

Uptake and elimination of radiocaesium in fish and the "size effect"

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Abstract

A number of hypotheses have previously been developed concerning the rates of uptake and elimination of radiocaesium (^{137}Cs) in fish. These include the influence of potassium and other water chemical parameters on both uptake and elimination, and the effect of fish size on accumulation. In order to test these hypotheses, we have assembled a data set comprising more than 1000 measurements of radiocaesium (^{137}Cs) in predatory fish (perch, pike and brown trout) in 9 European lakes during the years after Chernobyl. These data have been analysed using simple models for uptake and excretion of ^{137}Cs in fish, showing that:

1. Fish-water concentration factors (CF) were inversely proportional to potassium $[\text{K}^+]$ concentration of the different lakes, in agreement with previous studies.
2. The uptake rate of ^{137}Cs in fish was negatively correlated with lake $[\text{K}^+]$, but excretion rate was independent of $[\text{K}^+]$.
3. Lower than expected CF values were found in one lake, Iso Valkjärvi, Finland. This is attributed to inhibition of the K^+ (and therefore ^{137}Cs) high affinity transport system in aquatic plants and fish by low pH and/or low Ca^{2+} .
4. The inclusion of fish weight as a parameter in our dynamic model significantly improves the ability of the model to fit the observed measurements of ^{137}Cs .
5. The model developed from the above hypotheses was able to fit the data from 9 different lakes to within approximately a factor of 3 of the observed values.

Keywords: Chernobyl, fish, model, aquatic, radiocaesium.

Introduction

Studies on the contamination of aquatic systems following the Chernobyl accident have shown that, in general, radiocaesium activity concentrations in water declined relatively rapidly (Vakulovsky, Nikitin, Chumichev, Katrich, Voitsekhovitch, Medinets, et al., 1994; Smith, Kudelsky, Ryabov, Haddingh, van der Perk, & Voitsekhovitch, 2000a). Bio-accumulation of radiocaesium in fish, however, resulted in activity concentrations (both in Western Europe and in the former Soviet Union) which were in many cases significantly above the maximum permissible level for consumption. In many lakes in Russia and Belarus these problems have continued to the present day and evidence suggests (Jonsson, Forseth & Ugedal, 1999; Smith, Comans, Beresford, Wright, Howard & Camplin, 2000b) that they will continue for several decades in some cases.

The level of radioactive contamination of aquatic biota is commonly defined in terms of a concentration factor (CF) where

$$CF = \frac{\text{Activity concentration per kg of fish (wet wt)}}{\text{Activity concentration per litre of water}} \text{ kg}^{-1} \quad (1)$$

Previous studies on the accumulation of radiocaesium in fish have focused on the prediction of *CF* (sometimes termed the bioaccumulation factor, *BAF*, or aggregated concentration factor, *ACF*). Previously developed hypotheses (Blaylock 1982, Rowan & Rasmussen, 1994; Smith, Kudelsky, Ryabov & Haddingh, 2000c) predict the water-fish *CF* using an inverse relationship between the *CF* and the potassium concentration of the surrounding water. Studies also predict activity concentrations of radiocaesium in predatory fish which are approximately two times higher than those in non-predatory fish (Rowan & Rasmussen, 1994; Smith et al., 2000c). These hypotheses have been quantified and tested against empirical data from a large number of rivers and lakes (Blaylock 1982, Rowan & Rasmussen, 1994; Smith et al., 2000c) and are widely used predictive tools.

The equilibrium *CF* modelling approach is appropriate for cases in which the radionuclide activity concentration in fish can be assumed to be in equilibrium with that in water, for example several years after radionuclide fallout, or for long time scale continuous releases of radionuclides. Shortly after radioactive contamination of an aquatic system, or where activity concentrations in water are changing relatively rapidly, a dynamic modelling approach may be more appropriate. Dynamic models for radiocaesium accumulation in freshwater fish have been developed using the results of controlled laboratory experiments (e.g. Garnier-Laplace, Vray & Baudin, 1997).

The processes which determine the accumulation of radiocaesium in fish are complex, and lead to wide differences in contamination levels according to water chemistry, fish type, size and feeding patterns (e.g. Kolehmainen, Häsänen & Miettinen, 1967; Haddingh, van Aerssen, Ryabov, Koulikov & Belova, 1997). The most important pathway of radiocaesium into fish is via intake of food (e.g. Elliott, Hilton, Rigg, Tullett, Swift & Leonard, 1992), and the high concentration factors observed are a result of accumulation of radiocaesium through the food chain. The $[K^+]$ concentration of lake or river water has been shown to influence the rates of accumulation (e.g. Kolehmainen, 1967) and excretion (Cocchio, Rodgers & Beamish, 1995) of ^{137}Cs in fish. The "size effect" of radiocaesium accumulation in fish results in an increasing contamination (per unit weight of fish) with increasing fish size (Elliott et al., 1992; Koulikov & Ryabov, 1992; Haddingh et al., 1997). These hypotheses concerning the dynamics of radiocaesium

accumulation, and the "size effect" are much less well quantified and tested than those concerning the prediction of equilibrium concentration factor.

There have been many studies of the mechanisms of radiocaesium transfer to fish: the purpose of this paper is not to develop new "concepts" and models, but to quantify and test existing hypotheses. We have used a large set of empirical data derived from nine European lakes to quantify and test the following hypotheses:

1. Fish-water concentration factors (*CF*) are inversely proportional to potassium [K^+] concentration of the different lakes, in agreement with previous studies (Blaylock 1982, Rowan & Rasmussen, 1994; Smith et al., 2000c).
2. The uptake rate of ^{137}Cs in fish is negatively correlated with lake [K^+] (e.g. Kolehmainen, 1967).
3. The excretion rate of ^{137}Cs in fish is positively correlated with lake [K^+] (Cocchio et al., 1995).
4. Lake pH or Ca^{2+} can influence accumulation of ^{137}Cs in fish.
5. The accumulation of ^{137}Cs in fish increases with fish size according to a power law relationship (Elliott et al., 1992; Koulikov & Ryabov, 1992; Hadderingh et al., 1997).

The above hypotheses were tested by incorporating them in a series of models which were used to fit an empirical data set. Those hypotheses which, when quantified in a model, succeeded in significantly improving the fit to empirical data were accepted.

Methods

Summary of symbols with units as used

<i>A, B</i>	Empirically determined fitting constants for lake water model	Bq l ⁻¹
<i>C_f</i>	Activity concentration of ^{137}Cs in fish	Bq kg ⁻¹
<i>CF</i>	Fish-water concentration factor	l kg ⁻¹
<i>C_w</i>	Activity concentration of ^{137}Cs in lake water	Bq l ⁻¹
<i>k₁, k₂</i>	Empirically determined fitting parameters for lake water model	d ⁻¹
<i>k_f</i>	Rate constant of ^{137}Cs uptake in fish	l kg ⁻¹ d ⁻¹
<i>k_b</i>	Rate constant of ^{137}Cs excretion from fish	d ⁻¹
<i>n</i>	Fitting parameter for relationship between fish ^{137}Cs and weight	[]
<i>w</i>	Wet weight of fish	kg
<i>x</i>	Fitting parameter for model relating <i>CF</i> to [K^+]	[]
<i>y</i>	Fitting parameter for model relating <i>CF</i> to [K^+]	mg kg ⁻¹

Modelling

Previous studies (e.g. Saxén, 1994; Smith, Leonard, Hilton & Appleby, 1997) have modelled the change in radiocaesium concentration in lakewater after Chernobyl using a double exponential decay model:

$$C_w = A \exp(-k_1 t) + B \exp(-k_2 t) \quad (2)$$

where C_w (Bq l⁻¹) is the ^{137}Cs activity in the lakewater, t is time (days) since the Chernobyl accident, and A (Bq l⁻¹), B (Bq l⁻¹), k_1 (d⁻¹), k_2 (d⁻¹) are constants. The first component of equation 2 represents the removal of the initial deposit of activity to the lake via losses to bottom sediments

and outflow. The rate of this initial removal is represented by the parameter k_1 (d^{-1}). The second component is a result of long-term secondary inputs of activity to the lakewater from the catchment and (usually less importantly) bottom sediments (Smith & Comans, 1996; Smith et al., 1997). For all of the lakes we studied, equation 2 describes the temporal change in ^{137}Cs activity during the years after Chernobyl. Equation 2 was curve-fitted to the measured ^{137}Cs activities in order to derive values of the constants for each lake.

The activity concentration of radiocaesium in fish, C_f (Bq kg^{-1}) may be modelled by a simple “two-box” model describing uptake from the water C_w (Bq l^{-1}) and release from the fish (Figure 1):

$$\frac{dC_f}{dt} = k_f C_w - k_b C_f \quad (3)$$

where k_f ($1 \text{ kg}^{-1} \text{ d}^{-1}$) is the rate constant describing transfers of ^{137}Cs to fish through its food and k_b (d^{-1}) is the backward rate constant describing excretion of radioactivity from the fish. The ratio of these rate constants gives the equilibrium concentration factor, CF (1 kg^{-1}), of ^{137}Cs in fish relative to water:

$$\frac{k_f}{k_b} = \frac{C_f}{C_w} (\text{at equilibrium}) = CF \quad (4)$$

Using equations 2 & 3 we can write:

$$\frac{dC_f}{dt} = k_f [A \exp(-k_1 t) + B \exp(-k_2 t)] - k_b C_f \quad (5)$$

which, for the case $C_f(t=0) = 0$, has solution

$$C_f = \frac{k_f A}{k_b - k_1} (e^{-k_1 t} - e^{-k_b t}) + \frac{k_f B}{k_b - k_2} (e^{-k_2 t} - e^{-k_b t}) \quad (6)$$

This equation will be called Model 1. Parameters A (Bq l^{-1}), B (Bq l^{-1}), k_1 (d^{-1}), k_2 (d^{-1}) describe the time change in activity concentration of radiocaesium in the water. k_f ($1 \text{ kg}^{-1} \text{ d}^{-1}$) and k_b (d^{-1}) are the rate constants between the fish and water. Figure 2 illustrates the general characteristics of this model. During the first 1 - 2 years after the Chernobyl fallout, ^{137}Cs activity concentrations in fish increased. After fish activity concentrations had peaked, they slowly declined over a period of years. This slow decline represents activity concentrations in fish which have reached approximate equilibrium with respect to the slowly declining ^{137}Cs activity concentrations in the lakewater.

It has previously been shown (Kolehmainen, 1967; Rowan & Rasmussen, 1994; Smith et al., 2000c) that the bioaccumulation of ^{137}Cs in fish is inversely related to the K^+ concentration of the surrounding water. To study the relationship between ^{137}Cs and K^+ , another two models were tested. Both of these models are adaptations of Model 1, which assumes that k_f and k_b are both constant, i.e. the accumulation and release of ^{137}Cs in fish is independent of K^+ concentrations in the surrounding water. These two additional models assume that the kinetic rate constants are

related to the potassium concentration (c.f. equation 4) using an inverse relationship between CF and potassium:

$$CF = \frac{k_f}{k_b} = \frac{y}{[K^+]^x} \quad (7)$$

where y is a fitted constant and $[K^+]$ is measured in mg l^{-1} .

Model 2 assesses the effect of modifying the rate of uptake of ^{137}Cs as a function of $[K^+]$ (ie assuming that $k_f \propto [K^+]^x$ and k_b is constant). From equations 6 & 7:

$$C_f = \frac{\left(\frac{yk_b A}{[K^+]^x}\right)}{(k_b - k_1)} \left(e^{-k_1 t} - e^{-k_b t}\right) + \frac{\left(\frac{yk_b B}{[K^+]^x}\right)}{(k_b - k_2)} \left(e^{-k_2 t} - e^{-k_b t}\right) \quad (8)$$

Model 3 assesses the effect of modifying the removal rate constant (assuming that $k_b \propto [K^+]^x$ and k_f is constant). Again from equations 6 & 7:

$$C_f = \frac{k_f A}{\left(\frac{k_f [K^+]^x}{y}\right) - k_1} \left(e^{-k_1 t} - e^{-\left(\frac{k_f [K^+]^x}{y}\right) t}\right) + \frac{k_f B}{\left(\frac{k_f [K^+]^x}{y}\right) - k_2} \left(e^{-k_2 t} - e^{-\left(\frac{k_f [K^+]^x}{y}\right) t}\right) \quad (9)$$

All three models were fitted using the non-linear least-squares regression procedure, PROC NLIN, in SAS (SAS Institute Inc.). This is an optimisation procedure (i.e. minimises the sum of squares of observed minus estimated values) to find the best parameters to fit the observed data.

Data sources

Analyses were performed on 3 species of fish: brown trout (*Salmo trutta*), perch (*Perca fluviatilis*), and pike (*Esox lucius*) in 9 different lakes across Europe. The data comprised measurements of samples during the period 1986-97 however the range and number of measurements varies for each lake (see Table 1). Data on the ^{137}Cs activity concentration in fish and water for lakes Crummock, Devoke, Ennerdale, Loweswater and Windermere (in the English Lake District, Cumbria, UK) were obtained from the UK Ministry of Agriculture, Fisheries and Food, Directorate of Fisheries Research (MAFF/DFR) post-Chernobyl monitoring programme (Camplin, Leonard, Tipple & Duckett, 1989 and pers. commun.). The water and fish data for lakes IJsselmeer (Netherlands), Iso Valkjärvi (Finland), Hillesjön (Sweden) were obtained from data collected in the IAEA Validation of Model Predictions (VAMP) project (IAEA, 2000). Water data from the Kiev Reservoir was compiled from measurements collected by the Ukrainian Hydrometeorological Institute and fish data was compiled from measurements reported in Haddingh et al. (1997), and unpublished data of I.N. Ryabov (Severtsov Institute, Moscow).

In addition to the data summarised in Table 1, we have used measurements of fish-water CF values in a number of lakes in Belarus, Russia and Ukraine. CF measurements have been used from lakes Tyumenskoye (perch); Svyatoye, Kostyukovichy (perch, pike); Svyatskoye (perch, pike); Kolpino (perch); Svyatoye, Chechersk (perch, pike); Stoyacheye, Petrovshchina (perch);

Kozhanovskoe (perch, pike) and the Chernobyl Cooling Pond (perch). Details of these lakes can be found in Smith et al. (2000c), in which the *CF* data for perch were presented. The additional data for pike are previously unpublished.

It can be seen from Table 1 that the number of measurements varies significantly for the different lakes: for example, 141 measurements of pike were available for the Kiev Reservoir whereas only 13-34 were available for the other lakes in which pike were caught. This could, potentially, cause a bias in our model fits when we fit general models to measurements from a number of different lakes simultaneously. The effect of this potential bias was checked by randomly selecting a smaller number of measurements from the exceptionally large data sets and re-fitting the models to the new data set. No significant bias was found. In addition, the Kiev Reservoir data spanned a much longer sampling period (up to 1997) than the other data sets (up to 1993), which could potentially introduce bias. We checked for this by fitting the (final) models against a data set in which data from the Kiev Reservoir from 1992 onwards was discarded. Again, no significant effect on model parameters was found.

Values for the parameters A , B , k_1 , k_2 for the Cumbrian lakes only, were taken from a previous publication (Smith et al., 1997). For the other lakes, they were estimated using SAS statistical software (SAS Institute Inc.) by modelling the water data using equation 2. All fish data is presented in Bq kg^{-1} wet weight and the water activity concentration was measured in Bq l^{-1} .

Results and discussion

The estimates for the water parameters A , B , k_1 and k_2 are summarised in Table 2. Using these parameter values, Model 1 was fitted to the measurements (Figure 2 shows an example of the model fit) to obtain the parameter values for k_b and k_f for each species of fish in each lake separately. It was therefore possible to determine the uptake and release rates of ^{137}Cs in each species of fish independently of differences, and changes in, water ^{137}Cs concentration in the different lakes.

Table 3 shows the coefficient of determination (R^2) values for the fits of the models to the fish data. Notice that the fish data for Lake Hillesjön is divided into two groups: group A denotes small fish (<20cm) and group B denotes fish larger than 20cm.

Parameter values for k_f and k_b for each species of fish in each lake are also shown in Table 3. The results in Table 3 indicate that Model 1 successfully predicted the ^{137}Cs activity concentrations in each individual lake for each of the species of fish. The low R^2 values are simply a result of the large variation in the fish ^{137}Cs concentrations in some lakes. The corresponding parameters k_f and k_b , are generally estimated very accurately with the exception of three cases (pike -Windermere, perch - Crummock and trout -Windermere) where the estimates had large errors (Table 3). These errors can be attributed to the fact that in these cases measurements were only available for a short period (10 to 12 months) after the fallout (Table 1).

Half lives of the release rate constant can be calculated from our data as $T_{1/2} = \ln 2/k_b$. Mean estimates of these (Table 3) are: 222 days for trout, 307 days for perch and 576 days for pike.

Relationship between Concentration Factor and water chemistry.

Previous studies on both weapons-test (Kolehmainen, 1967; Blaylock, 1982; Rowan & Rasmussen, 1994) and Chernobyl (Smith et al., 2000c) derived ^{137}Cs have shown that the fish-water CF is inversely proportional to the concentration of potassium in the lakewater. To further test this hypothesis, we have estimated the CF for the measurements we have assembled using the ratio of the fitted k_f and k_b values (equation 4). Figure 3 shows plots of our estimated CF values against $[\text{K}^+]$. For perch (Figure 3a), the results of a previous study (Smith et al., 2000c) of CF values in lakes in Belarus, Russia and Ukraine are shown alongside our estimates, indicating good agreement between the two studies. For pike (Figure 3b), previously unpublished data from lakes in Belarus, Russia and Ukraine are shown alongside the estimates from the dynamic model fits (i.e. from the ratio k_f/k_b using data in Table 3), again indicating good agreement between the different European lakes. Our data for trout are only from lakes in the English Lake District and cover only a limited range in $[\text{K}^+]$ concentrations (Figure 3c), but the CF values are in general agreement with those for perch and pike for lakes with low $[\text{K}^+]$.

The measurements for both perch and pike show that CF values in fish from lake Iso Valkjärvi, Finland (shown as an outlier in Figure 3 a,b) are significantly lower than would be expected for a lake of low (0.4 mg l^{-1}) potassium concentration. We hypothesise that this is due to the low pH of this lake. As shown in Figure 4, multiple regression of our CF measurements against both $[\text{K}^+]$ and pH showed a significant improvement over the regression against $[\text{K}^+]$ alone. For perch, R^2 improved from 0.68 ($\log(CF)$ vs. $\log[\text{K}^+]$) to 0.79 ($\log(CF)$ vs. $\log[\text{K}^+]$, pH) and for pike the improvement in R^2 was from 0.70 to 0.83. Notice that regression of logged values was carried out (pH is already a log scale), as appropriate for the large range (2 orders of magnitude) of variation in CF values.

Previous research has shown that the accumulation of ^{137}Cs by the fry of atlantic salmon and brown trout from water decreases with decreasing pH (Morgan, Tytler & Bell, 1993). In the fish species studied by Morgan and coworkers this effect was related to a decreased uptake while the elimination rates were not significantly affected.

A possible explanation for this effect is the inhibition of Cs^+ uptake by the increased H^+ concentration due to protonation of the negative charges within the Cs^+ transporting K^+ channels in the gill. However, the direct uptake of ^{137}Cs via water is less important than uptake via food. The lower than expected ^{137}Cs levels in the fish from the low K^+ and acid lake Iso Valkjärvi may therefore be the effect of pH on the activity of K^+ transporters in the primary producers at the base of the food-chain. At low $[\text{K}^+]$, plants close “passive” channels for $[\text{K}^+]$ (and therefore Cs) transport in order to avoid K^+ leakage to the environment. $[\text{K}^+]$ is instead taken up by a different “high affinity” transport system (Maathuis, Verlin, Smith, Sanders, Fernández & Walker, 1996). This high affinity transport system is not affected by pH (Rodríguez-Navarro, Blatt & Slayman, 1986), but the “passive” channels open at low pH, releasing Cs^+ to the environment (Blatt, 1992; Müller Röber, Ellenberg, Provart, Willmitzer, Busch, Becker et al., 1995). The net effect is a reduced CF of radiocaesium in aquatic plants at low pH.

A further possible mechanism for the observed reduced CF in Iso Valkjärvi is the low concentration of dissolved calcium in the lake. Calcium has also been shown to be a regulator of the potassium channel activity in aquatic plants, maximum concentration factors being obtained when potassium concentrations are low and calcium concentrations are high (J.A. Fernandez, unpubl. results). When calcium concentrations are low, the concentration factors are lower. This

calcium regulation would only be expected to operate under conditions of $[K^+]$ deficiency. A high concentration of potassium means that the high affinity transport systems (which can be influenced by Ca) are not required since the aquatic plants and fish have enough potassium to function normally. The effect of low $[Ca^{2+}]$ and/or low pH may therefore only influence fish-water CF in low $[K^+]$ scenarios.

Multiple regression of our CF measurements against both $[K^+]$ and $[Ca^{2+}]$ gave an improvement over the $CF-[K^+]$ model: $R^2 = 0.71$ for perch and 0.77 for pike, though the improvement was lower than for multiple regression with $[K^+]$ and pH. Multiple regression of our CF measurements against pH, $[K^+]$ and $[Ca^{2+}]$ gave little improvement in R^2 over the $CF-[K^+]$, pH model: $R^2 = 0.83$ for perch and 0.83 for pike (c.f. $R^2 = 0.77$ and 0.83 respectively for the pH, $[K^+]$ model). Because of the strong co-correlation of pH and $[Ca^{2+}]$ in the lakes we studied ($R^2 = 0.72$), we cannot unambiguously determine which of these water chemical variables is influencing the CF : possibly both are. The measurements, however, support the hypothesis that conditions of low pH and low $[Ca^{2+}]$, as seen in Lake Iso Valkjärvi, reduce the accumulation of ^{137}Cs in fish. We hypothesize that this mechanism may only operate in conditions of low $[K^+]$ concentration.

Influence of $[K^+]$ on uptake and excretion rates

Figure 5 shows correlations of the fitted uptake and excretion parameters against $[K^+]$ concentration in the lakewaters for all lakes excluding Iso Valkjärvi (which was classified as an “outlier”, see above). The uptake rate constant was significantly ($p < 0.005$) inversely correlated with $[K^+]$, whereas the excretion rate constant was not correlated with $[K^+]$. This observation is seemingly in contradiction to experimental studies of ^{137}Cs elimination in fish. Cocchio and coworkers (1995) argued, from studies on juvenile rainbow trout, that their experimental results “demonstrate that $[K^+]$ would influence accumulation of ^{137}Cs by affecting elimination and not assimilation”. Their results showed a factor of approximately 2 increase in ^{137}Cs elimination rate when water $[K^+]$ was increased from 0.39 mg l^{-1} to 133 mg l^{-1} . This latter value, however is around a factor of 10 higher than $[K^+]$ concentrations in most freshwaters and in the lakes we have studied. Elimination rates did not vary significantly among fish maintained at more typical K^+ concentrations of $0.39 - 12 \text{ mg l}^{-1}$. In addition, in these experiments (Cocchio et al., 1995), fish were fed with food labelled with a constant activity concentration of ^{137}Cs , so accumulation rates would be unlikely to show the same effect of potassium as observed in the environment. Since the majority of ^{137}Cs in fish is via food intake (Elliott et al. 1992), we hypothesise that higher K^+ concentrations reduce ^{137}Cs activity concentrations in food, and hence reduce rates of uptake by fish.

Generalised dynamic model.

To obtain a generalised model for the uptake of ^{137}Cs in fish in different lakes, the data for each lake was compiled together into one set for each species of fish. Models 1, 2 and 3 were then fitted to these three sets of data. The results of these fits are shown in Table 4 as R^2 values. Notice that two extra fits were carried out on perch and pike: with and without the inclusion of data from lake Iso Valkjärvi (an outlier, see above). Overall, Models 2 and 3 were an improvement over Model 1. This was to be expected since the concentration of potassium in the water has a significant effect on ^{137}Cs accumulation (Figure 3). The statistical package (SAS) was unable to estimate parameter values for Model 3 in some cases which suggests that the model was over-parameterised with respect to the data set. Model 2 provided the best results, in agreement with the finding above (Figure 5) that the influence of potassium on the forward rate constant is much greater than on the

backward rate constant. As before, the parameter estimates for k_f and k_b for all three models and species of fish were statistically significant at the 1% level. We also attempted to fit the data using a model which allowed for variation in both uptake, k_f , and release, k_b rate constants in fish. This model, however, failed to converge suggesting that it was over parameterised with respect to the data set. We conclude that Model 2 (equation 7), in which the ^{137}Cs uptake rate was inversely proportional to $[\text{K}^+]$, was best able to fit the empirical data we have assembled.

The influence of fish size on ^{137}Cs accumulation.

The “size effect” of radiocaesium accumulation in fish tends to result in an increasing contamination (per unit weight of fish) with increasing fish size (Elliott et al., 1992; Hadderingh et al., 1997). In the Kiev Reservoir and an old channel of the river Pripyat, near Chernobyl, Hadderingh et al. (1997) found that perch and pike (predatory species) showed a size effect in ^{137}Cs accumulation, whereas non-predatory species did not. We have modified Model 2 to account for variation in CF as a function of wet weight of fish and tested whether the new model (termed Model 4) gives an improved fit to the empirical data set. Elliott et al. (1992) showed a logarithmic relationship between fish ^{137}Cs accumulation and wet weight. We have therefore chosen to modify Model 2 (equation 8) using a power law relationship between ^{137}Cs in fish, C_f , and weight, w (kg):

$$C_f = \left[\frac{\left(\frac{yk_b A}{[\text{K}^+]} \right)}{(k_b - k_1)} (e^{-k_1 t} - e^{-k_b t}) + \frac{\left(\frac{yk_b B}{[\text{K}^+]} \right)}{(k_b - k_2)} (e^{-k_2 t} - e^{-k_b t}) \right] w^n \quad (10)$$

where n is an arbitrary constant. Note that we have dropped the coefficient x from Model 2 since we assume that its value is equal to 1 (i.e. $CF \propto 1/[\text{K}^+]$, see Figure 3). Table 5 shows model fitted parameters and coefficients of determination (R^2) for the models with (Model 4) and without (Model 2, with $x = 1$) accounting for the effects of fish weight.

The fits to the measurements of both perch and trout showed a clear improvement in R^2 when the effect of fish weight was incorporated in the model. The fits of pike data showed no such improvement. This may be because for pike we only have measurements of mature (> 0.1 kg) fish. We believe that, for perch, it is possible that much of the “size effect” can be explained by differences in feeding habits between small and large fish. As perch mature, their diet changes from eating mainly invertebrates to being primarily piscivorous, though the size at which they do this varies depending on available food, competition and other environmental parameters. In Windermere, perch of size less than 13cm (approx. < 0.025 kg) were observed to feed mainly on plankton, in the size class 13-18cm (approx. $0.025 - 0.1$ kg) they fed mainly on benthic invertebrates and perch longer than 18cm (approx. > 0.1 kg) fed mainly on small fish (Le Cren, 1958).

We grouped the measurements of perch into “small” (here defined as < 0.1 kg) and “large” (here defined as > 0.1 kg) size categories, and applied Model 2 (i.e. without weight as an input parameter) to each of the size categories separately. As expected, fitted parameter values were significantly different for the two size groups, showing much greater ^{137}Cs accumulation in the large fish. Applying Model 2 to the two groups separately gave an overall R^2 of 0.67, compared to $R^2 = 0.69$ for Model 4 (i.e. including weight). The similar R^2 values indicate that the size effect can largely be explained by differences in ^{137}Cs accumulation between large and small fish, and is

not necessarily a continuous increase in ^{137}Cs activity concentration with increasing fish size. This change is possibly as a result of changes in feeding habits (from plankton and invertebrates to small fish), however fish age at the time of peak water contamination may also be important.

Figure 6 shows plots of the measured ^{137}Cs activity concentrations in fish against those “predicted” by the best fit model for each species (Model 4 for perch and trout, Model 2 for pike). Note that we here use the word “prediction” in the sense that the models are estimating measured values. This does not represent a true prediction since the models are derived from the data set. Model “predictions” are generally within a factor of 3 of measured values. A significant fraction of this remaining unexplained variation in activity concentrations is expected to be variation in behaviour (for example, feeding habits, growth rates) of individual fish. For example, 71 measurements of ^{137}Cs in perch in Devoke Water for a given year (1987) and corrected for differences in fish weight still showed a variation of a factor of 2 either side of the mean value. This could not be attributed to seasonal variation since no time-dependence was observed in the corrected values for the given year.

Over and above this statistical variation, the model for brown trout (Figure 6c) significantly overestimates ^{137}Cs activity concentrations in some fish (*i.e.* there are a number of data points outside the lines representing a factor of 3 error in the model “predictions”). This, we believe, is probably due to fish stocking in these lakes. In some cases stocked fish were identified (Camplin et al., 1989) and could be removed from the data set, but this was not always possible. We believe that the low measured values (Figure 6c) were recently stocked fish since single, anomalously low, values were often observed in groups of fish caught in the same lake at the same time.

Conclusions

We have used a large set of empirical data derived from nine European lakes to quantify and test a number of hypotheses concerning uptake and excretion of ^{137}Cs in fish. The results of these tests were as follows:

1. The fish-water concentration factor (CF) of ^{137}Cs was confirmed to be inversely proportional to potassium $[\text{K}^+]$ concentration of the different lakes, in agreement with previous studies (Blaylock 1982, Rowan & Rasmussen, 1994; Smith et al., 2000c).
2. The uptake rate of ^{137}Cs in fish was negatively correlated with lake $[\text{K}^+]$, as hypothesised by Kolehmainen et al. (1967).
3. The excretion rate of ^{137}Cs in fish was not found to be positively correlated with lake $[\text{K}^+]$. This is in contradiction to the hypothesis of Cocchio et al. (1995), probably because this effect was observed by these workers only at very high $[\text{K}^+]$ (133 mg l^{-1}) which is not typical of most freshwaters.
4. Lake pH or Ca^{2+} may reduce accumulation of ^{137}Cs in fish. We were unable to accurately quantify this hypothesis. However, we predict that our models for fish-water CF may overestimate ^{137}Cs accumulation in conditions of low pH and low $[\text{Ca}^+]$
5. The results confirm the hypothesis that accumulation of ^{137}Cs in fish increases with fish size according to a power law relationship (Elliott et al., 1992; Koulikov & Ryabov, 1992; Haddington et al., 1997). We have quantified this hypothesis and shown that for predatory fish, change in feeding patterns with fish size may strongly influence this “size effect”.

The models developed from the above hypotheses were able to fit data on three different fish species from 9 different lakes to within approximately a factor of 3 of the observed values.

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Figure 3. Graphs of fish-water concentration factor of (a) perch; (b) pike; (c) trout, vs. potassium concentration in the water of different lakes. (●) Estimates from uptake and excretion rates (equation 3) calculated in this study; (○) Estimate from L. Iso Valkjärvi, considered to be an outlier; (□) Estimated from measurements in additional lakes in Belarus, Russia and Ukraine using equation 1: for perch these data were taken from Smith *et al.* (2000c). Solid lines show the inverse relationship between CF and $[\text{K}^+]$. Dotted lines show factor of 3 error above and below the best estimate.

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Figure 6. Correlation of fitted models with measured ^{137}Cs in (a) perch model (Model 4, *i.e.* including weight), $R^2 = 0.69$; (b) pike (Model 2, *i.e.* not including weight), $R^2 = 0.51$ and (c) trout (Model 4, *i.e.* including weight), $R^2 = 0.60$. Parameters are given in Table 5. Dotted lines show factor of three difference either side of model predicted values.

Table 1 Summary of all fish data used in model fits.

Lake	Species of Fish	Sampling start	Sampling end	Time Period (days)	Weight data?*	Number of obs.
Crummock, UK	Trout	18/07/86	24/03/88	615	✓	26
Crummock, UK	Perch	18/07/86	21/06/87	338	✓	14
Devoke, UK	Trout	05/09/86	28/05/93	2457	✓	228
Devoke, UK	Perch	05/09/86	13/09/93	2565	✓	104
Ennerdale, UK	Trout	19/08/86	16/06/93	2493	✓	303
Loweswater, UK	Trout	11/05/86	11/09/91	1949	✓	193
Loweswater, UK	Perch	18/07/86	11/09/91	1881	✓	76
Loweswater, UK	Pike	25/09/86	11/09/91	1812	✓	34
Windermere, UK	Trout	15/05/86	01/05/87	351	✓	32
Windermere, UK	Perch	15/05/86	01/05/87	351	✓	12
Windermere, UK	Pike	17/07/86	12/05/87	299	✓	27
Hillesjön, Sweden	Small Perch	15/01/87	15/10/91	1734	✗	11
Hillesjön, Sweden	Large Perch	15/07/86	15/10/91	1918	✗	13
Hillesjön, Sweden	Pike	15/06/86	15/10/91	1948	✗	14
IJsselmeer, Netherlands	Perch	15/06/86	15/09/90	1553	✓	15
Iso Valkjärvi, Finland	Perch	15/07/87	15/10/91	1553	✗	19
Iso Valkjärvi, Finland	Pike	15/07/87	15/12/91	1614	✗	13
Kiev Res., Ukraine	Perch	01/08/90	09/09/97	2596	✓	279
Kiev Res., Ukraine	Pike	25/07/87	02/06/97	3600	✓	141

* ✓ = weight data available; ✗ = weight data not available.

Table 2 Summary of water parameters A , B , k_1 , k_2 used in the model fits, and water chemical parameters, $[K^+]$, $[Ca^{2+}]$ and pH of the lakes.

Lake	A $Bq\ l^{-1}$	B $Bq\ l^{-1}$	k_1 d^{-1}	k_2 (d^{-1})	$[K^+]$ $mg\ l^{-1}$	pH	$[Ca^{2+}]$ $mg\ l^{-1}$
Crummock	0.187	0	0.0073	0	0.35	6.6	2.5
Devoke	0.477	0.083	0.0089	0.00071	0.55	6.3	3.0
Ennerdale	0.338	0.014	0.0089	0.0005	0.39	6.5	2.0
Loweswater	1.03	0.024	0.015	0.0011	0.82	6.9	5.8
Windermere	0.051	0.073	0.0060	0.00071	0.59	7.0	6.3
Hillesjön	16.61	13.53	0.0153	0.0004	3.0	7.3	1.0
IJsselmeer	0.250	0.019	0.0044	0.001	7.0	8.5	80
Iso Valkjärvi	21.34	3.36	0.0057	0.0004	0.4	5.1	0.9
Kiev	21.13	0.514	0.018	0.00081	3.1	8.2	51

Table 3. Parameter values for k_f and k_b for trout; perch; pike in each lake (bold type indicates greater than 50% standard error in parameter estimate.). Mean half times of the excretion rate constant (in days) are calculated for each species.

Lake	$k_f \pm \text{SE}\%$	$k_b \pm \text{SE}\%$	R^2
Trout			
Crummock	22.75 ± 15%	0.0023 ± 24%	0.43
Devoke	24.70 ± 15%	0.0048 ± 16%	0.35
Ennerdale	25.08 ± 5%	0.0029 ± 8%	0.66
Loweswater	23.43 ± 10%	0.0039 ± 11%	0.52
Windermere	8.59 ± 20%	0.0017 ± 76%	0.37
Mean $T_{1/2}$ (d)	-	222	-
Perch			
Crummock	28.15 ± 16%	0.0014 ± 67%	0.02
Devoke	29.07 ± 8%	0.0016 ± 20%	0.24
Loweswater	23.46 ± 7 %	0.0019 ± 12%	0.69
Windermere	13.48 ± 14%	0.0024 ± 43%	0.74
Hillesjön (A)	13.97 ± 26%	0.0043 ± 26%	0.88
Hillesjön (B)	5.64 ± 19%	0.0013 ± 33%	0.41
IJsselmeer	8.41 ± 36%	0.0044 ± 33%	0.69
Iso Valkjärvi	3.12 ± 9%	0.0016 ± 10%	0.83
Kiev	5.3 ± 28%	0.0014 ± 15%	0.45
Mean $T_{1/2}$ (d)	-	307	-
Pike			
Loweswater	14.37 ± 15%	0.0019 ± 29%	0.50
Windermere	11.65 ± 13%	0.0013 ± 62%	0.59
Hillesjön	2.79 ± 16%	0.00086 ± 37%	0.55
Iso Valkjärvi	6.13 ± 13%	0.0013 ± 18%	0.57
Kiev	1.35 ± 7%	0.00066 ± 7%	0.55
Mean $T_{1/2}$ (d)	-	576	-

Table 4. R^2 values for the full data set for each species of fish for each of the three different models. (1) indicates Lake Iso Valkjärvi included in the data set, (2) indicates Lake Iso Valkjärvi excluded from the data set. ‘-‘ indicates that SAS was unable to estimate parameters.

Species	Model 1	Model 2	Model 3
Trout	0.47	0.47	0.47
Perch (1)	0.62	0.73	-
Perch (2)	0.60	0.78	0.64
Pike (1)	0.91	0.96	0.95
Pike (2)	0.70	0.78	-

Table 5. Model fits with (Model 4) and without (Model 2) fish weight as an input parameter. NS indicates that one or more of the fitted values was not significantly different to zero. Notice that R^2 values for the fit without fish weight are slightly different to those in Table 3. This is because the input data sets were slightly different since not all of the measurements of ^{137}Cs in fish were accompanied by information on fish weight (Table 1).

Species	Model	Y	k_b	n	R^2
Perch	2	$11330 \pm 2.3\%$	$0.0013 \pm 3.0\%$	-	0.57
Perch	4	$18000 \pm 24\%$	$0.0014 \pm 3.0\%$	$0.34 \pm 12.1\%$	0.69
Pike	2	$2840 \pm 7.3\%$	$0.0022 \pm 6.8\%$	-	0.51
Pike	4	NS	NS	NS	-
Trout	2	$2260 \pm 4.4\%$	$0.0033 \pm 11\%$	-	0.49
Trout	4	$5000 \pm 18\%$	$0.0029 \pm 8.3\%$	$0.38 \pm 7.9\%$	0.60

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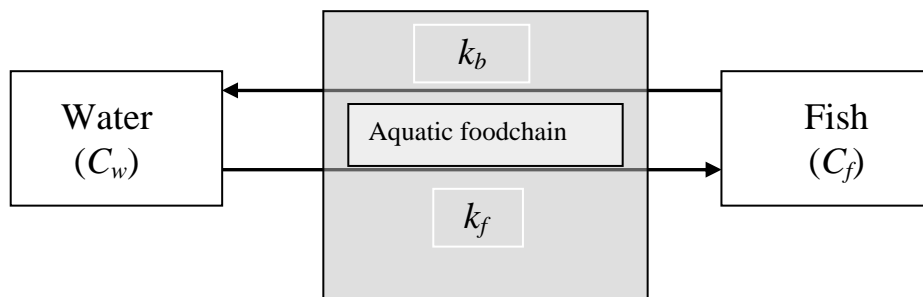


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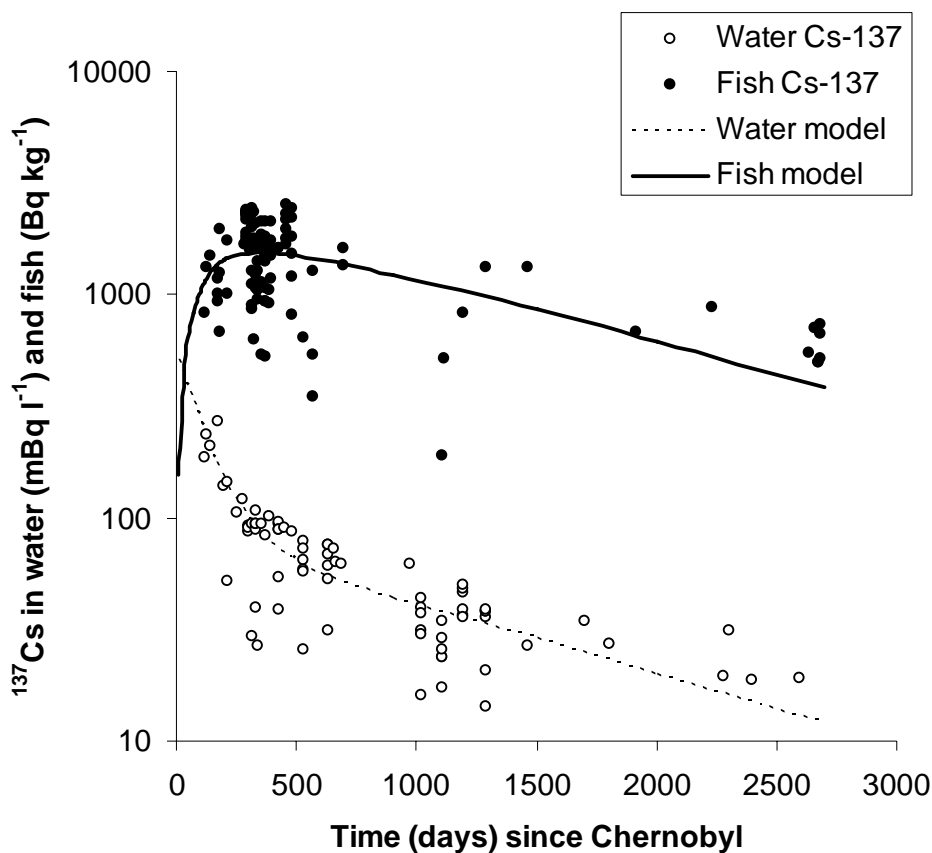


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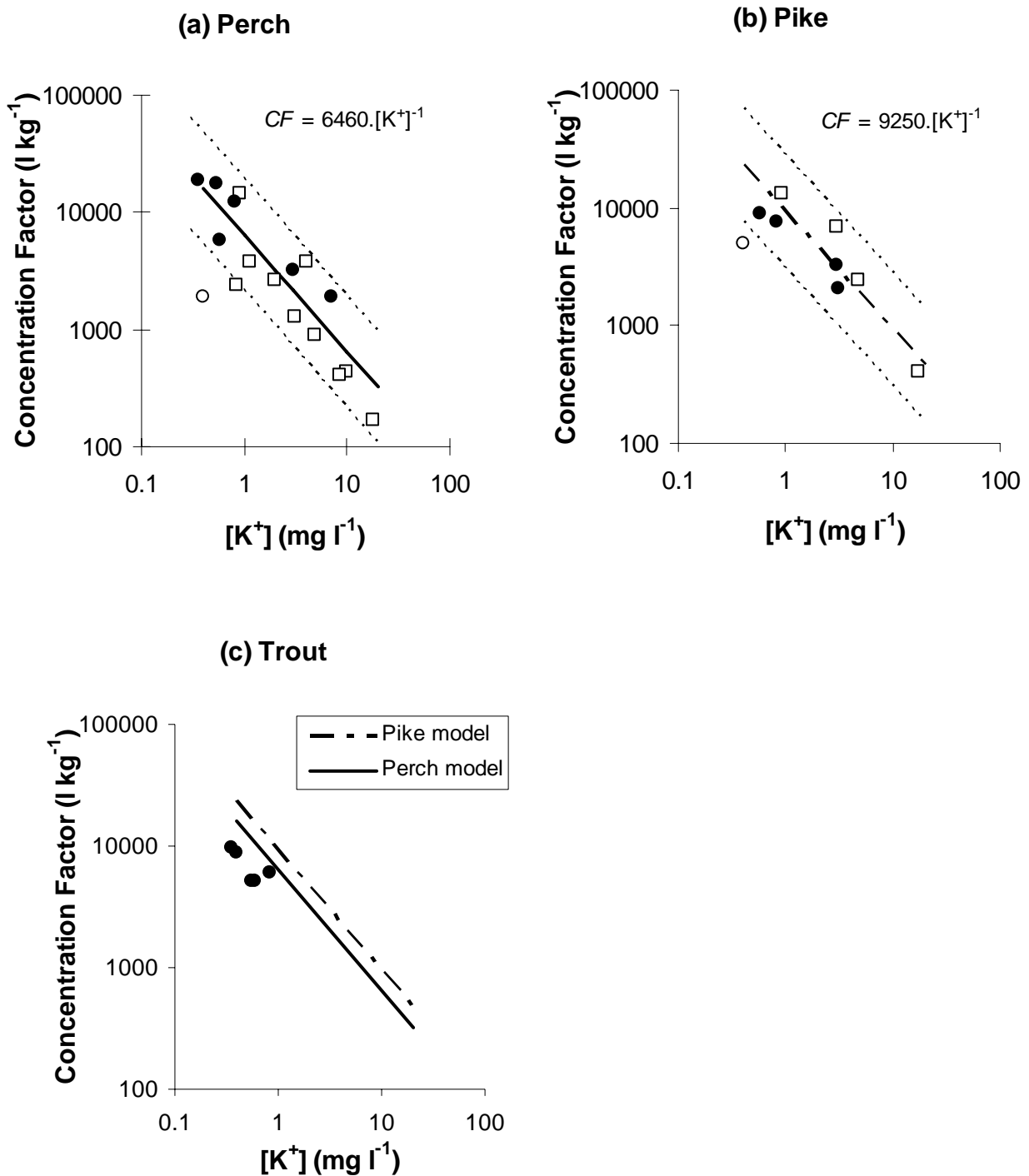


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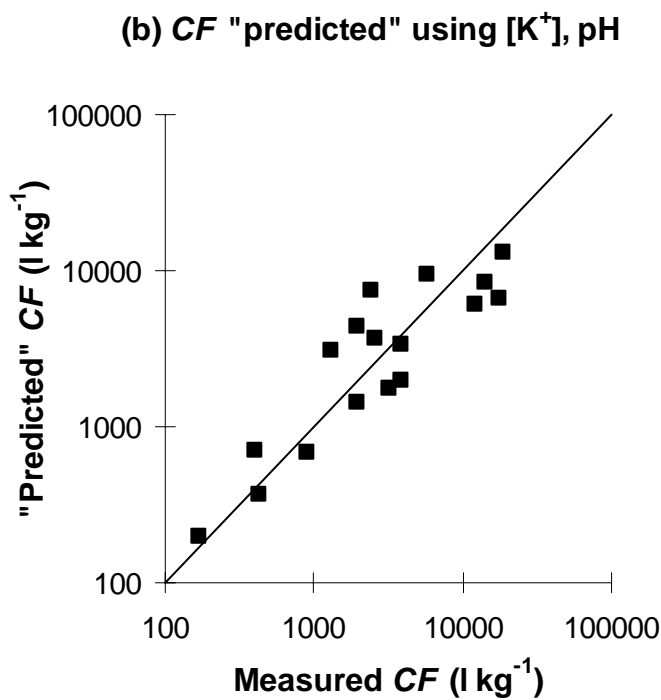
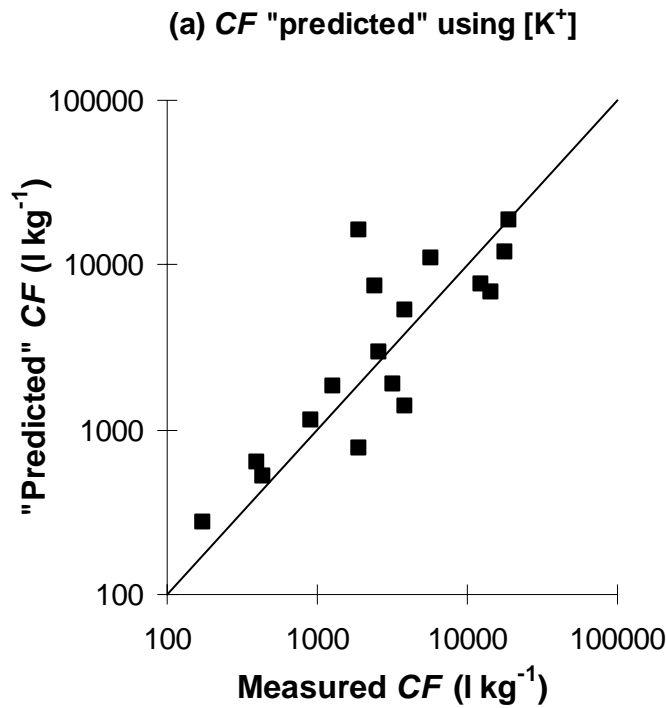


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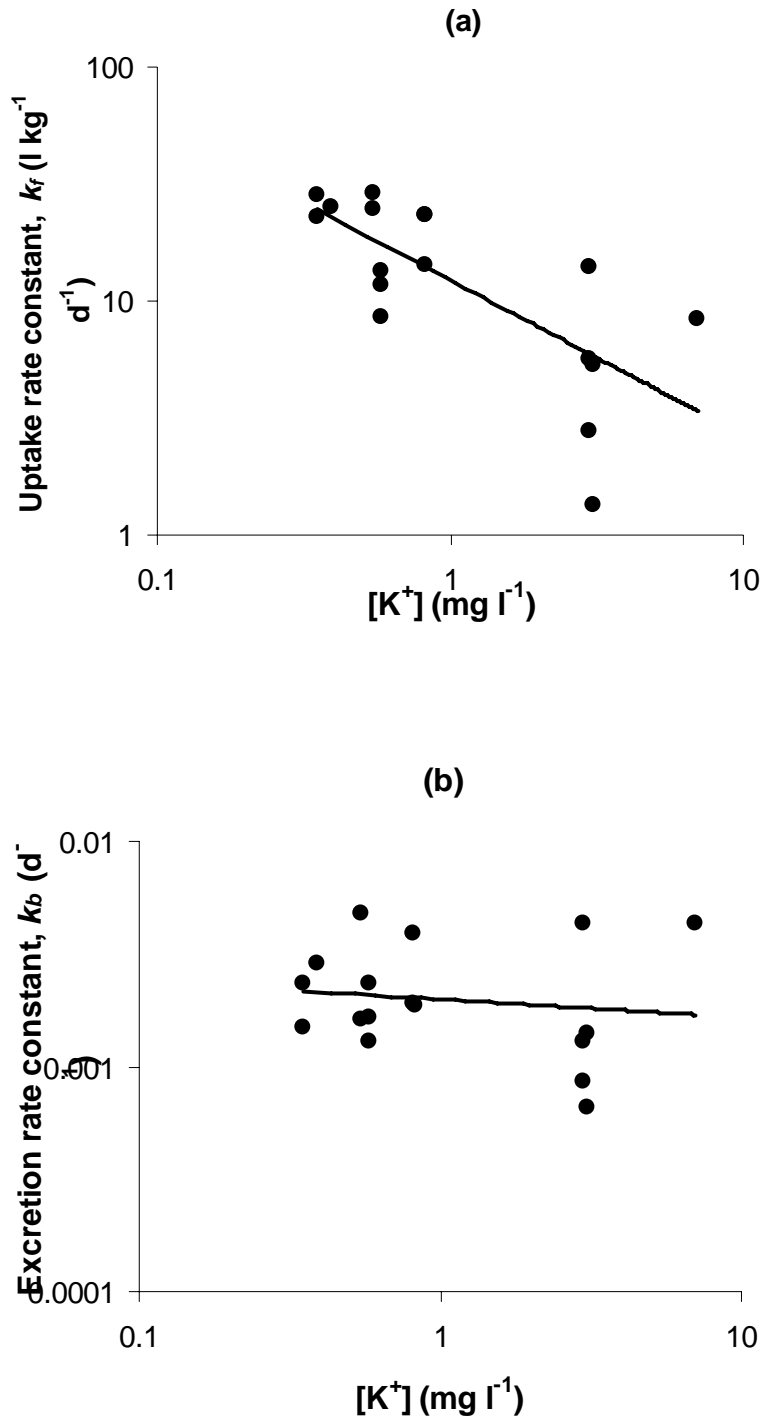


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