Environmental Science & Technology

Article

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Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe

Danielle M. Tendall,*[,†,‡] Stefanie Hellweg,[‡] Stephan Pfister,[‡] Mark A. J. Huijbregts,[§] and Gérard Gaillard[†]

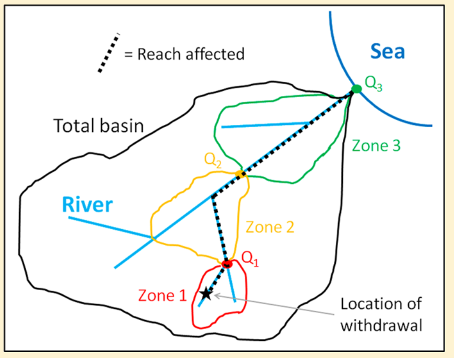
[†]Biodiversity and Environmental Management, Research Station Agroscope Reckenholz-Tänikon ART, 8046 Zürich, Switzerland

[‡]Institute of Environmental Engineering, ETH Zürich, 8093 Zürich, Switzerland

[§]Institute for Water and Wetland Research, Radboud University, 6500GL Nijmegen, The Netherlands

Supporting Information

ABSTRACT: In the context of climate change and food provisioning for a growing global population, the impacts of water consumption on aquatic biodiversity (e.g., river water consumption for irrigation) should be considered in Life Cycle Impact Assessment (LCIA). A previous LCIA method quantifying the potential impacts of river water consumption on fish biodiversity, using a species-discharge relationship (SDR), constituted an essential first step. This method is however limited in terms of regionalization and taxa considered, and predicts the potential risk of local species loss only. Here, we address these shortcomings by developing region-specific SDRs for Europe at various scales (continent, country, and eco-region), and including macro-invertebrate biodiversity. SDR exponents vary from 0.06 to 0.45 between regions, underlining the importance of such regionalization. Furthermore, we provide a new regionalized method which considers the location of water consumption within a river basin, by integrating the concept of longitudinal river zonation. This involves the use of a novel measure of potential loss of species richness, standardizing local species loss to an equivalent of global extinction and reflecting species vulnerability. The new method is applied in a Swiss case-study. The consideration of the location of water consumption within a basin was found to be of high importance in the assessment: potential species loss varied between 4.22×10^{-3} and 3.95×10^{-1} species (2 orders of magnitude) depending on location. This work thus provides enhancements in the assessment of potential impacts of river water consumption on aquatic biodiversity and contributes to the ecological relevance of the method.



■ INTRODUCTION

Water abstraction for crop production is expected to increase during the next century in many regions of the world, due to climate change and the increasing food demand of a growing population.[1] Globally, 71% of agricultural irrigation water is sourced from surface water,[2] thus it is important to reflect the impacts of river water consumption in life cycle impact assessment (LCIA), including the impacts on river ecosystems themselves. Different approaches exist for assessing the potential impacts of freshwater consumption;[3] in particular, characterization factors for the impact of river water consumption on freshwater species richness have been calculated by Hanafiah et al. (2011):[4] the fate factor relates consumption (in m^3) to reductions in river discharge as a one-to-one relationship (1 m^3 withdrawn and consumed on average per year results in 1 m^3 reduction in yearly average discharge at the river mouth). The effect factor is based on a relationship between the species richness of fish for whole river basins, to the average discharge at the mouth of the basins (the so-called species-discharge relationship or SDR[5]). The characterization factor (CF) represents the potentially disappeared fraction of species in the ecosystem, weighted by the volume of ecosystem affected for a certain time period per unit of water consumed [$PDF \cdot m^3 \cdot y \cdot m^{-3}$].

The method by Hanafiah et al. (2011)[4] contributed an important key idea in the assessment of freshwater biodiversity loss, but it has several limitations: the SDR used was developed for latitudes below 42° and near-natural rivers, hence it is not applicable to large parts of Europe (as well as Canada, the main part of Russia and Australia). The SDR used was furthermore developed using basins across the world; higher precision in the SDR might be achieved by developing SDRs specific to smaller regions. Approaches to model regionalized SDRs exist, including using sub-basins,[6] river archetypes,[7] and longitudinal river zones (e.g., for several regions in the U.S.[8]). However, such regionalized SDRs are currently not available globally, nor for Switzerland and Europe. In addition, local effects and influence of the location of water consumption within the basin are not addressed in the approach by Hanafiah et al. (2011).[4]

Received: March 1, 2013
Revised: February 7, 2014
Accepted: February 7, 2014
Published: February 7, 2014

dx.doi.org/10.1021/es4048686 | Environ. Sci. Technol. 2014, 48, 3236−3244

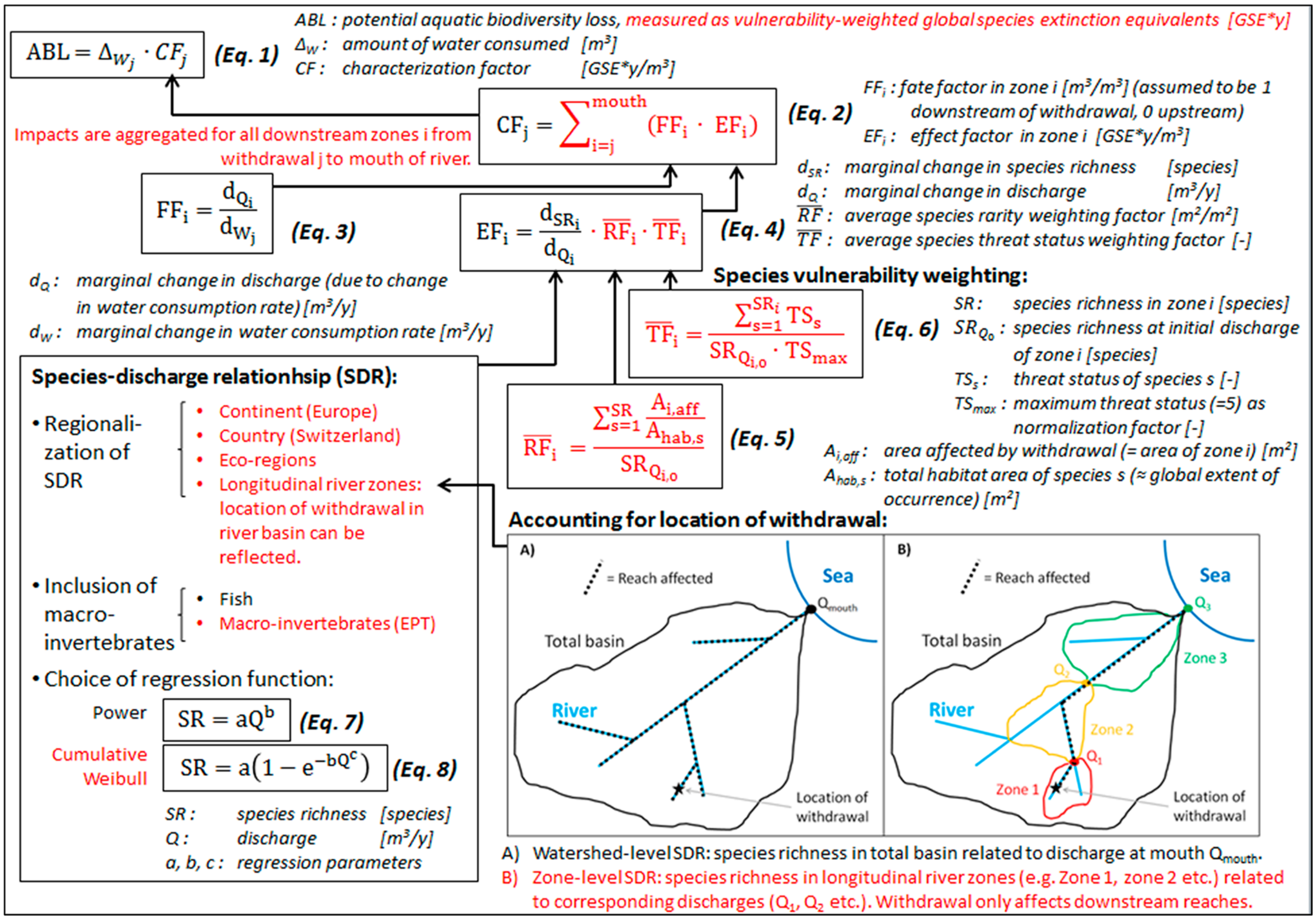


Figure 1. impact assessment framework and relevant equations. The novel components addressed in this paper are highlighted in red.

They estimated species loss in the entire watershed (using discharge at mouth), and weighted it by the river volume, regardless of location of the water consumption (thus assuming that the entire river is affected equally, no matter whether water consumption occurs at the spring or the mouth of the river). The weighting by river volume causes the magnitude of the characterization factor to be higher for large rivers than for small rivers, although actual species loss is often larger in smaller rivers. Finally, only fish species were included. Although fish are commonly used and recommended as indicators of aquatic ecosystem health,[9] using species richness of just one taxon remains an important limitation.[10] Macro-invertebrate diversity in particular could be added as a suitable indicator of changes in hydrology, particularly for smaller streams.[11,12]

We addressed the above-mentioned issues as follows: (1) provision of regionalized fish SDRs for Europe, at the continental, country, and eco-region scale; (2) test of an alternative regression function that can better capture SDR behavior at latitudes above 42°; (3) development of macro-invertebrate SDRs for Switzerland; (4) development of a refined method to calculate characterization factors, reflecting the location of consumption in a given basin: this approach relies on longitudinal zonation of rivers within their watersheds, and uses equivalents of global species extinction as a new measure of biodiversity loss. This method can be applied worldwide according to a tiered approach (described in the Methods section). Through a case study in Switzerland and Europe (which are also expected to be affected by increased water consumption for irrigation in future,[13,14] and where 75% of irrigation water is withdrawn from surface water[14]), we analyze the sensitivity of potential impacts on freshwater ecosystems to the above-listed developments. Finally, we discuss the implications and limitations of the new method proposed.

MATERIALS AND METHODS

Framework. Figure 1 gives an overview of the impact assessment framework and the developments provided here (in red), with the corresponding equations. The individual elements are explained below.

The calculation of potential impacts using a characterization factor is given by eq 1 (Figure 1). The characterization factor is composed of a fate and effect factor (Figure 1, eq 2). The fate factor (Figure 1, eq 3) is taken directly from Hanafiah et al. (2011),[4] and is assumed to be 1 for all parts of the river affected. This is valid for water consumption, which we understand as water withdrawn from the river and not returned to that river (e.g., evaporated, integrated in a product, or returned to another watershed or nonconnected water body) (for cases of nonconsumptive withdrawal, see Supporting Information, SI, section 12). Our developments principally address the effect factor (Figure 1, eq 4), where the potential change in species due to a change in discharge is calculated using the derivative of the species-discharge relationship, assuming marginal changes in discharge (if assuming non-marginal changes, then an average approach can be used, see SI

dx.doi.org/10.1021/es4048686 | Environ. Sci. Technol. 2014, 48, 3236−3244

section 1). The new developments (shown in red in Figure 1) are explained in the following paragraphs.

Regionalization of SDR. Hanafiah et al. (2011)[4] used an SDR that relates discharge of a total river basin to species richness within the basin (count of unique species occurring within the basin).[5] We hereafter refer to this approach as "watershed-level" (illustrated in Figure 1, Map A). We applied the watershed-level approach to Europe, and developed region-specific SDRs for several eco-regions in Europe (according to the Water Framework Directive:[15] defined as an area with relatively homogeneous ecological conditions, within which comparisons and assessments of biodiversity are meaningful), as well as for Switzerland and the Swiss lowlands orographic region[16] (considered representative for the Swiss lowlands biogeographic eco-region[17]). These eco-regions are shown in Figures S1 and S2 in the SI; the SDRs for each eco-region are developed using only the watersheds contained within the eco-region.

Inclusion of Macro-Invertebrates. Furthermore, we developed watershed-level SDRs for Switzerland for a subgroup of macro-invertebrates consisting of ephemera, plecoptera, and trichoptera taxa (commonly referred to as EPT). EPT are generally regarded as sensitive to disturbances, and 62% of Swiss EPT species are considered threatened or near threatened according to the IUCN Red List criteria.[18]

Choice of Regression Function. Species-area relationships and likewise species-discharge relationships are often assumed to follow a power regression function[19] (Figure 1, eq 7). This was used here as default regression function. However, applying this function to latitudes above 42° or non-natural rivers may overestimate species richness, due to an asymptotical behavior in the SDR at these latitudes[4] or within disturbed rivers (the SDR curve flattens out at large discharges, indicating a maximum limit in species richness). This is relevant for Switzerland and Europe, since actual maximum species richness is lower than predicted by nonregionalized models. This could be due to recent glaciations[20] or high human disturbance levels. Therefore, other regression functions for species-area relationships suggested in literature[21] were tested for the examples of Switzerland and Europe. In particular, the cumulative Weibull function (Figure 1, eq 8) was retained as a possible alternative and applied for the Swiss SDRs, since it can simulate an asymptote. All regression modeling and statistical tests were performed using the statistical software package "R".[22] Regression fitting objective was maximizing Pearson's R^2.

Accounting for Location of Water Consumption. In order to increase spatial detail and account for the location of water consumption in the basin, we applied another approach for developing SDRs (illustrated in Figure 1, Map B): species richness is counted in distinct longitudinal zones (subdivisions of the whole basin), and is related to the discharge at the outlet of each zone, providing what is hereafter referred to as a "zone-level" SDR.[8,23,24] Longitudinal zones are in essence defined to distinguish different species assemblages, with the assumption that each zone contains different species. Zonation may vary according to region and taxon.[25] This approach allows aggregation of the downstream effects from the point of water consumption to the river mouth: the loss of species in each zone affected by the water consumption can be summed, without double-counting of species (assumed to be distinct in each zone; the plausibility of this assumption is addressed in the Discussion section). Thus, the impact of a water consumption is sensitive to its location. We developed a zone-level SDR for fish in Switzerland, using a longitudinal zonation proposed for Europe[25,26] which defines four fish zones (trout, grayling, barbel, and bream) based on the slope and width of the river. Using the zone-level approach, the marginal CF providing the impact on aquatic biodiversity for a consumptive withdrawal of river water in a river zone j is given by eq 2 (Figure 1), and includes impacts in all the subsequent downstream zones i from the water consumption to the river mouth.

Species Vulnerability Weighting. The species loss estimated according to the zone-level approach is assumed to affect the river system downstream of the water consumption; however, this "local" loss does not inform us on the gravity of the impact from a global perspective. We therefore suggest weighting the local species loss in a zone by the ratio of the area affected (= area of zone) to the total global habitat area for each species occurring in the zone (in a similar way to previous work).[27] The total habitat area is estimated from global occurrence points (see below for data sources) in a very simple way using GIS (illustrated in SI Figure S3): cells of a raster grid are attributed a non-null value if they contain at least one occurrence point. Thus, the probable extent of habitat area per species is the summed area of all non-null cells (regardless of the density of points occurring in each cell). Consistently, the same approach is applied to the river segments which constitute a zone, by summing the area of cells which the segments traverse (with identical resolution, in this case 1 arc-degree; this is a coarse resolution but can be chosen differently for subsequent applications). This converts local species loss into global species extinction equivalents and replaces the previous weighting of species loss by total river volume. We use the average rarity of all the species present in a zone. (Figure 1, eq 5).

In addition to weighting the species loss by their rarity, we further weight it by the normalized extinction threat status of the species in that zone[27] (based on the IUCN Red List[28]), also averaged for all species within a zone (Figure 1, eq 6). The IUCN threat status of species occurring in the wild reflects further vulnerability of species by considering multiple criteria, such as abundance, turnover rate, fragmentation, dynamics in geographic extent, etc.[29] The IUCN threat status is qualitative: we convert it to a quantitative value, assuming a linear increment in vulnerability (as is often used per default in ecology[30]). The threat status is thus interpreted here as a scale of one to five, one being the category "least concern" and five being the category "critically endangered" (conversion in SI Table S1). Thus the loss of a critically endangered species is weighted five times higher than the loss of a species of least concern (note that this scale may be adapted if justified in future). We use the maximum threat status (= 5) as a normalization factor. Together, the rarity factor and the threat status factor provide a weighting of species loss by an indicator of their vulnerability. We use the current extent of occurrence and threat status of the species concerned as the reference state.

Note that this approach differs from the previously used $PDF \cdot m^3 \cdot y$: we take vulnerability-weighted equivalents of global extinction $[GSE \cdot y]$ as an indicator of potential impacts on biodiversity, in the form of biodiversity loss. This uses the absolute number of species potentially lost, weighted by the fraction of their habitat affected and threat status, which implies that the more taxa considered, the higher the impact. A normalization of different taxa, for example by the global characteristic species richness of the taxa (as suggested in SI eq S2), may be a way of addressing this issue.

dx.doi.org/10.1021/es4048686 | Environ. Sci. Technol. 2014, 48, 3236−3244

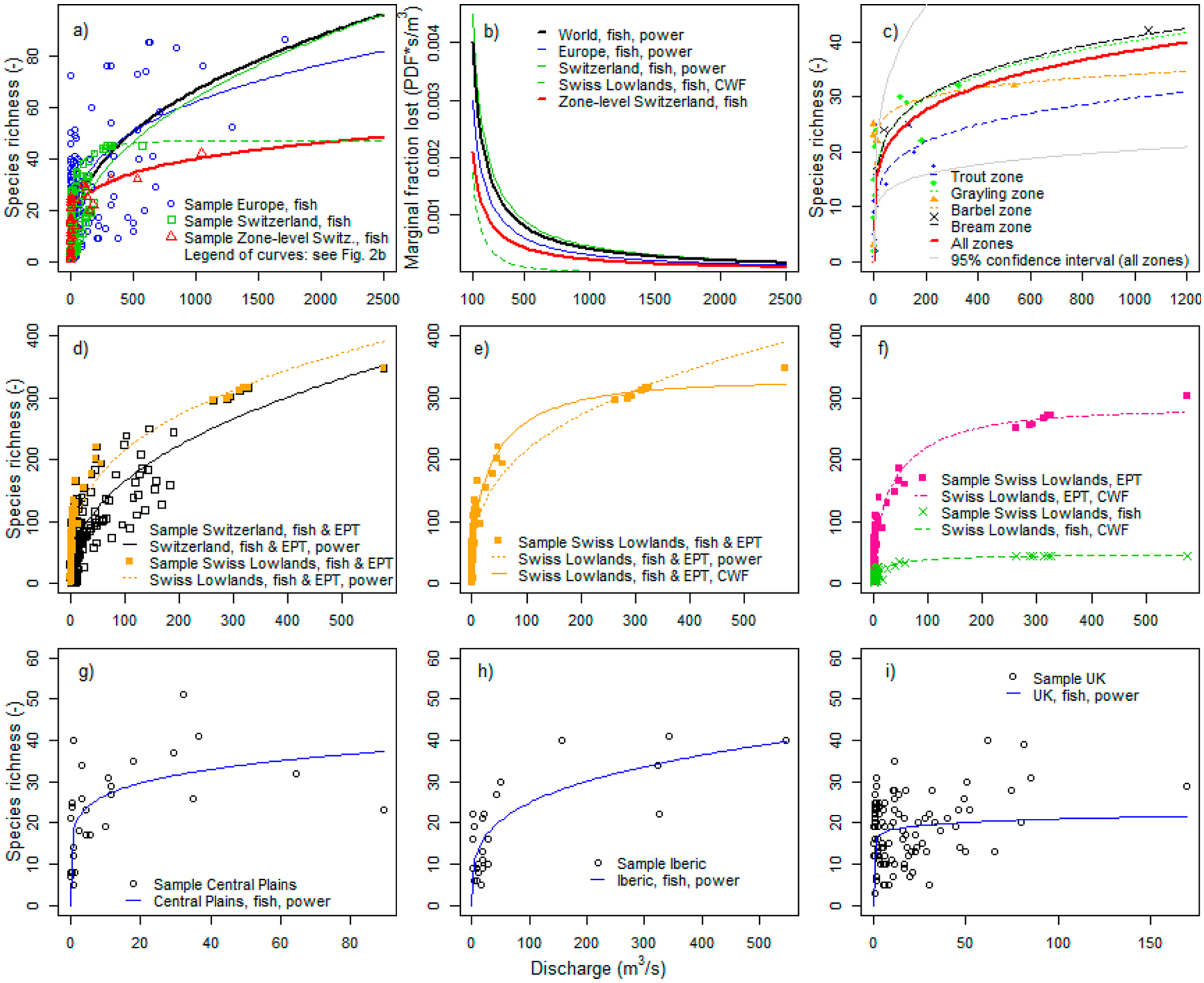


Figure 2. (a) Comparison of selection of new SDRs and the nonregionalized SDR,[5] and overview of data samples used for each SDR (note that the zone SDR is lower since it does not aggregate species richness in the entire watershed; rather, impacts are aggregated for each zone); (b) comparison of marginal species loss (PDF·s) per unit water consumption (m^3) for these different SDRs, according to initial discharge ($m^3 \cdot s^{-1}$); (c) zone-level SDRs for fish in Switzerland, for all longitudinal fish zones combined ("all zones"), and with distinction of the four zones used ("trout", "grayling", "barbel", and "bream"); (d) comparison of regionalization for total fish and EPT SDRs in Switzerland; (e) comparison of regression function for total fish and EPT in Swiss Lowlands; (f) comparison of fish and EPT single-taxa SDRs for Swiss Lowlands; (g) SDR for Central Plains European eco-region; (h) SDR for Iberic European eco-region; and (i) SDR for United Kingdom (U.K.) European eco-region. All SDRs are at watershed-level, unless specified otherwise.

The data sources are as follows:

- Species occurrence (for habitat area): Swiss Center for Biological Records[31] (Switzerland), Global Biodiversity Information Facility[32] (Global).
- Discharge: Swiss Federal Office for the Environment[33] (Switzerland), Global Runoff Database[34] and European Water Archive[35] (Europe).
- Catchment delimitation: Swiss Federal Office for the Environment[36] (Switzerland), Joint Research Center[37] (Europe).
- Eco-regions: Agroscope[16] (Switzerland), Water Framework Directive[15] (Europe).
- River width: Swiss Federal Office for the Environment[38] (Switzerland).
- Slope: Swiss Federal Office of Topography[39] (Switzerland), European Environmental Agency[40] (Europe).

Tiered Approach for Practical Application and CF Calculation. While this work only provides SDRs for Europe, this paragraph proposes an approach to select SDRs and derive characterization factors in other regions. SDRs on the zone level are not always available and sometimes other SDRs must therefore be used in practice. We provide the following recommendations according to a tiered approach in the selection of SDRs:

- Tier 1: The use of SDRs fitted for the zone-level, specific to the region concerned, and including as many taxa as possible, is recommended in order to capture significant intrabasin spatial variability, and decrease uncertainty.
- Tier 2: If such a zone-level SDR is not available, we suggest using a regionalized watershed-level SDR for fish, in combination with the zone-level CF method suggested here. This means that the same approach as in the zone-

level approach is followed (eq 2, Figure 1), but instead of using the slope of the zone-level SDR at the respective discharges of the zone outlets in eq 4, the slope of the watershed-level SDR at the respective discharges would be taken as an approximation. The location of water consumption and potential sensitive zones can thus be reflected, despite a larger error due to partial double-counting in the predicted impacts: indeed, the watershed-level SDR estimates species loss in the entire upstream catchment of a zone; adding this estimated loss for each zone thus "double-counts" some of the potential impacts, since upstream catchments of zones downstream of each other will overlap. This would be a conservative approach, as the watershed-level SDR predicts larger values of species loss than the zone-level approach (see Figure 2b).

- Tier 3: If regionalized SDRs are not available or cannot be developed, the nonregionalized SDR[5] can be used, although the uncertainty of the results may be significant.
- Tier 4: If longitudinal zonation is not possible (e.g., a river does not exhibit longitudinal species zonation), a regionalized watershed-level SDR can be used to estimate the loss of species within the whole basin, which can then be converted to GSE·y using the average rarity and threat status factors for the whole basin. The location of water consumption within the basin cannot be reflected in this case.
- Tier 5: If longitudinal zonation is not possible and a regionalized watershed-level SDR not available, a non-regionalized SDR can be used.

RESULTS AND DISCUSSION

Species-Discharge Relationships. Table 1 provides the equations of all the watershed-level SDRs developed along with R^2 as a measure of goodness of fit, and sample size n (graphical representations of the principle SDRs are available in Figure 2). All our SDRs show positive exponents as expected,[24] suggesting that fish and EPT species richness in these regions will be negatively affected by reductions in discharge. Our results generally compare well with reference values in literature: the exponents of the power functions in Table 1 range from 0.06 to 0.45; literature provides an average exponent of 0.24 for a worldwide range of comparable aquatic and terrestrial species-area relationships,[41] and the non-regionalized fish SDR has an exponent of 0.40.[5] Our range of exponents shows a high variability according to the region: similar variability has also been found for the U.S., with exponents ranging from 0.02 to 0.20.[6] This illustrates that region-specific SDRs can differ from average SDRs. Our exponent of 0.30 (95% confidence interval = [0.25; 0.35]) for Europe is higher than a previous value of 0.23 for western Europe,[42] which may be due to different data samples, or to changes in conditions. Higher exponents can indicate a stronger association between species richness and discharge, which in turn can indicate a higher vulnerability to changes in discharge[6,43] and potentially more rapid extinctions.[8] The exponents for the Swiss SDRs are generally higher than for the European SDRs; the exponents for the Swiss Lowland SDRs are generally lower than for the whole of Switzerland; and the exponents for the EPT SDRs are higher than for the fish SDRs. The latter finding suggests that EPT taxa may be more vulnerable to changes in discharge compared to fish, as has also been suggested for mussels.[6] A notable outlier in the SDR exponents is the case of the United Kingdom: this region shows a weak SDR with the lowest exponent of 0.06, similar to that found in certain regions of the U.S.[6] This may be due to the perturbing effect of other drivers of aquatic biodiversity, to historical biogeographic constraints,[6] or to the unfavorable sampling in this eco-region (with very few data points for large discharges).

Table 1. Equations of the Watershed-Level SDRs Developed (y = species richness, x = average discharge in $m^3 \cdot s^{-1}$), with R^2 as a Measure of Goodness of Fit, Sample Size n, and the Figure Where the SDRs are Displayed[a]

SDR (region, taxa, regression function)	equation	R^2	n	Figure
Europe, fish, power	$y = 7.82x^{0.30}$	0.35	271	2a
Central Plains, fish, power	$y = 18.75x^{0.15}$	0.39	29	2g
Iberic, fish, power	$y = 6.92x^{0.28}$	0.59	25	2h
United Kingdom, fish, power	$y = 15.93x^{0.06}$	0.04	104	2i
Europe, fish, CWF	$y = 6211.01(1 - e^{-0.001x^{0.30}})$	0.35	271	
Switzerland, fish, power	$y = 2.84x^{0.45}$	0.42	662	2a
Swiss Lowlands, fish, power	$y = 9.31x^{0.28}$	0.70	145	
Swiss Lowlands, fish, CWF	$y = 47.13(1 - e^{-0.20x^{0.47}})$	0.73	145	2a, 2f
Switzerland, fish and EPT, power	$y = 22.54x^{0.43}$	0.69	582	2d
Swiss Lowlands, fish and EPT, power	$y = 45.07x^{0.34}$	0.90	121	2d, 2e
Swiss Lowlands, fish and EPT, CWF	$y = 326.7(1 - e^{-0.12x^{0.56}})$	0.93	121	2e
Swiss Lowlands, EPT, power	$y = 35.98x^{0.35}$	0.89	121	
Swiss Lowlands, EPT, CWF	$y = 280.9(1 - e^{-0.11x^{0.57}})$	0.92	121	2f
Zone-level Switzerland all zones, fish, power	$y = 9.32x^{0.21}$	0.53	53	2a, 2c
Zone-level Switzerland − trout zones, fish, power	$y = 7.75x^{0.20}$	0.21	35	2c
Zone-level Switzerland − grayling zones, fish, power	$y = 10.88x^{0.19}$	0.79	11	2c
Zone-level Switzerland − barbel zones, fish, power	$y = 19.26x^{0.08}$	0.46	5	2c
Zone-level Switzerland − bream zones, fish, power	$y = 10.63x^{0.20}$	0.95	3	2c

[a]EPT = ephemera, plecoptera, trichoptera; CWF = cumulative Weibull function; power = power function.

For Switzerland, the cumulative Weibull function (CWF) provides a better fit than the power function, as measured by Pearson's R^2 (Table 1) and confirmed using Akaike Information Criterium (AIC; Swiss Lowlands, fish: AIC = 930 for power, 918 for CWF; Swiss Lowlands, fish and EPT: AIC = 1119 for power, 1075 for CWF; Swiss Lowlands, EPT: AIC = 1097 for power, 1061 for CWF). This suggests that the CWF may indeed be preferable for modeling SDRs in cases where the SDR tends toward a limited maximum number of species. In the case of Europe, this was not observed, and the CWF did not improve fit.

Figure 2 compares a selection of the SDRs developed here with the nonregionalized SDR[5] and shows details of the zone-level SDR, as well as details of selected SDRs for different taxa, regression functions and regions.

According to Figure 2a, the regionalization of the SDR (nonregionalized – Europe – Switzerland) causes smaller differences in species richness prediction than changing the function used (e.g., CWF rather than power function). The asymptotical behavior of the SDR in the Swiss Lowlands can be seen through the shape of the SDR using the CWF. The consequence of this behavior is that marginal changes in discharge no longer affect the predicted species richness above a threshold discharge. As previously mentioned, recent glaciations, or degraded quality of the rivers could explain this behavior. The SDR at the zone-level shows a similar shape to the SDRs at the watershed-level, although predicted species richness is larger at the watershed-level (as can be expected, since species are aggregated over all zones within the whole watershed). Figure 2b shows that differences in predictions of marginal fraction of species lost ($PDF \cdot s \cdot m^{-3}$) are largest for rivers with lower discharges (e.g., maximum difference 2.76×10^{-3} $PDF \cdot s \cdot m^{-3}$ at 100 $m^3 \cdot s^{-1}$). This difference, however, rapidly decreases with increasing discharge (e.g., 8.13×10^{-4} $PDF \cdot s \cdot m^{-3}$ at 500 $m^3 \cdot s^{-1}$ and 1.79×10^{-4} $PDF \cdot s \cdot m^{-3}$ at 2500 $m^3 \cdot s^{-1}$).

The zone-level SDR of fish for Switzerland was developed both per zone, and for all zones combined (i.e., all data points together with no distinction of zones) (Figure 2c, equations in Table 1). According to our hypothesis that more homogeneous regions should show a better fit, the SDRs per zone are expected to be more precise. The confidence intervals of the SDRs per zones were, however, relatively large due to small data samples; the SDR for all zones combined showed the smallest interval (and encompassed all the SDRs per zone at the 95% confidence level, with regression parameters intercept a = [7.07; 11.63] and slope b = [0.15; 0.26]). We therefore used the SDR for all zones combined ($y = 9.32x^{0.21}$; $R^2 = 0.53$), which represents the general relationship between zone discharge and species richness, without distinction between zone type. Zone-level SDRs in various regions of the U.S. show exponents ranging from 0.13 to 0.45,[8,24,43] and these exponents are generally higher than watershed-level SDRs for the same regions. Our zone-level SDR exponent (= 0.21) fits within this range, and is lower than that of the corresponding watershed-level SDR (i.e., 0.45). We verified that the area of each zone observed (each point in Figure 2c) is unrelated to its species richness ($R^2 = 0.005$, $n = 53$; data available in SI Table S2). This supports the hypothesis that species richness is related to discharge rather than observation area in Switzerland.

Case Study. We conducted a case study in order to provide an application example of the method proposed. The case study consists of an agricultural scenario maximizing productivity for the climate in 2050 in the Swiss watershed "Broye" (a subcatchment of the Rhine) (see Klein et al. (2012)[44] and the SI for details). This implies an increase in irrigation, resulting in consumptive withdrawals of 32.23 mio $m^3 \cdot y^{-1}$ (equivalent to a decrease in average discharge of 1.02 $m^3 \cdot s^{-1}$). This is roughly 9% of the current Broye river discharge (initial yearly average discharge 11.73 $m^3 \cdot s^{-1}$) and 0.05% of the current total Rhine river discharge (initial yearly average discharge 2254.06 $m^3 \cdot s^{-1}$). The sensitivity of the results to the location of the water consumption was assessed: using the method proposed here and the zone-level SDR developed for Switzerland, impacts were calculated in GSE·y according to two hypothetical locations of water consumption (see SI Figure S4), and the contributions of each zone were analyzed. Implications of applying the different tiers proposed above are discussed.

Absolute local species loss in the different river zones varies by 2 orders of magnitude, from 4.22×10^{-3} in the bream zone to 3.95×10^{-1} in the grayling zone. Therefore, if the same water consumption were to occur further downstream, for example in the agricultural plains of The Netherlands (see SI Figure S4), the impact in terms of local number of species lost would be much lower compared to the impact if withdrawn upstream (see SI Table S5, rows 1 and 2). This difference is explained by two elements captured by our new method. First, if assuming water consumption in the upstream subcatchment Broye, then the potential impacts of the water consumption in the first zones affected are high (the SDR is steeper at lower discharges: rivers with smaller discharge have a higher loss of species per unit water consumption) (see SI Table S4). If assuming water consumption in the agricultural plains of The Netherlands, then the SDR is flatter for the larger initial discharge. Second, the number of zones affected if assuming water consumption upstream is higher than if assuming water consumption downstream, for which only the bream zone is considered affected. This difference would not be captured if using a method at the watershed-level, which would apply a single CF for the whole of the Rhine basin.

Weighting local species loss with the rarity factor and threat level for each zone (Figure 1, eqs 4–6; see SI Table S3 for rarity and threat status factors of each species concerned in the case study), the potential impacts of river water consumption modeled for the case study amount to 1.8×10^{-5} GSE·y (the relative contribution of each zone affected is provided in SI Table S4). The difference of threat- and rarity-weighted species loss between zones is much smaller (factor of 2.5, SI Table S4) compared to local species loss (2 orders of magnitude), because the rarity factor tends to be higher for larger zones in this case study (SI Table S3) and therefore counteracts the larger number of local species lost in smaller rivers. However, this may be different in other case studies, as the rarity factor also depends on the extent of occurrence of the species, in addition to zone area. The contribution of the threat status weighting in this case study is small, since all zones show a similarly low threat status. However, we expect that this factor will show major differences for hotspots of biodiversity where many threatened species are located. Note that translating potential local species loss to an equivalent of global extinction generally results in very small values for a marginal water consumption; however, impacts are expected to reach high values in case of a nonmarginal water consumption or in case of river systems with endemic species.

The uncertainty of the result related to the inherent uncertainty in the SDR was assessed: using the 95% confidence interval of the zone-level SDR, the calculated impact increases or decreases by a factor 2.2, for the 2.5% and 97.5% confidence intervals respectively. Further uncertainty may arise if applying tiers 2 or 3 in practice. For the present case study, a quantitative comparison of impacts according to the tier used is provided in SI Table S5. It confirms that the impacts are overestimated if using watershed-level SDRs (tier 2) rather than a zone-level SDR (when aggregating impacts in all downstream zones, as proposed in Figure 1), due to the related double-counting. The uncertainty of the result related to the assumption of marginality was assessed by calculating the impact using the average approach in the grayling zone (where water consumption represents 9% of the available discharge) instead of the marginal approach: the calculated impact increases by a factor 5 (from 1.8×10^{-5} GSE·y to 8.78×10^{-5} GSE·y).

Note that in this theoretical case study, impacts were estimated based on current discharges rather than discharges simulated for 2050: indeed, consistent discharge predictions for the climate scenario considered were only available for the grayling zone. In the latter, discharge would decrease by 26% in the climate scenario considered; the estimated impacts in this case study would increase by 24%. The impacts in the other zones would very probably also increase if considering the future discharge for the same climate scenario.

Critical Appraisal. The approach proposed in this work enables a regionalized estimation of potential impacts of river water consumption on aquatic biodiversity, with differentiation of the location of consumption within a river basin, and consideration of sensitive zones in the basin. Potential species loss is no longer weighted by total river volume, avoiding characterization factors dominated principally by river volume. Local species loss is rather converted to an equivalent of global species extinction via a weighting that reflects species' vulnerability.

For the application of the approach presented here, the location of river water consumption within a basin must be known, since it can cause substantial differences in the estimated impact.

The cumulative Weibull function may be preferable to the power function for modeling SDRs for cases where species richness tends toward a limited maximum number of species.

The rarity factor and threat status reflect different aspects of species occurrence (indeed, the first considers total habitat extent, whereas the second considers dynamic changes in habitat extent in addition to other vulnerability criteria). The two weighting factors were averaged individually for each zone in this work and then multiplied; they could also be multiplied for each species individually, and then averaged for the zone. The disaggregation of the two factors allows a higher flexibility for inclusion or omission in the calculation if necessary. For this case study, the sensitivity of the CFs and results to this aggregation choice was very low (<1% change).

Several limitations in our proposed approach remain: (1) we have neglected the effect of temporal variability or seasonality in water use and exposure of aquatic organisms (using only annual averages). It is unclear how seasonality would influence the impacts (higher seasonal stress may be counterbalanced by recovery of the ecosystem after a temporary disturbance).[45,46] (2) The causal link between discharge and species richness is not well supported yet. Field-survey literature[9,47] is inconclusive, with direction of response even being inverted in some cases. (3) We assume that species are distinct within each longitudinal zone, although some species are able to survive in several such zones. For instance, the European eel (*Anguilla anguilla*) occurs over a broad range of longitudinal zones.[48] Additionally, migratory species may inhabit several zones during their life cycle. Literature shows two schools of thought in this respect, which either accept or reject this zonation assumption.[25] This is an issue only if aggregation of species loss in zones does not use the habitat rarity weighting factor (as proposed in Figure 1, eq 5), which corrects for double-counting of species even if they occur and are counted in several zones. Indeed the rarity factor weights the loss of a species in a zone by the area of its habitat affected in that zone: thus if a species is affected in several zones, its habitat affected in each zone can be aggregated, and the rarity weighted species loss can likewise be aggregated. Aggregation of potential species loss in downstream zones without rarity weighting is therefore adequate only for regions which exhibit longitudinal zonation with mainly distinct species. (4) Our method only accounts for effects in the zone of water consumption and strictly downstream, excluding feedback effects in upstream zones (such as loss of connectivity and isolation of upstream populations[49]). (5) Our weighting by the rarity factor relies on the use of the most complete species occurrence data set available to date, which nevertheless does not guarantee that all occurrences have been registered: this can introduce a bias in the results. (6) The reference state is the current extent of occurrence and threat status of the species concerned. This reference should be updated regularly, when the extents of occurrence and threat status data are updated.

Outlook. The SDRs developed here apply not only to near-natural rivers, but a broad range of rivers occurring in these regions. Therefore, they may also be assumed valid for rivers strongly affected by human interventions. The zone-level SDR we provide is valid for Switzerland. The existence of such a zone-level SDR should also be verified for other regions (such as done for certain regions in the U.S.[24,43]). Zone-level SDRs specific to individual zone types could be more precise than zone-level SDRs for all zone types together (as used here).

The suggested approach using zone-level species richness is highly relevant for other regions in the world exposed to river water consumption. It may be applied to calculate CFs for further regions, for which longitudinal zonation exists (as demonstrated here for a case study). Operationalization of this approach for application in LCA requires an expansion of CF calculation to a global spatial coverage, and an increase in spatial resolution of inventory data (for details, see SI section 13).

The suggested weighting of species loss by rarity and threat status reflects the gravity of the impact at a global scale and enables conversion to a weighted equivalent of global species extinction, providing a quantifiable relation with biodiversity loss. A consistent use of this approach for other ecosystem damages would enable aggregation across biodiversity indicators where desirable; it has been done for wetlands,[27] and is also possible for terrestrial and marine ecosystems.

■ ASSOCIATED CONTENT

Supporting Information

Characterization factor for nonmarginal water consumption, maps of the eco-regions, an extended analysis of the developed watershed-level SDRs, an illustrated explanation of the rarity weighting, the conversion of IUCN criteria used, the data for the zone-level SDR, details of the case study, calculated weighting factors and impacts, a comparison between impact units, a suggested approach for nonconsumptive river water withdrawals, and inventory requirements. This material is available free of charge via the Internet at http://pubs.acs.org.

■ AUTHOR INFORMATION

Corresponding Author

* Phone: +41 44 362 85 56; e-mail: danielle.tendall@usys.ethz.ch.

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

We warmly thank Yves Gonseth and Joerg Strackbein for providing precious data, Christian Stamm, Thomas Walter, Armin Peter, and Joachim Hürlimann for their critical expertise, Francesca Verones for insightful discussions and Ruth Freiermuth-Knuchel for initial support. We also thank four anonymous reviewers for their excellent comments. This work is part of the AGWAM Project (No. 406140_125957/1) financed by the Swiss National Science Foundation in the framework of the National Research Program 61.

■ ABBREVIATIONS

AIC	Akaike information criterion
CF	characterization factor
CWF	cumulative Weibull function
EF	effect factor
EPT	ephemera, plecoptera, and trichoptera
FF	fate factor
GSE	global species extinction equivalents (weighted by vulnerability)
IUCN	International Union for the Conservation of Nature
ABL	potential aquatic biodiversity loss (in terms of species richness)
PDF	potentially disappeared fraction of species
RF	rarity factor
SDR	species-discharge relationship
SR	species richness
TS	threat status

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