

The radiation exposure of fish in the period of the Techa river peak contamination



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ABSTRACT

Waterborne radioactive releases into the Techa River from the Mayak Production Association in Russia during 1949–1956 resulted in downstream contamination of the river ecosystem. The discharged liquid waste contained both short-lived isotopes (⁹⁵Zr, ⁹⁵Nb, ^{103,106}Ru, ^{141,144}Ce, ⁹¹Y, ⁸⁹Sr and ¹⁴⁰Ba with half-life from 3 days to 1.02 years) and the long-lived ⁹⁰Sr and ¹³⁷Cs (half-life - 28.79 y and 30.07 y, respectively). Even now, when two half-lives of ⁹⁰Sr and ¹³⁷Cs have passed, the contamination in the upper river region (about 70 km from the source of releases) is still relatively high. Current anthropogenic dose rates calculated for the fish of the Techa River depend on the distance along the stream and decrease from 150 to 3 μGy day⁻¹. Radiation exposure of fish is expected to have been much higher at the time of the releases. The aim of the study was to evaluate the dose rates for the most common fish species of the river, viz., roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and pike (*Esox lucius*), in the period of peak contamination of the upper reaches of the Techa River from 1950 to 1951. To achieve this objective, calculation of both internal and external dose rates was performed. For dose rate calculation, the contamination of the river compartments was modeled, body-size dependent dose coefficients were evaluated, morphometric data were analyzed. Maximum dose rates were obtained for roach; minimum – for pike, it depends on fish lifestyle (time spent at the bottom). In the period before September 1950, fish of the upper reaches are assessed to have been exposed to dose rates exceeding the screening level equal to 240 μGy day⁻¹. From September 1950 up to the end of 1952 the fish dose rates along the Techa River were found to be close to the UNSCEAR threshold equal to $9.6 \times 10^3 \mu\text{Gy day}^{-1}$ or even much more higher (up to $1.9 \times 10^5 \mu\text{Gy day}^{-1}$). Extremely high historical dose rates did not lead to the difference in fish size and fish growth rate currently observed in the Techa River and in the comparison waterbody (the Miass River). Discussion includes the description of radiation effects observed currently in the river fish. Today the effects observed in hematopoietic system may be the consequence of radiation exposure of fish over several generations. For example, long term dwelling of fish in the radioactively contaminated environment leads to their adaptation to chronic radiation exposure. At the same time, an increase their sensitivity and decrease their adaptive capacity to respond to other stress factors can be observed.

1. Introduction

The monitoring of wildlife populations inhabiting the contaminated territory is the key source of information about community response to radiation exposure (NEA, 2007). Fish are the most widespread

bioindicators of water pollution, since they are very sensitive to changes in the aquatic environment compared with other hydrobionts. Fish accumulate toxicants in their bodies up to several orders of magnitude greater than those measured in ambient water (Kryshev and Ryabov, 2000; Fesenko et al., 2011; et al., 1974; Yancheva et al., 2015;

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de Lapuente et al., 2015). Therefore, fish are used effectively for assessing the health of aquatic ecosystems and environmental risk (Sazykina and Kryshev, 2003; Brown et al., 2008; Dalzochio et al., 2016; López-López and Sedeño-Díaz, 2015). Being higher-level consumers themselves, many species of fish are consumed as food by people and the radionuclide accumulation in fish can lead to an increased risk to human health as a result of migration through food chains (Puffer et al., 1982; Dhanakumar et al., 2015). Therefore, the studies of contaminated aquatic ecosystems are important from both radiobiological and radiation protection perspectives (IAEA, 2002; ICRP, 2008). In this connection, the Techa River, most significantly contaminated in 1949–1956, could be considered as a “testing ground” to verify our understanding of high-level chronic radiation effects on biota. Radioactive releases into the Techa River were the result of old technologies at the beginning of operation of the Mayak plutonium facility (Alexandrov et al., 1951; Degteva et al., 2016). The total amount of released activity was equal to 114 PBq; about 98% of the radioactivity was released within a short time period from September 1950 through October 1951 (Degteva et al., 2016). The discharged waste contained both short-lived isotopes (^{95}Zr , ^{95}Nb , $^{103,106}\text{Ru}$, $^{141,144}\text{Ce}$, ^{91}Y , ^{89}Sr and ^{140}Ba with half-life from 3 days to 1.02 years) and long-lived ^{90}Sr and ^{137}Cs (half-life - 28.79 y and 30.07 y, respectively). After 1952, the radiation exposure of aquatic biota was mostly determined by long-lived radionuclides. In 1956–1964, several dams and storage reservoirs were built to control and minimize radioactive contamination of the Techa River (Mokrov, 2003; Shagina et al., 2012). Nonetheless, even now, when two half-lives of ^{90}Sr and ^{137}Cs have passed, the contamination in the upper river region (about 70 km from the source of releases) is still relatively high (Trapeznikov et al., 1993; Balonov et al., 2007; Melnikov et al., 2012). The ratio of ^{137}Cs -to- ^{90}Sr concentration, as well as the activity concentrations themselves, are decreasing with distance from the site of releases. Since 2011, the extensive study of the Techa River biota has been performed focusing on fish fauna. The general aim was to investigate whether fish exposure to high dose rates and subsequent chronic exposure of their offspring (dozens of generations) at low dose rates results in radiation effects which can be observed 60 years after beginning of releases. One of the important tasks of the study is the radiation dosimetry.

The ranges of current activity concentrations in the river ecosystem compartments presented in Shishkina et al. (2016) for ^{137}Cs and ^{90}Sr (measurements of 2011–2013) are as follows:

- in water: 0.02–0.41 Bq l⁻¹ and 6–35 Bq l⁻¹, respectively;
- in sediments: 0.2–220 kBq kg⁻¹ (dry weight) and 0.04–4.4 kBq kg⁻¹ (dry weight), respectively;
- in benthic organisms: 0.3–28 kBq kg⁻¹ (dry weight) and 4.5–20 kBq kg⁻¹ (dry weight), respectively;
- in aquatic plants: 0.1–15 kBq kg⁻¹ (dry weight) and 1–3 kBq kg⁻¹ (dry weight), respectively;
- in fish: 0.01–0.3 kBq kg⁻¹ (fresh weight) and 0.1–1.4 kBq kg⁻¹ (fresh weight), respectively.

Maximum values were detected in the Upper Techa River (about 10 km from the historical site of releases). Minimum values are typical of the Low Techa River (about 200 km downstream of the site of releases) near the point of the confluence with the Iset River.

Current dose rates calculated for the fish of the Techa River also depend on the distance along the stream and decrease from 150 to 3 μGy day⁻¹ (Shishkina et al., 2016; Tryapitsina et al., 2017). These values are significantly lower than dose rate of $\dot{D}_1 = 9.6 \times 10^3 \mu\text{Gy day}^{-1}$. According to UNSCEAR (UNSCEAR, 2011), this is a maximum individual dose rate to a small fraction of a population of aquatic organisms that would not have adverse effects at the population level. Moreover, the maximum dose rates in the Techa River are still lower than the screening level $D_2 = 240 \mu\text{Gy day}^{-1}$ (Garnier-Laplace et al., 1997) established in the ERICA Tool to be used in screening out

environmental exposure situations of negligible concern.

Radiation exposure of fish is expected to have been much higher at the time of highest releases. To understand and accurately interpret biological effects of chronic radiation exposure in fish, it is necessary to know the history of dose formation in fish populations over the whole period of exposure.

The aim of the study was to evaluate the dose rates for the most common fish species of the Techa River, viz., roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and pike (*Esox lucius*), in the period of 1950–1952. To achieve the stated objective, calculation of both internal and external dose rates was performed. Unfortunately, for the initial period of contamination (1949–1952), environmental-monitoring data are scarce, and the reconstruction of radiation doses required modeling of the Techa River contamination using data on radionuclide releases. Therefore, one of the tasks to solve was estimation of radionuclide activity concentrations in water and bottom sediments at various distances along the stream using the model of radionuclide transport in the Techa River (Shagina et al., 2012) elaborated for human dose assessment. Another important task was to analyze the relationships between morphometric parameters of fish (mass and linear body sizes) and fish age (assuming that fish morphometric parameters were the same in the mid of XXth century). This is important in terms of dimension-dependent dose coefficients converting the activity concentration of a radionuclide (incorporated in the body or environment) into the body-average dose rate value. Morphometric parameters are also necessary for biokinetic modeling of radionuclide turnover, taking into account fish growth rate.

Therefore, the objectives of the presented study are as follows:

1. Analysis of morphometric data for fish species;
2. Description of dimension-dependent species-specific dose factors for both internal and external exposure;
3. Modeling of the Techa River contamination in the period of maximum releases;
4. Dose rate calculation and comparison with the UNSCEAR threshold ($\dot{D}_1 = 9.6 \times 10^3 \mu\text{Gy day}^{-1}$) for expected effects on freshwater organisms and with ERICA-derived incremental screening dose rate of $D_2 = 240 \mu\text{Gy day}^{-1}$ for aquatic ecosystems.

2. Materials and methods

2.1. Sampling

Fish from the Techa River and the Miass River (control waterbody) were examined in terms of morphometry. Fish were sampled at the end of April–the beginning of May and at the end of August–September at three stations (RT1, RT2 and RT3) located in the upper, middle and low reaches of the Techa River and in the Miass River (RM) in 2011–2013 (Shishkina et al., 2016). The RT1, RT2 and RT3 are located at 33 km, 109 km and 237 km, respectively, downstream of the historical site of releases. Fig. 1 illustrates the location of sampling sites. Table 1 describes the sample sizes for fish species at each station studied.

2.2. Morphometrics, age and sex determination

Linear dimensions of fish (length, width, depth) as well as fish mass and sex were measured in-situ. The weight of the fish (m) was measured with a precision balance to the nearest 0.01 g. Width of fish body (w) was taken manually as the maximum distance between two lateral sides with a Vernier caliper to the nearest 0.1 mm. Digital photographs were taken from the lateral side of the fish. Fish body length (l) and depth (d) were measured processing the digital images using freely distributed morphometric software tpsDig2 (Rohlf, 2005). All l were taken as maximum distance in parallel to the anterior-posterior body axis. All d were taken as maximum distance in perpendicular to the anterior-posterior body axis. Sex was determined by internal gonad inspection.

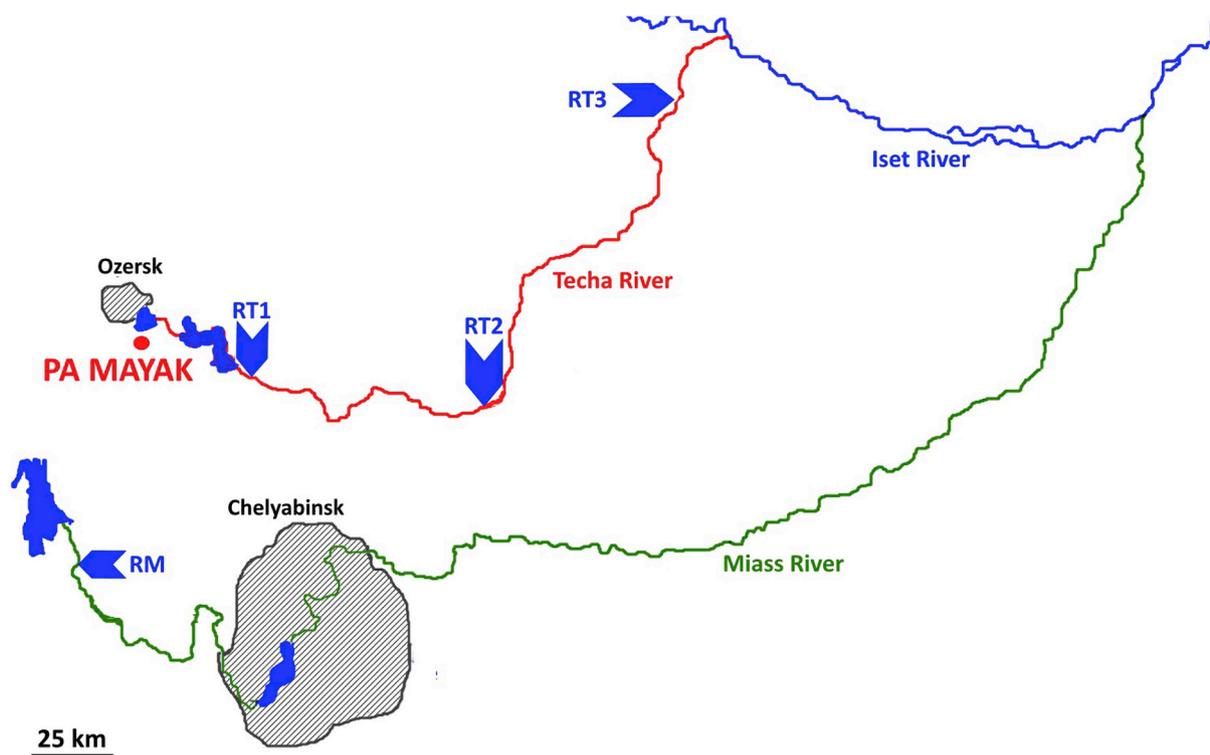


Fig. 1. Schema demonstrating the sampling sites on the Techa River (RT1, RT2 and RT3) and the Miass River (RM).

Table 1
Number of fish sampled at the Techa River (RT1, RT2 and RT3 sampling sites) and the Miass River (RM).

Sampling site	Roach	Perch	Pike
RT1	103	76	55
RT2	48	43	26
RT3	49	54	15
RM	111	85	40

Age was firstly determined by counting the annuli on scales sampled in the central zone of fish flank above the lateral line. Scale flakes were washed with liquid ammonia and, next, examined under the microscope SteREO Discovery.V12 (Carl Zeiss Microscopy, Germany). Additionally, the method of counting the annuli of opercular bone was performed. The results obtained with these two methods for each fish were compared. In case of disagreement, one more method, viz., otolith annuli count, was involved. If the measurement result (the age) of at least 2 independent methods coincided, then this value was regarded as the real age of a specimen (in terms of full fish years).

The obtained value was assumed as the age (τ) of all fish caught during the spawning season. Fish caught during the feeding period were assumed to be 5 months older ($\tau + 0.42$).

2.3. Dimension-dependent species-specific dose factors

Dose factors (k^e for external exposure and K^i for internal exposure) converting the activity concentrations of a radionuclide, r , (incorporated in the environment or homogeneously throughout the fish body) into the body-average dose rate value depend on fish body dimensions. Values of k^e and K^i typical of different sizes of fish bodies were calculated using the ERICA Assessment Tool (version 1.2) (Brown et al., 2008). This tool allows dose factors to be derived based on the ellipsoidal representation of organisms, based on (d , w , l and m) parameters. Dose factors for both internal and external exposure of

organisms of different sizes were calculated for $^{90}\text{Sr}/^{90}\text{Y}$, $^{137}\text{Cs}/^{137\text{m}}\text{Ba}$, ^{95}Zr , ^{95}Nb , $^{103,106}\text{Ru}$, $^{141,144}\text{Ce}$, ^{91}Y , ^{89}Sr and ^{140}Ba using the results of the morphometric data analysis. It should be noted that body dimensions and body masses are coupled variables, therefore, the mass dependence of dose factors was fitted by simple analytical expressions for species under study exposed to different radionuclides.

2.4. Modeling of the Techa River contamination in 1950–1951

The model of radionuclide transport in the Techa River (Shagina et al., 2012), which was originally elaborated for human dosimetry (the Techa River Dosimetry System – TRDS, Degteva et al., 2006), is applicable to fish dose rate reconstruction too. The Techa River model includes three major segments: (1) the upper-river segment adjacent to source term, (2) the up-river ponds, and (3) the free-flowing river downstream of the ponds. It should be noted that all sampling sites, viz., RT1, RT2 and RT3, are located at 3rd segment of the Techa River model. Each model segment consists of 3 compartments, viz., water, bottom sediments and floodplain. Model variables are time and distance from the site of releases. The model in the form of a system of differential equations describes the following main processes: (1) transport of radionuclides adsorbed on solid particles, originally contained in the discharges or originating in the up-river ponds (stirring up of contaminated bottom sediments) and (2) transport of radionuclides in soluble form. Since, sedimentation and resuspension depend on the size of solid particles and the water-flow rate, archive data on hydrological conditions and granulometric composition of the bottom sediments specific to the Techa River in 1949–1951 were considered in sedimentation, resuspension and sorption–desorption parameters of the model (Shagina et al., 2012).

The releases were reconstructed and published in Degteva et al. (2012) based on the combination of historical data on daily discharges of certain liquid radioactive waste and their radiochemical composition for each type of routine release.

Model output is radionuclide activity concentrations in the water and bottom sediments calculated for different values of variables (time

and distance from the site of releases). In the framework of current study, the daily radionuclide activity concentrations in water and sediments were calculated for each of radionuclides under consideration at three distances from the source of releases: 33 km, 109 km and 237 km (RT1, RT2 and RT2).

2.5. Dose rate calculation

Dose rates were calculated using the following assumptions:

- sedentary lifestyle, viz., for all species fish populations occupied only those parts of the river where sampling sites were set (it should be noted, there are no data available on fish migration along the stream of the Techa River, therefore the rough assumption was used as a first approximation);
- homogeneous body-burden for all radionuclides;
- perch spends half of lifetime at the bottom; roach spends 75% of lifetime at the bottom; pike is a mid-water dweller; the data on fish behaviour in the Techa River are not available, the assumptions were fixed expertly based on literature data (Kottelat and Freyhof, 2007; Shvenk, 2000; Nikol'sky, 1963; 1974; Manteifel, 1965, 1974; Pavlov and Gusar, 1989; Sudakov, 1977; Chebotina et al., 1992; Imbrock et al., 1996);
- only adult species (> 1 year old) were considered.

2.5.1. External dose rates

The basic expression for calculation of body-average dose rates \dot{D} due to external exposure of species, f , with mass, m , can be formulated as a system of equation (1).

$$\begin{cases} \dot{D}_{roach} = \sum_r k_{f,m}^r (0.25A_w^r + 0.75A_s^r) \\ \dot{D}_{perch} = \sum_r 0.5k_{f,m}^r (A_w^r + A_s^r) \\ \dot{D}_{pike} = \sum_r k_{f,m}^r A_w^r \end{cases} \quad (1)$$

where $k_{f,m}^r$ is the dose factor; A_w^r and A_s^r , are activity concentrations of a radionuclide, r , in the water and in the bottom sediments, respectively.

2.5.2. Internal dose rates

Assuming steady-state between the organism and the surrounding water, the body-average radionuclide activity concentration can be calculated as a product of radionuclide activity concentration in the water, A_w^r , and the radionuclide-specific body-to-water concentration ratio, CR_f^r , estimated for fish at steady-state. The steady-state is possible if the radionuclide intake rate equals the excretion rate and, therefore the concentration of radioactivity in the body remains constant. However, in the initial period of releases the water contamination was changing significantly day-by-day (Degteva et al., 2012). Therefore, the daily body-average radionuclide activity concentration, $A_f^{r,T}$ (Bq kg⁻¹), for each species, f , was modeled as a discreet function represented by the linear combination of intake and retention. The intake could be expressed based on the mean intake rate I^r (L kg⁻¹ year⁻¹) due to water contamination with mean activity concentration, $A_w^{r,T}$ (Bq L⁻¹) during the Δt (year), where t is continuous variable of time. The retention is determined by the body-average radionuclide activity concentration, $A_f^{r,T-1}$ (Bq kg⁻¹), accumulated at the previous time point, $T-1$, and retention function $R^r(\Delta t)$ describing the time dependence of the elimination of activity concentration. Thus, the internal dose rates could be expressed according to equation (2).

$$\dot{D} = \sum_r K_{f,m}^r A_f^r; A_f^{r,T} = A_w^{r,T} I^r \Delta t + A_f^{r,T-1} R_f^r(\Delta t) \quad (2)$$

where K_m^r is the species-specific dose factor converting the activity concentration A_f^r of a radionuclide, r , in the fish body into the body-average dose rate value \dot{D} .

The daily body-average radionuclide activity concentration in Eqn (2) can be easily transformed as follows (Eqn. (3)):

$$A_f^{r,T} - A_f^{r,T-1} R_f^r(\Delta t) = A_w^{r,T} I^r \Delta t \quad (3)$$

When the steady-state is achieved, the activity concentrations in the body becomes time-independent ($A_f^{r,T} = A_f^{r,T-1} = A_f^r$).

$$A_f^r (1 - R_f^r(\Delta t)) = A_w^r I^r \Delta t$$

$$\frac{A_f^r (1 - R_f^r(\Delta t))}{A_w^r \Delta t} = I^r$$

where $\frac{A_f^r}{A_w^r} = CR_f^r$ - the body to water concentration ratio of radionuclide-specific activity concentration, which is the standardized radioecological parameter (Hosseini et al., 2008). Therefore, the intake rate can be formulated according to Eqn (4).

$$I^r = CR_f^r \frac{(1 - R^r(\Delta t))}{\Delta t} \quad (4)$$

The retention function, $R^r(\Delta t)$, was assumed (according to Kryshev and Ryabov, 2000) as exponentially decreasing function (Eqn. (5)).

$$R_f^r(\Delta t) = e^{-\left(\lambda_r + \mu_f + \varepsilon_{r,f}\right)\Delta t} \quad (5)$$

where λ_r - the radionuclide-specific constant of radioactive decay; μ_f - the average annual rate of increase in fish weight; $\varepsilon_{r,f}$ - the rate of biological elimination of radionuclide. In other words, the intake function can be reformulated according to Eqn. (6).

$$I^r = CR_f^r \frac{\left(1 - e^{-\left(\lambda_r + \mu_f + \varepsilon_{r,f}\right)\Delta t}\right)}{\Delta t} \quad (6)$$

Actually, the elimination rate $\varepsilon_{r,f}$ should be species-specific, age and sex dependent. Other factors affecting the elimination rate are reproductive activity, environmental temperature, water mineral composition, etc. (Kulikov and Chebotina, 1988; Kryshev and Ryabov, 2000; Smith et al., 2002).

Finally, Eqn. (2) describing the internal dose calculation was reformulated to Eqn. (7).

$$\dot{D} = \sum_r K_{f,m}^r A_f^r; A_f^{r,T} = A_w^{r,T} CR_f^r \left(1 - e^{-\left(\lambda_r + \mu_f + \varepsilon_{r,f}\right)\Delta t}\right) + A_f^{r,T-1} e^{-\left(\lambda_r + \mu_f + \varepsilon_{r,f}\right)\Delta t} \quad (7)$$

The steady-state species-specific concentration ratios (CR) for Sr and Cs isotopes were taken from Shishkina et al. (2016), where the estimates were performed directly for the Techa River species (Table 2). The data on CRs of other radionuclides were derived from the Erica Tool database. Since, no data on bioaccumulation of Yttrium were found, the value of CR_{Y-91} was assumed to be equal to mean CR of Cesium (5501 kg⁻¹) because of metabolism similarity.

Table 2 describes the literature-derived physical and biological half-life ($T_{r1/2}$ and $T_{b1/2}$) for different radionuclides and corresponding constants of elimination. Physical half-lives were derived from JANIS (Java-based nuclear information software) freely available at NEA Data Bank (Soppera et al., 2014). Species-specific biological half-life was available only for isotopes of Cs, Sr and partially for Ru. In the calculations given below, the unknown values of $\varepsilon_{r,f}$ for bone-seeking (^{141,144}Ce, ⁹⁵Zr and ¹⁴⁰Ba) and soft-tissue distributed (^{103,106}Ru, ⁹⁵Nb and ⁹¹Y) radionuclides were assumed to be equal to those of Sr or Cs isotopes, respectively.

The average annual rate of increase in fish weight (μ_f) was

Table 2
Literature-derived parameters of biokinetic model (Equation (7)).

Radionuclide	Physical half-life ^a T _{r1/2} , y	λ _r , y ⁻¹	CR, l kg ⁻¹	Basis for CR assessment	Biological half-life T _{b1/2} , day ⁻¹	ε _r , y ⁻¹	Basis for T _{b1/2} assessment
¹³⁷ Cs	30.07	0.023	Roach – 550 Perch - 590 Pike – 500	Shishkina et al., 2016	Roach – 76 Perch - 187 Pike - 500	3.33 1.35 0.51	Hasanen et al. (1967) Carlsson, 1978
⁸⁹ Sr	0.138	5.02	Roach – 58		Roach – 70	3.61	Tjahaja et al. (2012) ^b
⁹⁰ Sr	28.79	0.024	Perch - 24 Pike - 15		Perch - 44 Pike - 44	5.75 5.75	
¹⁴¹ Ce	0.089	7.79	171 ^c	The Erica Tool database	n/a ^d		
¹⁴⁴ Ce	0.78	0.89					
¹⁰³ Ru	0.088	7.88	102 ^c		Roach – 37	6.84	Vray (1994) ^b ; Garnier-Laplace et al. (1997) ^a
¹⁰⁶ Ru	1.02	0.68					
⁹⁵ Zr	0.175	3.96	932 ^c		n/a ^d		
⁹⁵ Nb	0.096	7.22	32 ^c				
¹⁴⁰ Ba	0.035	19.8	172 ^c				
⁹¹ Y	0.16	4.33	n/a ^d				

^a NEA Data Bank (<https://www.oecd-nea.org/databank/>).
^b Obtained using the database for wildlife (Bersford et al., 2015) freely available at website of Center for Ecology & Hydrology (<http://doi.org/10.5285/b95c2ea7-47d2-4816-b942-68779c59bc4d>).
^c Species-unspecific data available.
^d No data available.

evaluated based on the analysis of morphometric data.

2.5.3. Statistical analysis

To determine if significant correlations exist between the body mass and the age, sex, sampling site or season of sampling, the Spearman rank correlations were calculated (*p* < 0.05). Non-parametric Mann-Whitney test (*α* = 0.05) was also performed to compare the datasets. Experimental data fitting was performed using the robust regression method of least absolute residuals to minimize the influence of outliers on the fit.

3. Results

3.1. Analysis of morphometric data

3.1.1. Age dependence of body mass

Table 3 shows the description of age, sex and mass distribution of all species under study. As can be seen from the table, most of the fish were younger than 3 years old. As expected, mean ages and masses of female tend to be higher than those for male. The variability of masses is large; CV ≥ 1. The Spearman rank correlations between body mass and the potential factors of influence such as age, sex and sampling site were calculated for perch, roach and pike. Coefficients of Spearman correlation are shown in Table 4, where gray cells correspond to *p* < 0.05.

According to Table 4, strong correlation between masses and ages was found for all species (as expected). In spite of the fact that female fish body mass tends to be greater than that of male (Table 2), no

Table 3
Age sex and mass distribution of sampled perch, roach and pike.

	perch		roach		pike	
	male	female	male	female	male	female
n	124	134	151	176	68	68
Age, years						
Average (CV) ^a , years	2.5 (0.4)	2.7 (0.5)	2.5 (0.3)	2.9 (0.4)	2.3 (0.5)	2.5 (0.6)
Median, years	2.3	2.3	2	3	2	2.1
25–75% ^a	2–3.3	2–3.3	2–3	2–3.3	1–3.1	2–3.3
Mass, g						
Average (CV) ^a , g	87 (1)	119 (1.1)	56 (1.7)	72 (1)	510 (0.9)	610 (1)
Median, g	52	66	36	55	350	350
25–75% ^a	20–125	28–159	22–56	34–83	130–820	220–740

^a CV – coefficient of variation.

Table 4

The Spearman rank correlations between body mass and age, sex and sampling site. Gray cells correspond to *p* < 0.05. The correlation coefficients > 0.5 (strong correlation) are shown in bold.

	Perch	Roach	Pike
Mass & age	0.83	0.69	0.89
Mass & sex	0.12	0.28	0.07
Mass & sample site	0.13	0.05	0.26

significant correlation between mass and sex was found for pike and perch and weak correlation was detected for roach.

Weak correlation between mass and sample sites was found for perch and pike. The difference was the result of non-equal age composition of the catches. The percentage of perch younger than 2 years old was ~30% at the RM and RT3 and < 10% at RT1 and RT2. More than 30% of pike at RT3 was ≤ 4 years old, whereas at other stations pike of that age made up ~13%. Comparison of the species-specific body masses within the same age groups did not prove the mass dependence on sample site.

Therefore, the data on male and female body masses as well as the data on different sampling sites can be combined for analysis of species-specific age dependence of body mass. Age dependences of combined body masses were well fitted (Table 5) by the exponential function (Eqn. (8)).

$$m_f = a_f e^{\mu_f \times t} \tag{8}$$

where μ_f – the average annual rate of increase in weight of species, *f*, which is the required parameter of the biokinetic model (Eqn. (7)).

Fig. 2 illustrates the curves of exponential fits of the obtained age dependences. The primary data are grouped in the plot according to age; the error bars correspond to standard deviations. When mass axis is

Table 5
Description of age dependences for perch, roach and pike body masses: parameters of exponential fits and corresponding standard errors as well as goodness-of-fit statistics (*r*², *p*).

	N	a _f ± err, g	μ _f ± err, year ⁻¹	r ² , <i>p</i>
Perch	258	9 ± 3	0.76 ± 0.03	0.995, < 0.0001
Roach	297	21 ± 5	0.30 ± 0.04	0.987, < 0.0001
Pike	135	140 ± 20	0.46 ± 0.01	0.993, < 0.0001

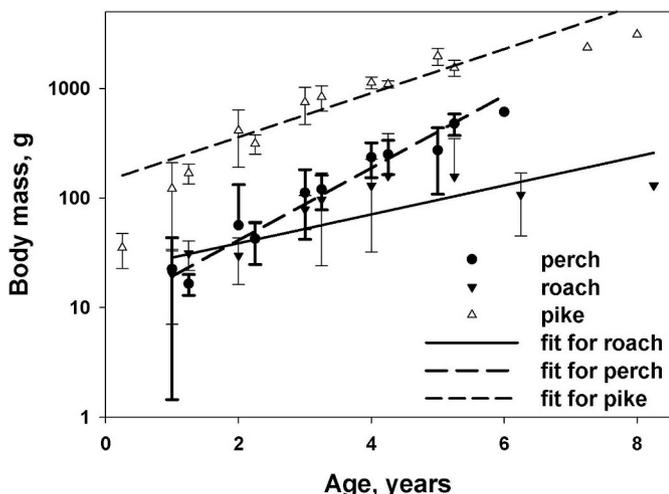


Fig. 2. Age dependence of body mass of perch, roach and pike. The curves correspond to the exponential fits of age dependences. Points correspond to the data grouped according to age; the error bars correspond to standard deviations.

in the log scale, the slope of the line of the exponential fitting is equal to μ_f . As can be seen from Fig. 2 and Table 4, the maximum annual rate of mass growth was observed for perch (about 80% per year). Pike's μ_f is 1.6 lower. The roach demonstrate minimum mass growth rate, which is 2.5 times lower than that for perch. Results obtained for μ_f are used for

calculations (Eqn. (7)).

3.1.2. Mass dependences of body sizes

Mass dependences of fish linear dimensions are well fitted by power functions (Eqn. (9)). Fig. 3 shows the example of such fitting for perch.

$$y = a \times x^b \tag{9}$$

Similar relationships were observed for roach and pike. Moreover, the mass dependences of l , w and d are practically the same for perch and roach. The difference of corresponding scale (a) and power (b) parameters is not statistically significant (t -test, $\alpha = 0.05$). This indicates that in spite of the fact that the body size of perch is somewhat bigger than that of roach (Table 3), the body proportions are similar and mass dependences of body dimensions can be described in the same way. This is confirmed by a comparison of the average fish size in the prey (Table 6). Pike is totally different in terms of both body sizes and body proportions. Table 6 presents the fitting parameters a and b of the power function for perch/roach and for pike.

3.2. Dimension-dependent species-specific dose factors

Body dimensions allow the specification of new organisms in the Erica Tool and, next, to calculate internal and external dose factors (k^I and K^E , respectively). Since perch and roach demonstrate similar body proportions, the mass dependences of dose factors were the same for these two species. Fig. 4 demonstrates an example of mass dependences estimated for dose factors of ^{137}Cs . The domains of smoothing functions are limited by observed values of fish body masses.

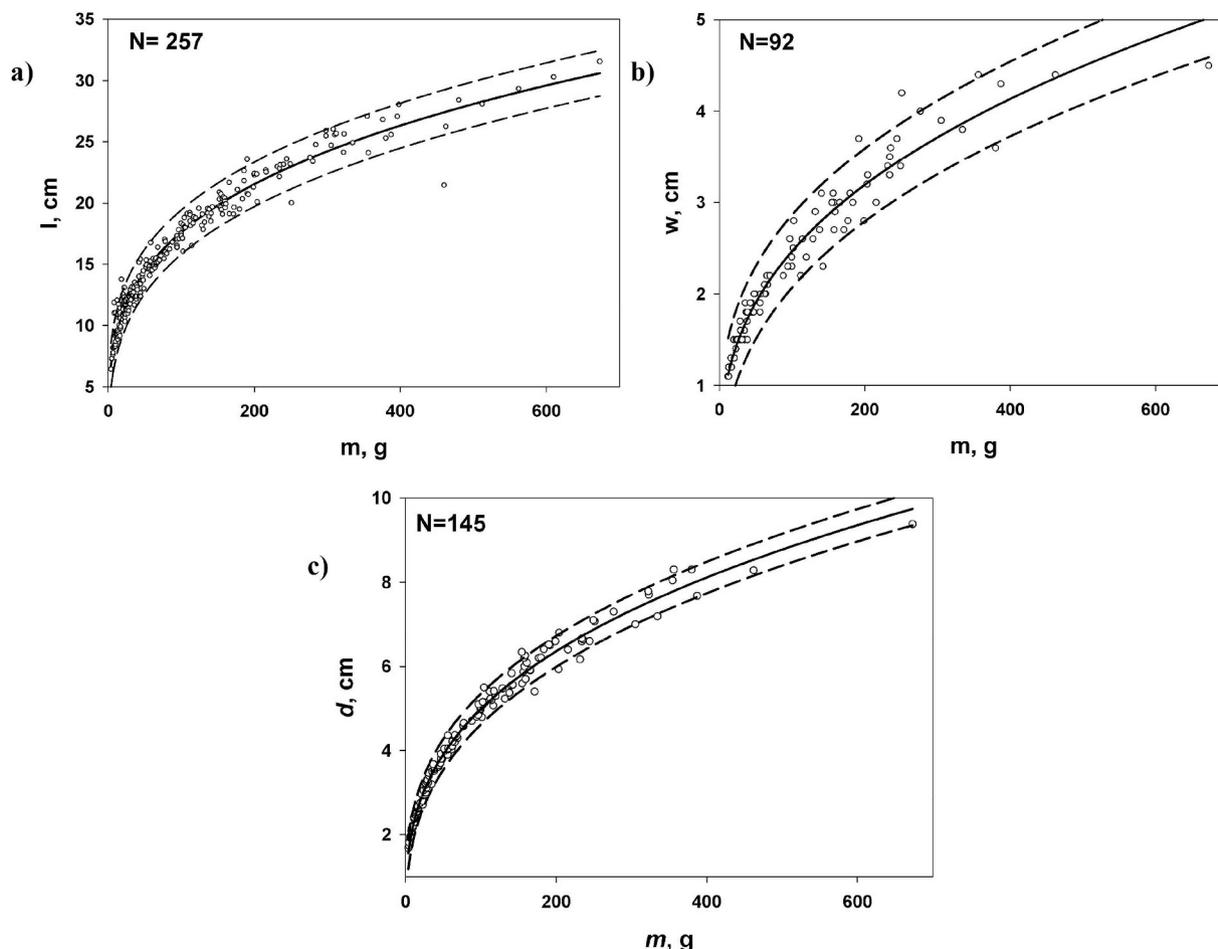


Fig. 3. The mass dependence of perch linear dimensions: (a) length – l ; (b) width – w ; (c) depth – d . Circles correspond to measurements, solid line is the fit with power function. Dashed gray lines bordered the 90% confidence interval for data scattering.

Table 6

Description of perch, roach and pike body dimensions and their mass dependences in terms of parameters of power function fits and corresponding standard errors as well as goodness-of-fit statistics (r^2 , p).

	N	Average, cm	Min-max, cm	$a \pm \text{err}$, g	$b \pm \text{err}$	r^2 , p
length (l)						
perch	257	16	6.4–32	4.42 ± 0.08	0.299 ± 0.003	0.9995, < 0.0001
roach	297	14	8–34			
pike	133	36	14–71	5.0 ± 0.2	0.328 ± 0.006	0.997, < 0.0001
width (w)						
perch	92	2.4	1.1–4.5	0.43 ± 0.02	0.375 ± 0.009	0.998, < 0.0001
roach	83	1.9	1.2–3.8			
pike	44	3.3	1.5–6.7	0.45 ± 0.04	0.342 ± 0.009	0.995, < 0.0001
depth (d)						
perch	145	4.5	1.7–9.4	0.98 ± 0.02	0.360 ± 0.004	0.989, < 0.0001
roach	173	4.3	2.4–8.5			
pike	91	5.7	2.0–12.5	0.75 ± 0.04	0.345 ± 0.009	0.998, < 0.0001

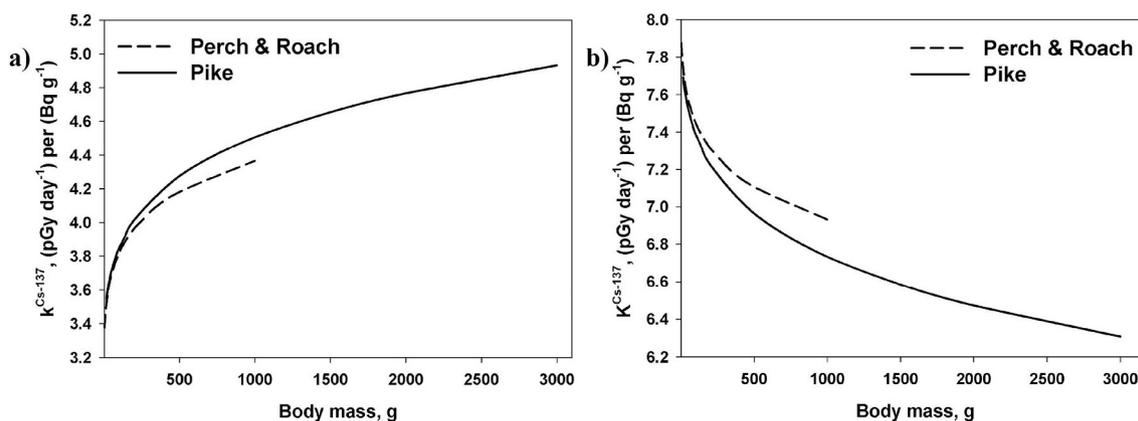


Fig. 4. Mass dependences of dose factors of internal (a) and external (b) exposure to ^{137}Cs .

As can be seen from Fig. 4a, internal doses of the fish with large masses (ages) should be higher than those of smaller (younger) individuals of the population. The inverse tendency is observed for external exposure (Fig. 4b). Nevertheless, the radionuclide-specific dose factors calculated for different species with typical masses (body sizes) taken as the average in the catch were found to be very similar: the mean difference was about 7% and 2% for external and internal exposure, respectively. As it was expected maximum influence of body mass on dose rate was obtained for external exposure to beta-emitters. Detailed description of mass dependences of dose factors is presented in Appendix A.

Therefore, it is reasonable to use the same dose factors for each of

species under study. Table 7 presents the dose factors for external and internal dose rate calculations and corresponding uncertainties which is the result of both propagation of the error of mean and the individual variability (in terms of variation coefficient).

According to Table 7, the dose factors for internal dosimetry (k^r) are more uncertain for isotopes with photon-dominated emission. In contrast, the dose factors for external dosimetry (k^r) calculated for isotopes with electron-dominated emission are very sensitive to body size and, therefore, their group-average value are quite uncertain. This is more evident for pike, for which the expected population-specific CVs of $k_{\beta\text{-emitter}}^r$ exceed 100%.

Table 7

Dose factors and corresponding uncertainties used in external and internal dose rate calculations for different species.

Radionuclide	K^r , (pGy day $^{-1}$) per (Bq g $^{-1}$)	Uncertainty of K^r , %		K^r , (pGy day) per (Bq g $^{-1}$)	Uncertainty of K^r , %	
		Perch & roach	Pike		Perch & roach	Pike
^{90}Sr	14.5	5	11	1.1	62	143
^{89}Sr	7.7	4	10	0.4	44	188
^{137}Cs	4.0	10	13	7.3	3	7
^{141}Ce	2.4	2	3	1.0	4	9
^{144}Ce	16.3	7	18	2.5	38	119
^{103}Ru	2.1	15	18	6.0	3	7
^{106}Ru	17.6	10	23	4.9	28	81
^{95}Zr	2.4	15	16	9.6	3	5
^{95}Nb	1.3	35	35	10.0	2	5
^{91}Y	7.9	3	9	0.5	47	127
^{140}Ba	14.4	10	13	37.7	3	5

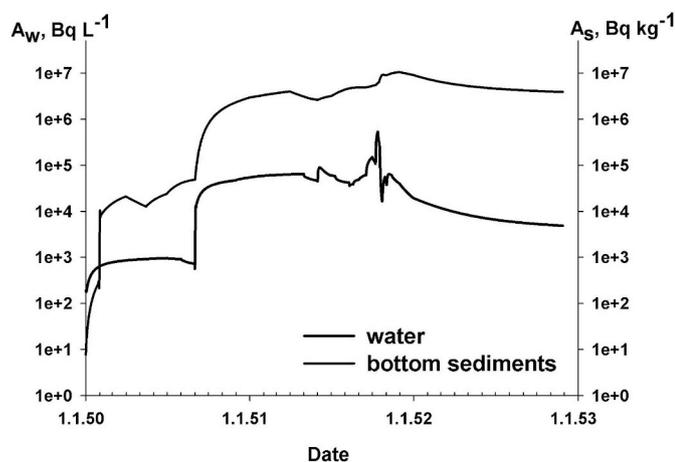


Fig. 5. Total radionuclide activity concentration in water and bottom sediments at RT1 in 1950–1952.

3.3. Contamination of the Techa River in the period of 1950–1952

Radioactive releases began in 1949. Initially low-active discharge was purified by adsorption after separation of plutonium and only after that it was released into the river. However, since January 1950, the decantate was dumped into the river without any treatment and water contamination increased (Degteva et al., 2016). Fig. 5 shows the total radionuclide activity concentration in water and bottom sediments at RT1. In the autumn of 1950, radionuclide activity in the water increased by two orders of magnitude, and then it remained approximately at the same level for about year. In addition to the routine releases, there were accidental discharges (in the autumn of 1951), which led to the local picks of radionuclide activity concentration in the water (Fig. 5). After October 28, 1951, the releases were significantly reduced; all technological waste were redirected to the Lake Karachay (Pryakhin et al., 2016). In Fig. 5 the decrease in water contamination at the end of the period under study reflects this fact. The plot of radionuclide activity concentration in bottom sediments shows smoother contour than that for water; there are no conspicuous local peaks. As expected, decrease in sediment contamination after the end of massive releases was smaller than that in water.

Table 8 presents the mean total radionuclide activity concentrations in water and bottom sediments as well as maximum and minimum values estimated for RT1, RT2 and RT3 for 3 time intervals: (1) low-active releases (LR) before September 1950; (2) massive releases (MR) in September 1950–October 1951 and (3) reduced releases (RR) after October 1951.

Radionuclide composition in the water was similar in the periods of LR and MR. When releases were reduced (RR period), the contribution of short-lived radionuclides was decreased. Mean radionuclide composition in the river water was calculated combining all daily-average data on LR and MR periods of releases; the data on RR period were

Table 8

Total radionuclide activity concentrations in water and bottom sediments at different stations in different periods of releases; means and the range of the values (in parenthesis).

Period of releases	Total radionuclide activity concentrations in					
	water, kBq L ⁻¹			bottom sediments, kBq kg ⁻¹		
	RT1	RT2	RT3	RT1	RT2	RT3
LR	0.81 (0.18–0.96)	0.37 (0.08–0.45)	0.2 (0.04–0.24)	20 (7.6–49)	0.9 (0 ^a – 2.1)	0.7 (0 ^a – 1.6)
MR	62 (0.7–535)	28 (0.3–237)	16 (0.18–131)	3340 (49–9240)	26 (2–1340)	19 (2–950)
RR	13 (4.9–64)	6.1 (2.3–29)	3.4 (1.3–16)	5960 (3900–10500)	630 (380–1300)	27 (9–880)

^a < 10⁻¹ kBq kg⁻¹.

Table 9

Radionuclide composition in the river water in two periods (low-active & massive releases in January 1950–October 1951 and reduced releases in November 1951–December 1952). The range of time and distance-dependent variation of radionuclide fraction is shown in parentheses.

Radionuclide	Radionuclide composition, %	
	LR & MR	RR
⁹⁰ Sr	13 (1–23)	37 (4–61)
⁸⁹ Sr	25 (9–45)	5 (0.3–30)
¹³⁷ Cs	2 (0.2–17)	4 (0.5–16)
¹⁴¹ Ce	0.1 (0 ^a – 1)	0 ^a (0 ^a – 0.2)
¹⁴⁴ Ce	1 (0.1–30)	4 (0.3–6)
¹⁰³ Ru	19 (4–31)	1 (0 ^a – 23)
¹⁰⁶ Ru	10 (1–21)	23 (8–29)
⁹³ Zr	7 (3–18)	7 (1–14)
⁹⁵ Nb	18 (7–40)	15 (2–40)
⁹¹ Y	5 (0.2–9)	5 (0.3–10)
¹⁴⁰ Ba	0.2 (0 ^a – 1)	0 (0 ^a – 1)

^a < 0.1%.

analyzed separately (combining the results for different distance). Table 9 presents the mean radionuclide fractions as well as the range of the daily-average values calculated for RT1, RT2 and RT3 with the model of radionuclide transport.

Radionuclide composition of the bottom sediments was different in different periods of releases; composition was also different at different locations along the stream. Fig. 6 illustrates this difference.

3.4. Dose rates

According to Table 7, gamma-emitting radionuclides (¹³⁷Cs, ^{103,106}Ru, ⁹⁵Zr, ⁹⁵Nb and ¹⁴⁰Ba) with high dose factors for external exposure are expected to be the main contributors to external dose formation. Beta emitting isotopes (^{89,90}Sr, ¹⁴⁴Ce, ¹⁰⁶Ru and ¹⁴⁰Ba) with high dose factors for internal exposure are candidates to be main contributors into internal dose rate. However, the activity concentration of ¹⁴⁰Ba in the water and bottom sediments was small (Table 9, Fig. 6) and, therefore its contribution into both external and internal dose rates was insignificant.

Activity concentration of ¹³⁷Cs in both water and bottom sediments was relatively small in the period of LR; its contribution to external dose rate became significant in the period of RR. Initially, ^{103,106}Ru, ⁹⁵Zr and ⁹⁵Nb were the main dose-forming radionuclides in terms of external exposure.

Similarly, activity concentration of ¹⁴⁴Ce in the water was low in the LR and MR periods. At that time, ^{89,90}Sr and ¹⁰⁶Ru were the main contributors into the internal dose rate. The contribution of ¹⁴⁴Ce into internal dose rate becomes significant in the RR period.

Table 10 shows daily doses (means and the range of the values) and external dose fractions at different stations in different periods of releases. Fig. 7 illustrate the dynamic of daily doses over the time period

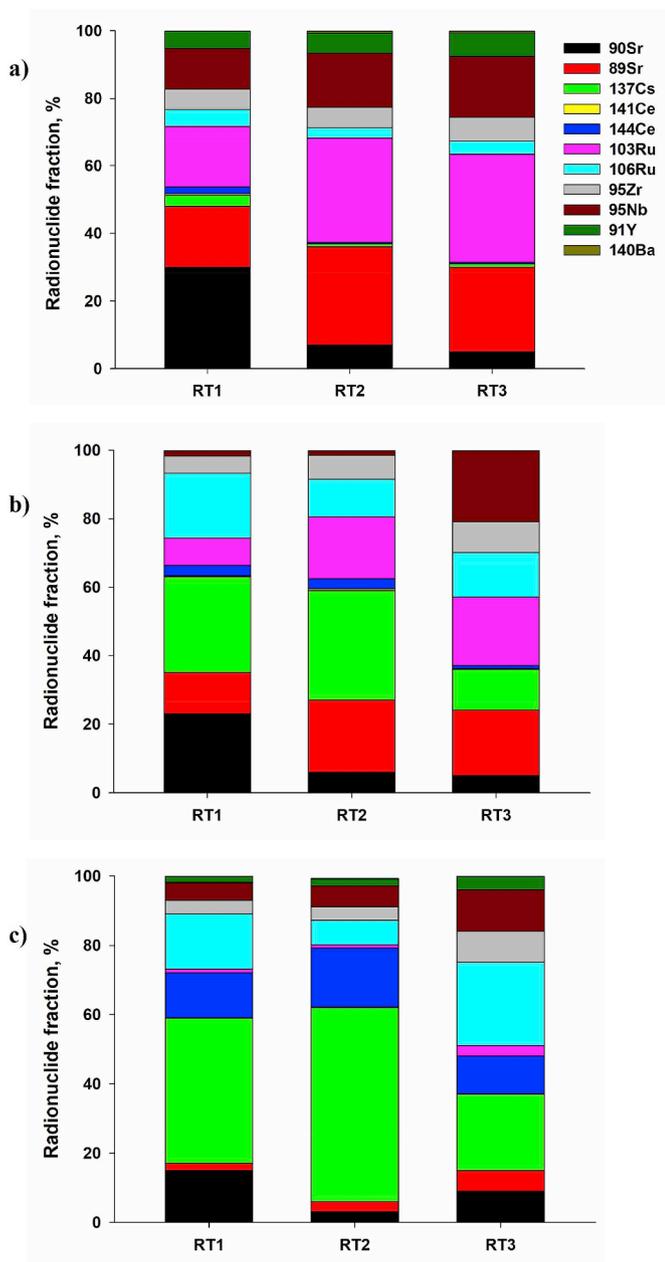


Fig. 6. Mean radionuclide percent compositions in the bottom sediments in the period of (a) LR; (b) MR and (c) RR.

Table 10

Species-specific dose rates (\dot{D}) due to combined external and internal exposure and external dose fraction (f_{ext}) calculated for different locations along the stream and for different periods of releases (means and the range of values are given in parenthesis). Dose rates exceeding the incrementing screening level for aquatic ecosystem ($\dot{D} > \dot{D}_2$) are given in italics; the calculations exceeding the UNSCEAR threshold ($\dot{D} > \dot{D}_1$) for expected effects on freshwater organisms are given in bold italics.

Period	species	RT1		RT2		RT3	
		\dot{D} , $\mu\text{Gy day}^{-1}$	f_{ext} , %	\dot{D} , $\mu\text{Gy day}^{-1}$	f_{ext} , %	\dot{D} , $\mu\text{Gy day}^{-1}$	f_{ext} , %
LR	Roach	5.7×10^2 (0.3–9 × 10 ²)	10.3	2.4×10^2 (0.1–3.4 × 10 ²)	1.5	1.3×10^2 (0.05–2 × 10 ²)	2.3
	Perch	4.55×10^2 (0.5–7.1 × 10 ²)	8.6	1.9×10^2 (0.2–2.8 × 10 ²)	2.8	1×10^2 (0.1–1.5 × 10 ²)	2.8
	Pike	3.7×10^2 (0.7–5.6 × 10 ²)	0.9	1.6×10^2 (0.3–2.5 × 10 ²)	0.9	0.9×10^2 (0.2–1.4 × 10 ²)	0.9
MR	Roach	6.6×10^4 (9 × 10 ² –1.9 × 10 ⁵)	28.6	2.2×10^4 (4 × 10 ² –7 × 10 ⁴)	5.5	1.2×10^4 (2 × 10 ² –3.9 × 10 ⁴)	7.7
	Perch	4.9×10^4 (7 × 10 ² –1.7 × 10 ⁵)	18.6	1.9×10^4 (3 × 10 ² –6.7 × 10 ⁴)	5.4	1.0×10^4 (1.5 × 10 ² –3.7 × 10 ⁴)	5.4
	Pike	3.5×10^4 (0.5 × 10 ² –1.4 × 10 ⁵)	0.6	1.6×10^4 (2 × 10 ² –6.1 × 10 ⁴)	0.9	8.6×10^3 (1 × 10 ² –3.4 × 10 ⁴)	0.9
RR	Roach	5.5×10^4 (1.8 × 10 ⁴ –1.7 × 10 ⁵)	45.4	1.6×10^4 (5 × 10 ² –5.7 × 10 ⁴)	18.2	8.6×10^3 (2 × 10 ² –3.3 × 10 ⁴)	14.3
	Perch	4.6×10^4 (1.8 × 10 ⁴ –1.4 × 10 ⁵)	44.3	1.4×10^4 (4 × 10 ² –5.3 × 10 ⁴)	10.7	7.8×10^3 (2 × 10 ² –3 × 10 ⁴)	10.7
	Pike	2.9×10^4 (8 × 10 ³ –1.1 × 10 ⁵)	0.2	1.3×10^4 (4 × 10 ² –4.6 × 10 ⁴)	0.2	7.0×10^3 (2 × 10 ² –2.6 × 10 ⁴)	0.2

under study for RT1, RT2 and RT3 in comparison with the screening level (\dot{D}_1) and UNSCEAR benchmark (\dot{D}_1). According to Table 10 and Fig. 7, exposure dose-rates were the highest for roach. Roach have higher accumulation (CR) and longer biological elimination period for Sr isotopes (Table 2), which is the reason of higher internal dose rates due to ^{90,89}Sr. Due to lower annual rate of increase in roach weight (Table 5), the impact of other contributors on internal dose rates (according to Eqn. (7)) is less than that of perch and pike. Moreover, roach spend more time at heavily contaminated bottom than other species that is why roach received higher external dose rates.

In the LR period, species- and site-specific dose rates were determined mainly by internal exposure; daily doses varied in the range of 3–4 orders of magnitude increasing over LR period (Fig. 7). Fish at RT1 and RT2 were mainly exposed to dose rates exceeding the screening level but not the UNSCEAR threshold ($\dot{D}_1 > \dot{D} > \dot{D}_2$). At RT3 calculated dose rates did not exceed \dot{D}_2 .

In the period of MR, species- and site-specific dose rates were found to be maximum. The difference between calculated daily doses reached 2–3 orders of magnitude (Fig. 7). The dose rates were dominated by internal exposure; contribution of external dose rate was significant only for roach and perch at RT1 (Table 10) in RR period. The calculated dose rates to fish of RT1 and RT2 exceed the UNSCEAR threshold, \dot{D}_1 . Roach and perch at RT3 were subject to similar dose rates of about 10⁴ $\mu\text{Gy day}^{-1}$ ($\dot{D} > \dot{D}_1$). Assessed dose rates to pike were mainly below \dot{D}_1 value, but exceeded the screening level, \dot{D}_2 .

In the period of RR, dose rates for fish of RT1 and RT2 were still very high ($> \dot{D}_1$) (Fig. 7). Dose rates at RT3 were found to be mainly in the range from \dot{D}_2 to \dot{D}_1 . The contribution of external dose rate to the exposure of fish spending part of the lifetime at the bottom increased essentially due to ¹³⁷Cs accumulation in the bottom sediments (Fig. 7).

4. Discussion

4.1. Uncertainty of dose rate estimates

Uncertainties of retrospective dosimetry are always high. In the case of the fish exposed in 1950–1952 the dose uncertainty is determined by the following factors:

- input parameters of the radionuclide transport model of the Techa River and, most of all, source term description;
- model simplifications of radionuclide transport;
- error of estimates of the radionuclide-specific water-to-organism concentration ratios and their individual variability;
- error of model for the uptake and elimination of radionuclides by fish;
- error of dose factors related to body shape simplification, approach of uniform radionuclide distribution as well as individual variability of body dimensions;

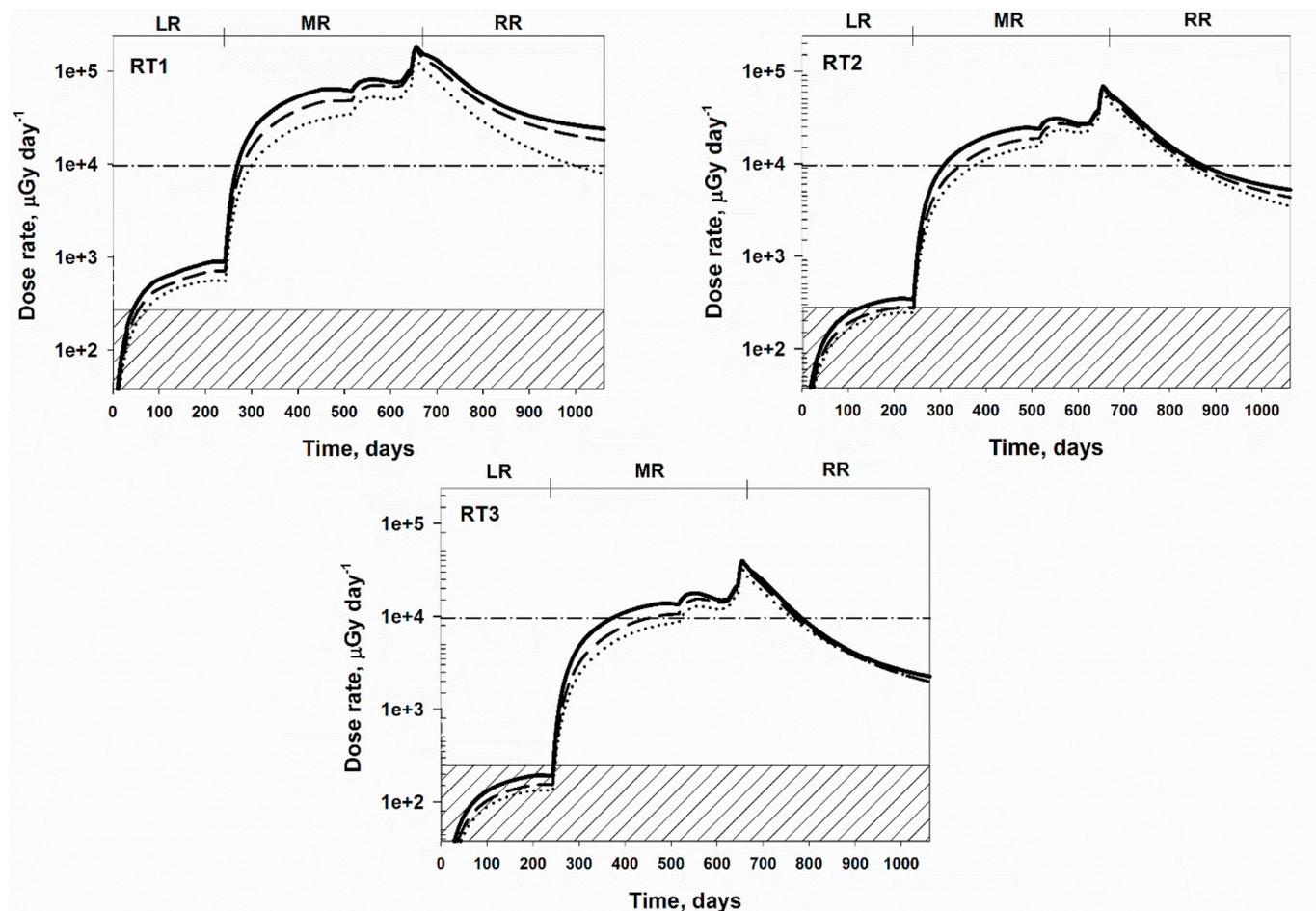


Fig. 7. Time dependence of daily doses for roach (solid bold line), perch (dashed line) and pike (dotted line) in the earlier period of radionuclide releases into the Techa River (LR – low-active releases before September 1950; MR - massive releases in September 1950–October 1951; and RR - reduced releases after October 1951) calculated for RT1, RT2 and RT3. The shaded area corresponds to doses below the screening level. Horizontal dash-dotted line corresponds to the UNSCEAR threshold.

- uncertainty due to assumptions about fish behaviour/habitat.

The source term was estimated according to the archive data on radiochemical compositions and daily discharges of separate liquid radioactive waste types (Degteva et al., 2012). Comparison of predicted discharges and results of monitoring of total daily releases in terms of total radionuclide activity concentrations shows a reasonable agreement with average relative deviation about 50%. This value can be assumed as an estimate of the uncertainty of the input parameters of the radionuclide transport model of the Techa River.

Comparison of model predictions with the results of measurements of specific total-beta activity of the river-bottom sediments made in August–October 1951 (Alexandrov et al., 1951) was undertaken by Shagina et al. (2012). The mean relative deviation of measurement results from calculations was found to be about 40%. In other words, the model simplification does not contribute significant to increase the overall uncertainty of source term and transport modeling.

These site-specific estimates can be considered as unbiased (no systematic error). However, the CRs for other radionuclides were derived from the Erica Tool. The differences between the CRs of Cs and Sr typical of the Techa River and respective values from the Erica Tool are by order of magnitude. Extrapolating this to CR of $^{141,144}\text{Ce}$, ^{95}Zr , ^{140}Ba , $^{103,106}\text{Ru}$, ^{95}Nb and ^{91}Y one can expect the induced uncertainty of corresponding internal dose rates would be also by about of order of magnitude.

External dose uncertainty induced by assumptions about the time spending at bottom sediments were preliminary estimated by variation

of this parameter based on literature data (Kottelat and Freyhof, 2007; Shvenk, 2000; Nikol'sky, 1963; 1974; Manteifel, 1965, 1974; Pavlov and Gusar, 1989; Sudakov, 1977; Chebotina et al., 1992; Imbrock et al., 1996). Variation in time spending at bottom from 0 to 30% for pike, from 30 to 70% for perch, and from 60 to 90% for roach results in 50%, 33% and 27% of relative uncertainty of external dose estimates, respectively. Migration activity influences on both external and internal dose forming. Calculation of annual dose rates assuming different scenario of exposure history allow to evaluate dose uncertainty. Some preliminary calculations were performed based on the literature data (Kottelat and Freyhof, 2007; Neuman et al., 1996; Sudakov, 1977; Nikol'sky, 1963, 1974; Orlova and Popova, 1976; Manteifel, 1965, 1974). It was assumed, that the time fraction spent at main place of exposure was 60% for perch, 80% for roach and 90% for pike. The length of pike and roach movement for the rest of the time was assumed to be not exceeding several tens of kilometers; perch is able to migrate over long distances (the length of perch movement was assumed to be 100 km). The preliminary estimates of relative uncertainties induced by the assumptions of sedentary for pike and roach do not exceed 15%; long-distance movements of perch result in about 400% uncertainty.

Ideally, modeling of radionuclide turnover results in calculations of time dependence of radionuclide activity concentrations in fish could be validated with the use of measurement results. However, there were limited number of the Techa River studies before 1952, which were focused mainly on the area near from the source of releases (Lemberg and Antipina, 1952). The first complex research of the whole river was done in June–August 1952 (Marey et al., 1952). Measurements of total

radionuclide activity in the bodies of 9 roach caught at the Muslyumovo village (close to RT2) gives the value of total specific activity equal to 340 kBq kg⁻¹. The model prediction is non-contradictory to the experimental data and equal to 430 ± 100 kBq kg⁻¹. Similar agreement was found for experimental data on 6 roach caught at Kluchevskoe village (close to RT3) and model prediction, viz., 111 kBq kg⁻¹ and 200 ± 80 kBq kg⁻¹, respectively.

The dose factors derived from the Erica Tool had been calculated under the approach of uniform radionuclide distribution in the whole fish body. No dose factors for specific source-organs (including body surfaces for description of a passive radionuclide sorption) is available to take into account inhomogeneity. Error of dose factors related to body shape simplification and the approach of radionuclide homogeneity in fish body should not play a significant role in dose estimates due to exposure to gamma emitters (Bersford et al., 2015; Gómez-Ros et al., 2008). It should be noted that ¹³⁷Cs and most short-lived radionuclides typical of the Techa River radiation situation are γ-emitters. Calculations of internal doses due to bone-seeking Strontium isotopes (under assumption of their homogeneity in fish body) may result in overestimation by about factor of 3 (Ivanov et al., 2015).

One more source of uncertainty is an external exposure to contaminated of aquatic plants of the Techa River. Detailed analysis of overall uncertainties of mean dose rates is the task to be solved. Ad interim, the results obtained can be assumed as order-of-magnitude estimates. The further direction of the dosimetry for the Techa River would be focused on cumulative dose calculation taking into account ontogenetic features of dose accumulation.

4.2. Radiation implications for health of the Techa River fish population

In fact, the current dose rates for fish inhabiting the Techa River (3–150 μGy day⁻¹) are below the screening levels of ERICA ($\dot{D}_2 = 240 \mu\text{Gy day}^{-1}$), which suggests that no radiation-induced changes should be registered in them. However, data on screening levels of radiation exposure, as well as UNSCEAR threshold levels, are based, mainly on the results of relatively short-term (by multi-generational point of view) laboratory experiments. In natural conditions, fish, in addition to chronic radiation exposure, are affected by a number of abiotic and biotic factors that could substantially modify fish reaction to radiation exposure (Tryapitsina et al., 2012; Akleyev et al., 2015). The Techa River is not an isolated ecosystem and the fish communities of 1950–1952 could be “diluted” with time by migrants. Nevertheless, one may assume that fish populations inhabiting currently this water course are the offspring, to some extent, of fish that have been affected by radiation exposure since 1949 with maximum dose rates in 1951–1952 (up to $1.9 \times 10^5 \mu\text{Gy day}^{-1}$). It is not unlikely that long-term radiation exposure of proportion of the individuals in aquatic populations of organisms - for several decades (20–30 generations in fish) - may lead to the development of physiological and/or genetic adaptations, or, vice versa, to changes in the adaptive capacity of fish.

Depending on the level of anthropogenic exposure (including radiation exposure), biological effects may occur on different levels of biological system organization: changes in ichthyofauna species composition, in sex and/or age structure of fish-populations, changes in the growth rate, changes in the number of cells and in cellular composition of the peripheral blood, increased frequency of chromosome aberrations, increased level of nuclear DNA damage.

Detail studies of species composition before the releases were provided in 1906–1908 (Kuchin, 1910) and 1941 (Revnivih, 1941) for the Techa River and connective waterbodies. Investigations performed in the earlier period of releases (in 1951–1952) (Marey et al., 1952) found no difference in species composition, which could be associated with radiation exposure. In 1956–1964 a cascade of reservoirs was constructed in the upper reaches of the Techa River, which was named the Techa Reservoirs Cascade (TRC) (Mokrov, 2003) to prevent the entry of radionuclides into the Techa River.

Recent studies of free-flowing area of the Techa River (current dose rates are $3\text{--}1.5 \times 10^2 \mu\text{Gy day}^{-1}$) (Akleyev et al., 2015) revealed no significant differences in the ichthyofauna species composition either when compared with the Miass River or when compared with the Techa River before radioactive contamination (Kuchin, 1910; Revnivih, 1941). This allow concluding the species composition has not changed substantially over time. In other words, no effects of chronic exposure of dozens fish generations to doses up to UNSCEAR threshold was found at the level of populations and communities.

Analysis of the data on fish growth rate, which was performed in June–August 1952 (Marey et al., 1952), in general, shows no radiation-induced changes for fish in spite of the dose rates 3 times exceeding $\dot{D}_1 = 9.6 \text{ mGy day}^{-1}$ at the upper course of the river (29–37 mGy day⁻¹ in RR period). However, the bottom-dwelling fish (such as gudgeon) demonstrated a slight slowdown in growth rates (Marey et al., 1952) with an increase in the radionuclide activity concentrations in water and sediments (under equal food resources).

Taking into account the fact that the accumulation of radionuclides in gudgeon hardly differs from that in roach (Marey et al., 1952), then it is safe to assume that in 1952 radiation exposure dose rate for this fish species was not less than that for roach due to a greater contribution of the external exposure dose from bottom sediments and, therefore, in 1952 it exceeded $6.6 \times 10^4 \mu\text{Gy day}^{-1}$. The obtained data on slowdown of the growth rate in bottom-dwelling fish suggest that: a) the revealed effects are caused by radiation exposure; b) in the communities of ichthyofauna, bottom-dwelling fish are the most exposed to radioactive contamination of watercourses; c) in natural fish populations, radiation-induced changes at the population level can be observed at radiation exposure levels of more than 66 mGy day⁻¹.

Analysis of the data on fish morphometry obtained in the current study (65 years after the onset of chronic radiation exposure) shows no differences in mean size as well as in growth rate of species from the Techa River and species from the Miass River (control waterbody). In other words, if there had been any severe perturbations in growth rate associated with the onset and prolonged exposure of fish to contamination in the Techa, these were not currently evident.

It is known that various stressors induce in a body a chain of general protective reactions aimed at the maintenance of homeostasis. Functioning of the body under strained adaptive systems may change the reaction to the effect of other stress-factors both towards increased adaptation to them and to decrease in adaptive capacities. The molecular, cellular and physiological changes could be the response on radiation to achieve the stabilization at organism level (with preserving adaptive potential or with decreased adaptive capacities) (Antonishkis et al., 2013).

The radiation-induced effects, viz., increase in the level of damaged nuclear DNA, registered using comet-assay, and increase in the frequency of erythrocytes with micronuclei, were detected in 2008–2015 for fish from TRC and the Techa River (Styazhkina et al., 2012; Styazhkina et al., 2015; Pryakhin et al., 2012).

Study of fish living in the Techa River (Tryapitsina et al., 2017; Akleyev et al., 2015) have shown that chronic exposure reduces the adaptive capacity of fish erythropoiesis. Spawning stress associated with hormonal changes) as well as stress due to changes in habitat conditions (keeping fish into cages in the *in situ* experiments) lead to a decrease in the adaptive capacity of erythropoiesis and ineffective erythropoiesis at a dose rate of 108 μGy day⁻¹.

In studies of the fish of the Techa River performed between 2012 and 2015, observed changes at the cellular level and at the level of the hematopoietic system did not result in changes at the level of the whole body/organism (fish growth rate), population (age and sex structure of a population), or at the level of communities (species composition of fish) at exposure dose-rates 0.9–150 μGy day⁻¹ (Akleyev et al., 2015; Tryapitsina et al., 2017). The occupation of contaminated waterbodies by fish communities for more than 65 y (20–30 generations) leads to their adaptation to chronic radiation exposure decreasing from 10⁵ to

$10^2 \mu\text{Gy day}^{-1}$. It may be thought to lower the adaptive potential to respond on other adverse factors affecting the fish.

5. Conclusions

Dose rates for fish exposed to radiation in the initial period of the Techa River contamination (1950–1952) were reconstructed as the order-of-magnitude estimates. Maximum dose rates were calculated for roach, minimum dose rates were observed for pike and perch. Fish were mainly exposed to dose rates exceeding the screening level ($D_2 = 240 \mu\text{Gy day}^{-1}$) at the upper reaches in the period of low-active releases (before September 1950). After September 1950 up to the end of 1952 the fish species along the stream were mainly exposed to dose rates exceeding the UNSCEAR threshold ($\dot{D}_1 = 9.6 \times 10^3 \text{Gy day}^{-1}$).

The radiation-induced effects on the growth rates were not observed for fish at dose rates below $6.6 \times 10^4 \mu\text{Gy day}^{-1}$ under the chronic exposure. The effects on hematopoietic system earlier reported in Tryapitsina et al. (2017) may be a consequence of radiation exposure to fish over several generations with dose rates in the initial period up to $1.9 \times 10^5 \mu\text{Gy day}^{-1}$.

The occupation of contaminated waterbodies by fish communities for more than 65 y (20–30 generations) leads to their adaptation to chronic radiation exposure decrementing from 10^5 to $10^2 \mu\text{Gy day}^{-1}$. It may be thought to lower the adaptive potential to respond on other adverse factors affecting the fish.

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