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Empirical Validation of MesoHABSIM Models Developed with Different Habitat Suitability Criteria for Bullhead *Cottus Gobio* L. as an Indicator Species

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Abstract: Application of instream habitat models such as the Mesohabitat Simulation Model (MesoHABSIM) is becoming increasingly popular. Such models can predict alteration to a river physical habitat caused by hydropower operation or river training. They are a tool for water management planning, especially in terms of requirements of the Water Framework Directive. Therefore, model verification studies, which investigate the accuracy and reliability of the results generated, are essential. An electrofishing survey was conducted in September 2014 on the Stura di Demonte River located in north-western Italy. One hundred and sixteen bullhead—*Cottus gobio* L.—were captured in 80 pre-exposed area electrofishing (PAE) grids. Observations of bullhead distribution in various habitats were used to validate MesoHABSIM model predictions created with inductive and deductive habitat suitability indices. The inductive statistical models used electrofishing data obtained from multiple mountainous streams, analyzed with logistic regression. The deductive approach was based on conditional habitat suitability criteria (CHSC) derived from expert knowledge and information gathered from the literature about species behaviour and habitat use. The results of model comparison and validation show that although the inductive models are more precise and reflect site- and species-specific characteristics, the CHSC model provides quite similar results. We propose to use inductive models for detailed planning of measures that could potentially impair riverine ecosystems at a local scale, since the CHSC model provides general information about habitat suitability and use of such models is advised in pre-development or generic scale studies. However, the CHSC model can be further calibrated with localized electrofishing data at a lower cost than development of an inductive model.

Keywords: modelling; habitat preference; suitability; MesoHABSIM; fish sampling; alpine streams

1. Introduction

One reason for the discrepancy between the needs of conservation, restoration of running waters, and water management practice is the lack of effective instruments for predicting the impact of these activities. This situation often leads to damaging of the river ecosystem [1]. One solution, which enables minimization of the negative impact of human activities, could be application of actions based on scenarios derived by instream habitat models. Such models allow prediction of the alteration to a river ecosystem caused by river training or hydropower operations and are therefore useful water management tools for restoring river ecological status as a step for the implementation of the Water Framework Directive demands [2–5].

Instream habitat models describe relations between fauna and flora and their physical environment. They allow for quantitative assessment of changes in habitat availability and suitable conditions for aquatic organisms under specific environmental circumstances. Such methods link numerical modelling of the physical environment (e.g., hydraulics, river channel geometry, substrate granulation) with probabilistic functions of habitat preferences of aquatic organisms (fish, benthic organisms). The first models were developed in the USA in the 1970s and became advanced and accurate tools for predicting river ecosystem reactions at a local scale [6]. Nowadays there are many models available, such as Computer-Aided Simulation Model for Instream Flow (CASiMiR) [5,7,8], Geographic Information System (GIS)-based predictive habitat suitability model [9], Habitat Suitability Index model [10], Physical Habitat Simulation Model (PHABSIM) [11], or Mesohabitat Simulation Model (MesoHABSIM) [12,13] and its modification [14]. To take into account the influence and interaction of physical variables on biological response, these models adopt a variety of statistical computation techniques: fuzzy logic, neural networks, classification tree, GIS, univariate or multivariate logistic regression, random forest [15–17]. Based on GIS classification of habitat suitability models [18–20], we may assign literature and expert models as well as empirical models to deductive and inductive groups, respectively. Both approaches suffer from some constraints. Statistical models (inductive) need a substantial amount of input data to generate reliable results or are restricted to some river types [21]. Literature- and expert-based models (deductive) provide more subjective and general results, which may need to be statistically validated [18,22]. As indicated by Mouton et al. [23], this subjectivity of expert models is a result of a 'knowledge acquisition bottleneck' and thus the results generated should be interpreted cautiously. The decision of which model should be applied is made by the user, with regard to the amount needed (number of parameters, minimum required input) and availability of data. With the appropriate structure of empirical data, a user may perform inductive modelling with many approaches such as those mentioned above. However, when empirical data is scarce and difficult to gather (i.e., not enough time, cost limited, lack of reference conditions, species with low detectability), application of literature- or expert opinion-based models is favored [18,22–24]. This raises the following questions:

(1). Which of the two methods (inductive or deductive) more accurately describes the habitat suitability of aquatic organisms?
(2). Are the results of both approaches even comparable or would they lead to drastically different conclusions and therefore management actions?

In this paper, we address these questions with the help of a model verification study. It is a frequently used approach that provides information about the accuracy and reliability of the results generated [18,25]. The model verification is performed by comparing independent fish observations with model predictions [18,24]. In our case, the approach was the comparison of MesoHABSIM modelling predictions calculated with inductive and deductive habitat suitability models for bullhead (*Cottus gobio* L.) with the results of fish sampling conducted on the Stura di Demonte River in Italy. The answer to the questions formulated above may determine in what context and scale both model approaches should be applied.

2. Materials and Methods

MesoHABSIM is a model describing river ecosystems at a mesoscale (the area used by an organism during the daytime) [26], in contrast to previous models operating at a microscale (the area used by an organism at the moment of observation, which in practice is a few meters) such as PHABSIM [11–13,27–30]. The advantage of coarsening the operating scale of the MesoHABSIM model is a greater reliability of electrofishing as, due to fish mobility, the probability of capture is greater in larger areas. As emphasized by Fausch et al. [31], some of the key factors for fish presence, such as cover structures, habitat complexity, or movement barriers are best identified at mesoscale. Another aspect is that mesoscale allows the investigation of larger sections of the river and observation of the connectivity between habitats. It allows for the identification of hydromorphological variability along an entire river in a manner suitable for assessing the ecological status of rivers or water bodies [4]. The MesoHABSIM method enables observation of the reaction of fish to changes in environmental conditions (anthropogenic or natural) and planning of water management at the river and catchment scales [32]. Mesohabitats are defined by hydromorphological units (HMUs) and their geomorphology and hydrology. HMUs are then evaluated with habitat suitability criteria calculated either with logistic regression models or with conditional habitat suitability criteria (CHSC), which are established with the help of the literature and complemented by expert knowledge. The degree of accordance of these approaches was tested in the present paper using fish sampling data.

2.1. Study Area

The Stura di Demonte River is an alpine river with a length of 112 km and basin area of 1472 km^2 [33], located in north-western Italy (southern Piedmont). The source of the Stura di Demonte River is located in Colle della Maddalena close to the Italian–French border in the Alps, at an elevation of 1996 m above sea level. It is a tributary of the Tanaro River; their confluence is near Cherasco. Stura di Demonte is a confined single thread river. Its main pressure comes from hydropower in the headwaters, but in the analyzed section this can be considered negligible and the river is in high hydromorphological and good ecological status [34].

The survey was conducted at low flow conditions (about 5.3 m^3/s) on 11 and 12 September 2014 at two study sites in the upper section of the Stura di Demonte River, about 23 km from the source. These two sites represent the Stura di Demonte morphology in its mountainous physiographic unit (before it enters the River Po plains). Local morphology is a single-thread, sinuous river with alternate bars. Morphology stays constant in the central reach of the river, where bullhead habitat was surveyed. In the high gradient headwaters of the river (upstream Sambuco), bullhead are not present, which is the same for the downstream plains. The first site investigated, with a length of 400 m, was located close to Vinadio (44.305848 N, 7.187842 E), 15 km downstream from the 13 MW hydropower station close to Pietraporzio. Nevertheless, the river in this stretch is geomorphologically rather unaltered. It is characterized by a sinuous morphology with alternate gravel bars and a sequence of 'fast habitats', i.e., HMUs such as riffles or rapids with boulder rocks, and 'slower habitats' like glides and pools with greater depth and smaller substrate granulation (Figures 1 and 2). The second site was placed 2.5 km downstream, just below the village of Aisone (44.310605 N, 7.220265 E). Site 2, with a length of 110 m, represents a slightly higher degree of morphological alteration; nevertheless, it also has a sequence of slow and fast habitats with numerous structures that may provide cover for fish. Habitat variability at studied sites represented the spectrum for entire central reach of Stura di Demonte River, so the data from both sites were combined for model evaluation.

Figure 1. Three pre-exposed area electrofishing grids (PAEs) at site 1 on the Stura di Demonte River, before electrofishing, September 2014. One PAE is deployed in a ruffle (upper part of the picture), and two in glides (lower part).

Figure 2. Electrofishing in rapids on the Stura di Demonte River, September 2014.

2.2. Habitat Mapping and Electrofishing

Using field computers and ArcGIS software, HMU types such as riffle, glide, complex-high, and backwater (Table 1) were delineated on digital photography. For each habitat unit, a series of hydraulic measurements were taken in at least seven stratified random locations. These measurements comprised substrate distribution, embedment rate, water depth, and mean current velocity in the water column. The measurements of velocity and depth were taken using a dipping bar [35]. Substrate classification was modified for the MesoHABSIM method by References [12,13] from Austrian Norm 6232 [36]. Important information on fish-relevant river characteristics, like the absence, presence, or abundance of cover, such as submerged vegetation, woody debris, boulder rocks, shallow margins, shoreline vegetation cover, was also noted during the field survey and incorporated into the GIS model. For further analyses, riverine habitats were divided into 'fast' and 'slow' categories (Table 1), according to the observed dominating flow characteristic for each HMU type [13,37–39].

The electrofishing survey was conducted with pre-exposed area electrofishing grids (PAEs) [40] and concurrently hydromorphological mapping of the Stura di Demonte River was performed. The electrofishing grid consists of two cables—electrodes that are parallel to each other (6 m long). These cables are attached to two PVC pipes at both ends of each cable to maintain the same distance of 1.5 m between electrodes. The PAEs were plugged into a 1000-W generator and 15-A transformer using a 400-V alternative current. After placing the grids in HMUs, it was necessary to wait at least 10–15 min (Figure 1) before electrofishing was started. This is the minimum period for fish to return to their habitat [40]. Depending on the mesohabitat area in each of identified HMUs, between two and five PAEs were located. The species of all captured fish was determined, and their length measured with an accuracy of 1 mm. After the measurements, fish were released into the river downstream to avoid catching them in the next samples. For each PAE location, environmental variables of HMU type and habitat characteristics were also recorded.

Table 1. Definition of HMUs—hydromorphological units (modified from [13,37,38]). HMUs with fast and slow water velocity are grouped. HMUs marked with bold font were delineated during the survey on the Stura di Demonte River. It is worth noting that some authors [39] treat shallow habitat patches as sub-units, although in this study the standard Mesohabitat Simulation Model (MesoHABSIM) classification was applied.

	HMU	Description of Hydromorphological Unit
Fast	**Riffle**	Shallow stream reaches with moderate water velocity, some surface turbulence, and higher gradient. Convex streambed shape.
	Rapid	Higher gradient reaches with faster water velocity, coarser substrate, and more surface turbulence. Convex streambed shape.
	Cascade	Stepped rapids with small waterfalls and very small pools behind boulders.
	Ruffle	Dewatered rapids in transition to either run or riffle.
	Plungepool	Main flow passes over a complete channel obstruction and drops vertically to scour the streambed.
	Fast run	Uniform fast-flowing stream channels.
	Run	Monotone stream channels with well-determined thalweg. Streambed is longitudinally flat and laterally concave.
Slow	**Pool**	Deep water impounded by a channel blockage or partial channel obstruction. Slow. Concave streambed shape.
	Glide	Moderately shallow stream channels with laminar flow, lacking pronounced turbulence. Flat streambed shape.
	Backwater	Slack areas along channel margins, caused by eddies behind obstructions.
	Sidearm	Channels around islands, smaller than half river width, frequently at different elevation than main channel.
	Complex-high	Shallow areas with water flowing through the stones, frequently at different elevation than main channel (more water than choriotop).
	Complex-low	Dewatered shallow areas with water flowing through the stones, frequently at different elevation than main channel (more choriotop than water).

2.3. Indicator Species

Bullhead (*Cottus gobio* L.) is a small demersal species that does not exceed 15–18 cm in length. It can be found in upland and mountain rivers, and sometimes in lowland rivers with a steeper gradient or even in oligotrophic lakes. For feeding habitats, bullheads prefer running, clear, cold waters abundant with oxygen and coarse substrate, using large stones, boulder rocks and tree root wads as cover [41–45]. This bottom-living species has a weak swimming ability and therefore substrate composition and shelter availability are crucial. Such shelters enable bullhead to avoid excessively high current velocity and to hide during the daytime from predators such as large brown trout *Salmo trutta fario* L. or piscivorous birds like kingfisher *Alcedo atthis* L. and dipper *Cinclus cinclus* L. [44]. Feeding habitats for bullhead, described above, are similar and overlap with spawning and nursery areas. It spawns under the large stones in running waters and young specimens choose slower and shallower parts of the riverbed, so adults and juveniles are often caught in the same HMUs. For wintering, this species chooses deeper units such as pools and plungepools [41–45]. In the present study adult bullhead foraging habitats, occupied by the species during most of the season, were investigated.

Bullhead often coexist in river ecosystems with other species like barbel (*Barbus barbus* L.), European grayling (*Thymallus thymallus* L.), and brown trout. During low flow with reduced wetted area, bullhead is rather a food competitor for brown trout [46]. The main component of the bullhead's diet is macroinvertebrates (e.g., mayflies, caddisflies, flies, chironomids, crustaceans) [43,47,48]. Bullhead is a stenotopic species that is especially vulnerable to oxygen deficit and increased temperature [49,50]. It has also been proven that the presence of well-structured bullhead populations is a good indicator of longitudinal connectivity between river sections [51–55]. These special features have resulted in incorporation of this species in indices used for calculation of ecological status of running waters like EFI+ [56] or in the Polish national method for assessment of ecological status of rivers based on fish (EFI+IBI_PL) [57,58].

2.4. Models

For the purpose of this study, two physical suitability criteria models were created for adult bullhead in alpine streams: a statistical model derived from empirical data, and a CHSC model. The statistical model was developed using an independent database containing information on fish presence and habitat conditions gathered from ten Italian mountain streams (Table 2) [14].

The statistical logistic regression (LR) model described by Vezza et al. [14] was based on data from 95 mesohabitats. A stepwise forward procedure along with Akaike's information criteria [59] was used to determine which parameters should be included in the following regression formula:

$$R = e^{z} \quad (1)$$

where e is the natural log base and $z = b_1x_1 + b_2x_2 + \ldots + b_nx_n + a$, where x_n are significant habitat attributes, b_n are regression coefficients, and a is a constant. A five-fold cross-validation procedure was used to increase model certainty; the procedure was repeated 20 times and, each time, a new set of randomly selected data was set aside for validation purposes. P-values were used to rank the selected variables in the models, while standard errors were evaluated to reduce the number of habitat descriptors and avoid over-fitting [60]. Correlation among selected numerical and categorical variables was tested using a heterogeneous correlation matrix (polycor package, version 0.7–8, [61]) to avoid collinearity effects on model performance. The analysis on the correlation revealed a weak correlation among selected variables (absolute value of Spearman's coefficient ranging from 0.08 to 0.40). To assign suitability classes to each mesohabitat observation, thresholds in probability were derived from the receiver operating characteristic (ROC) curve [13]. Exceeding the probability threshold for the presence or high abundance model deemed the HMU suitable or optimal, respectively. In this way, two statistical models were developed: an absence/presence model (further named 'presence') and a presence/abundance model (further called 'high abundance').

The CHSC model (attributes listed in Table 3) was derived from a literature review [41–45,48,52–54] interpreted by experts from the Stanislaw Sakowicz Inland Fisheries Institute. It includes the suitable ranges for each of the environmental attributes measured. The determination of suitability for each HMU in the survey database was based on comparison of the observed values for the HMU's depth and velocity distribution, choriotop distribution, HMU type, and cover with the suitable ranges identified. With reference to the measurements taken (velocity, depth, and choriotop descriptions), a unit was considered to have acceptable ranges (i.e., suitable) for the target fish species if at least two of the seven measured values collected during the survey fell within suitable limits. In the case of HMU type and cover, a unit was considered acceptable if those attributes were annotated during the data collection. An HMU was presumed suitable when the selected attributes were in acceptable ranges. A suitable habitat needs to have three of five attributes acceptable; for an optimal habitat, at least four attributes should be in this range [13].

Table 2. Logistic regression-based suitability coefficients for bullhead, derived from Italian mountain streams. The coefficients are multipliers of the attribute values. Positive numbers indicate a positive reaction and vice versa. The receiver operating characteristic (ROC) cutoff values for the presence and abundance models are 0.435 and 0.42, respectively. The % values for depth and velocity refers to the percentage of measurements in each HMU with values in a given range.

Presence	Regression Coefficient	Abundance	Regression Coefficient
Constant	−5.9359	Constant	−0.3185
Run (yes/no)	2.3823	Ruffle (yes/no)	−3.0067
Depth 15–30 cm (%)	−1.8063	Depth 15–30 cm (%)	3.2451
Velocity 0–15 cm/s (%)	−2.3177	Velocity 0–15 cm/s (%)	−6.7445
Macrolithal (yes/no)	7.3479		
Mesolithal (yes/no)	9.7954		

Table 3. Physical habitat attributes used in the conditional habitat suitability criteria (CHSC) model, based on expert knowledge and supplemented by literature data for bullhead.

Conditional Habitat Suitability Criteria	
Choriotop:	
Microlithal	Present
Mesolithal	Present
Macrolithal	Present
Velocity range (cm/s)	(30–105)
Depth range (cm)	(25–75)
Cover:	
Undercut banks	Present
Boulders	Present
Woody debris	Present
HMU:	
Rapids	Present
Riffle	Present
Ruffle	Present

To determine the relation between the LR model results and actual bullhead observation in grids, calibration plots were created (Figures 3 and 4). On the X axis, classes of the predicted probability of fish occurrence are highlighted, while on the Y axis, bullhead occurrence as a proportion of the grids surveyed is shown. Calibration plots are scatter plots of histograms analyzed with linear regression [62]. To prepare the bullhead observation data (Y axis), the PAEs were split into three groups (empty grids, grids with ≤two individuals, grids with ≥three individuals). The threshold for bullhead 'presence' in a grid was set at 0.11–0.22 ind./m^2 (which refers to one or two individuals per grid), while for 'high abundance', the value was set at ≥0.3 ind./m^2 (which refers to three or more individuals per grid). Separate plots were created for the presence and high abundance models, using the proportion of grids with fish to all grids, and the proportion of grids with ≥ three individuals to grids with fish, respectively.

To create histograms for the model predicted probabilities, sensitivity analysis was conducted to select the most appropriate class width in iterative manner. Accordingly, several combinations of various class width were tested. At first, the class width was set to 0.1 for all intervals; subsequently, a width of 0.2 or 0.3 was chosen for the first class, and 0.15 for all other intervals. Ultimately, the best calibration was achieved for presence and high abundance models with 0.15 and 0.3, respectively, as the width of the first class, and 0.15 for other class intervals, producing five bins of data [62].

Calibration plots cannot be produced for the CHSC model due to its data structure (HMUs are assigned non-numerical categories: not suitable, suitable, and optimal, without calculating the probability value). Hence, the validation was performed with the help of stacked bar diagrams (Figures 5 and 6). To assess the similarity of the inductive and deductive models, Spearman rank correlation was applied.

3. Results

During two days of survey on the Stura di Demonte River, in 80 pre-exposed grids, 142 fish representing three species were captured, including: 116 bullhead—*Cottus gobio*, 24 brown trout—*Salmo trutta fario*, and two souffia—*Leuciscus souffia* Risso (see Table A1 in Apppendix). Fish were caught in 18 selected mesohabitats, including: rapid (four), riffle (four), ruffle (three), sidearm (two), glide (two), plungepool (one), pool (one), and complex-high (one). No fish were found in the six HMUs represented by: pool (two), complex-high (two), sidearm (one), or backwater (one), the HMUs ascribed to the group of slow habitats (Table 1). Those HMUs were characterized by a lack of fish refuges, and lower depths and velocities (in comparison to those where fish were present).

In reference to bullhead, this species was present in 44 grids, located in 16 mesohabitats; 72% of those individuals were found in fast habitats represented by rapid, riffle, ruffle, and plungepool types, and only 28% were captured in slow habitats such as glide, pool, sidearm, backwater, and complex-high types. The ranges of depth and velocity that were noted during survey in rapids varied from 10–50 cm and 0–122 cm/s, while in glides they were 5–52 cm and 0–46 cm/s, respectively. It should be stressed that in the group of 'fast habitats', boulder rocks were abundant in more than 90% of cases and present in 100%. In the group of 'slow habitats', these numbers were much lower—abundant in 20% and present in 60%. A similar tendency was observed for shallow margins—in a group of 'fast habitats', those features were ascribed as abundant in 45% of cases and present in 100%, while in 'slow habitats' these numbers were lower: 40% and 60%, respectively. The highest number of bullhead individuals—nine per grid (1 ind./m^2) was observed in a rapid mesohabitat where, in addition to a high number of large stones, woody debris and shallow areas were also present.

The results of regression analysis between the statistical models (absence/presence and presence/abundance) and fish occurrence in grids are presented in Figures 3 and 4. The initial class of predicted fish occurrence was set at 0.3 with an interval value of 0.15, resulting in five bins of data. All of the combinations tested showed a high level of agreement between the predictions of those LR models and bullhead observations in pre-exposed grids. The presented results trend line for the presence and abundance models was statistically significant, with $R^2 = 0.90$ and $R^2 = 0.88$, respectively.

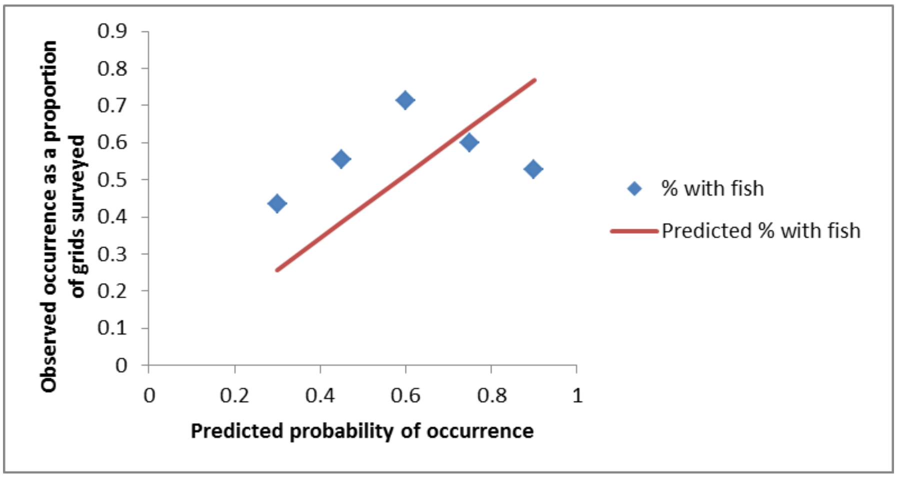

Figure 3. Calibration plot of logistic regression model of fish presence for bullhead (first class 0.3 with 0.15 intervals); y = 0.8556x, $R^2 = 0.90$, $p < 0.01$.

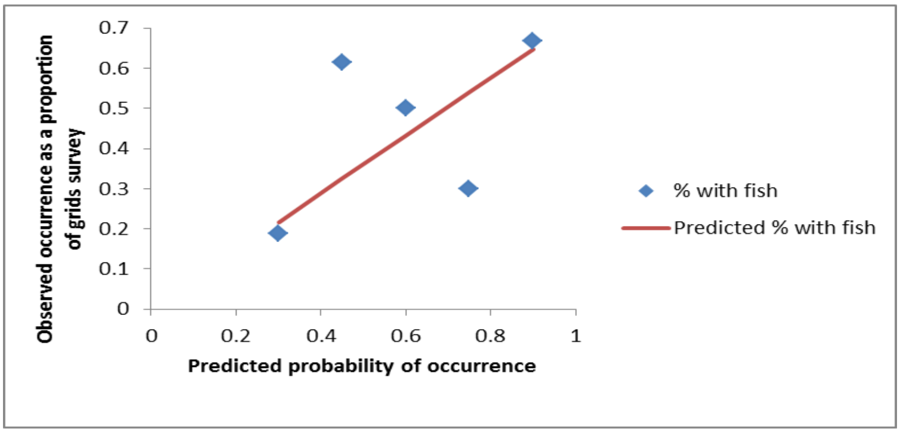

Figure 4. Calibration plot of logistic regression model of fish abundance for bullhead (first class 0.3 with 0.15 intervals); y = 0.7201x, $R^2 = 0.88$, $p < 0.05$.

Collation of the predictions of the statistical and CHSC models against bullhead presence in grids is shown in Figures 5 and 6.

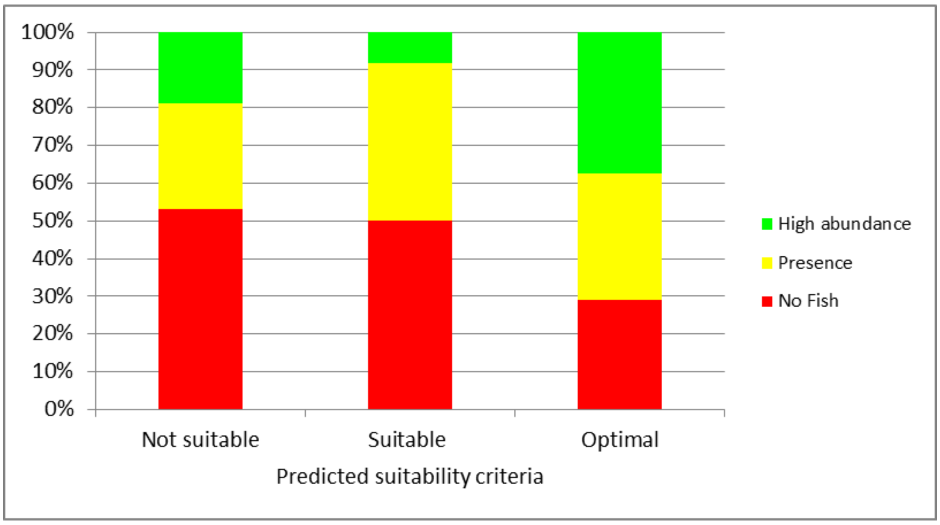

Figure 5. Comparison between results of the statistical model and results of bullhead sampling in the Stura di Demonte. The x-axis shows predicted suitability criteria in three different classes: not suitable, suitable, and optimal. Fish presence class is shown by colors.

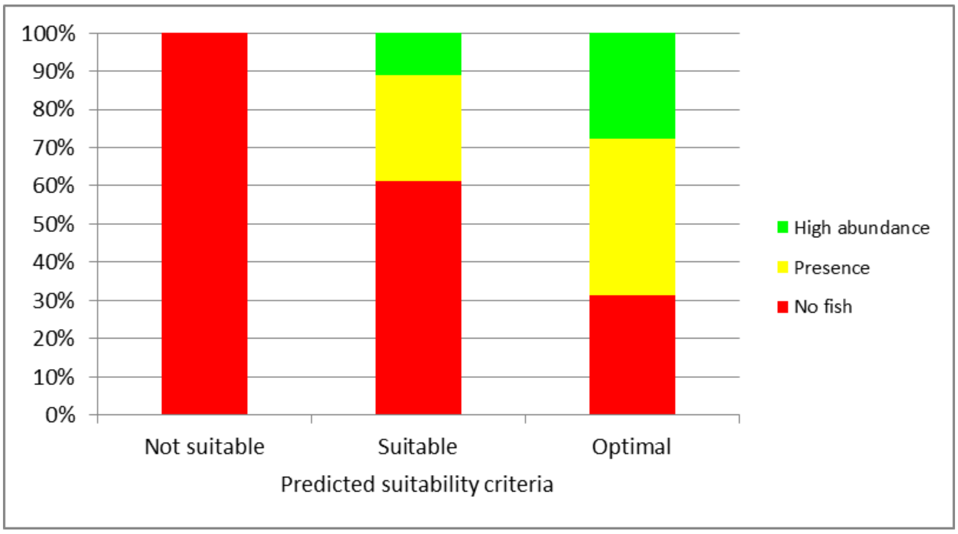

Figure 6. Comparison between results of the conditional habitat suitability criteria (CHSC) model and results of bullhead sampling in the Stura di Demonte. The x-axis shows predicted suitability criteria in three different classes: not suitable, suitable and optimal. Fish presence class is shown by colors.

The similarity level of the standard statistical and CHSC models expressed by the Spearman coefficient was low ($r_s = 0.26$) and not significant ($p = 0.22$).

4. Discussion

Application of a specific modelling method (inductive or deductive) is often connected with the type and stage of a project (planning documents or construction works) but practice shows that the decisive factors are: time, costs, and the possibility of collecting sufficient data. To meet these demands, habitat suitability models (i.e., CASiMiR, MesoHABSIM) offer special modules adapted for both theoretical and empirical data [8,13]. Such models are important approaches that enable investigation of the present state of a river's hydromorphology and its ecological potential.

The calibration plots present the goodness of fit between two sets of variables [62]: Bullhead observation in grids, and model predictions of fish occurrence/abundance. The best reliability, with significant correlation ($R^2 = 0.90$), was recorded for the 'presence' model (Figure 3). This confirms good selection of regression coefficients for bullhead, which were derived from different (but still similar) streams located in the Piedmont region in Italy [14]. The data complements the bullhead database for Italian mountain rivers and can be used for modelling of different types of human activity (including hydropower or water abstraction) for assessment of the impact on this indicator species in mountain rivers in the EIA procedure.

According to the abundance threshold for bullhead used in the models, it varies from 0.3 ind./m^2 in CHSC to 0.05 ind./m^2 in the statistical model. These values differ due to the different techniques used for taking the electrofishing samples. The method of electrofishing using pre-exposed grids applied in this study is more accurate than the classical wading against the current technique (used in ten Italian mountain streams), especially in areas with a depth not exceeding 1 m [13]. Other authors offer a wide range of bullhead abundance in alpine streams, varying from 0.002 to 0.41 ind./m^2 [52].

A specific combination of observed physical variables is a key factor for habitat quality [63]. The higher number of bullhead individuals (72%) present in 'fast habitats' (i.e., riffle, rapid, ruffle, plungepool) refers not only to the more suitable conditions of depth and velocity reflecting also better oxygen saturation but is connected to the greater abundance of shelters such as boulder rocks, woody debris, or canopy cover. The habitat factor determining bullhead occurrence is cover availability. In Alpine rivers, most common covers are boulder rocks. Such covers were present in 88% and abundant in 69% of HMUs where bullhead was observed. An abundance of shallow areas provides greater diversity of habitats. They were present or abundant in all the 'fast habitats', while they were only present in more than half of the 'slow' ones. Habitat variability provided by the undisturbed river stretch of Stura di Demonte meets the requirements of the bullhead population. These results should be taken into account while planning river management practices. For example, installation of traps for large sediment may not only disrupt river continuity, but also reduce availability of covers, strongly affecting fish populations. The best solution for assessing potential effects of management actions planned is the use of the modelling approach.

Bullhead was assigned by Welcomme et al. [55] into the riffle guild, and our observations confirm that bullhead prefer to exist in those habitats (28% of individuals). Other authors [44,64–66] also note that the most suitable habitat units for bullhead are riffles. With regard to the group of slow HMUs, the highest number of bullhead individuals was found in glides (20%), which Gosselin et al. [67] specify as the most occupied habitat by bullhead in lowland rivers in the United Kingdom. This could be simply a consequence of fast HMU availability. Vezza et al. [14] did not find bullhead in glides. The greatest abundance was observed by him in runs. In the sections of the Stura di Demonte analyzed, there were no run HMUs delineated during the survey; nevertheless, Vezza et al. [14] also described high abundance in rapids, riffles, and ruffles, which corresponds to our observations. Due to the fact that the statistical model based on the Italian database (Table 2) considered runs as a preferable habitat for bullhead, the absence of this HMU type in this study may weaken the nonetheless high correlation of the fish presence model.

The statistical and CHSC models have different cutoff values separating suitable and optimal habitats (Tables 2 and 3). In consequence of the statistical dissimilarity between the CHSC and statistical model verification results (Figures 5 and 6), it was decided to reduce the cutoff value of the statistical model. The original cutoff value used by the statistical model as obtained from ROC analysis was 0.435 for the 'presence' model and 0.420 for the 'high abundance' model (Table 2). Experimentally, the value of cutoff coefficients in both models was reduced to 0.3 and put against the fish sampled in grids to verify if the results will be closer to the CHSC model. The result of comparison with this less rigorous model is shown in Figure 7. A new calculation of similarity expressed by Spearman coefficient between the statistical model with reduced cutoff and the CHSC model is significant ($p < 0.05$) and correlating at $r_s = 0.42$. Still, this value is relatively low, indicating that the modified statistical and CHSC models are different.

For additional comparison between the results for the two techniques, another analysis was performed, where the predictions of each model were put against the fish data. For the presence/absence case based on confusion matrix analysis, sensitivity and specificity were calculated [23,68] for every model (Table 4). Correct classification means here that fish are present in suitable and optimal habitats, and absent in not suitable habitats. The highest correct classification (65%) for fish presence/absence was achieved by the CHSC model (Table 4, Figure 6). This may indicate that since this model is based on more generic data (wider criteria of physical attributes for bullhead i.e., higher velocities, greater depths, more cover types than specified in the literature were accepted basing on experts' knowledge), it can more easily match observations. The sensitivity parameter for the CHSC model was 1.0 while the specificity was 0.22, indicating a more generic approach. Hence, the CHSC model should be applied in case of a lack of available data from fish sampling [18,22], and there is a need for information on habitat suitability for target riverine species. The statistical models, standard statistical model and statistical model with modified cutoff value, showed a higher misclassification rate (42% and 41%, respectively) with lower sensitivity (0.66 and 0.77, respectively) (Table 4, Figures 3 and 5) than the CHSC model: misclassification rate 35%, sensitivity 1.0 (Table 4, Figure 6).

On the other hand, the correct classification by statistical models in exact values (agree to the class: no fish—not suitable, presence—suitable, high abundance—optimal) was higher than in the CHSC model. Better correlation of CHSC and statistical models with the cutoff modified to 0.3, shown by Spearman correlation rank, is reflected by the greater sensitivity of the latter (Table 4). Overall, the CHSC model indicates more unoccupied locations as suitable habitats and, consequently, no unsuitable habitats are occupied, i.e., it may be overestimating suitable habitats. On the other hand, high occupation of all suitable habitats can be expected only when all niches are saturated with individuals. This is rarely the case, and therefore it may actually be that statistical models, although more precise but still affected by the same phenomenon, are underestimating suitable habitats. Hence, such a conservative approach is most appropriate when investigating habitats for endangered species but may not be necessary for community analysis.

Table 4. Classification of the results of the models used and fish observations in grids for presence/absence and exact values. In the case of presence/absence confusion, matrix parameters—sensitivity and specificity—are shown; sensitivity shows the probability that the model will correctly classify a presence, specificity shows the probability that the model will correctly classify an absence [23,68].

	Statistical Model		Statistical Model with Cutoff 0.3		CHSC Model	
	% of grids					
	Misclassification	Correct	Misclassification	Correct	Misclassification	Correct
Presence/absence	42	58	41	59	35	65
Sensitivity	0.66		0.77		1	
Specificity	0.47		0.36		0.22	
Exact values (agree to the class)	55	45	56	44	65	35

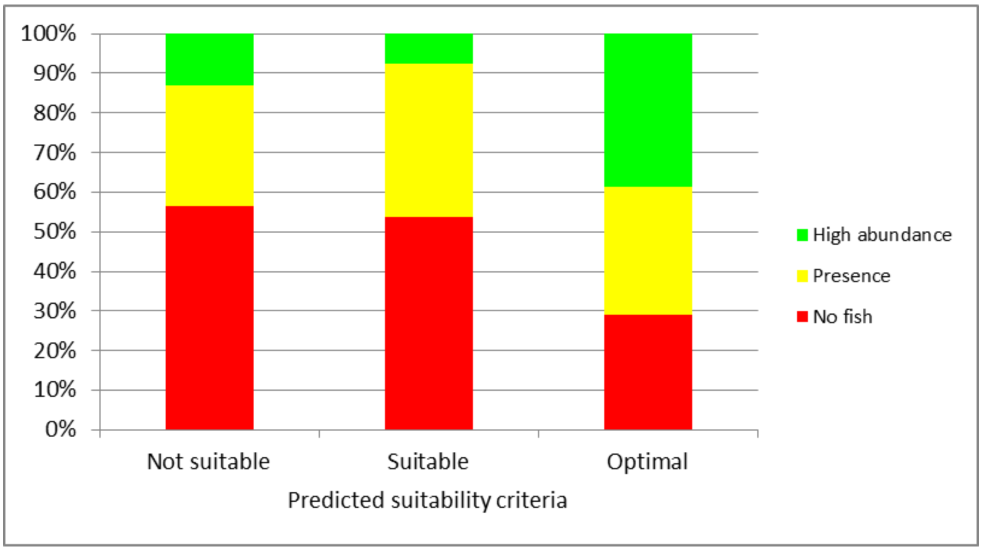

Figure 7. Comparison between results of the statistical model with a lower cutoff value and results of bullhead sampling in the Stura di Demonte. The x-axis shows predicted suitability criteria in three different classes: not suitable, suitable, and optimal. Fish presence is shown by colors.

The validation study presented in this paper shows good accuracy of the tested models, and they can be applied for assessing habitat value for bullhead in mountain rivers and streams. However, we would not recommend using them interchangeably or comparing two data sets analyzed with different models, unless the fact of potential habitat underestimation by statistical approach is taken into account. The choice of appropriate method should be determined by the purpose for which the user wants to use the results and by the availability of data. The CHSC model might be implemented in studies of maintenance and regulation works in rivers for which there is no concern of endangered species habitats. Application of the CHSC model for target species typical for a specific type of river determines the main factors that might reduce fish abundance. This model will be especially helpful where some of the limiting factors causing fish absence (such as water quality or water temperature [69,70]) exist or there are no possibilities for sufficient data collection [18,23,24]. One option for improving the performance of the CHSC model is to use field data for more detailed calibration and verification. It is still much less effort and less cost intensive than collecting hundreds of samples for statistical model development.

For investigating measures that could potentially impair sensitive, unabundant members of the ecosystem, it may be advisable to do modelling based on an inductive approach with habitat-specific electrofishing samples. This modelling method provides the most rigorous results, reflecting site- and species-specific factors that may be overlooked in literature-based models [24].

Although the conclusions above are quite generic, it needs to be remembered that they are based on investigation of one species only. The difficulty is that gathering appropriate data for developing statistical models can be cost- and effort-intensive. Luckily, efforts are underway to establish databases that can be used for such purposes and help to organize model choices more systematically. There are databases created for a specific region, catchment or river type, i.e., bullhead in Piedmont in Italy [14] or the Rushing Rivers Institute database for north-eastern USA rivers [24]. They contain information from multiple rivers, which is very valuable as the more data gathered the better, and the more precise the model. Local water management authorities or environmental agencies equipped with appropriate tools (like habitat suitability models) may use existing data sets for planning water management. Nevertheless, verification studies such as the one presented here are necessary in order to assure data and model compatibility as well as appropriate interpretation of the results.

5. Conclusions

The Stura di Demonte River offers optimal and suitable physical habitats for bullhead. Fish were distributed most numerously in those lotic habitats that provided appropriate shelters. The verification performed in this study documented that all models tested well represent habitat use by bullhead, leading to the points listed below:

(1). Inductive and deductive models do not offer statistically identical results, and the choice of model can have some influence on the results.
(2). Lowering the cutoff value in a standard statistical model results in greater similarity to the CHSC model (Spearman rank correlation).
(3). The statistical model underestimates suitable habitats and is more appropriate for endangered species studies according to precautionary principle.
(4). Application of properly constructed MesoHABSIM literature-based models complemented by expert opinion like the CHSC model provides more generic information about habitat suitability.
(5). Nevertheless, the CHSC model validated very well and beyond expectations. It could easily be better calibrated with a relatively small sample of field data.

For further development of predictive models existing habitat preference databases can and should be used in similar verification studies. Those actions will allow systematization of the information available and standardization of the applications. Complementing water management planning by application of modelling studies is recommended as a standard of good practice, which is essential for conservation of endangered species, or other management goals such as minimizing expansion of some invasive species.

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Appendix

Table A1. List of fished HMU with grid number and number of fish caught: brown trout (*Salmo trutta fario*), bullhead (*Cottus gobio*), and fallfish (*Leuciscus souffia*). Data from both sites were combined together.

HMU No	HMU Type	GRID No	SPECIES			Sum
			Brown trout	Bullhead	Fallfish	
11001	RAPID	1		9		9
		2		5		5
		3		3		3
		4		1		1
11002	RIFFLE	1	1	4		5
		2				0
		3		4		4
		4				0
		5		1		1
11003	SIDEARM	1				0
		2				0
		3	1			1
		4	1			1
		5				0
11004	RIFFLE	1	1	3		4
		2		2		2
		3				0
		4		2		2
		5	1	2		3
11005	RIFFLE	1	1	4		5
		2				0
		3	2	4		6
11006	COMPLEXHIGH	1				0
		2				0
11007	GLIDE	1		5		5
		2		5		5
		3		1		1
		4		2		2
		5		1		1
11008	PLUNGEPOOL	1	1	4		5
		2		4	2	6
		3	2	3		5
		4	3	2		5
11009	RAPID	1		2		2
		2	2	1		3
11010	RIFFLE	1		5		5
		2	1			1
		3		1		1
11011	GLIDE	1	1	1		2
		2		5		5
		3		2		2

		4		1		1
11012	POOL	1		1		1
		2		2		2
		3				0
		4		3		3
		5				0
11013	RUFFLE	1		2		2
		2				0
		3	1			1
11014	RAPID	1		1		1
		2		5		5
21001	BACKWATER	1				0
		2				0
21002	SIDEARM	1				0
		2				0
21003	POOL	1				0
		2				0
21004	RUFFLE	1		1		1
		2		2		2
		3		2		2
		4				0
		5		1		1
21005	RAPID	1				0
		2				0
		3		2		2
		4		2		2
		5				0
21006	COMPLEXHIGH	1		2		2
		2				0
		3	1			1
21007	SIDEARM	1	1	1		2
		2				0
21008	RUFFLE	1	1			1
		2	2			2
		3	1			1
21009	COMPLEXHIGH	1				0
		2				0
21010	POOL	1				0
		2				0
Sum	**24**	**80**	**25**	**116**	**2**	**143**

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