essence in terms of PEMU of proportional rules (e.g. Scenarios 3–5) and the allocation law proposed in this paper by means of a linear EMBF (Scenario 6). In both cases two constraint are to be defined: the lower boundary of the competition range (in this case the MFR), and the shape of the EMBF. Varying the control parameter is then possible to change the relative importance assigned to the environment and to the hydropower.

	Scenarios 3–5	Scenario 6
Constraint 1 (MFR)	MFR respected	MFR respected
Constraint 2 (EMBF)	Proportional to hydropower	Linear
Control parameter	Allocation %	Q_M

to the one of the hydropower. Based on common sense, practitioners have adopted a technique that can be seen as a special case of PEMU. Choosing a proportionality between the allocated flows corresponds to fix a nonlinear EMBF implicitly connected to that of the hydropower (Fig. 10a), to impose the minimal flow Q_{MFR} required by law, and to decide the relative importance between the competitors evaluated through the constant percentages of the allocated flows. Thus, although from a formal viewpoint the proportional approach and the PEMU (non-proportional) can be considered similar (see Table 2), the constraint of fixing the redistribution law generally reduces the flexibility of such policies. Yet, this simple practice is able to generate dynamic and ecologically sustainable releases, and given that it is actually operating in several countries, it leaves place to studies aiming at ameliorating its performances. Let us now discuss once more the linear shape of the EMBF and explain how does it work when combined with a PEMU repartition law. The goal was not to adopt the best function for maximizing environmental indicators (specific information for the study area are needed for this purpose), but to show an application easily comparable to real scenarios, proposed by practitioners. Notice that a linear b_2 conserves scale-properties (the first unities of water allocated to the environment in correspondence of small flows make its benefit grow faster than in correspondence of higher flows) and does origin water allocations which are not linear in the total inflow incoming. This agrees with the general agreement that ecosystem damages are characterized by non-linearity (see e.g. Shafroth et al., 2002) and can be seen from equations in Appendix: Q_2 depends on I and r (I), so it is not proportional to I. Despite what concerns dams control, in run-of-river power plants the flow left to the river (Q_2) strictly depends on the natural inflow (I) and the marginal benefit function of the other user (b_1). The former (I) automatically provides the flow variability, the latter (b_1) allows for a competition in terms of PEMU: as a result of the competition between a linear (b_2) and a nonlinear (b_1) marginal benefit function a nonlinear (or *non-proportional*) water repartition policy is gained (Fig. 10c and d). Fig. 10b and d show instead how the combination of two nonlinear marginal benefit functions (when proportional, like in scenarios 3–5) can yield a linear (or proportional) water repartition policy.

3.3. Operational feasibility of PEMU and recommendations

For the sake of completeness we briefly discuss here how the PEMU methodology can be put into practice in a cost-effective way. Fig. 10b shows how practitioners have quantitatively implemented a simple solution to withdraw a constant percentage of the total incoming flow. Flow releases generated on a PEMU basis require flow-dependent repartition, which at least ideally, can be obtained by shaping a water deflector that withdraws water

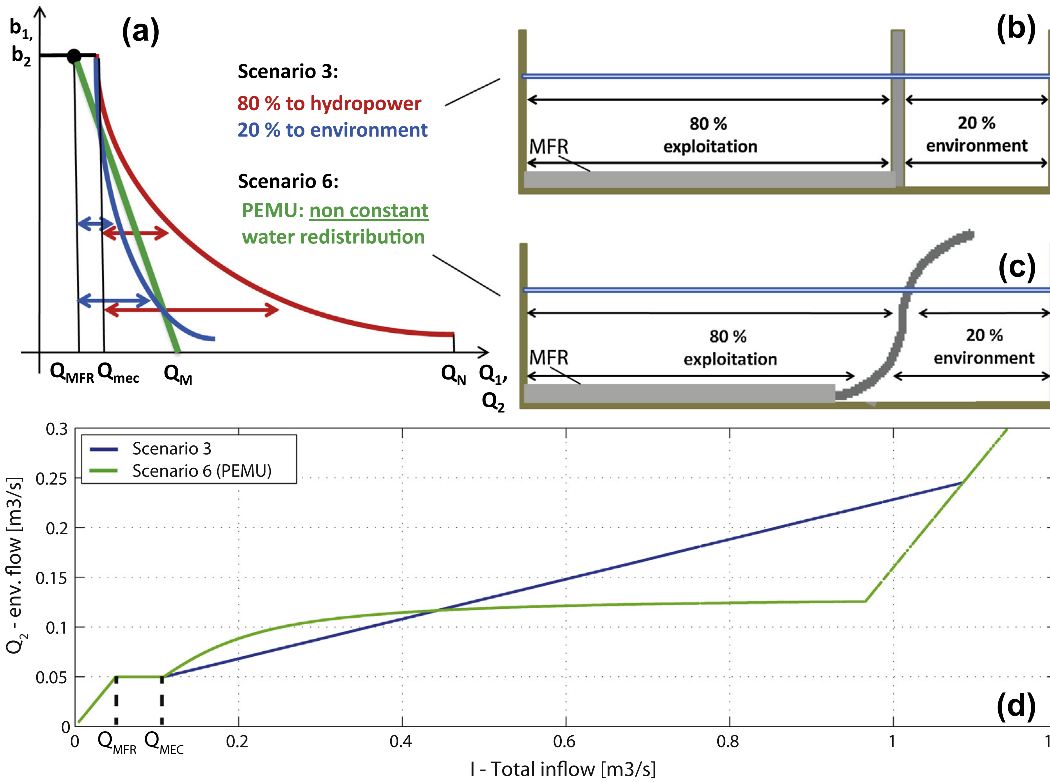

Fig. 10. (a) Qualitative comparison between Scenario 3 (blue), and Scenario 6 (green); the red line represents the marginal benefit function of the power plant. Horizontal arrows highlight that water repartition rates between red and blue line are constant in the entire competition space (e.g. 4 and 1, 8 and 2 if the 80% of the inflow to the hydropower and the 20% to the environment); between red and green lines they are not (they are in fact inflow-dependent). (b) Qualitative layout of a transect with a classic water deflectors thought to put Scenario 3 in practice: the fraction of water withdrawn does not depend on the inflow (on the level). (c) Qualitative layout of a transect with a shaped water deflector thought to put a general scenario obtained by PEMU in practice: for increasing inflows incoming the fraction of water withdrawn increases in a nonlinear way. (d) Repartition law in terms of total inflow (I) and water left to the river (Q_2) for scenarios 3 and 6: it shows how the result of a linear EMBF can be a non linear repartition low and viceversa. The gray step which appears in both panels (b) and (c) is imposed to take into account the MFR. For the sake of simplicity Q_{mec} was neglected in panels (b) and (c). Both solutions (b) and (c) withdraw on average the 80% of the inflow to the hydropower, but in subpanel (c) the percentage varies with the water level.

according to the allocation rules (Fig. 10c). However, an accurate study of the hydraulics and the possible problematics arising during the design phase (Niadas and Mentzelopoulos, 2008), for putting in practice such devices is recommended, though being outside the scopes of this paper.

To conclude, we summarize a simple procedure that can be adopted in order to help the evaluation of different flow release scenarios:

1. numerical simulation of possible release scenarios and evaluation of ecological indicators (Eco1, Eco2), economical benefits (energy production) and relative importance (Γ);
2. plot of results as Fig. 6 and selection of the interesting alternatives;
3. comparison of interesting alternatives by checking single ecological indicators, and visually controlling the resulting allocate flows (Fig. 8 and 9);
4. return to point (1) should a deeper comparison with alternatives including other specific ecological indicators (e.g. habitat indicators for fishes) be required;
5. selecting and implementing one of the allocation rules;
6. monitoring and assessment of long-term ecological impacts and specific-site data collection when practically and economically feasible.

Although the present paper stopped at step 3, we recommend steps 4 and 5 to be planned accurately given that the development and analysis of measures for ecological sustainability are still in their infancy.

4. Conclusions

We considered the riparian ecosystem from a holistic point of view (Tharme, 2003; King et al., 2003), espousing the idea that ecological sustainability of flow releases can be obtained by preserving the environment as close as possible to its natural state. Accordingly, two synthetic and dimensionless ecological indicators (*Eco1* and *Eco2*) have been introduced, which are useful for a first ranking of alternatives. We analyzed both the ecological and economical efficiency of several flow release policies for a typical mini-hydropower plant in Switzerland.

We showed that the simple practice of releasing a fixed percentage of the inflow to the environment, which has already been introduced in the eastern part of Alps, is an actual step towards eco-sustainability. Moreover, this strategy finds a mathematical correspondence with the method proposed by Perona et al., 2013. This similarity allows for an economical interpretation of environmental water use based on optimal redistribution following the principle of equal marginal utility. The choice of a linear EMBF already allows to improve proportional approaches (scenarios 3–5), because it generates non-proportional and inflow-dependent water allocation policies. We are able to generalize these results even when modifying the weights assigned to IHA or introducing new ecological indicators: it is in fact the wide range of inflow-dependent solutions, obtainable by either a linear or a non linear EMBF, which enables to improve the proportional repartition rules.

In conclusion, similarities with practical techniques (e.g. proportional repartition rules) and the general higher efficiency of the policies based on the principle of equal marginal utility encourage the choice of adopting the latter as a mean to improve both economic and ecological performances, without necessarily entailing higher installation costs or requiring advanced automatization.

Acknowledgements

We thank the Swiss National Science Foundation for funding the project ADAMANT (Grant No. *PP*00*P*2-128545/1), as well as Renato Gaggini of EcoControl SA, Giordano Favaro of Solesuisse SA and Reto Bolgiani of Premel SA for openly discussing practical details. Gregory Characklis, Benoît Crouzy and Katharina Edmaier are also acknowledged for useful comments and observations. We acknowledge the two anonymous reviewers for their detailed comments and constructive criticisms, which helped to clarify the work.

Appendix A

A.1. Water redistribution following PEMU

The optimal water allocation rule can be analytically computed by solving the system of Eqs. (1) and (2) for the variables Q_1 and Q_2 and the marginal benefits (4) and (5), which results in

$$Q_1^* = \begin{cases} 0 & I \leqslant (Q_{MFR} + Q_{mec}) \\ \frac{2c\Delta Q}{d(-I+Q_M+r(I))-b\Delta Q} & (Q_{MFR} + Q_{mec}) \leqslant I \leqslant I_{COMP} \\ Q_N & I > I_{COMP} \end{cases}$$

$$Q_2^* = \begin{cases} I & I \leqslant (Q_{MFR} + Q_{mec}) \\ \frac{dI-bQ_M+dQ_M-dr(I)+bQ_{MFR}}{2d} & (Q_{MFR} + Q_{mec}) \leqslant I \leqslant I_{COMP} \\ I - Q_N & I > I_{COMP} \end{cases}.$$

Therein,

$$\Delta Q = Q_M - Q_{MFR}, \tag{11}$$

$$I_{COMP} = Q_N + Q_{2N}, \tag{12}$$

$$r(I) = \frac{\sqrt{d^2(I-Q_M)^2 + 2d(2c+bI-bQ_M)\Delta Q + b^2\Delta Q^2}}{d}, \tag{13}$$

$$Q_{2n} = \frac{-cQ_M + cQ_{MFR} - bQ_MQ_N + dQ_MQ_N + bQ_{MFR}Q_N}{dQ_N}. \tag{14}$$

Q_N is the maximum turbine capacity, Q_{MFR} corresponds to the MFR imposed by law and Q_M is the point where the EMBF becomes zero. I_{COMP} and Q_{2N} are respectively the total inflow and the quantity left into the river when hydropower gets Q_N and competition ends. Q_{mec} is the minimum flow required by turbines to start producing (then fixed equal to $0.08Q_N$ as suggested by constructors).The parameter d is defined by

$$d = b_{mec}\alpha, \tag{15}$$

where b_{mec} represents the marginal benefit of the hydropower at the threshold Q_{mec} and α is a parameter which allows for change in the allocation rule between Q_{MFR} and Q_{mec}. We fixed it equal to one, in order to have d equal to b_{mec} to soon overcome the mechanical threshold (Fig. 4).

References

Alfieri, L., Perona, P., Burlando, P., 2006. Optimal water allocation for an alpine hydropower system under changing scenarios. Water Resour. Manage. 20, 761–778.

Anagnostopoulos, J.S., Papantonis, D.E., 2007. Optimal sizing of a run-of-river small hydropower plant. Energy Convers. Manage. 48, 2663–2670.

Arthington, A.H., Bunn, S.E., Poff, N.L., Naiman, R.J., 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecol. Appl. 16, 1311–1318.

Bartholow, J.M., 2010. Constructing an interdisciplinary flow regime recommendation1. JAWRA J. Am. Water Resour. Assoc. 46, 892–906.

Bizzi, S., Pianosi, F., Soncini-Sessa, R., 2012. Valuing hydrological alteration in multi-objective water resources management. J. Hydrol. 427–473, 277–286.

Brown, T., daniel, T., 1991. Landscape aesthetics of riparian environments – relationship of flow quantity to scenic quality along a wild and scenic river. Water Resour. Res. 27, 1787–1795.

Brown, T., Duffield, J., 1995. Testing part-whole valuation effects in contingent valuation of instream flow protection. Water Resour. Res. 31, 2341–2351.

Brown, T., Harding, B., Payton, E., 1990. Marginal economic value of streamflow – a case-study for the colorado river basin. Water Resour. Res. 26, 2845–2859.

Costigan, K.H., Daniels, M.D., 2012. Damming the prairie: human alteration of great plains river regimes. J. Hydrol. 444, 90–99.

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1–13.

King, J., Brown, C., Sabet, H., 2003. A scenario-based holistic approach to environmental flow assessments for rivers. River Res. Appl. 19, 619–639.

Mathez, C., 2011. Ria de Carnac in Buseno, Calancatal (GR) Erhebung der Abflussdaten 2010/2011 und der entsprechenden Dauerkurven fnr den Standort der geplanten Zentrale und fnr denjenigen der geplanten Fassung.

Milhous, R., Wegner, D., Waddle, T., Energy, W., Team, L., Flow, I., Energy, W., Team, L., Energy, W., Team, L., 1984. User's Guide to the Physical Habitat Simulation System (PHABSIM). Department of the Interior, US Fish and Wildlife Service.

Niadas, I., Mentzelopoulos, P., 2008. Probabilistic flow duration curves for small hydro plant design and performance evaluation. Water Resour. Manage. 22, 509–523.

Paish, O., 2002. Small hydro power: technology and current status. Renew. Sustainable Energy Rev. 6, 537–556.

Perona, P., Camporeale, C., Perucca, E., Savina, M., Molnar, P., Burlando, P., Ridolfi, L., 2009. Modelling river and riparian vegetation interactions and related importance for sustainable ecosystem management. Aquatic Sci. 71, 266–278.

Perona, P., Characklis, G., Duerrenmatt, D.J., 2013. Obtaining natural-like flow releases in diverted water reaches from simple riparian benefit economic models. J. Environ. Manage 118, 161–169.

Petts, G., Morales, Y., Sadler, J., 2006. Linking hydrology and biology to assess the water needs of river ecosystems. Hydroecol. Ecohydrol. 20, 2247–2251.

Petts, G. E., 2008. Hydroecology: The scientific basis for water resources management and river regulation. In: Wood, P. J., Hannah, D. M. Sadler, J. P. (Eds.,) Hydroecology and Ecohydrology: Past, Present and Future, John Wiley & Sons, Ltd, Chichester, UK, http://dx.doi.org/ 10.1002/9780470010198.ch13.

Petts, G.E., 1996. Water allocation to protect river ecosystems. Regulated Rivers: Res. Manage. 12, 353–365.

Petts, G.E., 2009. Instream flow science for sustainable river management1. JAWRA J. Am. Water Resour. Assoc. 45, 1071–1086.

Poff, N., Ward, J., 1989. Implications of streamflow variability and predictability for lotic community structure – a regional-analysis of streamflow patterns. Can. J. Fisheries Aquatic Sci. 46, 1805–1818.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., Stromberg, J.C., 1997. The natural flow regime. Bioscience 47, 769–784.

Poff, N.L., Olden, J.D., Merritt, D.M., Pepin, D.M., 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. Proc. Nat. Acad. Sci. 104, 5732–5737.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, J.H., Olden, J.D., Rogers, K., Tharme, R.E., Warner, A., 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biol. 55, 147–170.

Richter, B., Baumgartner, J., Wigington, R., Braun, D., 1997. How much water does a river need? Freshwater Biol. 37, 231–249.

Richter, B.D., Baumgartner, J.V., Powell, J., Braun, D.P., 1996. A method for assessing hydrologic alteration within ecosystems. Conservat. Biol. 10, 1163–1174.

Richter, B.D., Thomas, G.A., 2007. Restoring environmental flows by modifying dam operations. Ecol. Soc., 12.

Rosenberg, D., McCully, P., Pringle, C., 2000. Global-scale environmental effects of hydrological alterations: introduction. Bioscience 50, 746–751.

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413, 591–596.

Shafroth, P., Stromberg, J., Patten, D., 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecol. Appl. 12, 107–123.

Suen, J., Eheart, J., 2006. Reservoir management to balance ecosystem and human needs: incorporating the paradigm of the ecological flow regime. Water Resour. Res., 42.

Tharme, R., 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Res. Appl. 19, 397–441.

Vismara, R., Azzellino, A., Bosi, R., Crosa, G., Gentili, G., 2001. Habitat suitability curves for brown trout (salmo trutta fario l.) in the river adda, northern italy: comparing univariate and multivariate approaches. Regulated Rivers: Res. Manage. 17, 37–50.

Wehse, H., Pazhepurackel, V., Wittwer, C., 2011. Raccomandazione relativa all'elaborazione di strategie cantonali di protezione e di utilizzo nel settore delle piccole centrali idroelettriche.