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Risk assessment of the harvested pike-perch population of the Azov Sea

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Abstract

We consider the fishery of pike-perch (*Lucioperca lucioperca* L.) in the Azov Sea, Russia. There are indications that this population might be threatened by the anthropogenic deterioration of environmental conditions. Since the species has an economic importance, extinction risk minimization is complemented by the consideration of optimal harvesting. The problem is therefore multicriteria. A stochastic simulation model is developed to assess simultaneously the annual yield and the extinction risk of the exploited population resulting from different harvesting strategies (i.e. combinations of harvesting efforts on yearlings and on adults). Another problem being considered is the assessment of the economic and ecological consequences of unintentional yearling catch. The model suggests that the current fishing rates induce a high risk of population abundance drop. Avoidance of yearling harvesting always improves both ecological and economical criteria. However, the extinction risk remains high even in the absence of fishing, suggesting that additional conservation measures are necessary to improve the environmental conditions. A simulation with slightly diminished salinity (by 0.5‰) drastically improves both the population viability and the annual yield. © 2002 Elsevier Science B.V. All rights reserved.

Keywords: Population; Harvesting; Stochasticity; Quasi-extinction; Risk; Simulation model; Salinity

1. Introduction

In this study, we use stochastic simulations to combine the comparative analysis of different harvesting strategies (Tyutyunov et al., 1993) with extinction risk assessment, applying it to the pike-perch population in the Azov Sea, Russia. Thus, we study population viability under the double threat of harvesting and stochastic environmental impacts. The consideration of stochasticity is im-

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portant, as it makes the optimal harvesting strategy more cautious and it permits the evaluation of the risk of population extinction.

We will use the notion of ‘extinction risk’ more specifically as the ‘quasi-extinction risk’ as defined by Ginzburg et al. (1982), i.e. the probability that population abundance drops under a given pre-assigned level. This approach has been applied in several case studies, using software specifically developed for numerical risk assessment (Akçakaya, 1991; Burgman et al., 1993).

Traditionally, optimal harvesting problems imply a single economic criterion (e.g. Beverton and Holt, 1957; Clark, 1976; Horwood and Whittle, 1986; Horwood, 1990; Getz and Haight, 1989; Arditi and Dacorogna, 1992) while conservation concerns have brought population ecology to probability models that can be applied to extinction risk assessment, e.g. Bartlett (1960), Soulé (1987), Burgman et al. (1993), Vucetich and Waite (1998) and Stephan and Wissel (1999). Since in most realistic cases harvesting will decrease population abundance and, therefore, increase the extinction risk, these two criteria—the catch and the extinction risk—cannot be combined into one. Thus, a trade-off problem arises. Pareto optimality (Vincent and Grantham, 1981) can be applied as a basis for multicriteria optimization (see Vincent, 1987; Taylor et al., 1975; Walters, 1986; Getz and Haight, 1989; Reynolds and Ford, 1999 for examples and further references).

As an alternative to multicriteria optimization (e.g. Cass and Riddell, 1999), a constrained problem can be considered: maximization of the harvest under a constraint about the probability of a catastrophic collapse of the exploited population (see, e.g. Reed, 1988). Such stochastic dynamic programming, where the control depends on the harvesting threshold, has recently been applied to the problem of optimal harvesting of a population with a risk of extinction (Lande et al., 1995; Ganguly and Chaudhuri, 1995). Similar applications to multiple species harvesting were developed by Horwood and Whittle (1986), Horwood (1990) and Pradhan and Chaudhuri (1999). However, this technique requires the precise knowledge of the population size every year and the perfect control over the fishing effort, which must be

adjusted dynamically every year. Both of these requirements are impossible to fulfill in the case of the Azov Sea, where monitoring of fishing is far from perfection. This is why a more realistic technique consists of optimizing a constant, the long term harvesting policy, which makes it impossible to combine the two above-mentioned independent criteria into a single utility function. Thus, in the case of pike-perch, the problem is multicriteria and stationary (in the sense that the harvesting effort is fixed to a constant and remains unchanged through the whole simulation horizon). Note that this does not mean stationarity of environmental factors.

We have applied this approach earlier to the case of two competing species (anchovy and sprat) being harvested simultaneously in the Azov Sea (Senina et al., 1999), and Ludwig (1998) performed a theoretical study of a similar system. The present study of a single population leads to a relatively simpler model with more robust parameter estimates and more robust predictions. It will also clearly illustrate our approach, which we believe can be useful for the risk assessment tasks needed for practical management of various ecological systems vulnerable to stochastic environmental impacts.

In this paper we will first describe the biology of the studied system, then develop a mathematical model of the population dynamics and harvesting. A stability analysis will reveal some relations between population viability and fishing effort or environmental variables. Parameters will be estimated and we will introduce a stochastic simulation scenario to evaluate the quasi-extinction risk and catch for all possible harvesting strategies in the model.

2. The biological system

The Azov Sea (46°N by 37°E) is a small (39 000 km², 360 km in longitude by 180 km in latitude), shallow (7 m average and 14 m maximal depth), brackish, and highly productive water body located to the east of the Crimea peninsula on the north of the salty Black Sea. It communicates with the latter by the Kerch straits (16 km wide)

which is the only outlet. The water exchange between the two seas and the resulting salinity regime depend strongly on the inflow of freshwater into the Azov Sea (Goptarev et al., 1991; Matishov et al., 2000). This freshwater is brought mainly by the rivers Don and Kuban.

Pike-perch (*Lucioperca lucioperca* L.) is a valued freshwater fish. Like many other such species, the pike-perch population started to decline after the regulation of rivers flowing into the Azov Sea in the 1950s and the increasing water salinity resulting from the decrease of the average annual freshwater inflow from 40.6 (which was equal to 1/8 of the sea volume) to 33.2 km³. Azov pike-perch is a semi-migratory fish: adults spawn (beginning of May) in the river and then drift (until the end of August) to the cooler parts of the sea (Kovtun et al., 1998). Depending on temperature conditions, the development of eggs takes from 3 to 10 days. At the age of 1.5 months, fry drifts to the sea (Berg et al., 1949). Before regulation of river inflow the population of pike-perch inhabited almost the whole area of the Azov Sea and was able to migrate very far into the upper Don. Presently, due to the fact that salinity higher than

11‰ is unfavourable for this species (Goptarev et al., 1991), it lives in the eastern part of the Azov Sea during the feeding period and in the eastern part of the Taganrog Bay and the delta of the Don during the reproduction period. The population does not have a pronounced vertical distribution due to the shallowness of these regions (depth does not exceed 10–12 m) and the absence of water stratification. Pike-perch lives up to 14 years, achieving maturity in its second year, while yearlings can already reproduce (Nicol'sky, 1974). The maximal individual size is 75 cm, corresponding to a weight of 5.5 kg although there are records about individuals of greater length (130 cm) and weight (20 kg) (Berg et al., 1949). Fecundity grows monotonically but not linearly with size (age) to a limit of 700 000 larvae per female older than 9 years.

Juveniles eat mainly zooplankton and, partially, insect larvae, while adults become active predators that eat small fishes (including their own juveniles, see Fortunatova and Popova, 1973). The growth of the pike-perch population is not limited by food in the very productive Azov Sea. Since pike-perch is a top predator, it is not controlled from above by any other species; harvesting is the only removal factor. This harvesting takes place over the whole year except when water is frozen from November to March and during spawning time. It intensifies after spawning when adults are vulnerable to harvesting due to aggregation in the freshwater regions (Goptarev et al., 1991). Regulation of river inflow and extensive poaching made annual yield decline in the 1980s to 500–1500 tonnes per year presently, while in the 1960s it reached 10 000 tonnes per year (Fig. 1A). Thus, there is a pressing need to obtain scientific knowledge for the rational harvesting of this species.

A special problem that needs to be addressed is the accidental catch of yearlings (i.e. 1-year-old individuals). They are caught together with the adults because of imperfect gear selectivity and are thought to represent a cause of the decline of the population. For this reason, yearlings will be considered separately from the adults in our modelling approach.

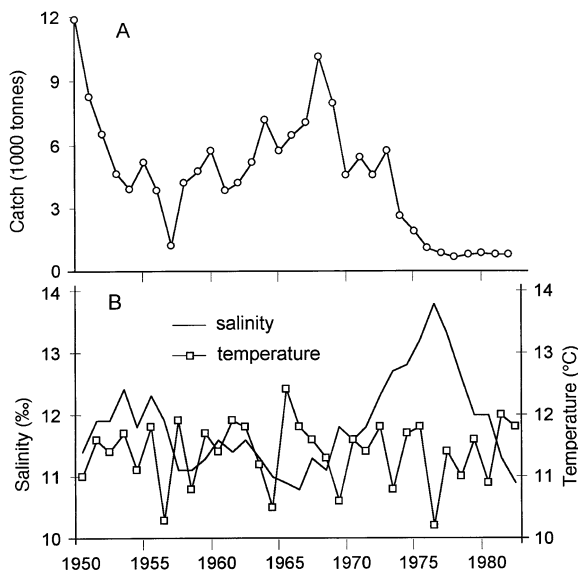


Fig. 1. (A) Annual catches of pike-perch in the Azov Sea. (B) Time series of average sea salinity (continuous curve) and average temperature (curve with squares).

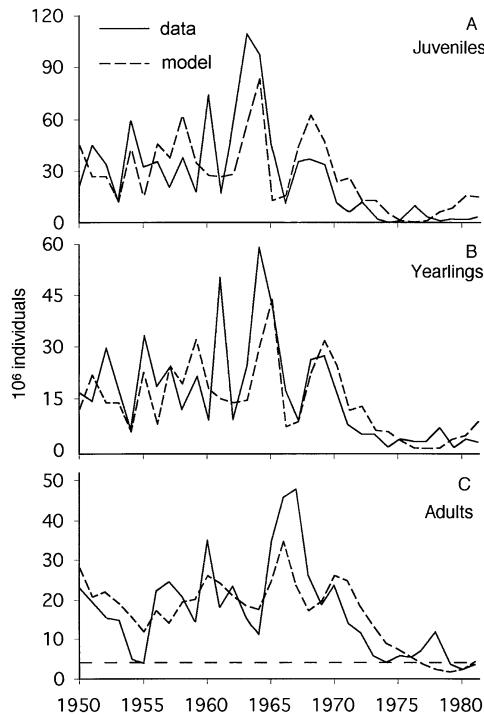


Fig. 2. Fit of the model trajectories to observations. Broken curves are the observed time series and continuous curves are the values predicted by the model. On graph C, the dashed line indicates the quasi-extinction threshold.

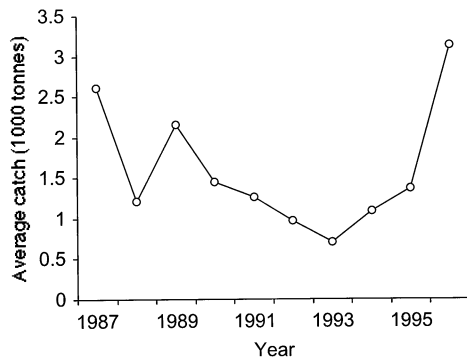


Fig. 3. Annual catches (10^3 tonnes) of pike-perch in the Azov Sea for the period 1987–1996 (after Zaidiner et al., 1998).

The available data consist of several time series running from 1950 to 1982 (i.e. 33 yearly observations) provided by the Azov Fisheries Research Institute (Rostov-on-Don). The series include values of average annual salinity of the Azov Sea,

average annual water temperature (Fig. 1B), annual abundance of juveniles, yearlings and adults (Fig. 2A–C) and yearly catches. The abundances and age structures were estimated by ichthyologists of the Azov Fisheries Research Institute according to the method of virtual populations using time series of commercial catches and data of regular scientific surveys. The errors of these estimates are not known. Temperature was measured near Berdyansk, a locality typical of the area where pike-perch live during most of the year, except for the spring spawning season when they migrate into the Don and Kuban rivers. The detailed data after 1982 are not available. There are only series of annual catches from 1987 to 1996 (Fig. 3), and intervals of river inflow and sea salinity. Unfortunately, this information cannot be used for independent validation of model results but, in general, it helps confirm the main result about the crucial influence of salinity (see Section 7).

3. The model

The first modelling task is to construct a model that, while being as parsimonious as possible (having the lowest possible number of parameters), can still be fitted to the data and reasonably reflects the observed values of the modelled variables. Then, the identified model can be used with MonteCarlo simulations to assess the trade-off between ecological and economical criteria for different harvesting strategies.

Three age groups are considered: juveniles, yearlings, and adults, N_0 , N_1 , and N_2 , respectively. The last two participate in reproduction. Juvenile recruitment is described by a modified Ricker stock–recruitment function (Ricker, 1954):

$$N_{0,t+1} = (a_1 N_{1,t} + a_2 N_{2,t}) e^{-b(N_{1,t} + N_{2,t})},$$

where a_1 and a_2 are the fecundity coefficients of yearlings and adults, respectively, and b quantifies intraspecific competition including cannibalism. Yearlings result from surviving juveniles and adults result from surviving yearlings and adults. The age-transition equations are therefore:

$$\tilde{N}_{1,t+1} = p_0 N_{0,t}, \quad \tilde{N}_{2,t+1} = p_1 N_{1,t} + p_2 N_{2,t},$$

where $0 < p_i < 1$ are the natural survival rates. The variables \tilde{N}_1 and \tilde{N}_2 denote the potential abundances in the absence of harvesting.

Yearlings and adults are subject to exploitation, although this is unintentional in the case of yearlings. Harvesting occurs after survival since natural mortality is more intensive during the winter because of low temperatures (the temperature of bottom layers can decrease to -1.0 °C) and intensive harvesting starts after the spawning period. Thus, we have:

$$N_{1,t+1} = \tilde{N}_{1,t+1}(1 - h_{1,t}),$$

$$N_{2,t+1} = \tilde{N}_{2,t+1}(1 - h_{2,t}),$$

where $0 \leq h_{i,t} \leq 1$ are the harvesting efforts.

Two environmental factors are accounted for: average annual salinity S_t and water temperature T_t . A statistical analysis has shown that these two factors are not correlated. Neither was there a significant correlation between abundance or yearly catch and temperature. There is a small but significant negative correlation between yearly catch and salinity. However, from our data alone we do not know the form of the relation between these environmental factors and yearly catch. Kozlitina et al. (1998) have studied the dependence between catch and salinity in long time series and found that this relation must be described by means of a non-linear model. Environmental stresses are known to affect first of all the most vulnerable processes of pike-perch population dynamics: reproduction and larval survival (Goptarev et al., 1991; Kozlitina et al., 1998; Kovtun et al., 1998). Arising from niche theory, a natural way to describe the influence of environmental factors on the reproduction rate is in the form of Gaussian functions, acting as multipliers on the recruitment term. This approach was successfully used earlier by Vorovich et al. (1981, 1989) and Senina et al. (1999). For S_t and T_t , these reducing factors are

$$G_{S,t} = e^{-((S_t - S_0)/\sigma_S)^2} \quad \text{and} \quad G_{T,t} = e^{-((T_t - T_0)/\sigma_T)^2} \quad (1)$$

where S_0 and T_0 are the optimal values and σ_S and σ_T characterize the tolerances.

Note that, as mentioned above, the temperature data used to calculate these factors on fecundity do not come from the areas where pike-perch spawns. However, since the model works with a yearly time step, it does not explicitly include the spatial details of spawning in the Don and Kuban rivers. Anyway, the annual average temperatures in different areas of the Azov Sea and in the inflowing rivers are strongly correlated (Goptarev et al., 1991).

Deviations of temperature from its optimal value affect above all reproduction and juvenile survival (Goptarev et al., 1991). Low temperature causes high energy losses at the expense of investment into fecundity; high temperature provokes oxygen starvation of most fish species in the shallow Azov Sea.

Salinity has constantly remained too high since the construction of the Tzimlyansk Reservoir in the 1950s that caused reduction of freshwater inflow and reduced the whole biological productivity of the brackish Azov Sea. High salinity is comparatively more detrimental to such freshwater predator species as pike-perch than to the sea fishes that inhabit the Azov Sea. Besides, there is an indirect influence of salinity, especially on the larval stage, due to the impact on lower trophic levels.

Using the symmetrical Gaussian function (1) for the salinity factor, we must emphasize that by ‘optimal’ salinity, ichthyologists mention the lowest observed average annual value as ‘the best’ for this freshwater species. This means that only the right side of the Gaussian curve has a real meaning and the model should not be used for salinity lower than the ‘optimal’ value. However, for the sake of uniformity, we write both reducing factors with the general form (1). Note that the average annual salinity (Fig. 1) is higher than the local salinity in the population habitat.

The complete model is:

$$N_{0,t+1} = (a_1 N_{1,t} + a_2 N_{2,t}) e^{-b(N_{1,t} + N_{2,t})} e^{-((T_t - T_0)/\sigma_T)^2} e^{-((S_t - S_0)/\sigma_S)^2}, \quad (2a)$$

$$N_{1,t+1} = p_0 N_{0,t} (1 - h_{1,t}), \quad (2b)$$

$$N_{2,t+1} = (p_1 N_{1,t} + p_2 N_{2,t}) (1 - h_{2,t}), \quad (2c)$$

The total catch during year t is

$$C_t = w_1 h_{1,t} \tilde{N}_{1,t} + w_2 h_{2,t} \tilde{N}_{2,t}, \quad (3)$$

with w_1 and w_2 being the specific average individual weights of yearlings and adults.

4. Parameter estimation and deterministic simulations

The value of adult survival p_2 and the optimal values of the environmental factors S_0 and T_0 (those providing the best conditions for recruitment) were taken from ichthyological sources (Bronfman et al., 1979; Goptarev et al., 1991). Annual harvesting rates $h_{1,t}$ and $h_{2,t}$ were calculated from the available catch data.

For the other parameters, we used a two-step estimation procedure. In the first step, parameters a_1 , a_2 , b , σ_S , σ_T were estimated by non-linear least-squares regression of the recruitment Eq. (2a) to the observed juvenile abundances N_0 , using the observed values of N_1 , N_2 , S , and T as inputs. These parameter estimates were then used as starting values in the next step. In this second step, the parameters a_1 , a_2 , b , σ_S , σ_T , p_0 , p_1 were

estimated by fitting (least-squares) simultaneously the three model trajectories $N_{0,t}$, $N_{1,t}$, $N_{2,t}$ (Eqs. (2)) to the data for the three age groups. In both steps, we used the non-linear minimization function provided by Microsoft Excel and we confirmed the results with the similar function in MATHEMATICA. Observed and simulated time series are shown in Fig. 2.

In the fitting procedure of the second step, the initial values of each age class were set to the observed values of 1949. We also tried to treat these initial values as additional parameters to be fitted together with the others, but this gave a fit qualitatively similar to the one in Fig. 2, while reducing the sum-of-squares only very slightly.

The parameter estimates are summarized in Table 1. To assess the quality of these parameter estimates, we used a residual bootstrap as described by Efron and Tibshirani (1993), creating 150 bootstrapped time series by taking the fitted trajectory as a basis and adding to each point a randomly drawn residual from the set of residuals (with replacement). The figure of 150 replications was chosen empirically in such way that the bootstrapped estimates of the standard deviations were obtained with two significant digits, i.e. we stopped adding replications when the second digit of the estimates did not change anymore.

The quality of the fit of the model to data was tested by the standard F -statistic (suggested by Zar (1999) also for the case of non-linear regression, though the significance values will have no absolute meaning) with 7 and 26 degrees of freedom. The values for N_0 , N_1 , and N_2 are, respectively, $F = 4.22$, $F = 4.01$, and $F = 4.40$ which are all very highly significant ($P < 0.005$).

We next examined the question whether, among the input variables, variations in fishing efforts play an important role. To answer this question, we simulated the population dynamics with constant fishing efforts \bar{h}_1 and \bar{h}_2 (fixed to their averages over the observed period) and found that this did not much worsen the general comparison of the simulated trajectories with the data. This outcome can be explained by the relatively small variation of harvesting efforts. In contrast, fixing either the salinity or the temperature to their average values \bar{S} and \bar{T} , made it impossible to obtain a good fit to the data. We conclude that, in

Table 1

Estimated parameter values and means \pm SD provided by the bootstrap procedure; parameters p_2 , S_0 , and T_0 were taken from ichthyological sources

Parameter	Regression estimate	Bootstrap estimate \pm SD
Yearling fecundity (a_1)	6.65	6.31 ± 3.35
Adult fecundity (a_2)	12.58	13.0 ± 4.7
Intraspecific competition (b) [ind ⁻¹]	3.9×10^{-9}	$(3.8 \pm 0.8) \times 10^{-9}$
Juvenile survival (p_0)	0.544	0.567 ± 0.072
Yearling survival (p_1)	0.474	0.467 ± 0.061
Adult survival (p_2)	0.622	
Salinity tolerance (σ_S) [%]	1.27	1.35 ± 0.19
Temperature tolerance (σ_T) [°C]	1.92	2.05 ± 0.35
Optimal salinity (S_0) [%]	11.0	
Optimal temperature (T_0) [°C]	10.0	

the proposed model, the recorded fluctuations of temperature and salinity play a much more important role in explaining the observed population dynamics than does harvesting.

Some parameter values need additional explanation. Looking at the set of parameters in Table 1, it may seem strange that juvenile survival is estimated to be higher than yearling survival. First, this might be explained by the unofficial fishing in the Azov Sea, which is assessed to be significant and which affects also yearlings (Zaidiner et al., 1998). Secondly, during the transition from yearling to adult stages, about 30% of yearlings expand their habitat by occupying more salty regions of the sea, a phenomenon that is not taken into account explicitly in the model, but which results in a higher natural mortality induced by the acclimatization stress.

The value $b = 3.9 \times 10^{-9} \text{ ind}^{-1}$ represents moderate intraspecific competition. For example, according to Eq. (2a), 20×10^6 yearlings and adults lead to recruitment reduction by approximately 8%. Model 2 is quite sensitive to b : as in any model with a Ricker recruitment function, this intraspecific competition parameter can be used for rescaling population abundance.

With the identified value of salinity tolerance ($\sigma_s = 1.3\%$), the model is extremely sensitive to salinity fluctuations: changing from 11 to 13‰ reduces recruitment as low as 20% of the optimum. This matches independent estimates of catches made by ichthyologists for these two salinity levels: 3.6×10^6 and 1.0×10^6 individuals correspondingly (Kozlitina et al., 1998).

4.1. Deterministic stability analysis

An analytical study of all possible equilibria of model 2 was performed with constant values for harvesting effort and environmental factors (fixed to their averages over the observed period, $\bar{h}_1, \bar{h}_2, \bar{S}, \bar{T}$). Writing model 2 in vector form as $\mathbf{N}_{t+1} = \mathbf{F}(\mathbf{N}_t)$, then the equilibria $\mathbf{N}^* = (N_0^*, N_1^*, N_2^*)$ can be found as solutions of the algebraic system $\mathbf{N}^* = \mathbf{F}(\mathbf{N}^*)$. The stability of equilibria characterizes the viability of the population. Namely, if the equilibrium is asymptotically stable, then, starting in its vicinity, the population

will tend to the equilibrium with time and will stay there indefinitely in the absence of external disturbance. The stability can be estimated with a standard technique (see, e.g. Edelstein-Keshet, 1988) by linearizing the system around the equilibrium and estimating the eigenvalues λ_i of the Jacobian matrix ($i = 1, 2, 3$). If $|\lambda_i| < 1$ for all i , then the equilibrium is asymptotically stable.

Two equilibria exist simultaneously in this autonomous system: the trivial equilibrium $\mathbf{N}^* = (0, 0, 0)$ and the non-trivial (non-zero) equilibrium

$$N_0^* = \frac{\ln[(a_1 p_0(1 - \bar{h}_1) + a_2 P)\bar{G}_S \bar{G}_T]}{b(p_0(1 - \bar{h}_1) + P)}$$

$$N_1^* = p_0(1 - \bar{h}_1), \quad N_2^* = P N_0^*,$$

where $P = \frac{p_0 p_1(1 - \bar{h}_1)(1 - \bar{h}_2)}{1 - p_2(1 - \bar{h}_2)}$ and \bar{G}_S, \bar{G}_T are the reducers (1) calculated for the average values of salinity and temperature, respectively. With the estimated set of parameters (Table 1), the non-trivial equilibrium is stable and the trivial one is unstable. This remains true in the absence of harvesting and with optimal environment conditions. However, if the average environmental conditions are far from their optimal values, the non-trivial equilibrium becomes unstable and the trivial one stable (deterministic extinction). In this case the absolute value of the dominant real eigenvalue λ_m becomes greater than one, while two complex conjugate eigenvalues always remain inside the unit circle. As an example, Fig. 4 shows how the stability (measured by the dominant eigenvalue λ_m) of the non-trivial equilibrium varies with salinity and with harvesting pressures. One can see that stability is much less sensitive to the harvesting pressure than to salinity.

5. Stochastic simulation

5.1. Environmental stochasticity

As an input to the simulation model, we built stochastic time series of salinity. First, time series of river inflows (Q_t) were generated as a Markov stochastic process (details can be found in Ratkovich, 1977). Then, following Bronfman and

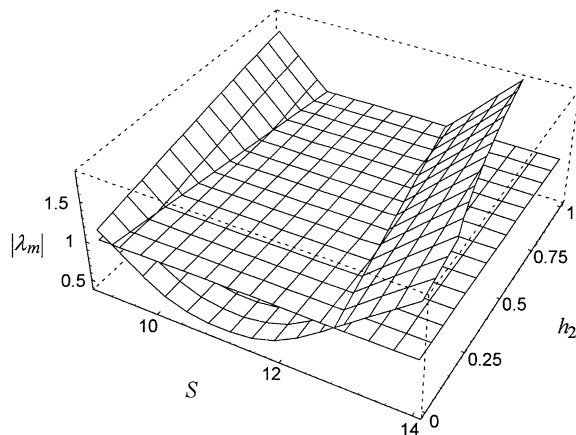


Fig. 4. Stability analysis of the model under fixed environmental conditions (see text). The surface gives the absolute value of the dominant (real) eigenvalue λ_m of the Jacobian at the non-trivial equilibrium, as a function of the sea salinity and the fishing effort on adults (h_2). The stability region corresponds to values of $|\lambda_m|$ below the flat surface (i.e. $|\lambda_m| < 1$). Note that it is only reasonable to consider that part of the surface where salinity is above the optimum $S_0 = 11\%$.

Surkov (1976), salinity was calculated with the equation:

$$S_t = 3.928 + 0.771S_{t-1} - 0.022Q_t - 0.015Q_{t-1}. \quad (4)$$

The temperature time series was simply generated by reshuffling (permutation) the observed values. These randomly generated environmental factors were then incorporated into the simulation program. For each set of harvesting parameters h_1 and h_2 , 50 stochastic simulations were replicated. Starting from the initial values of 1950, the time series of environmental factors keep the same serial correlation structure as the observed data. As for the bootstrapping above, this number of simulations was determined empirically, obtaining statistics with 5% accuracy.

5.2. Estimation of extinction risk and average catch

The simulated abundances of age groups were used to investigate numerically the problem of extinction risk minimization together with optimal harvesting under fixed effort. Specifically, we

consider the two-criteria problem of finding the combinations of fixed fishing efforts (h_1, h_2) that maximize the total expected harvest while minimizing the probability that the stock size will fall below the quasi-extinction threshold within the time horizon H . For each replication, the catches were averaged over the same period $H = 35$ years:

$$\bar{C} = \frac{1}{H} \sum_{t=1}^H C_t$$

with C_t given by Eq. (3) and with the average individual weights of yearlings $w_1 = 0.5$ kg and adults $w_2 = 2.5$ kg (calculated as a weighed average of all adult age groups). The quasi-extinction threshold was set to 10% of the maximum number of adults in the observed time series, i.e. 4.78 million individuals (Fig. 2C). This threshold value corresponds to the lowest abundance of adult pike-perch observed in the seventies, which was judged to be alarmingly low (Goptarev et al., 1991). An extinction event in the simulations consists therefore in the fall of adult abundance under this threshold at any time during the simulation period H . It is true, of course, that this does not necessarily mean real extinction of the population, but one should interpret the model results qualitatively rather than quantitatively: a high probability of falling under the quasi-extinction threshold indicates a high vulnerability of the population to various external impacts not accounted for by the model.

6. Results

The results of the multicriteria analysis are given as isopleths of catches (Fig. 5), of risks (Fig. 6), and as a Pareto diagram (Fig. 7). The fishing efforts h_1 (on yearlings) and h_2 (on adults) were varied independently from 0 to 0.7 by increments of 0.02.

These figures show that the policy of the 1980s ($h_1 = h_2 = 0.07$, the square on Fig. 7) was somewhat suboptimal. Avoiding yearling harvesting while keeping the effort on adults unchanged ($h_1 = 0$; $h_2 = 0.07$) would bring a moderate improvement of each criterion (see Figs. 4 and 5, and the star on Fig. 7). With respect to the

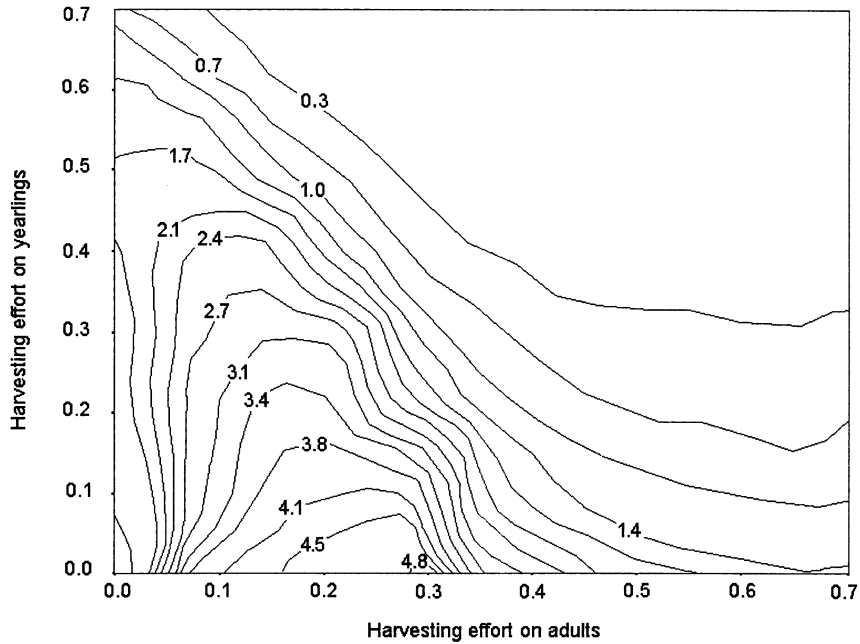


Fig. 5. Isopleth diagram of the average annual catch (10^3 tonnes) as a function of the harvesting efforts on yearlings and adults.

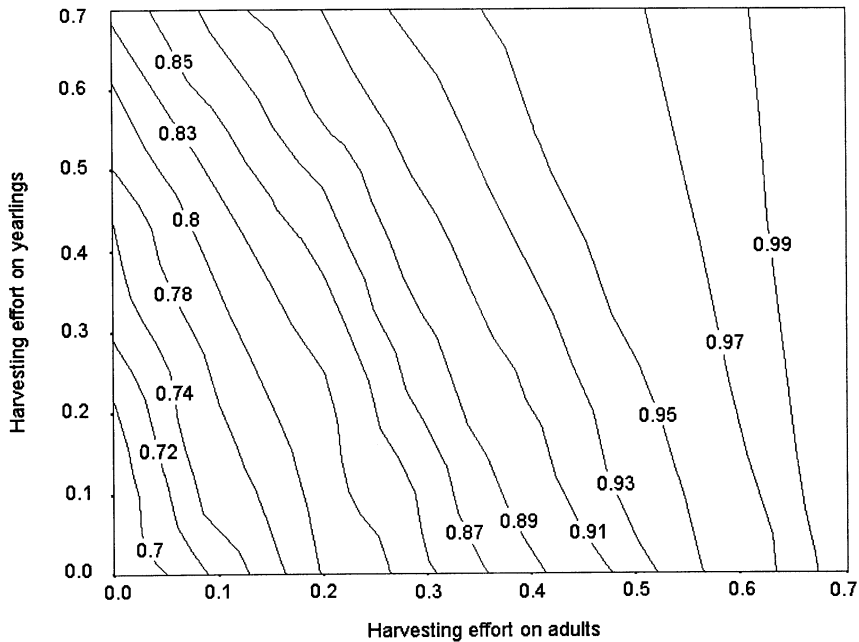


Fig. 6. Isopleth diagram of the extinction risk (probability that adult abundance drops below the quasi-extinction threshold, fixed at 4.78 million individuals) as a function of the harvesting efforts on yearlings and adults.

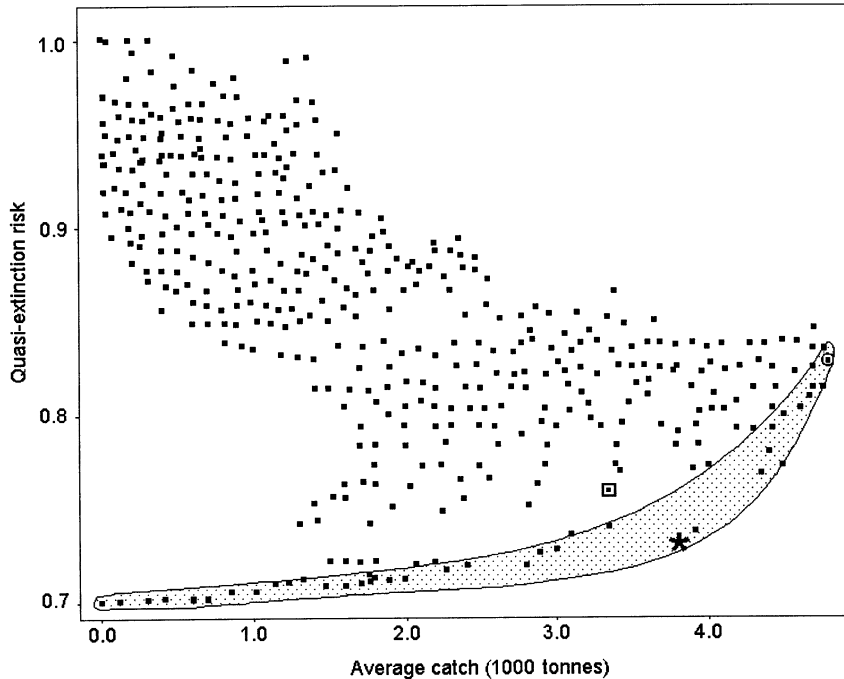


Fig. 7. Pareto diagram of quasi-extinction risk versus average catch. Each point results from 50 stochastic simulations with a given fishing strategy (a pair of constant fishing efforts on yearlings and adults). The Pareto frontier (indicated by the shaded zone) gives the trade-off between economic and conservation criteria. The square is the current strategy ($h_1 = h_2 = 0.07$); the star is the strategy with avoidance of yearling harvesting ($h_1 = 0$; $h_2 = 0.07$); the circle is the strategy for maximization of harvesting alone ($h_1 = 0$; $h_2 = 0.3$).

economic criterion alone (maximization of catches), Fig. 5 shows that the harvesting efforts should be $h_1 = 0$ and $h_2 = 0.3$. However, this would cause a strong increase of the extinction risk (see Fig. 6 and the circle on Fig. 7). Anyway, Fig. 6 shows that, even if harvesting were stopped, the extinction risk would remain high, close to 0.7.

Analyzing further the effect of age-specific harvesting, one can notice that the extinction risk is not as sensitive to yearling harvesting h_1 as it is to adult harvesting h_2 (Fig. 6). This might be explained by observing that the total size of the yearling and adult classes are similar (Fig. 2), while the adults have a much higher fecundity. Therefore, removal of adults has a comparatively higher negative effect on recruitment (which increases the extinction risk) than removal of yearlings. The simulations indicate that this fecundity effect is stronger than the reduced recruitment to

the adult age class through yearling harvesting.

On the Pareto graph (Fig. 7), the extinction risk over the simulation period H (y -axis) and the average total catch (x -axis) are represented as various points, each characterized by a given fishing strategy (h_1, h_2). The strategy corresponding to a given point cannot be read from the graph but can be known with help of our software. The Pareto frontier gives the trade-off between the two criteria. For a given catch, the Pareto frontier gives the lowest quasi-extinction probability that can be obtained; or, for a given risk level, it gives the best catch that can be obtained. Examining these points with our software, we found that points lying near the Pareto frontier all correspond to very small yearling harvesting efforts ($h_1 \leq 0.03$). Besides, due to stochasticity of the simulations, the frontier should be considered as a fuzzy approximation (dashed contour in Fig. 7). All points in the vicinity of the

Pareto frontier can be considered as optimal solutions. The above-mentioned strategy of selective adult harvesting ($h_1 = 0$; $h_2 = 0.07$) is an example of a Pareto-optimal point (the star in Fig. 7).

7. Discussion

The main strength of the present analysis, from our point of view, consists in the relative simplicity of the simulation model. Model 2 combines concepts that are very well known and widely used in fish population ecology: an age structure, fixed effort harvesting, and a Ricker reproduction function that accounts for two environmental factors. The level of model complexity is a function of research objectives and of data availability. More detailed simulation models could account for fish migration and spatial structure (Pet et al., 1996; Bartholow, 1996) or for the influence of genetic differences on population dynamics (McKenna, 2000). However, as it was shown by Bartholow (1996) who examined the effects of habitat spatial resolution on population-level responses, taking into account fine details of population functioning can be redundant: the aggregated description of spawn habitat gave a satisfactory approximation. Additionally, detailed description requires detailed information including knowledge about small-step temporal variations of the model state variables. In this paper, we present a model with age structure only, because the pike-perch population in the Azov Sea is harvested with a low size-selectivity. Thus, there is no need to use a model dealing with size- and sex-structure for evaluating the gradual size-selective impact on fish stock (e.g. Staub et al., 1987; Tyutyunov et al., 1993; Pet et al., 1996). Using the time step of 1 year, we avoid the explicit description of complex seasonal and small-scale details. This aggregated approach demonstrates its effectiveness on the ‘macro-level’ of population functioning (Fig. 2) and, therefore, can be considered as a reliable aide for the assessment of fishery policies from both economical and ecological aspects. Thus, the model and the software are applicable in the field of conservation biology and can

be useful for fishery experts, giving additional suggestions and arguments to make their decisions.

Regarding the way of accounting for environmental factors, the proposed model 2 is typically a ‘recruitment-driven’ model, contrary to the basic hypothesis of ‘mortality-driven’ models (e.g. Bartholow, 1996). Thus, natural survival rates p_i are supposed to be constant. This choice is supported by ichthyological observations on the modelled species (Goptarev et al., 1991; Kozlitina et al., 1998; Kovtun et al., 1998) and allows to obtain a good fitting of the model to the data (Fig. 2). Besides, it is easily seen from Table 1 that the identified values of p_i are characterized by low variations.

Inadequate information regarding population dynamics in stochastic environment can lead to worthless estimates of population viability (Burgman et al., 1993; Vucetich and Waite, 1998). Taking into account salinity fluctuations as the most important source of environmental stochasticity, we use the approach developed earlier for the Azov Sea by Bronfman and Surkov (1976). That model simulates precisely stochastic time series of salinity in order to obtain the same serial correlation as the observed values. This point is very important because the risk of depopulation depends directly on the number of consecutive high salinity values in the simulated series. Year-to-year autocorrelations in environmental fluctuations are known to decrease the extinction risk (Gottschalk and Griebeler, 2000). This is a typical property of population models with environmental (‘external’) stochasticity: once population size is small, it might be reduced even further if the environmental conditions remain bad (Stephan and Wissel, 1999).

The constant value of adult mean weight $w_2 = 2.5$ kg corresponds to 3-year-old individuals. Since adult survival is only 60% per year, moderate harvesting will not decrease the mean age too much, i.e. the economic criterion (which is the total weight) will be correctly estimated by the model (which works with population numbers rather than biomass). However, if the model is used to assess a pillage strategy, the biomass of the catch will be overestimated. Such strategy

should therefore be examined with double caution: firstly, because it results in a high extinction risk, secondly, because of the overestimation of the economic value of the catch.

The final choice of catch norms should belong to the set of Pareto-optimal strategies. Strictly speaking, even the strategies that maximize the single economic criterion should not include yearling harvesting at all. In addition, suppression of yearling harvesting would improve the conservation criterion as well (Fig. 6). Unfortunately, with the current fishing techniques, it seems impossible to be perfectly selective. Moreover, considering the trade-off between commercial profits and population safety, one can suggest not to exploit the population at all for some period of time, in order to allow recovery of the population density.

Of course, the recommendation to stop harvesting is not realistic. However, the Pareto analysis allows making some optimistic conclusions even with continued harvesting. Indeed, analysing the Pareto frontier in Fig. 7, it is easy to see that increasing the harvesting effort on adults (h_2) from 0 to the critical value $h_2 = 0.3$ (yield maximization) would cause a much faster increase of the average annual catch than that of the extinction probability. Only a further increase of the harvesting pressure results in a notable fall of catches, which can be recognized by fishery managers as an obvious manifestation of over-catch. As a result, the risk of extinction also rises rapidly.

As already mentioned, Fig. 6 shows that, even if exploitation is stopped, the quasi-extinction probability will remain high (≈ 0.7). Certainly, the quasi-extinction risk is only qualitative due to the arbitrary choice of the threshold. Nevertheless, this high value is alarming and emphasizes once again the importance of environmental factors for the pike-perch population. Thus, one should suggest that additional measures to improve the environmental conditions might be necessary to save the population. In other words, during the period of population recovery, economical investments should be made to increase the inflow of freshwater and to support the development of restocking factors, improvement of fishing gear selectivity, and other conservation

measures intended to increase population viability.

The analytical study of the model shows that, with values of sea salinity close to the optimum, the non-trivial equilibrium is stable. Otherwise, with high salinity (see, e.g. the peak of 1976 in Fig. 1) the origin becomes attractive, meaning deterministic extinction of the population. At the same time, it was seen that increasing the harvesting effort on adults does not influence much the stability of the non-trivial equilibrium (Fig. 4). On the basis of this non-intuitive result, we can suggest that it would be more useful to edict regulations on the river inflow in order to avoid strong fluctuations of sea salinity, rather than just reducing the harvesting effort. For example, dealing with different problems of salmon stock management, Bartholow (1996) notes that streamflow manipulation to influence the quality and quantity of habitat is one of the most quickly and easily achieved restoration methods. The critical role of salinity in the Azov Sea was already demonstrated with a simulation model devoted to the dynamics of the whole ecosystem (Vorovich et al., 1981).

To further demonstrate the comparative importance of environmental factors, we have used the model to simulate a hypothetical conservative action of salinity regulation: on the assumption that the annual inflow of the Don and Kuban rivers is increased by 5 km³ (15% of average annual inflow), random time series of annual sea salinity were generated by the model that was described above (Eq. (4)). According to this model, such increase in river inflow diminishes the average salinity by approximately 0.5‰ to an average of 11.5‰, which is considered as the upper bound for conditions of sustainable harvesting of pike-perch in the Azov Sea (Kozlitina et al., 1998).

The simulations show that harvesting of yearlings is not desirable, but harmless if not intensive. The dependence of both economic and ecological criteria on the harvesting effort on adults (h_2) is shown in Fig. 8 (assuming no harvesting of yearlings). Comparing Figs. 4 and 5, it can be seen that the situation improves considerably for both annual catch and quasi-extinction risk, under such slight improvement of environmental conditions.

We want to emphasize that this simulated scenario is not unrealistic. Such an increase of river inflow can be obtained with a rational use of water in the Azov basin. Furthermore, after 1986 the annual river inflow actually rose to the ecologically admissible level of 34–36 km³. Consequently, after the peak of 1976, the average sea salinity declined by about 0.15‰ per year and in 1994–1997, it fluctuated near 10.6–11.0‰ (Kuropatkin, 1998). At the end of May of the rainy year 1998, a minimum (over the period 1960–1998) of 9.8‰ was recorded (Kuropatkin, 1998). This is the probable cause for the increase of pike-perch larval survival observed annually since 1989 (Kovtun et al., 1998). Fig. 3 shows that after reaching its minimum in 1993, the catch also has started to grow.

In summary, our analysis makes a pessimistic forecast if the environmental conditions and the catch techniques do not improve, but it suggests that the combined effects of improved salinity and selective catch gear should result in a sustainable pike-perch exploitation of considerable economic value.

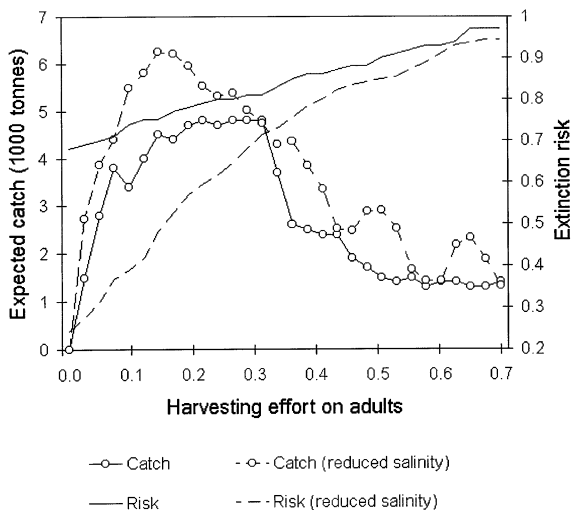


Fig. 8. Results of simulations with slightly increased freshwater inflow. The average annual salinity is now 11.5‰ instead of 12‰. Harvesting of yearlings is assumed to be perfectly avoided.

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