

Chapter 46

Population Viability Analysis of the European Sturgeon (*Acipenser sturio* L.) from the Gironde Estuary System

Ivan Jarić, J. Knežević-Jarić, G. Cvijanović, and Mirjana Lenhardt

Abstract This study presents a population viability analysis (PVA) conducted on the Gironde population of *Acipenser sturio* using the Vortex software package. As identified by the model, the most important objects of the future research efforts related to *A. sturio* life history should be the determination of the mean fecundity, age at which females reach maturity, female spawning frequency and population sex ratio, as well as the intrinsic population growth rate and the age-specific natural mortality, especially of the youngest age classes. The model has confirmed a high population susceptibility to unsustainable fishery, and a slow recovery potential that can span over a number of decades. The detected lag between the stocking activities and the initiation of the actual population recovery should be taken into account in the planning of restoration projects. PVA should be recognized as an important tool, and integrated within future research, management and policy development efforts.

I. Jarić (✉)

Institute for Multidisciplinary Research, Kneza Višeslava 1, 11000 Belgrade, Serbia
e-mail: ijaric@imsi.rs

J. Knežević-Jarić

Ecological Society “Endemit”, Oračka 42, 11080 Belgrade, Serbia
e-mail: jelak@isp.b92.net

G. Cvijanović

Institute for Multidisciplinary Research, Kneza Višeslava 1, 11000 Belgrade, Serbia
e-mail: mitrandir@imsi.rs

M. Lenhardt

Institute for Biological Research, Despota Stefana 142, 11000 Belgrade, Serbia
e-mail: lenhardt@ibiss.bg.ac.rs

46.1 Introduction

European sturgeon (*Acipenser sturio* L., 1758) is considered to be the most endangered sturgeon species in the world (Paaver 1999). It was once one of the most widely spread sturgeon species, but due to a number of different anthropogenic impacts, such as overfishing and pollution, it became extirpated from the majority of the European watersheds (Williot et al. 2002, 2009a; Jarić et al. 2009a). It is believed that the only remaining populations are the ones inhabiting the Gironde–Garonne–Dordogne Basin in France and the Rioni River in Georgia (Williot et al. 2002). Since 1982, following the collapse of *A. sturio* stocks in the Gironde Estuary, it has become a fully protected species in France (Trouvery et al. 1984; Williot et al. 2009a). The population, however, continued to decline, mainly due to by-catch mortality (Rochard et al. 1997). These negative impacts have been further aggravated by the lack of knowledge concerning its demography and life history.

Due to the above-described problems, the development of population models that are able to assess viability and the extinction risks of a population could be a tool that would bridge the existing gaps in understanding of sturgeon ecology and life history (Jarić et al. 2010). Population viability analysis (PVA) represents a method that is able to cope with uncertainties present in ecological data, and it is considered to be an indispensable tool in conservation biology (Boyce 1992; Akçakaya and Sjögren-Gulve 2000). PVA program packages have become very popular, since they are able to provide sufficient reliability and replication ability, as well as having low programming skill requirements. Vortex simulation software has been routinely used by the World Conservation Union (IUCN) to establish quantitative classification of endangered species (Brook et al. 1997).

Within the present study, we developed a model of the Gironde *A. sturio* population by means of the Vortex software package. We assessed the sensitivity of the model with regard to changes in different life-history parameters, and also conducted simulations using different fishing pressures and stocking dynamics, to assess patterns of their influence on population viability. Outcomes of the present study could be important both for the scientific community and for *A. sturio* management and policy development authorities.

46.2 Model Parameterization and Scenario Development

Population viability analysis of the *A. sturio* population from the Gironde Estuary system was conducted using the Vortex software package (version 9.72). The package consists of a computer model that runs an individual-based Monte Carlo simulation of a population through the action of deterministic factors and environmental, demographic, and genetic stochasticity (Lacy 1993). For a more detailed description of Vortex, see Lacy (1993, 2000) and Miller and Lacy (2005).

As a first step in model parameterization, a literature survey was conducted in order to define the population parameter values that are necessary for scenario-building in Vortex. For parameters that have been provided by different authors as ranges of values the most probable mean value was determined, and the ranges were included in the model as an environmental variation. If the ranges provided by different authors proved to be inconsistent, the values that were given in the majority of the publications were applied, and the other values were included in the sensitivity analysis. The values of all life history parameters used in the model are presented in Table 46.1.

In general, the available data on the deterministic growth rate (the r value) and the natural mortality for sturgeon populations are very poor. Based on the limited available information, we applied a number of different population growth rates ranging between 0.05 and 0.15 (see Jarić et al. 2010). Other authors have applied either a uniform mortality in their population models across all age classes following the age 0+ (Jager 2005, 2006; Heppell 2007; Beamesderfer et al. 2007; Kennedy and Sutton 2007; Doukakis et al. 2010), or fitted the mortality to reach the expected population growth, by placing a higher mortality on the younger age classes and significantly lower mortality on adults (Jarić et al. 2010). We applied both approaches in this study, but placed a greater emphasis on the latter approach. This decision was made because the former one was shown to be less ecologically realistic when applied in Vortex. Namely, a uniform distribution of age-specific mortality allowed the existence of only a small number of adults in a population, and prevented them from reaching the old ages that have been reported for this species.

To define a starting point for the determination of age-specific natural mortality, we used Rikhter and Efanov's method for the estimation of natural mortality for fish species in temperate regions (FISAT II 2000), which requires age at maturity of females as the input variable. This approach yielded 3.8% as a natural mortality value. Thereafter, we applied three different population growth rates (i.e., 0.05, 0.10 and 0.15) and fitted the age-specific natural mortality in such a way that each of the r values was met in different scenarios. We assumed that the younger age groups are characterized by higher annual variability in natural mortality, which decreases towards the adult age groups. We have applied the general rule for the standard deviation of the natural mortality of each of the groups for ages 1–3 to be equal to 40% of the natural mortality of those age groups. With regard to the age groups from age 4 to adulthood it was equal to 25%, while for adults it was 10% of the age-specific natural mortality. However, if the mean mortality of a certain age plus two standard deviations resulted in a higher value than 100 (mean + 2x SD > 100), the standard deviation was reduced to the level where the outcome was equal to 100. Variation in the natural mortality of the age 0–1 was incorporated through the standard deviation of the mean number of the offspring.

For the uniform mortality scenarios, we determined a starting value based on those applied in population models of sturgeon species that share similar life history with *A. sturio* (Beamesderfer et al. 2007; Heppell 2007). Thereafter, the value was modified to make it possible to reach the expected Gironde *A. sturio* life expectancy

Table 46.1 Population parameter estimates used in a Vortex simulation model of the European sturgeon from the Gironde Estuary

Variable	Value	Source
Age of first offspring for females	17	Magnin 1962 cited in Rochard et al. (1991), Magnin 1963 cited in Gessner et al. (2007)
Age of first offspring for males	14	Magnin 1962 cited in Rochard et al. (1991), Magnin 1963 cited in Gessner et al. (2007)
Maximum age of reproduction	40	Magnin 1962 cited in Williot personal communication
Maximum number of progeny per year	160 ^a	–
Sex ratio at birth – in % males	50	–
Percent adult females breeding	29.15%	(Every 3–4 years) Williot personal communication
Environmental variation in % adult females breeding	2 percentage units	–
Mean number of offspring per year	70 ^b	–
Standard deviation in number of offspring per year	10	–
Percent males in breeding pool	33.3%	(Every 3 years) Williot personal communication

Natural mortality fitted to the three different values of intrinsic population growth rate, and values of the uniform mortality applied:

Age	$r = 0.05$	$r = 0.10$	$r = 0.15$	Uniform mortality ($r = 0.129$)
age 0–1	0.0 ± 0.0^c	0.0 ± 0.0^c	0.0 ± 0.0^c	0.0 ± 0.0^c
age 1–2	85.0 ± 7.5	69.0 ± 15.5	40.0 ± 16.0	10.0 ± 2.5
age 2–3	60.0 ± 20.0	45.0 ± 18.0	30.0 ± 12.0	10.0 ± 2.5
age 3–4	35.0 ± 8.75	25.0 ± 6.25	20.0 ± 5.0	10.0 ± 2.5
age 4–5	10.0 ± 2.5	10.0 ± 2.5	10.0 ± 2.5	10.0 ± 2.5
age 5–17	3.8 ± 0.95	3.8 ± 0.95	3.8 ± 0.95	10.0 ± 2.5
age 17+	3.8 ± 0.38	3.8 ± 0.38	1.0 ± 0.1	10.0 ± 2.5

^a $400,000 \times 0.0004$: maximum fecundity 400,000 (Williot personal communication); 0.0004 is the maximum natural survival of age 0–1 (Pine et al. 2001; Jager 2005, 2006; Bajer and Wildhaber 2007; Kennedy and Sutton 2007)

^b $350,000 \times 0.0002$: average fecundity 300,000–400,000 (Williot personal communication); 0.0002 is the average natural survival of age 0–1 (Pine et al. 2001; Jager 2005, 2006; Bajer and Wildhaber 2007; Kennedy and Sutton 2007)

^cAge 0–1 natural mortality is incorporated in the mean and maximum number of offspring; see model parameterization and scenario development for clarification

and, at the same time, to produce the r value that falls within the range 0.05–0.15. As a result, a uniform mortality rate of 0.10 was reached, resulting in an r value of 0.129. In this case, we applied uniform standard deviation of the natural mortality across all age classes.

The age 0–1 natural mortality was incorporated in the model through a proportional reduction of the number of offspring produced. As a result, the

young-of-the-year that did not survive up to the age of 1 were not included in the simulation (see Miller and Lacy 2005). Therefore, it is important to note that in the presented results of the model, the population size (N) does not include embryos and young-of-the-year that do not survive up to the age of 1.

As proposed by a number of authors (Beissinger and Westphal 1998; Akçakaya 2000), the pseudo-extinction criterion was applied in this study, instead of scoring the risk of reaching absolute extinction, and the pseudo-extinction threshold was set to ten individuals. Sex ratio was set to be equal, and effects of an unequal sex ratio were tested in the sensitivity analysis. Age distribution at the beginning of each simulation was set to be stable, since the stochastic fluctuations in age distribution and year-class strength already represent an integral part of simulations in Vortex. The percentage of individuals of each sex that participate annually in spawning was established as $S = 100/I$, where I represents the time between two spawning migrations. For the sake of simplicity, as well as to reduce the number of assumptions already made in model parameterization, parameters such as density dependence in reproduction and the existence of catastrophic events were not included in the model. While demographic stochasticity is incorporated in the Vortex model, the problem of other Allee effects was partly resolved by introducing pseudo-extinction (Akçakaya 2000).

To increase the precision of the results, the number of iterations was set at 1,000 for all simulations and, occasionally, at 10,000. Excluding sensitivity analysis and the stocking scenarios, carrying capacity was set to be equal to the initial population size, (see Reed et al. 2003 for similar approach).

Four basic groups of scenarios were developed within this study: (1) assessment of the minimum viable population size (MVPs), (2) sensitivity analysis, (3) harvest scenarios, and (4) reintroduction and supportive stocking. The MVPs was assessed by carrying out a series of simulations with varying initial population size. The simulations covered 500 years, in order to assess long-term viability, while carrying capacity was retained at the level of the initial population size.

Sensitivity analysis was conducted to test the dependence of the model behaviour on changes in each of the life-history parameters, as well as to span the ranges of life-history parameter values provided by different authors. As stated by Miller and Lacy (2005), sensitivity analysis can reveal the effect of uncertainty in the applied parameters, as well as the relative influence of different parameters on the population dynamics and model projections. To test the effect of change in any single parameter, only one parameter was varied at a time. The age distribution of the initial population was kept constant in all scenarios, to avoid the effect of change in age distribution when life-history parameters are being modified. The tested changes in individual parameters either reflected the ranges provided for that parameter by different authors, or a particular range of change was applied (e.g., $\pm 50\%$).

The results of the sensitivity analysis were assessed through the use of a ranking approach. Absolute changes in output variables (i.e., pseudo-extinction probability, deterministic and stochastic population growth rates, expected heterozygosity, and the mean time to recovery) that were caused by changes in each of the life history parameters were ranked from the highest to the lowest. The ranking was performed

Table 46.2 Results of the sensitivity analysis in a Vortex simulation model, presenting the influence of each life-history parameter through the total rank value (where higher value signifies higher influence) for each of five resulting parameters – pseudo-extinction probability (P), deterministic population growth rate (r_{det}), stochastic population growth rate (r_{stoch}), heterozygosity level (H) and the population recovery time (R)

Parameter ^a	P	r_{det}	r_{stoch}	H	R	Total
Mean offsp ^b	143	237	240	208.5	241.5	1,070
Fem age	176.5	221.5	225.5	170.5	232.5	1,026.5
Sex ratio	113.5	237	224.5	175	169	919
1–4 mort	123	200.5	202	141	169	835.5
Fem breed	91.5	186	184	116.5	174.5	752.5
N	192	69	92	216	177.5	746.5
4-adult mort	119	164	161	139	145.5	728.5
Adult mort	125.5	165	166.5	135	135.5	727.5
Max age	129.5	149	139	124	102	643.5
K EV	166.5	69	77	142.5	114.5	569.5
Mort EV	159.5	69	98.5	164	74.5	565.5
Male age	128	69	72	93	91	453
Mean offsp EV	142.5	69	52	73.5	68	405
Male breed	109	69	58	82	68	386
Max offsp ^b	111	69	60	65	74.5	379.5
Fem breed EV	82	69	60	66.5	74.5	352

^aMean offsp – mean number of offspring (also represents egg-to-age-1 natural mortality); fem age – time needed for females to reach maturity; 1–4 mort – natural mortality of year 1–4 age groups; fem breed – female spawning frequency; N – initial population size; 4-adult mort – natural mortality of year 4 to adulthood age groups; adult mort – natural mortality of adult individuals; max age – maximum reproductive age; K EV – environmental variation in carrying capacity; mort EV – environmental variation in natural mortality of all age groups; male age – time needed for males to reach maturity; mean offsp EV – environmental variation in mean number of offspring; male breed – male spawning frequency; max offsp – maximum fecundity; fem breed EV – environmental variation in female spawning frequency

^bAge 0–1 natural mortality is incorporated in the mean and maximum number of offspring; see model parameterization and scenario development for clarification

separately for each of the four fitted age-specific natural mortalities (i.e., for three population growth rates applied and uniform mortality fitting – see Table 46.2) and the ranks were thereafter summed up for each life-history parameter. The life-history parameters were ranked separately for each output variable to reveal the relative influence of each parameter on the output variables, and then summed up across the output variables to determine the life-history parameters that have the highest overall influence on population viability.

The basic question addressed in the harvest scenarios was the determination of the threshold level of the harvest pressure below which population viability would still be preserved. The harvest was introduced in the model as the percentage of adults taken every year.

Additional scenarios were developed to test the effectiveness of reintroduction because of the ongoing restoration efforts across its former range (Kirschbaum and Gessner 2000; Williot et al. 2009a, b). The main goal was to test different

approaches in supportive stocking, and their influence on the population viability. In general, the stocking dynamics varied between different scenarios along three main dimensions: the total number of individuals that were released, age of released individuals (whether they were juveniles or adults), and the sex ratio of released individuals.

Output variables tracked in simulations were pseudo-extinction probability, population size, deterministic and stochastic population growth rates, expected heterozygosity, and the time to recovery (i.e., the mean time needed for population to reach half of the carrying capacity). As suggested by other authors (Ebenhard 2000; Beissinger 2002; Hanski 2002; Reed et al. 2002), the results of all scenarios were predominantly used to compare their relative effects, for instance to compare different management scenarios, rather than to be employed for absolute predictions. An extinction risk of $P < 0.05$ was applied as the threshold to evaluate viability (Beissinger and Westphal 1998).

46.3 Results and Discussion

Estimated population age structure of the undisturbed population is presented in Fig. 46.1. The ratio of adult individuals in a population was generally low for all population growth rates applied, and ranged from 3.5 to 7.8%. As depicted in

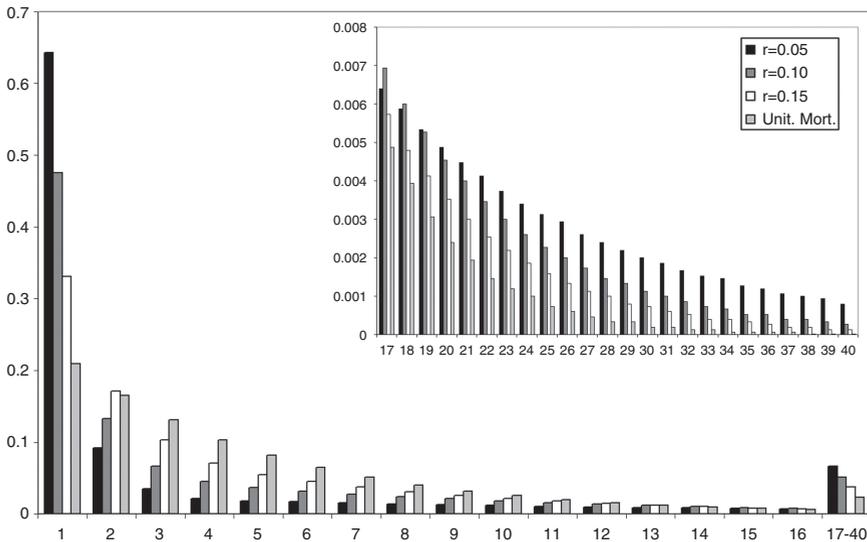


Fig. 46.1 Age structure of the undisturbed *A. sturio* population in the Gironde system, estimated by the Vortex population model according to four different population growth rates (r) applied. Small plot in the *upper right* corner presents the age structure within adult age groups. *Unif. mort.* –10% mortality rate applied across all age groups ($r = 0.129$)

Fig. 46.1, the largest ratio of adult individuals was reached at the lowest population growth rate ($r = 0.05$), while the lowest value was recorded when the uniform mortality across all age groups was applied ($r = 0.129$). This was caused by a high juvenile mortality applied at the lower population growth rates, thus resulting in a larger relative number of adults. The mean female generation time varied across different scenarios between 22 and 25.5 years, and was the longest at low population growth rates. These values are in accordance with findings of other authors (e.g., the range of 20–25 years according to Gessner 2000).

As expected, the scenarios for the assessment of the MVPs showed a high sensitivity of the population persistence probability to the population size. A long-term viability ($P < 0.05$) became more likely only for the population sizes of 200–500 individuals (representing all age classes at the stable age distribution, but without eggs and young-of-the-year that do not survive to age 1). However, the loss of genetic variability was still very high at such population sizes, and only at a population size of 2,000 or more individuals was the loss of heterozygosity lower than 10%. It is important to emphasize that the presented MVPs were estimated under zero negative impact scenarios. Consequently, pressure from any negative impact (e.g., pollution, by-catch mortality) would result in an increase of the MVPs.

Results of the sensitivity analysis are presented in Table 46.2. It revealed similar results across different population growth rates applied. The most influential parameters on the model output were the mean number of offspring, egg-to-age-1 natural mortality, sex ratio, and the age at which females reach maturity. The changes in juvenile and adult natural mortality, female spawning frequency and the initial population size also resulted in high model sensitivity. The results are similar to the population models of other sturgeon species (Pine et al. 2001; Beamesderfer et al. 2007; Kennedy and Sutton 2007), which have determined highest model sensitivity to egg mortality, overall natural mortality, age at which females reach maturity, and the spawning frequency. When all output variables are ranked together, the most influential parameters overall were the mean number of offspring and the egg-to-age-1 natural mortality. However, since these two life history parameters were combined within the model, it was not possible to make a distinction between their individual effects. If the influence on pseudo-extinction is considered separately, the single most influential parameter was the age at which females reach maturity. In general, male-related traits and the environmental variation of parameters had a relatively weak effect on the model behaviour. The low influence of male-related traits has also been recognized by other authors, and as a result, many sturgeon population models have been restricted to females only (e.g., Bajer and Wildhaber 2007; Kennedy and Sutton 2007).

The weaker influence of adult mortality, when compared with the influence of other life-history parameters, could probably be explained by a low ratio of adults in the population. The lower ratio of adult individuals means that the total number of individuals affected by a change in adult mortality would be also low, especially when compared with the ratio of youngest age classes in a population (Fig. 46.1). Findings of other authors have significantly varied with regard to the sensitivity of their models to adult mortality. While Kennedy and Sutton (2007) found a lower

influence of adult mortality in comparison to egg and juvenile mortality, Pine et al. (2001), Boreman (1997) and Heppell (2007) found the opposite.

The relationship between the deterministic growth rate and the other model output variables (pseudo-extinction probability, expected heterozygosity and the population recovery time), based on the results of sensitivity analysis, is presented in Fig. 46.2. A strong correlation between these variables was also confirmed by the

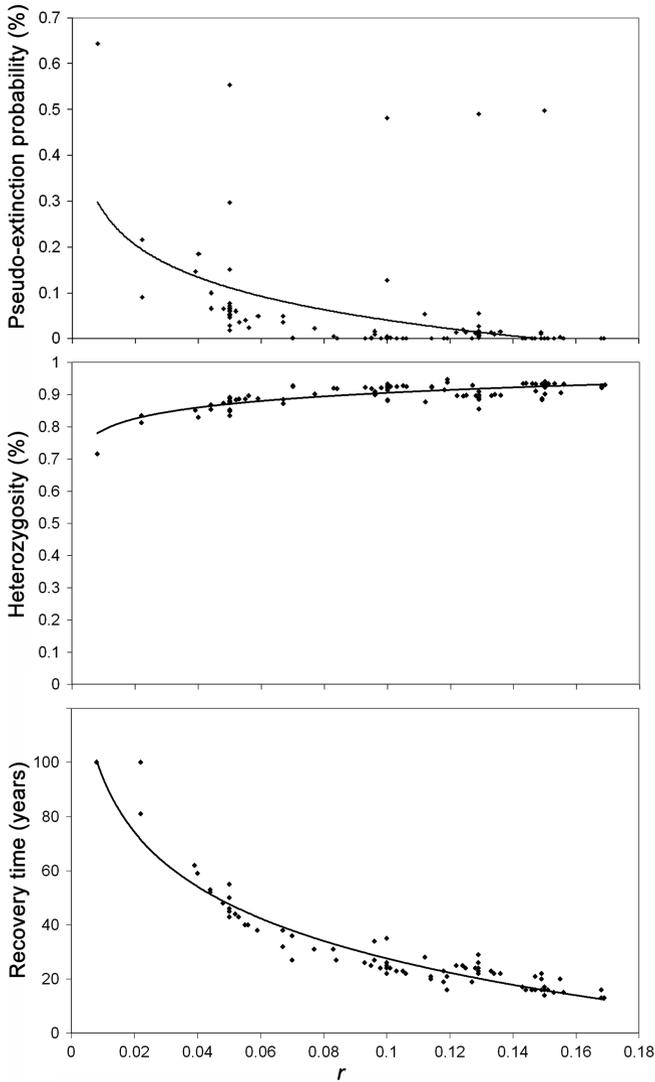


Fig. 46.2 Relationship between the deterministic population growth rate (r) and the pseudo-extinction probability, expected heterozygosity and population recovery time, based on the results of sensitivity analysis

use of Spearman's non-parametric correlation test (correlation coefficients ranging between 0.641 and 0.928, $P < 0.001$). Although the deterministic population growth rate does not consider stochasticity, male-related traits, and genetically linked demographic traits, it has a potential to be used as a predictor or indicator of the population state (Jarić et al. 2010).

The life-history parameters that are the most influential upon the population viability are usually proposed for the highest management and/or research priority (Mills and Lindberg 2002). Although fecundity, female age at maturity, female spawning frequency, sex ratio, and egg-to-age-1 and juvenile natural mortality represented the most influential parameters in this study, they have been poorly studied in the Gironde *A. sturio* population. There is a virtually complete lack of data on *A. sturio* spawning frequency in the Gironde Estuary system (Williot personal communication), and the same is true for age-specific natural mortality and the intrinsic population growth rate. The data on age at maturity and mean fecundity provided by different authors is inconsistent, and the reports on sex ratio have been often biased for different reasons (Williot personal communication). As a result, given the importance of these parameters and the current lack of knowledge, they should be widely recognized by the scientific community as future research priorities. Another issue which has been poorly addressed within the present literature on sturgeon ecology is the existence of senescence in sturgeons, the decline in fecundity and spawning success before the maximum longevity has been reached (Jarić et al. 2010). While the senescence has been detected in aquaculture (Williot et al. 2005), there are no data available for wild populations.

In harvest scenarios, unsustainable fishing pressures (i.e., where pseudo-extinction probability was $P > 0.05$) have always reduced population to the half of its initial size for no more than 2 or 3 decades, and resulted in a completely depleted stock within a single century of fishing (Fig. 46.3). The caviar industry within the Gironde

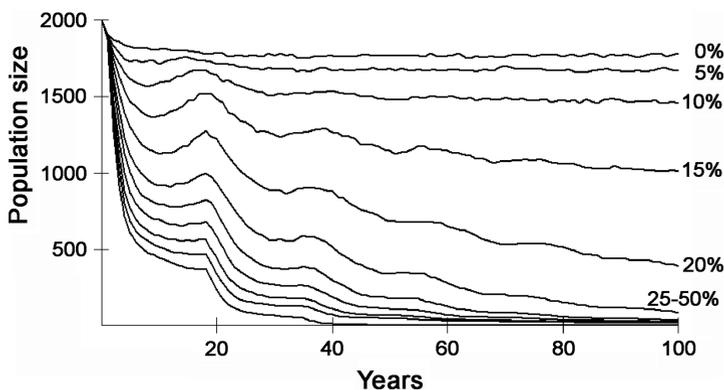


Fig. 46.3 Effects of different fishing pressures applied on the Gironde *A. sturio* population ($r = 0.05$). Y-axis – average population size among all iterations in a simulation; X-axis – years of the simulation; percentage values represent different fishing pressures. Initial population size and carrying capacity in all scenarios were set to 2,000 individuals

system was established in 1920, and by the early 1970s, *A. sturio* was no longer an economically important resource (Williot et al. 1997; Holčík 2000). Such a time frame for complete stock depletion (i.e., within 50 years) would correspond in this model to fishing pressures far above the sustainable ones (i.e., 15–20% units higher). The maximum sustainable harvest rates at the population growth rates of 0.05 and 0.10 were, respectively, 0.15 and 0.40, signifying percent of adults taken from the population each year. These threshold levels are similar to those determined for other sturgeon species, which ranged from 0.05 to 0.35 (Beamesderfer et al. 1995; Quist et al. 2002; Colombo et al. 2007; Kennedy and Sutton 2007). However, these values are somewhat higher than those determined for the historical *A. sturio* population in the Danube River basin at the same population growth rates (0.10 and 0.25, respectively; Jarić et al. 2010), which can be explained by different parameter values used in the model, due to significant differences in their life history among different river basins. The threshold harvest level determined at the population growth rate of 0.15 was rather high (0.60), which could be an indication that such a population growth rate might be unrealistically high for such slow-maturing species. A total fishing ban in France was introduced in 1982 (Williot et al. 2009a), but the accidental by-catch continues to threaten the species (Limburg and Waldman 2009). According to Gessner (2000), these catches alone might be sufficient to extirpate the population in the course of approximately 15 years. The harvest scenarios described thus have a potential to be applied for the evaluation of harvest through by-catch.

Results of some of the reintroduction scenarios are presented in Fig. 46.4. At lower population growth rates (0.05 and 0.10), the number of juveniles that needed to be released to ensure population viability ($P < 0.05$) had to be in the range from a few thousands to a few tens of thousand specimens. When compared with the number of adults needed to produce the same model outcomes, the juvenile/adult ratio ranged approximately from 10/1 to 100/1, being on average 42/1 across all scenarios. The juvenile/adult ratio generally decreased towards higher population growth rates. The optimal sex ratio of released individuals (i.e., the one that produces quickest recovery and smallest extinction risk and heterozygosity loss) was shifted towards a greater female ratio. The best outcomes, when juveniles were used for stocking, were reached with 65–70% of individuals being females.

In both adult and juvenile stocking scenarios, there was a clearly defined lag before the actual population recovery, which clearly corresponded to the time needed for females to reach maturity (Fig. 46.4). At lower numbers of stocked individuals, there was another lag in population recovery following the first one, prolonging the phase of a more rapid population recovery to twice the female age at maturity (17×2 in present scenarios). Such a lag, caused by the time needed for juveniles to reach maturity and start sustaining population through reproduction, has been found in other sturgeon species as well (Jarić et al. 2010). This should be taken into account in all restoration project planning, and confirms the claim that the recovery of sturgeon populations is a multi-decadal affair (Lenhardt et al. 2006). As stated by Gessner (2000), management programmes might prove their efficiency

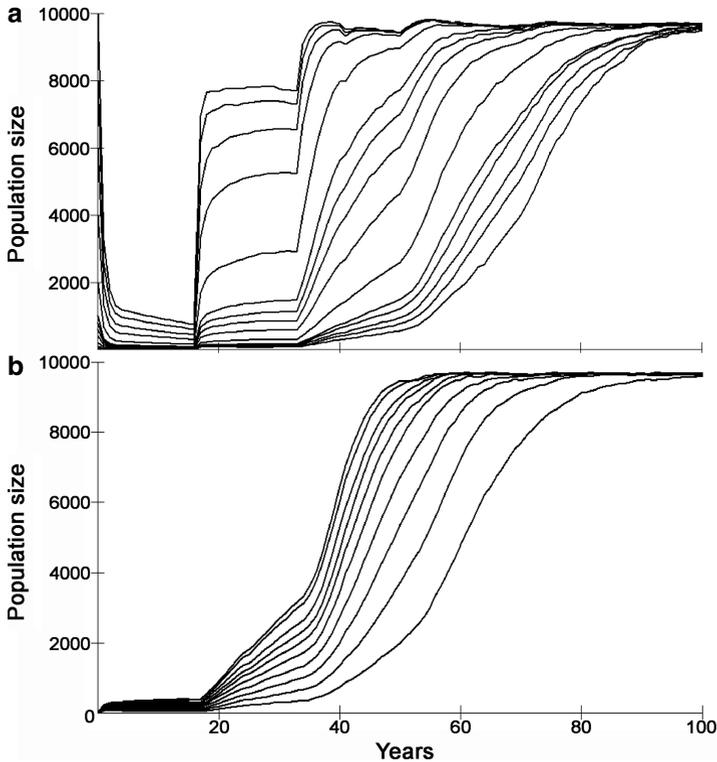


Fig. 46.4 Recovery of the *A. sturio* population after its reintroduction ($r = 0.10$). From the lowest to the highest one, respectively, full lines in each graph represent (a) the release of 20, 40, 60, 80, 100, 200, 400, 600, 800, 1,000, 2,000, 4,000, 6,000, 8,000 and 10,000 juvenile (age 1) individuals, and (b) the release of two, four, six, eight, ten, 12, 14, 16, 18 and 20 adult individuals; Y-axis – average population size among iterations with surviving populations in a simulation; X-axis – years of the simulation

only after 15–25 years, which unfortunately represents a time frame hard to cover by consistent administrative policies.

The decision to use the Vortex simulation model as a suitable tool for the assessment of *A. sturio* population viability was based on a number of its advantages: (1) it is able to perform sensitivity analysis and identify key life-history parameters, (2) it has options for simulating both harvest and reintroduction scenarios, and (3) as an individual-based model, it is able to track population genetic diversity and include demographic and environmental variation within the model. However, there are a number of other available PVA and fishery assessment software packages that might be applied for the assessment of *A. sturio* population viability and the major anthropogenic impacts. For instance, RAMAS software is able to handle very large population sizes, which is an advantage when it is applied to species with high fecundities, such as fish (Brook et al. 1997). It is also suitable for cases when there are spatial and habitat suitability factors that should be

incorporated in the model. On the other hand, RAMAS is not an individual-based model, but an age- or stage-structured model that therefore lacks genetic considerations and more complex demographic patterns. Furthermore, while some fishery assessment program packages have been successfully used for the assessment of sturgeon fishery sustainability (e.g., Doukakis et al. 2010), they are less suitable for the *A. sturio* population, since it is no longer an object of commercial fishery. In the end, none of the existing PVA software packages can be recognized as a superior one, since each one has its advantages and drawbacks, and studies that involve comparison of efficiency of different models are common (Brook et al. 1997, 1999, 2000; Lindenmayer et al. 1995).

Ongoing *A. sturio* restoration efforts are and will be hindered to an extent by the lack of knowledge regarding its life history and demography. As stated by Waldman (2000), currently there is a limited species-specific knowledge and experience available regarding *A. sturio* and, given its scarcity, there are few specimens to be used for scientific purposes. However, due to an expressed need for urgent conservation measures, the development of *A. sturio* PVA models should not wait for better data. On the contrary, PVA can provide directions to investigators with regard to the research activities of greatest importance (Jarić et al. 2009b). Therefore, PVA should be recognized as an important tool to be included in the process of research and management planning and policy development. As has been stated by Williot et al. (2009b), PVA studies can also help managers dealing with broodstock and supportive stocking management to choose the main lines for future breeding programmes. Figure 46.5 presents general principles of the integration of PVA within these processes. The greatest potential of PVA models is their ability to provide insights regarding the parameters that could have the strongest impact on population persistence and thus define research priorities (Jarić et al. 2009b), and to compare the efficacy of different management scenarios (Ebenhard 2000; Reed et al. 2002). At the same time, new insights acquired through research and monitoring could be used to evaluate and refine the models, as well as to define further modelling directions. A productive interaction among the PVA developers, scientific community,

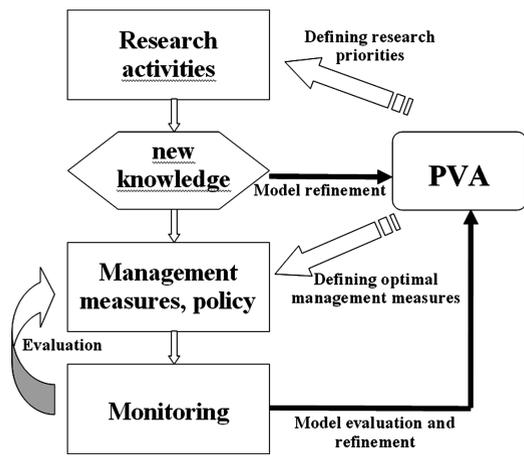


Fig. 46.5 Principles of population viability analysis (PVA) integration within the research and management planning and policy development processes

management authorities, policy developers and the general public will be essential for the general effectiveness of this approach. At the same time, the availability of PVA program packages such as VORTEX, which have already proved their usefulness and reliability (Brook et al. 2000), could diminish the need for experience in programming and developing PVA models (Jarić et al. 2009b).

46.4 Conclusions

The population viability analysis presented in this paper is one of the few models that have been developed for the Gironde *A. sturio* population. While there is an urgent need to establish effective recovery measures for the population, the existing knowledge regarding its life history and demography is still severely limited. In addition to some commonly recognized factors, such as political will and public awareness, improving knowledge with regard to this species could be one of the key factors for its successful preservation and recovery. The most important direction of future research efforts related to *A. sturio* life history (as was identified by the model) should be the determination of the mean fecundity, age at which females reach maturity, female spawning frequency and population sex ratio, as well as intrinsic population growth rate and age-specific natural mortality, especially of the youngest age classes. The question regarding the existence of senescence in wild populations should be also resolved.

The model confirmed a high population susceptibility to unsustainable fishery, and a slow recovery potential, which could span over a number of decades. The lag between stocking activities and the initiation of actual population recovery, which corresponds to the time the juveniles need to reach maturity, should be taken into account when planning the restoration projects.

The development of *A. sturio* PVA models should not wait for better data. On the contrary, they should be recognized as an important tool, and be integrated within the research, management, and policy development efforts. If used with necessary caution, PVA can provide invaluable help for the successful recovery of this endangered species.

Acknowledgements This study represents a part of the activities within Project No. 173045, funded by Ministry of Science of Republic Serbia. The authors would like to thank Dr Patrick Williot for his help in resolving different questions of *A. sturio* life history, as well as Dr Torbjörn Ebenhard for his guidance regarding use and model development in the Vortex simulation model.

References

- Akçakaya HR (2000) Population viability analyses with demographically and spatially structured models. *Ecol Bull* 48:23–38
- Akçakaya HR, Sjögren-Gulve P (2000) Population viability analyses in conservation planning: an overview. *Ecol Bull* 49:9–21

- Bajer PG, Wildhaber ML (2007) Population viability analysis of Lower Missouri River shovelnose sturgeon with initial application to the pallid sturgeon. *J Appl Ichthyol* 23:457–464
- Beamesderfer RCP, Rein TA, Nigro AA (1995) Differences in the dynamics and potential production of impounded and unimpounded White Sturgeon populations in the Lower Columbia River. *T Am Fish Soc* 124:857–872
- Beamesderfer RCP, Simpson ML, Kopp GJ (2007) Use of life history information in a population model for Sacramento green sturgeon. *Environ Biol Fish* 79:315–337
- Beissinger SR (2002) Population viability analysis: past, present, future. In: Beissinger R, McCullough DR (eds) *Population viability analysis*. University of Chicago Press, Chicago, pp 5–17
- Beissinger SR, Westphal MI (1998) On the use of demographic models of population viability in endangered species management. *J Wildl Manage* 62(3):821–841
- Boreman J (1997) Sensitivity of North American sturgeons and paddlefish to fishing mortality. *Environ Biol Fish* 48:399–405
- Boyce MS (1992) Population viability analysis. *Annu Rev Ecol Evol S* 23:481–506
- Brook BW, Lim L, Harden R, Frankham R (1997) Does population viability analysis software predict the behaviour of real populations? A retrospective study on the Lord Howe Island woodhen *Tricholimnas sylvestris* (Sclater). *Biol Conserv* 82:119–128
- Brook BW, Cannon JR, Lacy RC, Mirande C, Frankham R (1999) Comparison of the population viability analysis packages GAPPS, INMAT, RAMAS and VORTEX for the whooping crane (*Grus americana*). *Anim Conserv* 2:23–31
- Brook BW, O'Grady JJ, Chapman AP, Burgman MA, Akçakaya HR, Frankham R (2000) Predictive accuracy of population viability analysis in conservation biology. *Nature* 404:385–387
- Colombo RE, Garvey JE, Jackson ND, Brooks R, Herzog DP, Hrabik RA, Spier TW (2007) Harvest of Mississippi River sturgeon drives abundance and reproductive success: a harbinger of collapse? *J Appl Ichthyol* 23:444–451
- Doukakis P, Babcock EA, Pikitch EK, Sharov AR, Baimukhanov M, Erbulekov S, Bokova Y, Nimatov A (2010) Management and recovery options for Ural River beluga sturgeon. *Conserv Biol* 24(3):769–777
- Ebenhard T (2000) Population viability analyses in endangered species management: the wolf, otter and peregrine falcon in Sweden. *Ecol Bull* 48:143–163
- FISAT II (2000) FAO-ICLARM Fish stock assessment tools. Version 1.2.2. FAO, Rome, Italy
- Gessner J (2000) Reasons for the decline of *Acipenser sturio* L., 1758 in central Europe, and attempts at its restoration. *Bol Inst Esp Oceanogr* 16(1–4):117–126
- Gessner J, Van Eenennaam JP, Doroshov SI (2007) North American green and European Atlantic sturgeon: comparisons of life histories and human impacts. *Environ Biol Fish* 79:397–411
- Hanski I (2002) Metapopulation of animals in highly fragmented landscapes and population viability analysis. In: Beissinger R, McCullough DR (eds) *Population viability analysis*. University of Chicago Press, Chicago, pp 86–108
- Heppell SS (2007) Elasticity analysis of green sturgeon life history. *Environ Biol Fish* 79:357–368
- Holčík J (2000) Major problems concerning the conservation and recovery of the Atlantic sturgeon *Acipenser sturio* L., 1758. *Bol Inst Esp Oceanogr* 16(1–4):139–148
- Jager HI (2005) Genetic and demographic implications of aquaculture in white sturgeon (*Acipenser transmontanus*) conservation. *Can J Fish Aquat Sci* 62:1733–1745
- Jager HI (2006) Chutes and ladders and other games we play with rivers I. Simulated effects of upstream passage on white sturgeon. *Can J Fish Aquat Sci* 63:165–175
- Jarić I, Lenhardt M, Cvijanović G, Ebenhard T (2009a) *Acipenser sturio* and *Acipenser nudiiventris* in the Danube – extant or extinct? *J Appl Ichthyol* 25:137–141
- Jarić I, Lenhardt M, Cvijanović G, Ebenhard T (2009b) Population viability analysis and potential of its application to Danube sturgeons. *Arch Biol Sci* 61(1):123–128
- Jarić I, Ebenhard T, Lenhardt M (2010) Population viability analysis of the Danube sturgeon populations in a VORTEX simulation model. *Rev Fish Biol Fish* 20(2):219–237

- Kennedy AJ, Sutton TM (2007) Effects of harvest and length limits on shovelnose sturgeon in the upper Wabash River, Indiana. *J Appl Ichthyol* 23:465–475
- Kirschbaum F, Gessner J (2000) Re-establishment programme for *Acipenser sturio* L., 1758: the German approach. *Bol Inst Esp Oceanogr* 16(1–4):149–156
- Lacy RC (1993) VORTEX: a computer simulation model for population viability analysis. *Wildl Res* 20:45–65
- Lacy RC (2000) Structure of the VORTEX simulation model for population viability analysis. *Ecol Bull* 48:191–203
- Lenhardt M, Jarić I, Kalauzi A, Cvijanovic G (2006) Assessment of extinction risk and reasons for decline in sturgeon. *Biodivers Conserv* 15:1967–1976
- Limburg KE, Waldman JR (2009) Dramatic declines in North Atlantic diadromous fishes. *BioSci* 59(11):955–965
- Lindenmayer DB, Burgman MA, Akçakaya HR, Lacy RC, Possingham HP (1995) A review of the generic computer programs ALEX, RAMAS/space and VORTEX for modelling the viability of wildlife metapopulations. *Ecol Model* 82:161–174
- Miller PS, Lacy RS (2005) VORTEX: A stochastic simulation of the extinction process. Version 9.50 user's manual. Conservation Breeding Specialist Group (SSC/IUCN), Apple Valley, MN, USA
- Mills LS, Lindberg MS (2002) Sensitivity analysis to evaluate the consequences of conservation actions. In: Beissinger R, McCullough DR (eds) *Population viability analysis*. University of Chicago Press, Chicago, pp 338–366
- Paaver T (1999) Historic and recent records of native and exotic sturgeon species in Estonia. *J Appl Ichthyol* 15:129–132
- Pine WE III, Allen MS, Dreitz VJ (2001) Population viability of the Gulf of Mexico sturgeon: inferences from capture–recapture and age-structured models. *T Am Fish Soc* 130:1164–1174
- Quist MC, Guy CS, Pegg MA, Braaten PJ, Pierce CL, Travnichek VH (2002) Potential influence of harvest on shovelnose sturgeon populations in the Missouri River system. *N Am J Fish Manag* 22:537–549
- Reed DH, O'Grady JJ, Brook BW, Ballou JD, Frankham R (2003) Estimates of minimum viable population sizes for vertebrates and factors influencing those estimates. *Biol Conserv* 113:23–34
- Reed JM, Mills LS, Dunning JB Jr, Menges ES, McKelvey KS, Frye R, Beissinger SR, Anstett MC, Miller P (2002) Emerging issues in population viability analysis. *Conserv Biol* 16(1):7–19
- Rochard E, Williot P, Castelnaud G, Lepage M (1991) Elements de systematique et de biologie des populations sauvages d'esturgeons. In: Williot P (ed) *Acipenser*. Cemagref, Antony, pp 475–507
- Rochard E, Lepage M, Meauzé L (1997) Identification et caractérisation de l'aire de répartition marine de l'esturgeon européen *Acipenser sturio* à partir de déclarations de captures. *Aquat Living Resour* 10:101–109
- Trouvery M, Williot P, Castelnaud G (1984) Biologie et ecologie d'*Acipenser sturio*. Etude de la pêche. Cemagref, Etude no 17, Série Esturgeon no 1, p 79
- Waldman JR (2000) Restoring *Acipenser sturio* L., 1758 in Europe: lessons from the *Acipenser oxyrinchus* Mitchell, 1815 experience in North America. *Bol Inst Esp Oceanogr* 16(1–4):237–244
- Williot P, Rochard E, Castelnaud G, Rouault T, Brun R, Lepage M, Elie P (1997) Biological characteristics of European Atlantic sturgeon, *Acipenser sturio*, as the basis for a restoration program in France. *Environ Biol Fish* 48:359–370
- Williot P, Arlati G, Chebanov M, Gulyas T, Kasimov R, Kirschbaum F, Patriche N, Pavlovskaya LP, Poliakova L, Pourkazemi M, Kim Y, Zhuang P, Zholdasova IM (2002) Status and management of Eurasian sturgeon: an overview. *Int Rev Hydrobiol* 87(5–6):483–506

- Williot P, Brun R, Rouault T, Pelard M, Mercier D, Ludwig A (2005) Artificial spawning in cultured sterlet sturgeon, *Acipenser ruthenus* L., with special emphasis on hermaphrodites. *Aquaculture* 246:263–273
- Williot P, Rochard E, Rouault T, Kirschbaum F (2009a) *Acipenser sturio* recovery research actions in France. In: Carmona R et al (eds) *Biology, conservation and sustainable development of sturgeons*. Springer, Dordrecht, pp 247–263
- Williot P, Rochard E, Kirschbaum F (2009b) Acceptability and prerequisites for successful introduction of sturgeon species. In: Carmona R et al (eds) *Biology, conservation and sustainable development of sturgeons*. Springer, Dordrecht, pp 369–384

