

Sources of variation in post-Chernobyl radiocaesium in brown trout, *Salmo trutta* L., and Arctic charr, *Salvelinus alpinus* (L.), from six Cumbrian lakes (northwest England)

J.M. Elliott¹

J.A. Elliott¹

J. Hilton¹

Keywords : radiocaesium, brown trout, Arctic charr, Chernobyl, English Lake District.

The objectives were to evaluate temporal variation in ¹³⁷Cs in trout and charr from Crummock Water, Wastwater, Ennerdale Water, and to compare levels in trout from six lakes (above three + Windermere, Loweswater, Devoke Water) and charr from four lakes (above three + Windermere). Fish were caught between June 1986 and October 1988, chiefly with fyke nets and gill nets. Overall means for ¹³⁷Cs in trout were markedly higher than those for charr in the three lakes. Values for individual trout exceeded 1000 Bq.kg⁻¹ in Ennerdale and 500 Bq.kg⁻¹ in Wastwater and Crummock Water, but values for charr never exceeded 350 Bq.kg⁻¹. ¹³⁷Cs content increased with fish weight for trout from Wastwater and therefore values were scaled to a standard weight in subsequent analyses. No similar relationships were found for trout in Ennerdale or Crummock Water, or for charr in all three lakes. Monthly mean values for ¹³⁷Cs in trout followed similar temporal patterns in the three lakes and a parabola provided a simple model for these changes. Chernobyl deposition occurred early in May 1986 and maximum values in trout were in December 1986 in Wastwater, January/February 1987 in Ennerdale Water and March 1987 in Crummock Water. The exponential rate of decrease from these maxima did not differ significantly between the three lakes, and ecological half-lives of ¹³⁷Cs in the trout differed slightly, but not significantly, between lakes : 180 days for Wastwater, 194 days for Crummock Water and 249 days for Ennerdale Water. The equivalent value for Ennerdale charr was lower at 132 days, and it was impossible to fit models or calculate half-lives for Wastwater and Crummock charr. Maximum monthly geometric mean ¹³⁷Cs values in trout from six lakes and charr from four lakes were related to the initial concentration of ¹³⁷Cs in both water and sediment, and the maximum monthly geometric mean ¹³⁷Cs values obtained from the routine water samples. The relationship was curvilinear and approximated, but was not identical to, an asymptotic curve. No similar relationships were found with mean Ca or K levels in the lake water. Metabolic and diet differences may explain the lower ¹³⁷Cs levels in charr compared with trout and differences for the same species from different lakes may be due to limnological variations and differences between lakes in the nature of their sediments and catchment soils. Both points are discussed in detail.

Sources de variation en radiocésium de la truite, *Salmo trutta* L., et de l'omble chevalier, *Salvelinus alpinus* (L.), après Chernobyl, dans six lacs de la région de Cumbria (nord-ouest de l'Angleterre)

Mots clés : radiocésium, truite, omble chevalier, Chernobyl, région des lacs anglais.

Les objectifs étaient de déterminer les fluctuations temporelles en ¹³⁷Cs dans les truites et les ombles de 3 lacs (Crummock Water, Wastwater, Ennerdale Water) et de comparer les niveaux de ¹³⁷Cs dans les truites de six lacs (les trois ci-dessus + Windermere, Loweswater, Devoke Water) et dans les ombles de quatre lacs (les trois ci-dessus + Windermere). Les poissons ont été capturés entre juin 1986 et octobre 1988, principalement avec des filets maillants et des « fykes ». Les moyennes totales de ¹³⁷Cs dans les truites étaient plus élevées que celles dans les ombles dans les trois premiers lacs. Pour chaque truite, la valeur dépasse 1 000 Bq.kg⁻¹ dans le lac Ennerdale et 500 Bq.kg⁻¹ dans les lacs Wastwater et Crummock Water, mais les valeurs pour les ombles n'ont jamais dépassé 350 Bq.kg⁻¹. La teneur en ¹³⁷Cs augmente avec le poids des truites du lac Wastwater et, par conséquent, les valeurs sont étalonnées à un poids standard dans les analyses suivantes. Il n'y a pas de rapports semblables pour les truites dans les lacs Ennerdale et Crummock Water ni pour les ombles dans trois lacs. Les valeurs moyennes mensuelles de ¹³⁷Cs dans les truites suivent des modalités temporelles similaires dans les trois lacs, et une parabole fournit un modèle simple pour ces changements. Le dépôt de Chernobyl a eu

1. NERC Institute of Freshwater Ecology, The Windermere Laboratory, Ambleside, Cumbria LA22 OLP, U.K.

lieu début mai 1986 et les valeurs maximums des truites se situent en décembre 1986 dans Wastwater, en janvier/février 1987 dans Ennerdale, et en mars 1987 dans Crummock Water. Le taux exponentiel de la baisse de ces maximums n'est pas significativement différent dans les trois lacs, et les demi-vies écologiques de ^{137}Cs dans les truites diffèrent légèrement, mais pas significativement, entre les lacs : 180 jours pour Wastwater, 194 jours pour Crummock Water et 249 jours pour Ennerdale. La valeur équivalente pour les ombles du lac Ennerdale est inférieure à 132 jours, et il est impossible d'ajuster les modèles ou de calculer les demi-vies pour les ombles de Wastwater et Crummock Water. La moyenne géométrique des maximums mensuels des valeurs en ^{137}Cs des truites des six lacs et des ombles des quatre lacs sont liées à la concentration initiale en ^{137}Cs de l'eau et du sédiment ainsi qu'à la moyenne géométrique des teneurs en ^{137}Cs mesurées en routine dans les échantillons d'eau. La relation est curvilinéaire et approximative, mais non identique à une courbe asymptotique. Aucune relation similaire n'a été trouvée avec les teneurs moyennes de Ca ou K de l'eau des lacs. Les différences de métabolisme et de nutrition peuvent expliquer les teneurs plus faibles en ^{137}Cs dans les ombles par comparaison avec celles des truites. Les différences pour la même espèce dans les différents lacs peuvent être attribuées à des variations limnologiques et aux différences dans la nature de leurs sédiments et des sols de leurs bassins-versants. Ces deux points sont discutés en détail.

1. Introduction

Following the reactor accident on 26 April 1986, the Chernobyl plume passed over England during 2-3 May 1986. Much of the remaining activity continued northwards over Scotland, but some moved westwards and then returned to traverse Wales and England during 7-8 May (Smith & Clark 1989). Limited deposition occurred throughout the UK and the highest values were recorded in Cumbria, North Wales and parts of Scotland, where passage of the contaminated cloud coincided with heavy rainfall creating maximum values of 20 000 Bq.kg⁻² for ^{137}Cs (Baker & Cawse 1990). The three radionuclides of greatest significance within the plume were ^{131}I (half-life 8 days) in both gaseous and particulate forms, ^{134}Cs (half-life 2.06 years) and ^{137}Cs (half-life 30.2 years) both almost entirely in particulate form (Smith & Clark 1989).

The Ministry of Agriculture, Fisheries and Food, has a statutory duty to monitor and assess the impact of radioactive discharges into aquatic environments and radiochemical measurements are made routinely by the Directorate of Fisheries Research at sites throughout the UK. This monitoring programme was intensified soon after the Chernobyl accident. Early results showed that whilst the radiological significance of the fallout on marine fish was negligible, radionuclide accumulation by freshwater fish was significant, especially in high deposition areas such as the Cumbrian Lake District (Camplin et al. 1986 ; Mitchell et al. 1986 ; Camplin et al. 1989 ; Leonard et al. 1990). It was soon found that there were wide variations between lakes, fish species and individuals of the same species in the same lake.

In a previous study (Elliott et al. 1992), several conclusions were reached concerning sources of variation in ^{137}Cs levels in muscle of brown trout (*Salmo trutta* L.), perch (*Perca fluviatilis* L.) and pike (*Esox lucius* L.) in two lakes. Overall geometric means for ^{137}Cs in trout and perch in Devoke Water were significantly higher than those in Loweswater ; means for perch were significantly higher than means for trout in both lakes, and the mean for pike in Loweswater lay between the values for trout and perch. ^{137}Cs content increased with fish weight for trout from both lakes, perch from Devoke Water but not perch or pike from Loweswater. As there was no indication of dietary differences between trout and perch, it was suggested that the variation between species may be due to different metabolic rates (e.g. gastric evacuation rates). It was also demonstrated that there were differences in the ecological half-life of ^{137}Cs (i.e. effective half-life observed in situ, see also section 3.3) in trout and perch in Devoke Water and Loweswater, and that dissimilarity between lakes could be due to variations in their limnology and, particularly, the nature of their catchments.

The general objective of the present study was to expand the above ideas and evaluate their relevance to brown trout and Arctic charr (*Salvelinus alpinus* (L.)) populations in another four lakes. There were sufficient data from three of the lakes (Crummock Water, Wastwater, Ennerdale Water) for detailed analyses but the data for Windermere were incomplete and could be used only for comparative purposes. The two major objectives were therefore to evaluate temporal variation of radiocaesium (^{137}Cs)

in brown trout and Arctic charr from three lakes ; and to compare levels in trout from six lakes (above four plus Devoke Water and Loweswater) and charr from four lakes in relation to five factors ; namely initial concentration of ^{137}Cs in the water, maximum monthly mean ^{137}Cs concentration obtained from the routine water samples, concentration of ^{137}Cs in the sediment after the rapid period of ^{137}Cs deposition from the water column, and mean potassium and calcium levels.

2. Sites, materials and methods

All six lakes are situated in the Cumbrian Lake District (north-west England) (Fig. 1). They are all within 35 km of each other, five to the west of the area and one (Windermere) to the east. Windermere North Basin, at an altitude of 39 m, is a mildly eutrophic lake with a surface area of 8.05 km², mean depth of 21 m and maximum depth of 64 m. Crummock Water is an oligotrophic, acid lake at an altitude of 98 m, with a surface area of 2.52 km², mean depth of 27 m and maximum depth of 44 m. Wastwater is an oligotrophic, acid lake at an altitude of 61 m, with a surface area of 2.91 km², mean depth of 40 m and maximum depth of 76 m. Ennerdale Water is also an oligotrophic, acid lake at an altitude of 112 m, with a surface area of 3.00 km², mean depth of 18 m and a maximum depth of 42 m (Ramsbotton 1976). Similar descriptions of Devoke Water and Loweswater are provided in Elliott et al. (1992). The deposition of ^{137}Cs on vegetation in the catchment of the lakes was between 2000 and 20000 Bq.m⁻² (Spezzano et al. 1993).

All fish muscle radioactivity data were obtained from Tables in Camplin et al. (1989), but had not been previously analyzed in detail. The fish were caught between June 1986 and October 1988, chiefly with fyke nets and gill nets. The number caught varied considerably between months, species and lakes (see values for individual fish in Fig. 2). Fish samples were usually frozen for transport to the laboratory and, after thawing, fork length (to nearest mm) and wet weight (to nearest g) were recorded for each fish. Muscle tissue was then removed, weighed and compressed into a standard geometry for counting in the wet state for 3600s by gamma-ray spectrometry. As ^{134}Cs has a short half-life of 2.06 years, only ^{137}Cs data (decay corrected to the time of sampling) were used in the analysis. Results

were expressed as Bq.kg⁻¹ wet weight, the detection limit being approximately 2 Bq.kg⁻¹, and the counting error decreased with count size.

The initial concentration of ^{137}Cs in the lake water, immediately after the deposition event, was estimated by dividing the estimated deposition per unit area by the mean depth of the lake, all values being expressed as Bq.m⁻³ (values given in Spezzano et al. 1993). This approach was justified as the studied lakes were all expected to have been completely mixed at the time of the deposition, i.e. in early May, 1986.

Routine water samples taken throughout the study period were used to determine monthly mean concentrations of ^{137}Cs in the water. All values were expressed as mBq.l⁻¹ and decay corrected to time of sampling (for basic data and methods, see Camplin et al. 1989). The maximum monthly geometric mean value for each lake was used in the comparisons with the fish data. This value occurred near the start of the routine sampling but as the latter did not commence until several weeks after the initial deposition, the concentration of ^{137}Cs in the water must have already decreased markedly. Values for the maximum monthly ^{137}Cs concentration obtained from the routine water samples were therefore markedly lower than those for the initial concentration of ^{137}Cs in the lake water. However, as shown later, both values were closely correlated for the six lakes of the present study.

Two sediment samples were collected from the deepest point within each lake using a 7 cm internal-diameter Jenkin corer (Ohnstad & Jones 1982). The top 5 cm of both cores were combined, oven dried at 60°C and ground in a porcelain mortar. The ^{137}Cs content (Bq.kg⁻¹ dry weight) was determined by gamma-ray spectrometry (values given in Spezzano et al. 1993). Activity levels were decay corrected to 3rd May 1986 because the samples were taken on different dates. Since the major deposition of activity from the water column occurred rapidly in the first year after deposition from the atmosphere and there was little change in concentration after the end of 1986, the data are comparable even though they span a period of four years (1988-1992).

Mean potassium and calcium values ($\mu\text{E. l}^{-1}$) for the six lakes were estimated from data collected between 1974 and 1978 (values given in Carrick & Sutcliffe 1982).

All statistical analyses were carried out on individual fish, except when it is stated clearly that geometric means were used ; samples of fish were never pooled. Geometric means were used in preference to arithmetic means because they are less biased by

extreme values and because they are the only mean values that can be used with logarithmic-transformed data (see Elliott 1977). Coefficients of determination (r^2) were all corrected for small sample sizes, as were also all statistical tests.

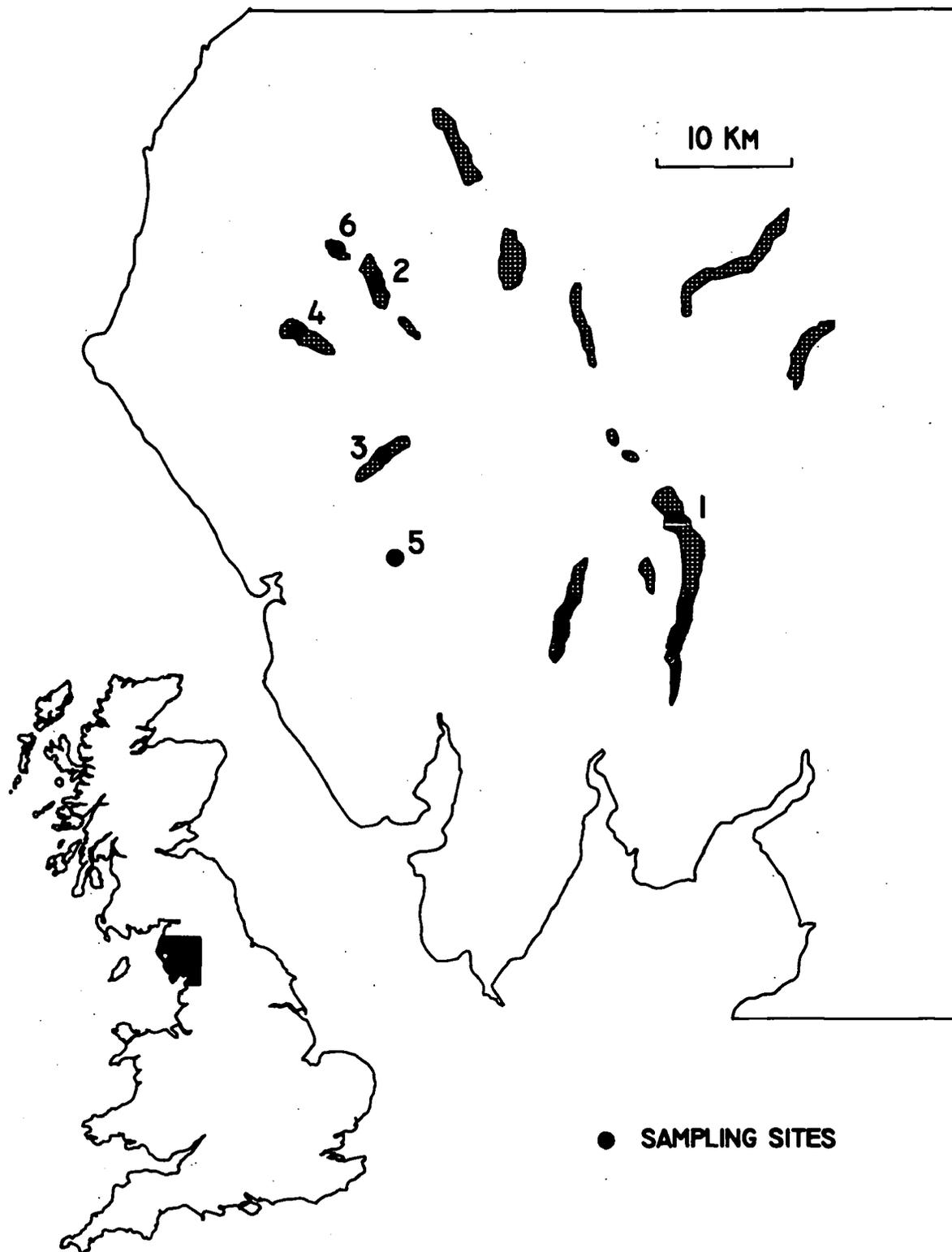


Fig. 1. Location of the sampling sites. Ordnance Survey grid reference in parentheses. 1) Windermere (north basin) (NY383010) ; 2) Crummock Water (NY160180) ; 3) Wastwater (NY160060) ; 4) Ennerdale Water (NY110150) ; 5) Devoke Water (SD157970) ; 6) Loweswater (NY125215).

Fig. 1. Position des sites d'échantillonnages. Les références de la grille de la carte « Ordnance Survey » sont mises entre parenthèses. 1) Windermere (bassin nord) (NY383010) ; 2) Crummock Water (NY160180) ; 3) Wastwater (NY160060) ; 4) Ennerdale Water (NY110150) ; 5) Devoke Water (SD157970) ; 6) Loweswater (NY125215).

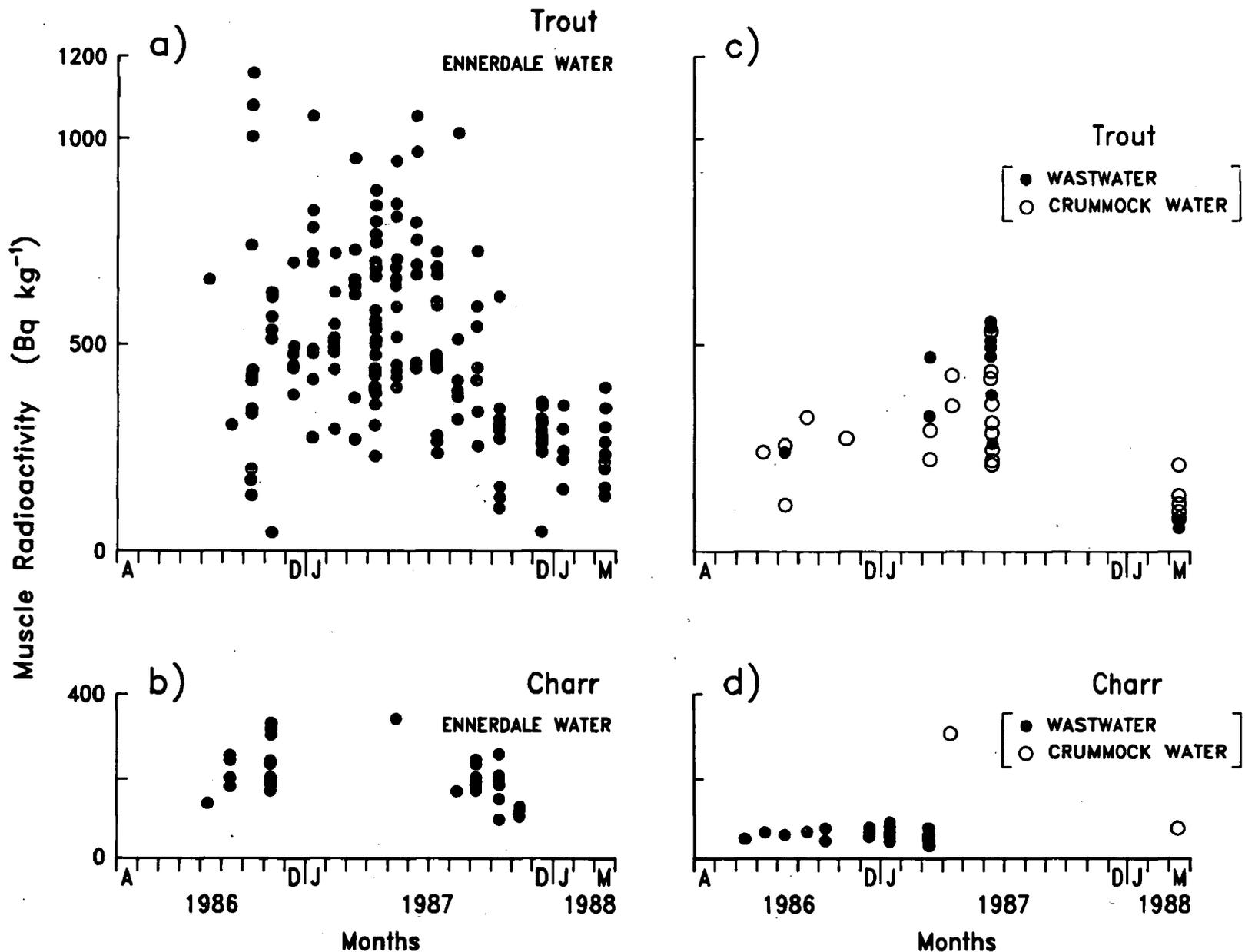


Fig. 2. Muscle radiocaesium values (^{137}Cs as Bq.kg^{-1} wet weight) for individual fish : (a) brown trout from Ennerdale Water ($n = 173$) ; (b) charr from Ennerdale Water ($n = 38$) ; (c) brown trout from Wastwater ($n = 12$) and Crummock Water ($n = 26$) ; (d) charr from Wastwater ($n = 24$) and Crummock Water ($n = 2$).

Fig. 2. Valeurs de radiocésium dans le muscle (^{137}Cs comme Bq.kg^{-1} poids frais) pour un individu : (a) truites du lac Ennerdale ($n = 173$) ; (b) ombles du lac Ennerdale ($n = 38$) ; (c) truites des lacs Wastwater ($n = 12$) et Crummock Water ($n = 26$) ; (d) ombles des lacs Wastwater ($n = 24$) et Crummock Water ($n = 2$).

3. Results

3.1. Major differences between species and lakes

^{137}Cs values increased markedly for trout in all three lakes, but in the case of charr, the increase was less spectacular, especially for the Ennerdale Water population (Fig. 2). There were insufficient data for the Crummock Water population of charr to draw any conclusions on temporal changes. Some values for trout in Ennerdale Water exceeded 1000 Bq.kg^{-1} , but charr values never exceeded

350 Bq.kg^{-1} (all values for kg fresh weight). Overall geometric mean values for trout (425 Bq.kg^{-1} , 95 % C.L. 391-462) and charr (183 Bq.kg^{-1} , 164-204) in Ennerdale Water were significantly higher ($P < 0.05$) than those for trout (304 Bq.kg^{-1} , 215-430) and charr (58 Bq.kg^{-1} , 48-82) in Wastwater, and mean values for trout in both lakes were significantly higher than that for trout in Crummock water (234 Bq.kg^{-1} , 191-287).

This preliminary analysis shows that the overall mean ^{137}Cs content in trout was always higher than

that in charr, and that mean values for both species were highest in Ennerdale Water, followed by Wastwater and finally Crummock Water. There were, however, large differences between individual fish of the same species in each lake.

3.2. Effect of fish size

There was a significant relationship ($P < 0.02$) between ^{137}Cs in the individual trout caught between August 1986 and March 1988 in Wastwater and their wet weight (Fig. 3). The relationship was well described by a power function (given in its logarithmic form in the legend to Fig. 3), and the r^2 value indicated that the variation in fish weight could account for 41 % of the variation in ^{137}Cs in individual trout. For the 26 and 173 trout caught between July 1986 and March 1988 in Crummock Water and between August 1986 and March 1988 in Ennerdale Water, respectively, the regression equations were not significantly different ($P > 0.05$) from a horizontal line through the geometric mean value (Fig. 3). Fish weight therefore had no significant effect on ^{137}Cs in trout from Crummock Water and Ennerdale Water. Similar analyses for charr revealed no significant relationships ($P > 0.05$) between ^{137}Cs in individual fish and their wet weight (see Fig. 2 for months in which charr were caught).

3.3. Temporal effects

The previous analyses have shown that any temporal comparisons must take into account the effects of fish weight on ^{137}Cs content. For the trout from Wastwater, the power-function relationship was used to scale each ^{137}Cs value to that of a fish of standard weight. The latter was the geometric mean wet weight of all fish taken in the samples; 401 g for Wastwater. The scaled values were then used to calculate monthly geometric means for ^{137}Cs in the fish. As fish weight did not significantly affect their ^{137}Cs levels, this procedure was unnecessary for Crummock Water and Ennerdale Water trout and charr, and for Wastwater charr, and therefore monthly geometric means were calculated from the original data.

Monthly mean values for trout followed similar temporal patterns in the three lakes (Fig. 4). They increased from low pre-Chernobyl values (see values in Elliott et al. 1992) to a maximum, and then decreased. Parabolas were fitted to give some indication of where the overall maximum ^{137}Cs levels

occurred (Fig. 4, Table 1). The maximum was in January/February 1987 for Ennerdale Water, December 1986 for Wastwater, and March 1987 for Crummock Water. However the peaks of the parabolas were very broad so that it is unlikely that these differences are significant. The months for this maximum value were used as the initial month when exponential models were fitted. The exponential rate of decrease ($b \pm 95\%$ C.L.) did not differ significantly ($P > 0.05$) between Ennerdale Water ($b = 0.0837 \pm 0.0332$), Wastwater ($b = 0.115 \pm 0.0598$), and Crummock Water ($b = 0.107 \pm 0.0276$). The close fit of the exponential model to the data for Wastwater and Crummock Water was remarkable; as there were only three geometric means, only one value markedly out of line would have produced a non-significant fit.

Geometric mean values for ^{137}Cs in lake water also decreased exponentially in the three lakes. Although the exponential rate of decrease ($b \pm 95\%$ C.L.) did not differ significantly ($P > 0.05$) between Crummock Water ($b = 0.109 \pm 0.073$) and either Ennerdale Water ($b = 0.131 \pm 0.032$) or Wastwater ($b = 0.0730 \pm 0.027$), the rate for Ennerdale Water was just significantly faster ($P < 0.05$) than that for Wastwater.

Mean values for charr in Ennerdale Water followed a pattern similar to the trout except that the maximum from the parabola occurred slightly later in March 1987 (Fig. 4a). Again, it is not possible from the data to determine if this later maximum is significantly different from that for trout. An exponential model was again fitted to the means during the decreasing phase giving; $b = 0.157 \pm 0.120$. This value is slightly, but not significantly ($P > 0.05$), higher than those obtained for trout in the three lakes. Neither a parabolic function nor an exponential curve could be fitted to the charr in Wastwater because ^{137}Cs values showed no significant decrease with time ($P > 0.05$) (Fig. 4b) and no fish samples were taken over the period in which values would be expected to decrease.

A summary of some of these differences between lakes and species is provided by the « ecological half-life » for ^{137}Cs in the fish (T_e days), i.e. the time taken for the ^{137}Cs content to fall to 50 % of its initial value ($T_e = (30 \cdot \ln 2)/b$, where $b =$ exponential rate of decrease, 30 simply converts months to days). The ecological half-lives for trout in all

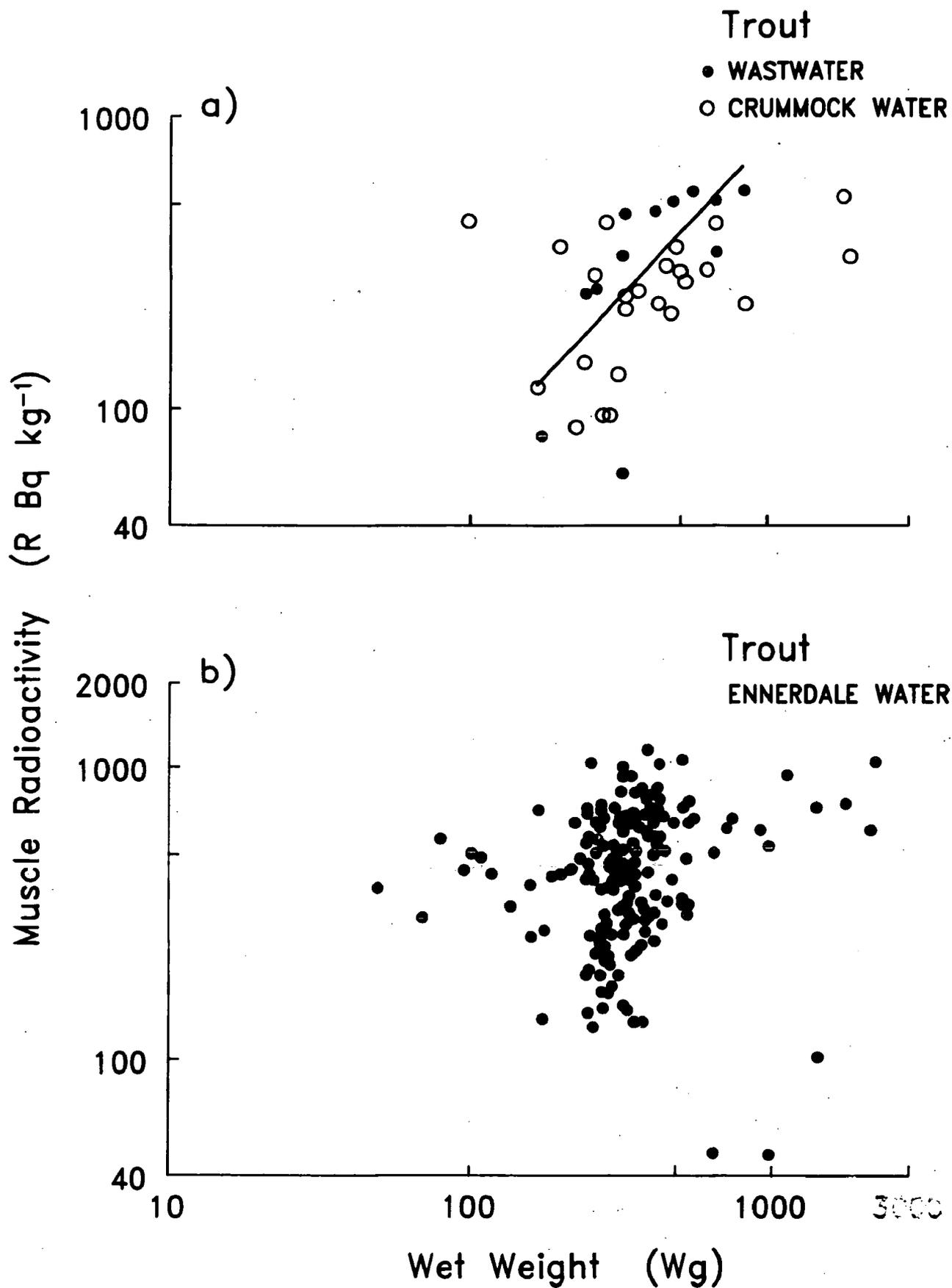


Fig. 3. Relationship between muscle radioactivity (R Bq.kg⁻¹) and wet weight (Wg) for : (a) individual brown trout from Wastwater and Crummock Water ; regression line for Wastwater : $\log_{10} R = -0.394 + 1.105\log_{10} W$ (n = 12, $r^2 = 0.414$, $P < 0.02$) ; regression line for Crummock Water was not significant (n = 26, $P > 0.05$) ; (b) individual brown trout from Ennerdale Water, regression line was not significant (n = 173, $P > 0.05$).

Fig. 3. Relations entre la radioactivité du muscle (R Bq.kg⁻¹) et le poids frais (Wg) pour : (a) une truite des lacs Wastwater et Crummock Water ; ligne de régression pour Wastwater : $\log_{10} R = -0.394 + 1.105\log_{10} W$ (n = 12, $r^2 = 0.414$, $P < 0.02$) ; ligne de régression pour Crummock Water non significative (n = 26, $P > 0.05$) ; (b) une truite du lac Ennerdale, ligne de régression non significative (n = 173, $P > 0.05$).

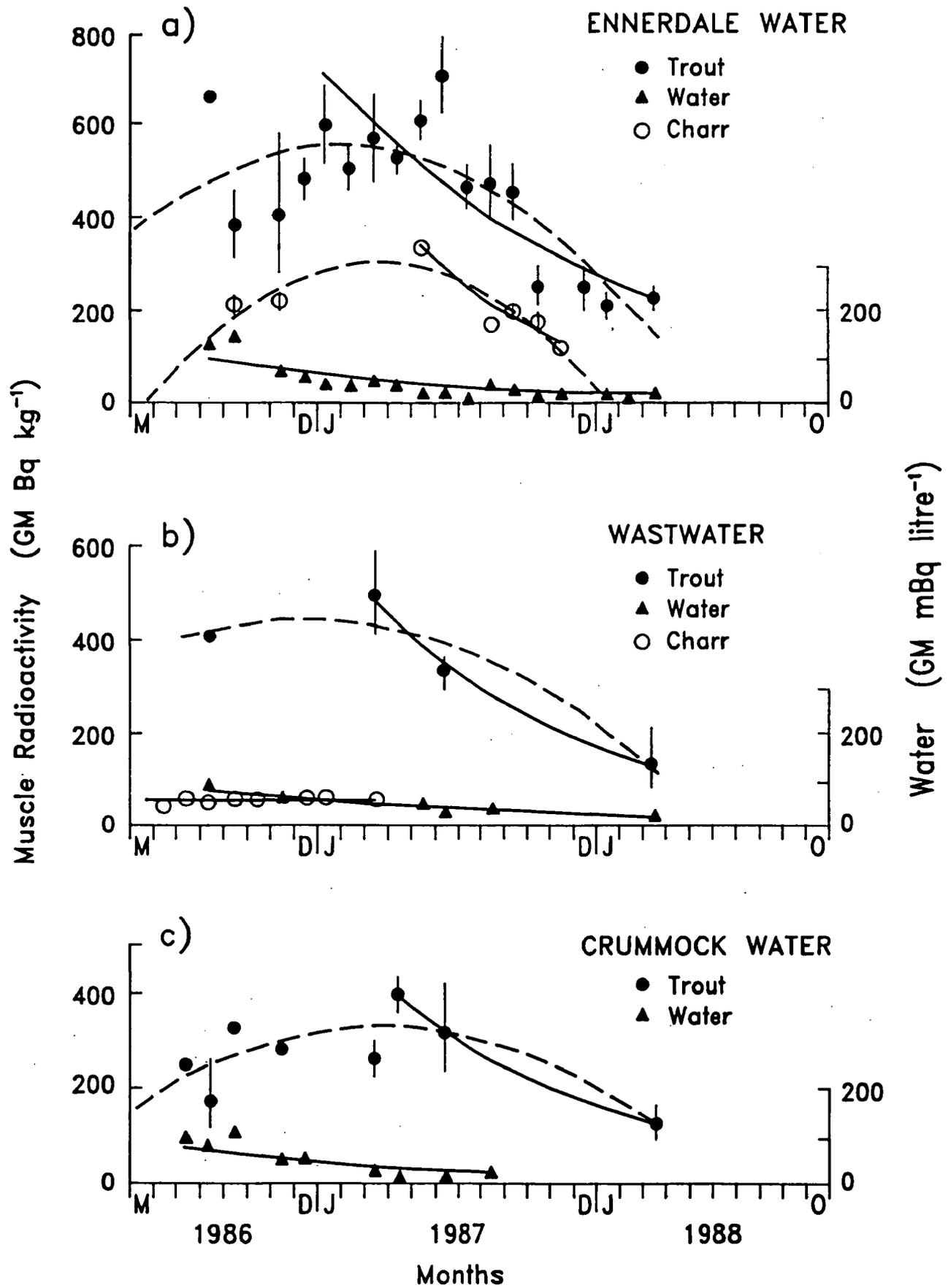


Fig. 4. Relationship between Geometric Mean (GM) radioactivity in fish (GM Bq.kg⁻¹ ± 1 S.E.) or in water (GM mBq.litre⁻¹) and time (t months from May 1986) for (a) trout and charr in Ennerdale Water, (b) trout and charr in Wastwater, (c) trout in Crummock Water. Solid lines are exponential curves given by : $GM = a \exp(-bt)$ where values of constants a and b are given in Table 1a. Broken lines are parabolas given by : $GM = c + dt - et^2$ where values of constants c, d and e are given in Table 1b. (Each GM for Wastwater trout is for values that were first scaled to a standard fish weight of 401 g).

Fig. 4. Relation entre la moyenne géométrique (GM) de la radioactivité dans les poissons (GM Bq.kg⁻¹ ± 1 S.E.) ou dans l'eau (GM mBq.litre⁻¹) et le temps (t mois à partir de mai 1986) pour (a) truites et ombles du lac Ennerdale Water, (b) truites et ombles du lac Wastwater, (c) truites du lac Crummock Water. Les lignes pleines sont des courbes exponentielles données par : $GM = a \exp(-bt)$ où les valeurs des constantes a et b sont données dans le tableau 1a. Les lignes tirets sont des paraboles données par $GM = c + dt - et^2$ où les valeurs des constantes c, d et e sont données dans le Tableau 1b. (Chaque GM pour les truites du lac Wastwater correspond à des valeurs qui ont été en premier lieu ramenées à un poids standard de poisson de 401 g).

Table 1. Equations for the exponential curves and parabolas in Fig. 4, together with the number of data points (n), coefficients of determination corrected for small sample size (r^2) and significance level for each relationship (P). (See legend in Fig. 4 for further details).

Tableau 1. Equations des courbes exponentielles et paraboles de la Fig. 4, avec le nombre de données (n), les coefficients de détermination corrigés pour peu d'échantillons (r^2) et le niveau significatif pour chaque relation (P). (Pour plus de détails, voir légende Fig. 4).

(a)		Exponential curve			
		GM= a exp(-bt)	n	r^2	P
Ennerdale	trout	GM= 1371 exp(-0.0837t)	13	0.73	<0.001
	charr	GM= 2162 exp(-0.157t)	5	0.80	<0.05
	water	GM= 148 exp(-0.131t)	18	0.82	<0.001
Wastwater	trout	GM= 1519 exp(-0.115t)	3	1.00	<0.001
	water	GM= 90 exp(-0.0730t)	6	0.91	<0.005
Crummock	trout	GM= 1253 exp(-0.107t)	3	1.00	<0.001
	water	GM= 93 exp(-0.109t)	9	0.59	<0.01

(b)		Parabola			
		GM= c + dt - et ²	n	r^2	P
Ennerdale	trout	GM= 389.0 + 37.7t - 2.22t ²	17	0.57	<0.001
	charr	GM= -12.2 + 62.3t - 3.09t ²	8	0.72	<0.005
Wastwater	trout	GM= 371.9 + 20.0t - 1.44t ²	4	0.85	<0.05
Crummock	trout	GM= 162.6 + 30.6t - 1.47t ²	8	0.57	<0.02

three lakes were not significantly different and were 249 days (95 % C.L. 180-402 days) for Ennerdale Water ; 180 days (119-344 days) for Wastwater ; 194 days (154-263 days) for Crummock Water. The charr ecological half-life for Ennerdale Water was lower at 132 days (75-553 days) ; it was not possible to calculate values for Wastwater and Crummock Water charr.

3.4. Comparisons between lakes in relation to five factors

The monthly geometric mean value nearest to the highest point of each parabola in Fig. 4 was used as an estimate of maximum muscle radioactivity (R Bq.kg⁻¹) for trout and charr in Ennerdale, Wastwater and Crummock Water. The highest monthly geometric mean value was obtained directly from the smaller data sets for charr and trout in Windermere. Maximum values for trout in Devoke Water and Loweswater were obtained from the earlier study (Elliott et al. 1992). The maximum values for trout from six lakes and charr from four lakes were then compared with five external factors (Figs 5, 6).

Several mathematical models were applied to the data (exponential, power law, asymptotic, logistic) but the only successful fits were obtained with a semi-logarithmic model ; the mean values for muscle radioactivity being on an arithmetic scale and the external factor used in the comparison being on a logarithmic scale (data are presented with X axes on both arithmetic and logarithmic scales in Fig. 5). Regression lines were fitted to the data and were significant for trout when the external factor was either the initial concentration of ¹³⁷Cs in the water or the maximum monthly geometric mean ¹³⁷Cs concentration from the routine water samples, and for charr when the external factor was the initial concentration of ¹³⁷Cs in the sediment (solid lines in Fig. 5 and equations in Table 2). Although the data in Fig. 5a appear to be a cloud of points with one extreme value for Devoke Water (value 5), the transformed values in Fig. 5b are evenly distributed along the regression line. As the latter was fitted in this form, there was no bias in the fitting procedure and it can be concluded the semi-logarithmic model was a good fit to the data.

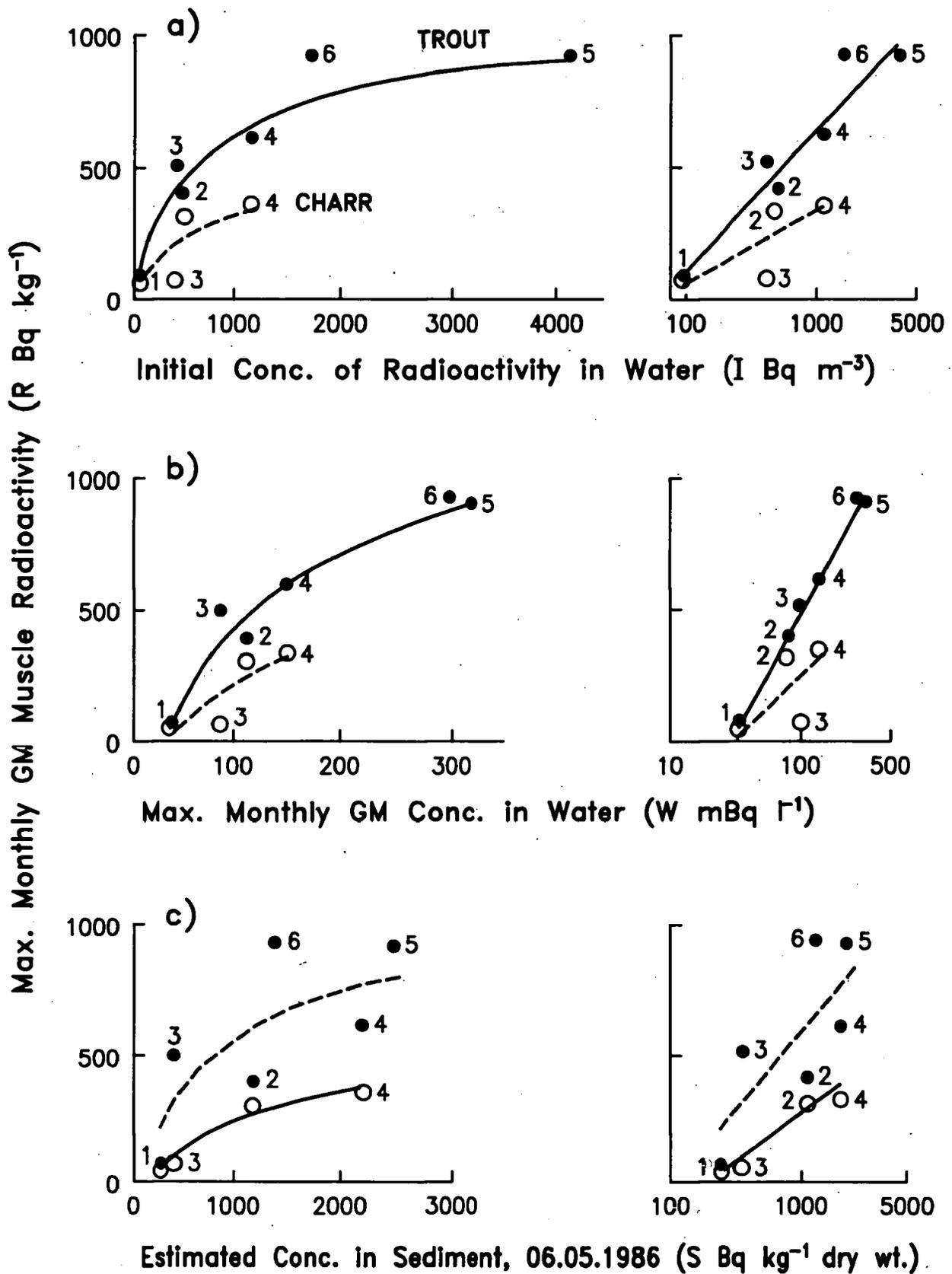


Fig. 5. Relationship between maximum monthly GM muscle radioactivity ($R \text{ Bq.kg}^{-1}$) for trout (\bullet) and charr (\circ) in different lakes and : (a) initial concentration of ^{137}Cs in the water ($I \text{ Bq.m}^{-3}$) ; (b) maximum monthly GM ^{137}Cs concentration obtained from the routine water samples ($W \text{ mBq.l}^{-1}$) ; (c) concentration of ^{137}Cs in sediment ($S \text{ Bq.kg}^{-1}$ dry weight). (Lakes : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). All data are presented with X axes on both arithmetic scales and logarithmic scales and relationships are indicated as solid lines (significant, $P < 0.05$) or broken lines (not significant, $P > 0.05$). Curves and lines given by $R = a + b \ln X$ where X is either $I \text{ Bq.m}^{-3}$ or $W \text{ mBq.l}^{-1}$ or $S \text{ Bq.kg}^{-1}$ dry weight and constants a and b are given in Table 2.

Fig. 5. Relation entre le maximum mensuel de la GM de la radioactivité du muscle ($R \text{ Bq.kg}^{-1}$) pour les truites (\bullet) et les ombles (\circ) dans les différents lacs et : (a) concentration initiale de ^{137}Cs dans l'eau ($I \text{ Bq.m}^{-3}$) ; (b) maximum mensuel de la GM de la concentration de ^{137}Cs obtenue des échantillons de routine de l'eau ($W \text{ mBq.l}^{-1}$) ; (c) concentration de ^{137}Cs dans le sédiment ($S \text{ Bq.kg}^{-1}$ poids sec). (Lacs : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Toutes les données sont présentées avec les axes X aux échelles arithmétique et logarithmique et les relations sont indiquées en lignes pleines (significatif, $P < 0.05$) ou en lignes tirets (non significatif, $P > 0.05$). Courbes et lignes sont données par $R = a + b \ln X$ où X est l'une de $I \text{ Bq.m}^{-3}$ ou $W \text{ mBq.l}^{-1}$ ou $S \text{ Bq.kg}^{-1}$ poids sec et les constantes a et b sont données dans le Tableau 2.

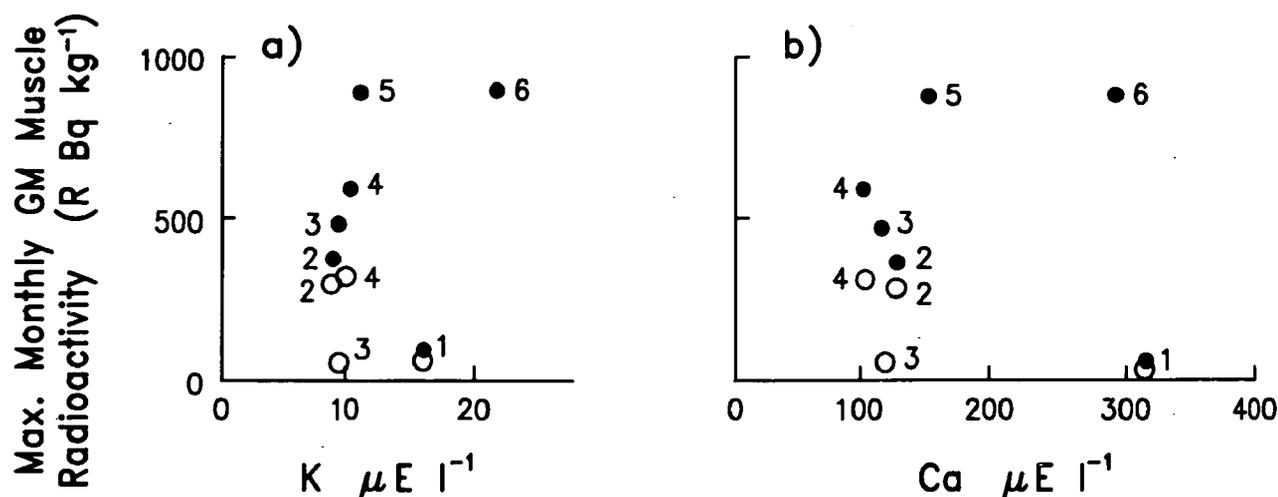


Fig. 6. Relationship between maximum monthly GM muscle radioactivity ($R \text{ Bq.kg}^{-1}$) for trout (\bullet) and charr (\circ) in different lakes and : (a) GM potassium level ($K \mu\text{E.l}^{-1}$) ; (b) GM calcium level ($\text{Ca} \mu\text{E.l}^{-1}$), (Lakes : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). All relationships were not significant ($P > 0.05$).

Fig. 6. Relation entre le maximum mensuel de la GM de la radioactivité du muscle ($R \text{ Bq.kg}^{-1}$) pour les truites (\bullet) et les ombles (\circ) dans les différents lacs et : (a) niveau de la GM de potassium ($K \mu\text{E.l}^{-1}$) ; (b) niveau de la GM de calcium ($\text{Ca} \mu\text{E.l}^{-1}$), (Lacs : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Toutes les relations n'étaient pas significatives ($P > 0.05$).

Table 2. Equations for the relationships in Fig. 5, together with the number of data points (n), coefficients of determination corrected for small sample size (r^2) and significance level for each relationship (P). (See legend in Fig. 5 for further details).

Tableau 2. Equations pour les relations de la Fig. 5, avec le nombre de données (n), les coefficients de détermination corrigés pour peu d'échantillons (r^2) et le niveau significatif pour chaque relation (P). (Pour plus de détails, voir légende Fig. 5).

X	Fish	Curve			
		$R = a + b \ln X$	n	r^2	P
I Bq m^{-3}	Trout	$R = -998 + 236.6 \ln X$	6	0.89	<0.01
	Charr	$R = -493 + 115.0 \ln X$	4	0.41	>0.1
W mBq l^{-1}	Trout	$R = -1349 + 392.9 \ln X$	6	0.96	<0.01
	Charr	$R = -740 + 210.2 \ln X$	4	0.48	>0.1
S Bq Kg^{-1} dry weight	Trout	$R = -1277 + 268.1 \ln X$	6	0.49	>0.05
	Charr	$R = -791 + 150.2 \ln X$	4	0.92	<0.05

None of the comparisons with mean potassium and calcium levels was significant (Fig. 6). Significant relationships were also obtained between the initial concentration of ^{137}Cs in the water ($I \text{ Bq.m}^{-3}$) and both the maximum monthly concentration in the routine water samples ($W \text{ mBq.l}^{-1}$) and the concentration in the sediment ($S \text{ Bq.kg}^{-1}$) (Fig. 7). A power law was the most appropriate model for these relationships ; this was expected from the

logarithmic transformations used earlier (Fig. 5). As also expected from the relationships in Fig. 7, there was a significant relationship between W and S ($\log_{10} W = -0.109 + 0.746 \log_{10} S$, $r^2 = 0.73$, $P < 0.02$).

Several conclusions can be drawn from these comparisons. The mean levels of muscle radioactivity in the fish varied considerably between lakes, even though the latter were all within a relatively small geographical area (Fig. 1). Mean levels for brown trout were generally higher than those for Arctic charr, except for the lowest values for both species in Windermere (Fig. 5). Differences in potassium and calcium levels between lakes have no obvious or significant effects on muscle radioactivity levels in either trout or charr (Fig. 6).

Differences in initial concentration of ^{137}Cs in the water, maximum mean concentration in the routine water samples, and concentration in the sediments were inter-related (Fig. 7), all being directly related to the initial deposition, and all affected the mean levels of radioactivity in the fish (Fig. 5). Although some of these relationships were not significant, this was probably due to the small sample for charr (Fig. 5a, b, $n = 4$, therefore there were only two degrees

of freedom) and the single non-significant relationship for trout (Fig. 5c) only just failed at the 5 % level (Table 2). The corrected r^2 values indicated that 49-96 % of the variation between lakes in levels of ^{137}Cs in trout could be explained by variations in ^{137}Cs concentrations in the water and sediment, the corresponding range being 41-92 % for charr (Table 2).

The curvilinear form of the relationships in Fig. 5 is intriguing because it indicates that as ^{137}Cs concentrations in the water and sediments increase between lakes, the rate of accumulation in the fish of each lake does not increase proportionally but clearly decreases. As an asymptotic curve was a poorer fit to the data, there was no evidence to suggest a maximum concentration for ^{137}Cs levels in each fish species. However, this poorer fit could be simply due to the small number of lakes and hence data

points. As mentioned above, the paucity of data could also be responsible for the non-significant fits in some cases, particularly since W and S are so highly correlated. Otherwise it is difficult to explain why mean levels of ^{137}Cs in trout were closely related to concentrations of ^{137}Cs in the water but not the sediments, and why the opposite was true for charr. The relatively high corrected r^2 values for correlations between initial concentration in the water and sediment concentration suggest that the non-significant fits for either species with these parameters should not be taken seriously.

4. Discussion

Prior to the Chernobyl accident, the main sources of artificial radionuclides in freshwater habitats were fall-out from atmospheric testing of nuclear weapons and nuclear power stations, with minor

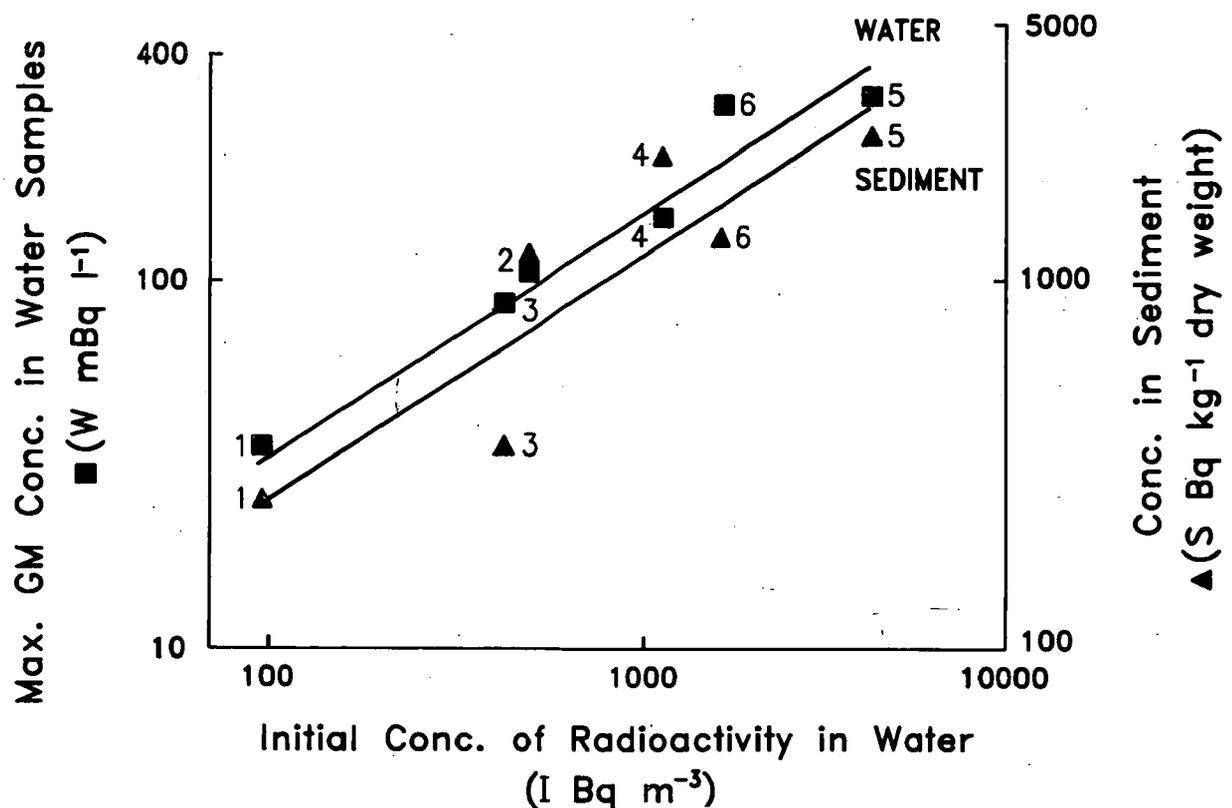


Fig. 7. Relationship between maximum monthly GM concentration obtained from the routine water samples (■) ($W \text{ mBq.l}^{-1}$) or concentration in sediment (▲) ($S \text{ Bq.kg}^{-1}$ dry weight) and initial concentration of radioactivity in water ($I \text{ Bq.m}^{-3}$). All data presented on logarithmic scales. (Lakes : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Regression lines given by : $\log_{10} W = 0.382 + 0.604\log_{10} I$ ($n = 6, r^2 = 0.952, P < 0.001$) ; $\log_{10} S = 1.160 + 0.635\log_{10} I$ ($n = 6, r^2 = 0.750, P < 0.02$).

Fig. 7. Relation entre le maximum mensuel de la GM de la concentration obtenue des échantillons de routine de l'eau (■) ($W \text{ mBq.l}^{-1}$) ou de la concentration dans le sédiment (▲) ($S \text{ Bq.kg}^{-1}$ poids sec) et de la concentration initiale de la radioactivité dans l'eau ($I \text{ Bq.m}^{-3}$). Toutes les données sont présentées en échelles logarithmiques. (Lacs : 1. Windermere, 2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Les lignes de régressions sont données par : $\log_{10} W = 0.382 + 0.604\log_{10} I$ ($n = 6, r^2 = 0.952, P < 0.001$) ; $\log_{10} S = 1.160 + 0.635\log_{10} I$ ($n = 6, r^2 = 0.750, P < 0.02$).

inputs from hospitals, research and radiochemical establishments. Discussion of this work would require a major review and is therefore beyond the scope of this paper, but some references are used when they are relevant to the findings of the present investigation. Of the long lived radionuclides in soluble form, Cs isotopes show the highest concentration factors in freshwater fish (Poston & Klopfer 1988). It is therefore not surprising that radio-caesium, especially ^{137}Cs with its long half-life, dominates the literature on post-Chernobyl radionuclides in freshwater fish. One problem with some of this information is that it is available only in unpublished theses or reports with limited circulation, rather than international journals. An excellent report from the Commission of the European Community is a valuable source of such information (Foulquier & Baudin-Jaulent 1989). In the following discussion, we have referred chiefly to work readily available in journals and cited reports only when it is essential. We have also emphasized work on salmonids because these are the subject of the present investigation.

4.1. Comparisons with other studies on Cs in freshwater fish, especially salmonids

^{137}Cs in brown trout increased from pre-Chernobyl values below 15 Bq.kg^{-1} to values above 500 Bq.kg^{-1} in the three lakes of the present study and above 1000 Bq.kg^{-1} in Ennerdale Water (Fig. 2). In marked contrast, values for charr in the three lakes never exceeded 350 Bq.kg^{-1} , and were generally lower than corresponding values for trout. Higher values, often exceeding 1000 Bq.kg^{-1} , were recorded in both perch and trout in Loweswater and Devoke Water with values in the latter lake being frequently higher than 2000 Bq.kg^{-1} (Elliott et al. 1992). The highest value recorded in the UK is 3000 Bq.kg^{-1} wet weight for brown trout from Loch Dee in Scotland (Leonard et al. 1990). Even higher values have been obtained from fish elsewhere in Europe with maximum values of 16030 Bq.kg^{-1} from Finland (Luukko & Särkkä 1989), 48000 Bq.kg^{-1} from Sweden (Hammar et al. 1987, Håkanson et al. 1989), 29572 Bq.kg^{-1} from central Norway (Lønvik & Koksvik 1990) and 55000 Bq.kg^{-1} from elsewhere in Norway (Strand et al. 1987). The highest value probably ever recorded is $410000 \text{ Bq.kg}^{-1}$ for perch from the cooling pond at Chernobyl (Kryshev 1992).

These maxima indicate that UK values were relatively low. The limit recommended for cessation of fishing in Sweden was 300 Bq.kg^{-1} before the Chernobyl accident but has now been increased to 1500 Bq.kg^{-1} (Håkanson et al. 1989), a value close to the maxima attained in the present study.

In contrast to wild fish, the ^{137}Cs content in stocked trout was low in both Devoke Water and Loweswater (Elliott et al. 1992), as found elsewhere in the UK (Mitchell et al. 1986, Camplin et al. 1989). This discrepancy between wild and recently stocked trout provides strong evidence that the food chain rather than the water is the main route of ^{137}Cs transfer to the fish. A laboratory study of the accumulation of ^{137}Cs by brown trout has shown clearly that uptake from water represents only a small proportion of the total caesium flux, most of which comes from the food (Hewett & Jefferies 1976). The latter authors also showed that if the activity in the fish was derived solely from the water, then an inverse relationship with size was observed. The lag between ^{137}Cs maxima in the water and maxima in both trout and charr (Fig. 4) was presumably due to the time required for ^{137}Cs transport through the sediments and food chain. Maximum values in lakes should therefore be attained first in the invertebrates, then in their chief predators (perch, trout, charr) and finally in the top carnivore in the lake (pike). Such an interpretation is supported by the lag in Loweswater between the maxima for trout and perch and the maximum for pike (Elliott et al. 1992). A similar lag in ^{137}Cs maxima in pike and their prey is reported for Swedish lakes (Håkanson et al. 1989) and Lake Constance on the German-Switzerland border (Linder et al. 1990).

When herbivorous and carnivorous fish have been compared in the same lake, ^{137}Cs values were markedly higher in the carnivores (Luukko & Särkkä 1989, Linder et al. 1990). Although several workers have stated that ^{137}Cs in fish increases with a progressive change of diet from zooplankton to benthos to fish (see references in Carlsson & Lidén 1978; Hammar et al. 1987, Hammar et al. 1991), this conclusion was not supported by data for perch and trout in Devoke Water, probably because of the overlap in their diets (Elliott et al. 1992). As perch ate more zooplankton than did trout, they should have had lower ^{137}Cs values, but the opposite occurred; ^{137}Cs being consistently higher in perch in both Devoke Water and Loweswater. ^{137}Cs values

were also higher in perch than trout from Llyn Trawsfynydd in North Wales (Preston et al. 1967, Hunt 1988), and were highest for perch amongst twelve species of fish from central Finland (Luukko & Särkkä 1989, Särkkä et al. 1991), four species in Lake Constance and nine species in Schreckensee (Linder et al. 1989).

The consistently higher values for perch are difficult to explain simply in terms of diet and could be due to different metabolic rates (Elliott et al. 1992). An example is provided by a comparison of the rates at which food is evacuated from perch and trout stomachs at different temperatures (Fig. 8). Although evacuation rates in both species are higher for a diet of invertebrates compared with fish, rates for trout are consistently higher than those for perch. The longer retention time in perch stomachs could therefore be one possible mechanism for their higher uptake of ^{137}Cs . Such an explanation

appears more plausible than differences in diet. Unfortunately, a similar explanation would not explain the lower values for charr compared with those for trout in the present study. Evacuation rates for charr feeding on invertebrates are consistently lower than those for trout with the same diet (Fig. 8). This suggests that levels of ^{137}Cs in charr should be closer to the high levels found in perch rather than the levels in trout. As this was not the case, it would appear that differences in diet were responsible for the lower levels in charr.

Other workers have recorded lower levels of ^{137}Cs in charr when compared with brown trout in the same lake in both Sweden (Hammar et al. 1987, Hammar et al. 1991) and Norway (Forseth et al. 1991). As mentioned above, differences in diet could be the explanation, provided that there were differences in radioactivity between the main prey types of the two fish species. Levels of radiocaesium have been shown to be higher in benthic invertebrates

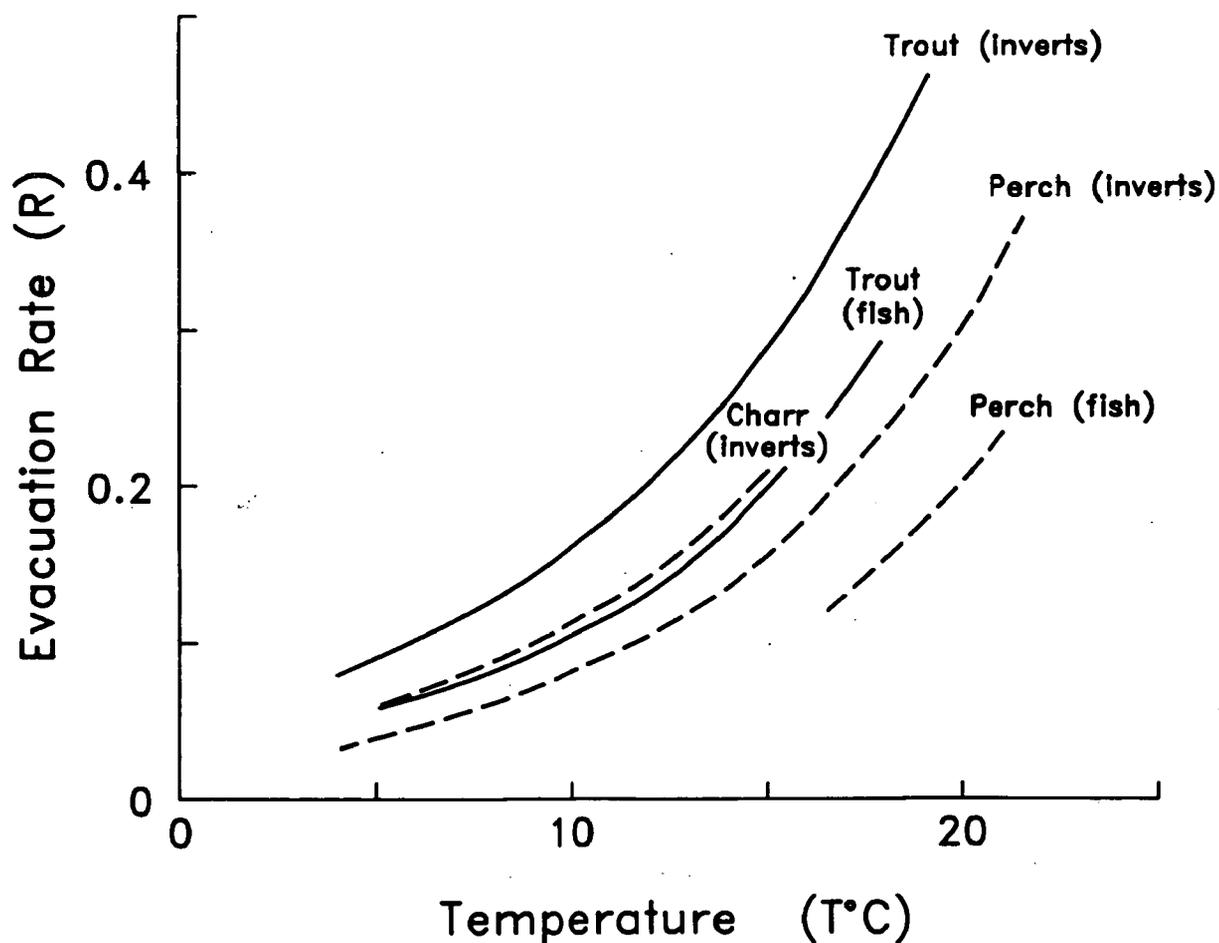


Fig. 8. Curvilinear relationship between gastric evacuation rate (R) and water temperature for brown trout feeding on invertebrates (from Elliott 1972) or fish (from Elliott 1991), perch feeding on invertebrates (from Persson 1979) or fish (from Persson 1981) and charr feeding on invertebrates (from Amundsen & Klemetsen 1988).

Fig. 8. Relation curvilinéaire entre le taux d'évacuation gastrique (R) et la température de l'eau pour des truites se nourrissant d'invertébrés (d'Elliott 1972) ou de poissons (d'Elliott 1991), et des perches se nourrissant d'invertébrés (de Persson 1979) ou de poissons (de Persson 1981) et des ombles se nourrissant d'invertébrés (d'Amundsen & Klemetsen 1988).

than in zooplankton in Swedish lakes (Hammar et al. 1991). However, in a detailed study in a Norwegian lake, Forseth et al (1991) have shown that although trout and charr had different diets in all years from 1986 to 1989, this caused major differences in the radiocaesium levels of their stomach contents in only one year (1986) with only minor differences in the other years. Their explanation for the differences between the two species is that brown trout have a higher food consumption, more than three times that of charr, and therefore have a higher intake of radiocaesium. Trout also have a faster rate of evacuation (Fig. 8) but presumably this does not balance the higher food intake.

Apart from brown trout in Wastwater, no significant relationships were obtained in the present study between ^{137}Cs content and fish weight (Fig. 3). This was in marked contrast to the results from the previous study of fish in Devoke Water and Loweswater (Elliott et al. 1992). In the previous study, there was a clear relationship between ^{137}Cs content and fish weight for trout from both lakes, perch from Devoke Water but not perch or pike from Loweswater. Although other workers have not always modelled this relationship, they have reported an increase in ^{137}Cs with fish size for several species, e.g. bluegills *Lepomis macrochirus* Raf. (Kolehmainen & Nelson 1969), large-mouth bass *Micropterus salmoides* (Lacépède) (Sprigarelli 1971), pike, perch and roach *Rutilus rutilus* (L.) (Carlsson & Lidén 1978), perch but not whitefish *Coregonus* sp. (Lindner et al. 1990), and several species from the cooling pond at the Chernobyl nuclear power plant (Koulikov & Ryabov 1992). As already mentioned in the results, such relationships must be taken into account when making temporal comparisons of ^{137}Cs in fish.

4.2. Ecological half-lives

The overall temporal patterns obtained for trout from the three lakes of the present study could be compared with those obtained from Devoke Water and Loweswater (Fig. 9). The peak in Devoke Water was clearly earlier than those for the other lakes but the rate of decrease in muscle radioactivity in Devoke Water was the lowest of the five lakes. Ecological half-lives for trout from Wastwater, Crummock Water and Ennerdale Water were not significantly different with mean values of 180 days, 194 days and 249 days respectively. The value for

Loweswater was significantly lower ($P < 0.05$) at 103 days and that for Devoke Water significantly higher ($P < 0.01$) at 825 days. Only one value was obtained for charr, that of 132 days for Ennerdale Water. High values have been obtained for trout (357 days) and charr (550 days) in a Norwegian lake (Forseth et al. 1991), for trout (range 344-610 days) and charr (range 331-816 days) in several Swedish lakes (Hammar et al. 1991), and for trout (3 years or 1095 days for ^{137}Cs , 1.3 years or 475 days for ^{134}Cs) in a Norwegian subalpine lake (Brittain et al. 1991). A high value was also obtained for perch from Devoke Water (2390 days) but not for either perch (94 days) or pike (112 days) from Loweswater (Elliott et al. 1992). Similar large differences are recorded in the pre-Chernobyl literature; for example, T_{50} was 500 days for brown trout from Trawsfynydd (Preston et al. 1967) and 804-950 days for perch, pike, roach and rudd *Scardinius erythrophthalmus* (L.) from Ulkesjon in Sweden (Carlsson & Lidén 1978).

These data are consistent with the range of 25-600 days given by Foulquier (1979) for ecological half-lives. A more recent report after Chernobyl by the same author (Foulquier & Baudin-Jaulent 1989) gave a more detailed breakdown of 100-200 days for rivers and 200 days to 3 years for lake fish. The large range is a consequence of the dependence of the ecological half-life on a number of factors. The ecological half-life (T_e) is derived from the difference in radioactivity in the flesh of fish taken from the natural environment, on two or more occasions. It can be represented by an equation:

$$C_f = C_f^0 e^{(-k_e t)}$$

where $k_e = (\ln 2)/T_e$, and C_f^0 is the initial concentration in the fish at the start of the observation period and C_f is the concentration at time t .

An equivalent equation can be derived from a mass balance around a fish assuming a logarithmic decay of activity in the food (for simplicity only input from the food is considered but the conclusion is not changed if input from the water is also considered):

$$C_f = k_{zf} \cdot C_z^0 \frac{(e^{-k_z t} - e^{-k_{fw} t})}{(k_{fw} - k_z)}$$

where k_{zf} is the rate of transfer of activity from food to fish; C_z^0 is the initial concentration in the food; k_z is the rate of reduction of activity in the

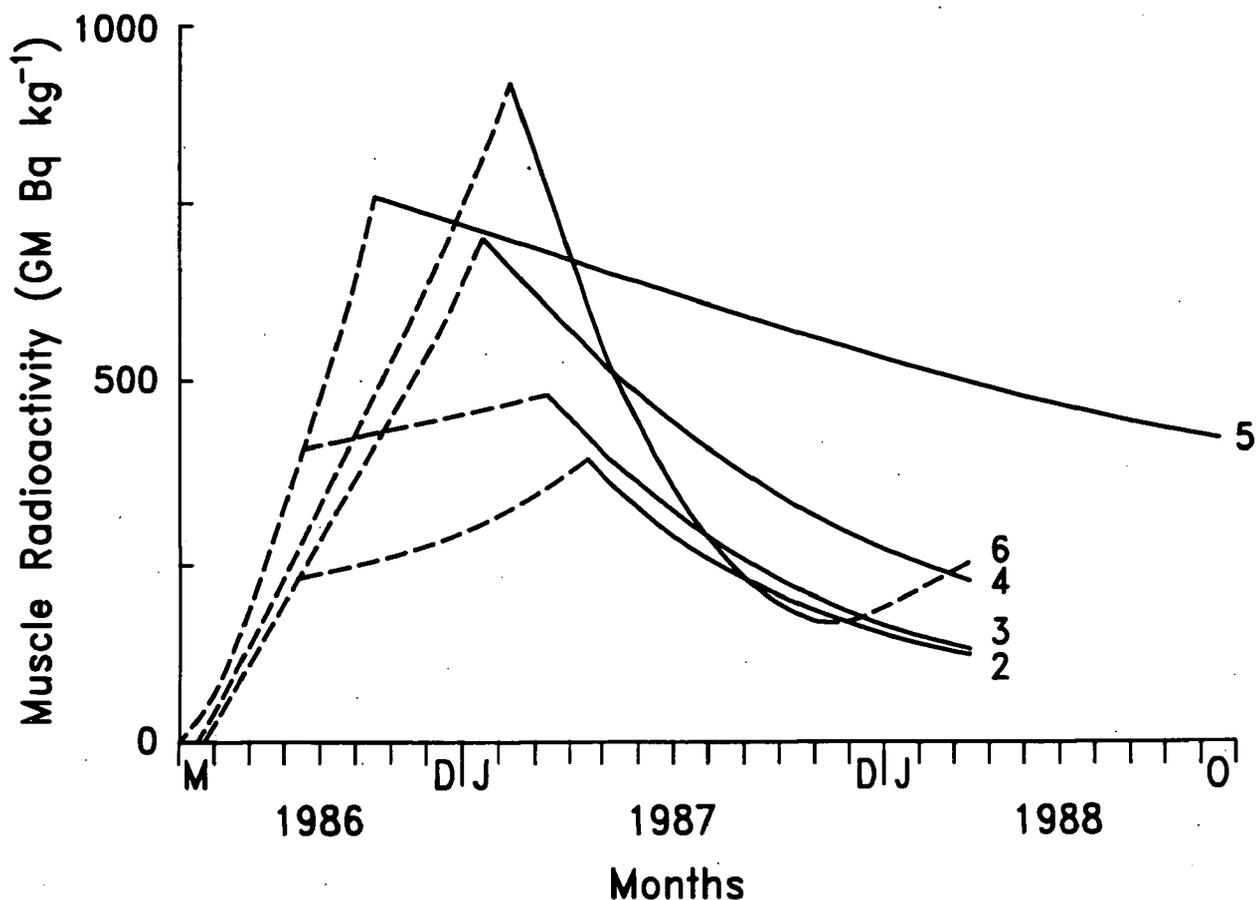


Fig. 9. Comparison of decreases in muscle radioactivity (GM Bq.kg^{-1}) in brown trout for 5 lakes (2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Broken lines simply show trends, solid lines are exponential curves given by equations in Table 1 (Lakes : 2, 3, 4) and in Elliott et al. (1992) (lakes : 5, 6).

Fig. 9. Comparaison de la diminution de la radioactivité du muscle (GM Bq.kg^{-1}) dans les truites pour cinq lacs (2. Crummock Water, 3. Wastwater, 4. Ennerdale Water, 5. Devoke Water, 6. Loweswater). Les lignes tirets montrent simplement des tendances ; les lignes solides sont des courbes exponentielles données par les équations du Tableau 1 (Lacs : 2, 3, 4) et par Elliott et al. (1992) (Lacs : 5, 6).

food with time ; and $k_{fw} = (\ln 2)/T_{50}$ where T_{50} is the biological half-life for the fish ; T_{b2} in the parlance of Foulquier (1979).

It is clear that k_e has at least two components. The first is a function of the rate of loss of activity from either the water or the sediment and the second is the biological half-life. Hence if the rate of removal of radioactivity from the environment is rapid, then the ecological half-life will approach the biological half-life. Conversely if the rate of loss from the environment is slow, then T_e will be much longer than T_{50} . Hence the much closer agreement between T_e and T_{50} observed in rivers, where the rate of removal of activity is high, compared to lakes, where loss rates can be slow resulting in T_e values up to several years.

Ugedal et al. (1993) have shown that biological half-lives for trout range from about 104-564 days depending on fish size and temperature. There is no

evidence for biological half-lives longer than 564 days for ^{137}Cs in trout, and this upper limit occurs only at low temperatures (4°C). The biological half-life decreases rapidly with increasing temperature so that the maximum value for large trout is of the order of 350 days by 8°C (Ugedal et al. 1993). Hence the very long ecological half-lives recorded in some studies must result from the maintenance of high activities of ^{137}Cs in the sediments and water, and hence in the food of the fish. Davison et al. (1993) showed that the half-life of radiocaesium in the water column of two lakes in the English Lake District after Chernobyl was of the order of tens of days, so that concentrations in the water column were at almost undetectable levels by the end of 1986. Calculations for a number of other lakes in the area, including Ennerdale Water, Wastwater and Crummock Water, suggest that their response would be similar and that high residual concentrations in

the water column could not result from slow rates of removal from the water column (Spezzano et al. 1993). Hence, the prolonged radiocaesium levels in the water, sediments and fish food could result from any one of, or combination of, three mechanisms: a long-term atmospheric or point source input; internal recycling of radionuclides within the lakes after a spike atmospheric input; or an extended radionuclide input from the catchment after a spike atmosphere input.

Fish from Trawsfynydd (Preston et al. 1967), and Ulkesjon (Carlsson & Lidén 1978) provide examples of the first of these three mechanisms. Trawsfynydd is the cooling pond of a nuclear power station from which it receives a continuous, licensed discharge of radioactivity. Similarly, data for Ulkesjon were for the period during which a continuing input of radiocaesium was received from the atmospheric deposition of nuclear weapons tests. As a result of these pseudo continuous inputs of radiocaesium, reported ecological half-lives are much higher than expected biological half-lives in these pre-Chernobyl studies. It is unlikely that a similar mechanism would explain the long ecological half-lives recorded in post-Chernobyl studies.

Although all the lakes except Devoke Water in the present study stratify seasonally, only Loweswater has a high enough productivity to allow the hypolimnium to become anaerobic and create conditions which might release radiocaesium from bottom sediments through displacement by ammonium ions (Evans et al. 1983; Comans et al. 1991). Even here, the non-equilibrium sorption of radiocaesium (Comans & Hockley 1992) suggests that Chernobyl radiocaesium was quickly immobilised onto clay minerals and would be unavailable in most sites for release by ammonium ions. For example, Hilton et al. (1992) found that only 20% of the absorbed ^{137}Cs in Windermere sediments collected in February 1987 was released after very violent chemical treatment in the laboratory. Hence internal recycling, the second of the three mechanisms suggested, is unlikely to be the cause of any prolongation of high dissolved radiocaesium levels in the water column.

A good example of the third mechanism is provided by a detailed study of Devoke Water (Hilton et al. 1993). The delayed input of radiocaesium from a peat bog in the catchment maintained ^{137}Cs levels

in the water column at high levels for several years after the Chernobyl deposition, when hydraulic flushing alone — regardless of any other loss processes which would normally operate — should have reduced the radiocaesium to unmeasurable levels. The continuing elevated ^{137}Cs levels in the water were probably the chief cause of the high levels of activity in the fish and the long ecological half-lives. A study of twenty-six lakes in the English Lake District showed that the phenomenon of extended inputs from peat bogs in the catchment was common for some western lakes with Ennerdale Water, Westwater and Crummock Water all showing evidence of the effect, but not as pronounced as in Devoke Water (Spezzano et al. 1993). Hence the ecological half-lives of ^{137}Cs in fish in these lakes are intermediate between those for fish in Loweswater and Devoke Water (cf. Fig. 9). Large catchment inputs may be also the ultimate cause of high ecological half-lives in Norwegian and Swedish lakes, but little direct evidence is available. However, the difference in the ecological half-life of ^{137}Cs and ^{134}Cs observed by Brittain et al. (1991) suggests that in at least one lake, significant input from the catchment occurred over a time scale which allowed a major part of the ^{134}Cs on the catchment to decay, while ^{137}Cs was transported into the lake. The third mechanism, extended radionuclide input from the catchment, would therefore appear to be the chief reason for very long ecological half-lives recorded for some fish species in post-Chernobyl studies.

4.3. Possible causes of variation in radiocaesium levels in the fish

The present investigation, and the earlier one on Devoke Water and Loweswater (Elliott et al. 1992), provide information on trout from six lakes and charr from four lakes, all within a relatively small area (Fig. 1). When the highest monthly mean values from these lakes were compared with five external factors, some relationships were significant whilst others were not (Figs 5, 6). Three of the external factors were interrelated, namely initial concentration of ^{137}Cs in the water, maximum monthly ^{137}Cs concentration obtained from the water samples, and concentration of ^{137}Cs in the sediment (Fig. 7). All three had a significant effect on mean values for trout or charr (Fig. 5). Hence the differences observed in the maximum activity in fish flesh are ultimately due chiefly to the different deposition received by each of the studied lakes.

The two remaining factors, namely mean potassium and calcium levels, showed no relationship with ^{137}Cs in fish (Fig. 6). Other workers have found that fish living in potassium-poor water accumulate more ^{137}Cs than do fish in potassium rich water so that the ^{137}Cs content of the fish is inversely related to potassium concentration in the water (earlier references summarized in Preston et al. 1967; later work by Kolehmainen et al. 1968, Thomann 1981, Luukko & Särkkä 1989, Särkkä et al. 1911). The absence of a significant relationship in the present study was probably due to the relatively low potassium concentrations in all six lakes. A similar explanation probably applies to the non-significant relationship for calcium levels in the six lakes.

The significant relationships between peak mean values in the fish and ^{137}Cs levels in the water and sediment illustrated the marked differences between the two fish species and between lakes (Fig. 5). Values were similar for charr and trout in Windermere but were consistently lower for charr in the other lakes. These differences between species were not due to faster gut clearance rates in charr and hence lower rates of uptake from the food. As shown above, gastric evacuation rates are slower, not faster, in charr compared with trout (Fig. 8) and therefore it would be expected, erroneously, that radioactive uptake should be higher in charr. Although no measurements were taken, it is probable that the zooplankton diet of the charr had a lower ^{137}Cs content than the zoobenthos diet of the trout. However, as discussed earlier (section 4.1), it has been shown for fish in a Norwegian lake that such an explanation may be too simplistic and that uptake of ^{137}Cs may be affected by other factors such as the higher food intake for either trout in summer or charr at lower temperatures in winter (Forseth et al. 1991).

As discussed earlier (section 4.2), Cs levels in the water and sediments can vary considerably between lakes in a small geographical area such as the English Lake District. This variation in the basis for the marked differences in ^{137}Cs levels in the fish of the six lakes of the present study. Differences in diet, in initial inputs of Cs to the lakes, in relative amounts of Cs in the water and the sediments, and in subsequent release of Cs from both the lake sediments and the catchment soils will all affect the uptake by the fish. Metabolic differences between fish species also play their part. It is therefore not surprising

that the modelling of changes in Cs concentrations in freshwater fish is a difficult and complex problem. The present study and the previous one on Devoke Water and Loweswater provide the first steps in a solution to this problem.

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