Sorption Studies of Radioiodine on Soils with Special References to Soil Microbial Biomass

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Abstract

In batch experiments with two types of soils, chernozem and podzol, radioiodine (^{125}I) showed an initial rapid sorption, followed by a long and slow further increase. Very little sorption ($R_d < 1$) was detected in clay minerals. Generally, higher R_d-values were observed for the chernozem soil, characterized by a higher amount of organic substance and of soil biomass. The sorption process was predominantly irreversible, the isotherms were linear at low ion concentrations and deviated from linearity starting at 10^{-5} mmol $\cdot$ ml^{-1}. Sorption ratio was found to increase with increasing volume to mass ratio. The composition of liquid phases (bidistilled water, synthetic soil water, rain water) highly affected iodine sorption. In experiments with KBr solution, the sorption of I$^-$ was found to be strongly preferred to Br$^-$.

Incubation of soil samples under varied conditions (decreased or increased soil biomass, O$_2$-concentration, incubation temperature, soil water content and storage conditions) delivered indications for the participation of soil microflora in iodine immobilization. Test with isolated soil bacteria and fungi showed that radioiodine can be incorporated by soil microorganisms under certain conditions only: Considerable uptake of radioiodine was found in washed (NaCl, CaCl$_2$) cells with both bacteria and fungi, but no incorporation was detected into cells incubated with radioiodine in the culture medium.

1. Introduction

The quantitative description of the migration rate of iodine isotopes, especially the long lived ^{129}I, in soil, requires the knowledge of the relevant soil parameters and their participation in iodine sorption. The understanding of the mechanisms of iodine immobilization and remobilization processes would help to introduce safety measures in order to minimize radioecological consequences.

Sorption studies of iodine on soils and minerals have been carried out by several investigators [1−6]. However, in some of these studies high concentrations of stable iodine were used and the results may not be applicable to the trace amounts of iodine in soils possible arising from ^{129}I depositions from nuclear facilities [3, 4]. In particular very little information is available on the process of desorption. In general, it was demonstrated, that soil organic matter is chiefly involved in the geochemistry of radioiodine [1−4, 7,

8, 9]. Because of the known interaction of soil organic matter and soil biomass the participation of microorganisms in iodine immobilization processes was suggested [10, 11, 12]. However, in these studies sterilization techniques such as autoclaving or gamma irradiation, have been applied which do not only eliminate microbial activity but also affect other nonliving soil constituents. Therefore the clear separation of geochemical and biological processes remained unsatisfactory.

In this work, the sorption and desorption behaviour of radioiodine on two different soil types was studied using a wide range of iodide concentrations. In particular, the contribution of soil biomass on radioiodine immobilization processes was investigated. The conceptual approach to elucidate the participation of soil microorganisms was the incubation of soil samples under conditions which are not expected to alter the non-living soil parameters (addition of glucose) or do so to very limited extent (CHCl$_3$-fumigatien, air drying of soil samples, different incubation temperatures) only.

As a further test for microbial effects, incorporation experiments were carried out with isolated soil bacteria and fungi.

2. Materials and methods

The soils used in the experiments were of the chernozem (clay silt soil) and podzol (humus sandy soil) types. The characteristics of the soils are given in Table 1. In each case the samples were taken from the upper 20 cm horizon of natural profile and were sieved (< 2 mm) before use. Clay minerals from Turkey used in sorption studies were koalinite, montmorillonite and chlorite-illite. ^{125}I was used as tracer in the initial concentrations of $1 \cdot 10^{-8}$ to $1 \cdot 10^{-3}$ mol $\cdot$ l^{-1}. It was obtained from Amersham International, Amersham Buchler GmbH, Braunschweig (Code IMS.30). The initial activity was 3.7 GBq $\cdot$ ml^{-1}, the volume 1 ml and I^{-1} ion concentration 7.6 µgI $\cdot$ ml^{-1}. 10 µl of the tracer solution were added to 100 ml of 0.1 M NaOH. The solution was divided into five 20 ml portions. Nothing was added to the first portion, 1 ml solutions containing 0.164 mg KI $\cdot$ ml^{-1}, 1.64 mg KI $\cdot$ ml^{-1}, 16.4 mg KI $\cdot$ ml^{-1} and 164 mg KI $\cdot$ ml^{-1} were added

Table 1. Characteristics of the soils used in the experiments

Soil type/location	Sand (%)	Silt (%)	Clay (%)	C_{org} (%)	pH (n-KCl)	Bio-mass $\mu gC \cdot g^{-1}$
Chernozem/ Jerxheim (FRG)	4.6	71.9	23.5	2.4	7.5	317
Podzol/ Didderse (FRG)	84.7	10.0	5.3	1.4	6.5	142

Table 2. Composition of synthetic soil water and rain water used in the sorption studies

Ions	Ion concentration $(mg \cdot l^{-1})$	
	Synthetic soil water (SSW)	Rainwater
Na^+	148.7	0.8
K^+	95.8	0.4
NH_4^+	–	1.1
Ca^{2+}	274.5	0.6
Mg^{2+}	44.0	0.2
Cl^-	486.4	1.5
NO_3^-	553.0	4.3
SO_4^{2-}	173.9	5.3
pH	6.1	4.2

to the second, third, fourth and fifth portions respectively. All samples were then filled to the 100 ml mark with bidistilled water. In other series of experiments the stock solutions were prepared using synthetic soil water (SSW) and rain water as the diluting solution. SSW has a composition similar to the soil water below maximum water holding capacity of the region where soil samples were taken. Table 2 gives the composition and characteristics of the water samples used.

The sorption experiments were carried out using the batch method [13]. 5 g of duplicate soil samples were suspended in centrifuge tubes in 10 ml of iodine ion solution prepared by 1:100 dilution of stock solution and shaken for 8 days or longer at 22°C. After separation of the two phases by centrifugation (6000 rpm) and filtration (0.22 μm) concentration change of the adsorbate in the aqueous phase was determined radiochemically using a NaI(Tl) detector. The distribution ratio R_d was calculated from the measured activities before and after shaking using the equations given in ref. [13].

For desorption studies, following the adsorption step, after separation of the two phases, 10 ml of water (bidistilled, SSW, or rain water) was added to the sample tube, shaken for the desired time, separated and counted as described above. Exchange experiments were carried out by shaking inactive solutions of various I^- concentrations for eight days. After phase separation carrier free iodine tracer solution was added to the solid phase and the samples were shaken for further eight days. From the decrease in the activity

of the liquid phase $R_{d, ex}$ of the I^- exchange was calculated.

To investigate the effects of soil biomass in iodine immobilization 10 g each of triplicate soil samples were suspended either in 20 ml of distilled water or synthetic soil water containing 63 kBq of ^{125}I ($\sim 10^{-12}$ mol I^{-1}). After shaking for at least 8 days solid and liquid phases were separated as described above. Standard errors in the batch experiments were below 5%, unless otherwise stated.

To investigate the effect of different incubation conditions soil samples were incubated with ^{125}I under aerobic and anaerobic (N_2) conditions. These effects were compared with soil samples which had been pretreated to reduce or eliminate soil biomass. Sterility was achieved by ^{60}Co gamma-irradiation (40 KGy) or by $CHCl_3$-fumigation. In another set of experiments tests about the influence of microorganisms on iodine immobilization were carried out to find a possible temperature optimum of the considered process. Soil samples were incubated for 8 d at 4, 10, 16, 20, 24, 28 and 36°C before determination of the R_d-values.

Indications on biological contributions to the iodine behaviour were also expected from the effects of different pretreatments of the two soil types. Soil samples were a) taken freshly from the field with moisture contents of 20 and 9% for chernozem and podzol, respectively, b) air dried within few days and c) stored in the laboratory (22°C) for about one year before use in the investigation of the R_d-values.

To study the influence of the quantity of soil biomass on iodine sorption soil samples of increased and reduced biomass were included in the batch experiments. The increase was achieved by the addition of glucose (about 20−30% of the added glucose-C is converted into biomass-C), and the reduction was obtained by fumigation with $CHCl_3$. Soil biomass was determined as described by Jenkinson and Powlson [14]. To test possible depletion effects, carrier iodine (KI) in concentrations of 10^{-8} to 10^{-3} mols $\cdot l^{-1}$ was added to distilled water and synthetic soil water. After incubation and separation of phases R_d-values were calculated for a chernozem soil as described above.

Further, incorporation experiments were carried out with isolated soil bacteria (Enterobacter sp., Bacillus sp. and Pseudomonas sp.) and fungi (Thielaviopsis sp., Verticillium sp., Fusarium sp., Trichoderma sp. and Penicillium sp.). The microorganisms were cultured either in complex or synthetic liquid medium. The synthetic medium was unbuffered or buffered with tris-buffer or phosphate-buffer for pH > 7. 12.5 kBq ^{125}I was added to 20 ml nutrient solution in 50 ml Erlenmeyer flasks either directly after the inoculation of the culture or after sufficient growth (bacteria = 10^8 cells $\cdot ml^{-1}$, fungi = 10 mg $\cdot ml^{-1}$ dry wt.). Adsorption was also investigated with harvested microbial cells which were either washed with H_2O or $CaCl_2$ (0.01 M) or NaCl (0.15 M) to remove extraneous materials. For incorporation experiments organisms were suspended in these liquids containing

the radioiodine. After incubation of dupliate samples for 3 days cells were separated by centrifugation (10,000 g).

3. Results and discussion

3.1. Sorption studies

The results of the kinetic studies are shown for chernozem and podzol soils in Fig. 1 and Fig. 2, respectively. In both soil types higher sorption is observed with synthetic soil water than with bistilled water. The initial sorption rate is much faster for podzol than for chernozem and reaches near steady state after about 4 days, while in chernozem the sorption process is slower and continues over the whole time period studied. This behaviour suggests the influence of organic components and microorganisms which are present in higher amounts in chernozem soil (Table 1). The adsorption as well as desorpiton results as a function of I^- ion loading in the solid phase for the two differ-

ent soil types are shown in Figs. 3 and 4 for chernozem and podzol, respectively. If the sorption values are taken to be true equilibrium constants, then they are not expected to show a variation with I^- concentration. It is seen in Figs. 3 and 4 that such constant sorption values exist for both soils. These regions extend to I^- ion concentrations of up to about 10^{-5} mmols $\cdot$ g^{-1} for chernozem and up to $5 \cdot 10^{-3}$ mmols $\cdot$ g^{-1} for podzol. The difference in the sorption properties of I^- ion with different sorption solutions is much more pronounced in the desorption than the adsorption results. Sorption is observed to be quite irreversible in the case of bidistilled water and reversible to a large extent for synthetic soil water and rainwater. It is interesting to observe the existence of an unexpected high R_d-region around $5 \cdot 10^{-3}$ mmols $\cdot$ g^{-1} both in the adsorption and desorption results for podzol soil. The pronounced increase in the sorption values suggests a change in the sorption mechanism which tends towards complete reversibility for all three types of sorption solutions used. To test the validity of this
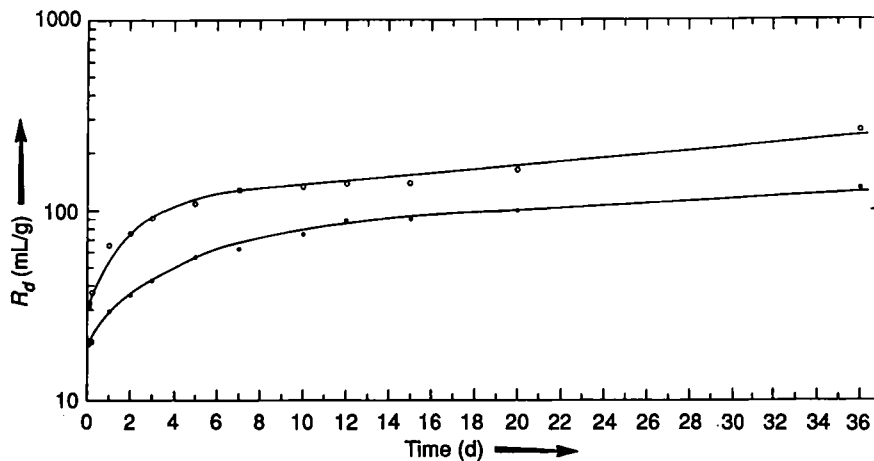

Fig. 1. Change of R_d with time for Chernozem soil. Initial iodine concentration, $(I^-)_0 = 1.0 \cdot 10^{-8}$ mol $\cdot$ l^{-1}. ● bidistilled water; ○ synthetic soil water.
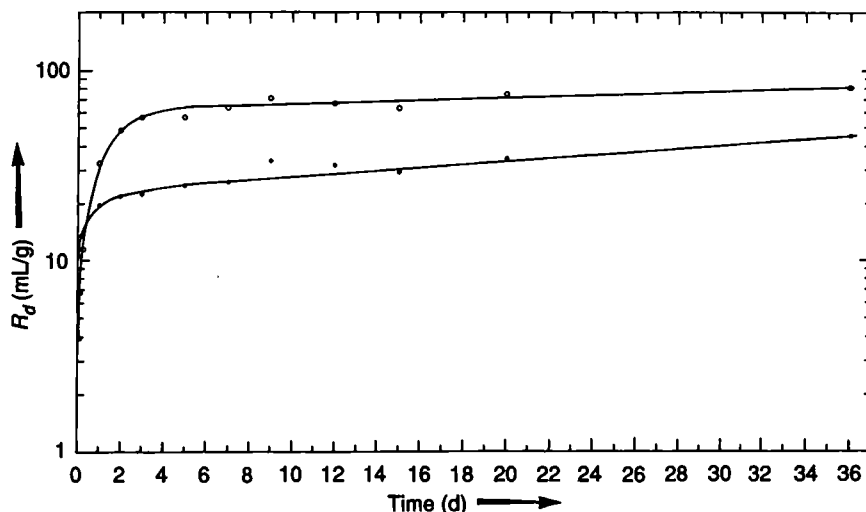

Fig. 2. Change of R_d with time for Podzol soil. Initial iodine concentration $(I^-)_0 = 1.0 \cdot 10^{-8}$ mol $\cdot$ l^{-1}. ● bidistilled water; ○ synthetic soil water.
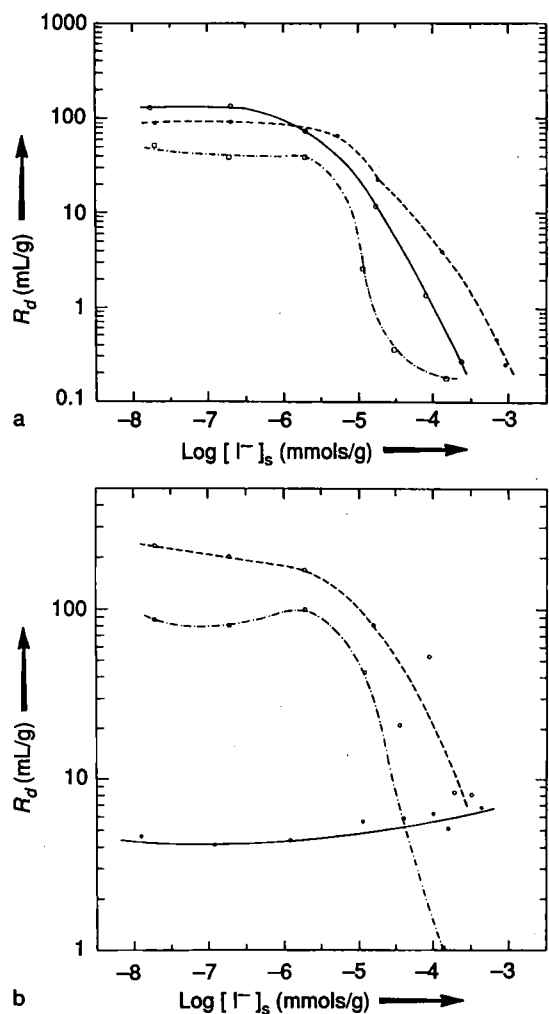
Fig. 3. Change of R_d values with I^- ion loading for Chernorem soil. ● bidistilled water; ○ synthetic soil water; ⊠ rainwater; a) adsorption; b) desorption.
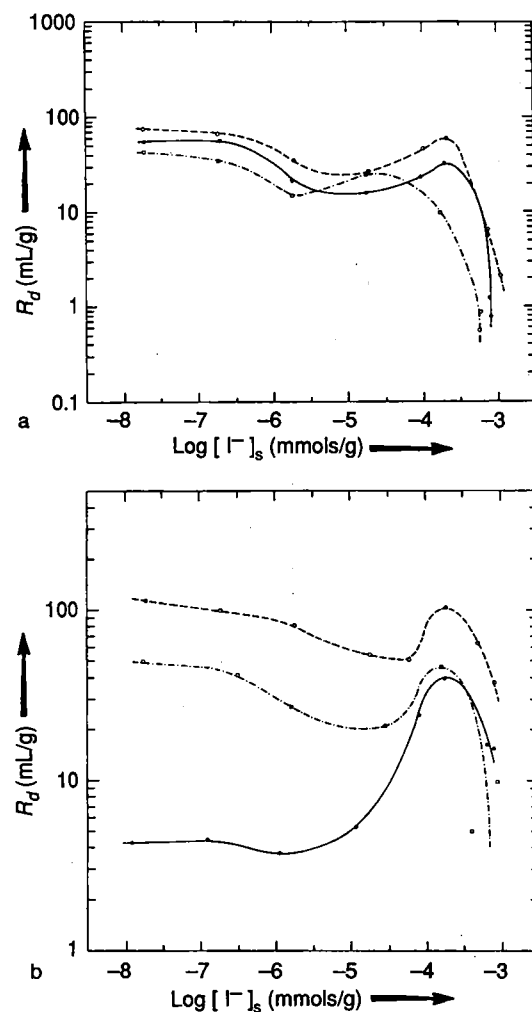

Fig. 4. Change of R_d values with I^- ion loading for Podzol soil. ● bistilled water; ○ synthetic soil water; ⊠ rainwater; a) adsorption; b) desorption.

observation tests were repeatedly carried out and re-vealed the same results. More work is needed to better understand and explain this experimentally observed high R_d-region.

The sorption isotherms for chernozem and podzol are shown in Figs. 5 and 6, respectively. The results for chernozem show deviations from linearity at high concentrations. Furthermore, reversible sorption is lower for chernozem than for podzol soils. Except for the high R_d-region discussed above the isotherms are linear for podzol. The adsorption and desorption re-sults are close to each other for synthetic soil water and rainwater, indicating reversibility.

The results of the sorption of I^- ion in synthetic soil water onto chernozem and podzol as a function of water volume to soil mass ratio, are shown in Fig. 7. An initial strong dependence of R_d on $V \cdot m^{-1}$ is ob-served between $V \cdot m^{-1} = 2$ to about 10 for both soil types. The gradual increase of R_d at $V \cdot m^{-1} = 10$ and higher is more pronounced for chernozem than podzol soil. Increasing the volume to mass ratio, probably better disperses and exposes the soil particles and their organic components to ions in solution resulting in

increasing sorption values. The presence of higher car-bon content and microorganisms lead to higher R_d values for chernozem.

The effect of the presence of different electrolytes in solution on the sorption of I^- has also been studied. Various concentrations of KBr, KCN, KNO_3 and K_2SO_4 have been used in the aqueous phase with no I^- carrier added. Contrary to loading experiments with inactive KI as shown in Figs. 3 and 4, the distri-bution ratio R_d remains constant at varying Br^-, CN^-, NO_3^- and SO_4^{2-} ion concentration. (Data not shown). The sorption process is reversible on podzol and mostly irreversible on chernozem. From the magni-tude of the R_d values the following order of influence on I^- ion sorption is deduced: $CN^- > NO_3^- \simeq Cl^- \simeq SO_4^{2-} > Br^-$ for chernozem, and $Cl^- > SO_4^{2-} > NO_3^- \simeq CN^- > Br^-$ for podzol.

The results of the exchange experiments are given in Table 3. The observation that in both soil types the ex-change ratio $R_{d,ex}$ is very small and of about the same magnitude, is quite striking. Not much exchange seems to take place between radioactive I^- and inactive I^- ion. One would have expected R_d values of comparable
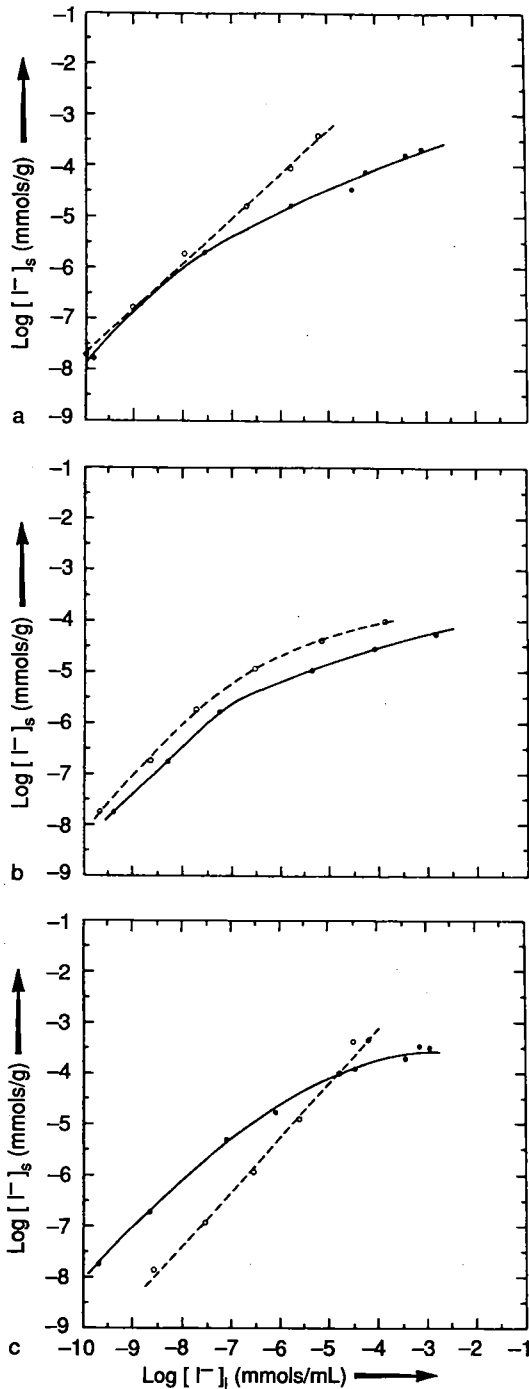
Fig. 5. Sorption isotherms for Chernozem soil. a) synthetic soil water; b) rainwater; c) bidistilled water; ● adsorption; ○ desorption.
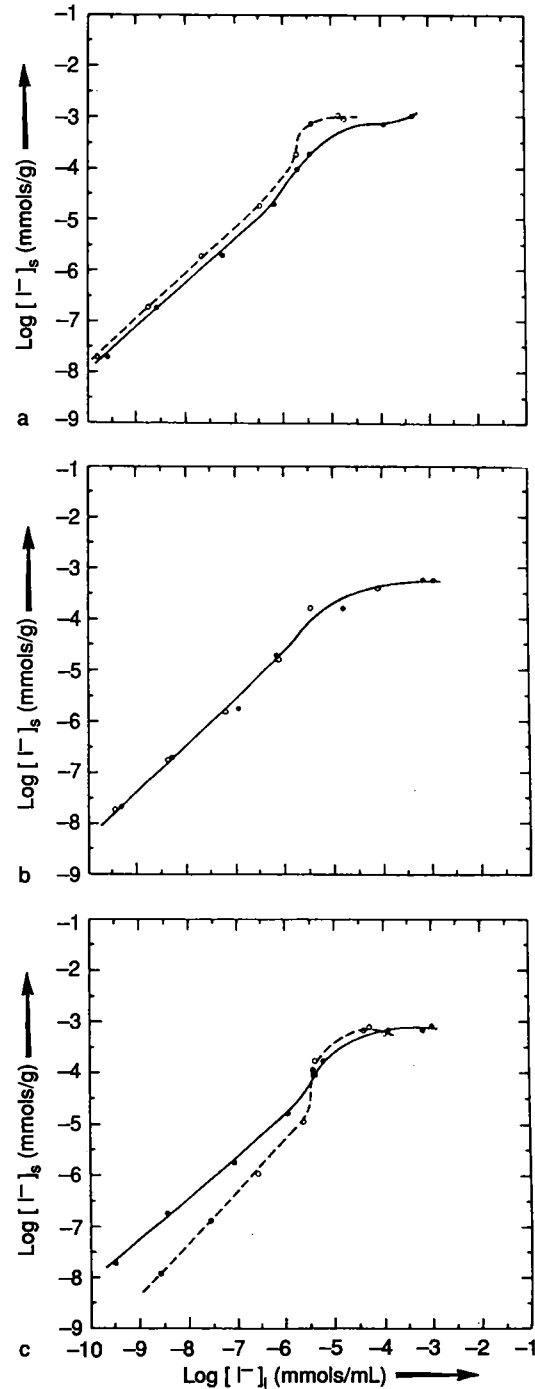
Fig. 6. Sorption isotherms for Podzol soil. a) synthetic soil water; b) rainwater; c) bidistilled water; ● adsorption; ○ desorption.

magnitudes as those obtained from the desorption experiments. Negligibly small $R_{d, \text{ex}}$-values observed are difficult to understand and to explain, especially for lower inactive I^- concentrations.

In another set of experiments the nature of the humic substances in the solid particles and their role in the adsorbing, desorbing and exchange processes were determined. Enough conc. HCl was added to the solutions to reduce the pH to about 2. After filtration the activity change in the liquid phases was determined. Part of the activity was precipitated with the

humic acids, while the humic substances not precipitated represent lower molecular weight fulvic acids. To determine the various components of the humic substances on the soil particles, 0.5 M NaOH solution was added to the solid phase following desorption. After phase separation, again concentrated HCl was added to the liquid phase. The precipitate consisted of humic acids and was separated from the solution containing fulvic acids by centrifugation. Humic substances present on the soil particles after NaOH treatment were insoluble humins. The results of humic
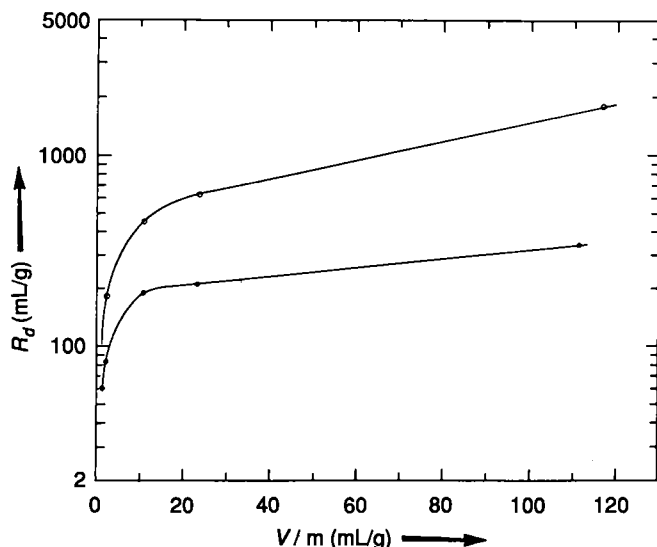
Fig. 7. Sorption of I^- ion in synthetic soil water onto Chernozem and Podzol as function of volume to mass ratio, $V \cdot m^{-1}$. ● Podzol; ○ Chernozem.

Table 3. Exchange behaviour of I^- ion on Podzol and Chernozem with no I^- carrier added radioactive ^{125}I tracer

Exchange solution	$(I^-)^0_{inactive}$ (mol $\cdot$ l^{-1})	Soil type	
		Chernozem $R_{d, ex}$	Podzol $R_{d, ex}$
Bidistilled	$1.0 \cdot 10^{-4}$	0.4	0.2
water	$1.0 \cdot 10^{-5}$	0.4	0.5
	$1.0 \cdot 10^{-6}$	0.1	0.4
	$1.0 \cdot 10^{-8}$	0.1	0.3
SSW	$1.0 \cdot 10^{-8}$	0.3	0.3

Table 4. Humic components in the sorption solution

Solution used	Soil type			
	Chernozem		Podzol	
	% Humic acid	% Fulvic acid	% Humic acid	% Fulvic acid
Adsorption solution	–	–	39	61
Desorption solution	39	61	30	70
Exchange solution	32	68	25	75

Table 5. Humic components of the soil particles after desorption (Original amount of humics of soil is taken as 100%)

Humic substances (%)	Chernozem	Podzol
Dissolved by NaOH (Humic and fulvic acids)	37	59
Undissolved by NaOH (Humins)	63	41
Precipitate with conc. HCl (Humic acid)	16	35
Not precipitated with conc. HCl (Fulvic acid)	21	24

Table 6. Sorption of I^- (R_d in ml $\cdot$ g^{-1}) on various clay minerals (Means of triplicate determination)

Clay type/C_{org} %	Bidistilled water		Synthetic soil water	
	R_d (Ads.)	R_d (Des.)	R_d (Ads.)	R_d (Des.)
Koalinite/ 0.04	0.5	1.4	0.6	1.7
Montmorillonite/ 0.13	0.2	0.2	0.6	1.7
Chlorite-illite/ 1.27	5.0	18.2	1.4	6.2

Table 7. Distribution ratios (R_d-values) of iodine in a Chernozem and in a Podzol incubated under different conditions

Incubation condition	R_d-values	
	Chernozem	Podzol
Insterile, aerobic	317.3 ± 25.0	73.0 ± 7.3
Insterile, anaerobic	26.7 ± 4.2	36.3 ± 6.2
Sterile (^{60}Co)	1.8 ± 0.03	4.3 ± 0.5
Sterile (CHCl$_3$)	6.2 ± 0.5	11.1 ± 2.6

component determination of the soil particles and in the adsorbing desorbing and exchange solutions are given in Tables 4 and 5, respectively. In Table 4 it is seen that the percentage of fulvic acid is higher in both soil types in the sorbing solution. This is expected as the lower molecular weight fulvic acids dissolve more easily in water. On the other hand in the study of humic components of the soil particles it is seen that podzol type soil contains more humic and fulvic acids than humins, whereas chernozem contains less acids and more humins. This can be explained by the higher amount of organio-mineral compounds due to the higher clay content and a higher biological activity and not at least to the history of the soil formation of the chernozem. Humins are very stable and are poorly extractable with NaOH. In absolute terms (Table 1) chernozem type soils contain more organic components and biomass than podzol soils. This reflects the higher R_d values in I^- ion sorption on chernozem soils.

The results of sorption of I^- ion on various kinds of clay minerals are given in Table 6. The very small R_d values observed, when compared to those obtained for chernozem and podzol soils, indicate that the role of inorganic components in the sorption of I^- is very

small. Chloride-illite type clays with a higher content of organic carbon sorb more iodide than koalinite and montmorillonite clays, the sorption process seems to be irreversible for clays.

3.2. Effects of soil biomass

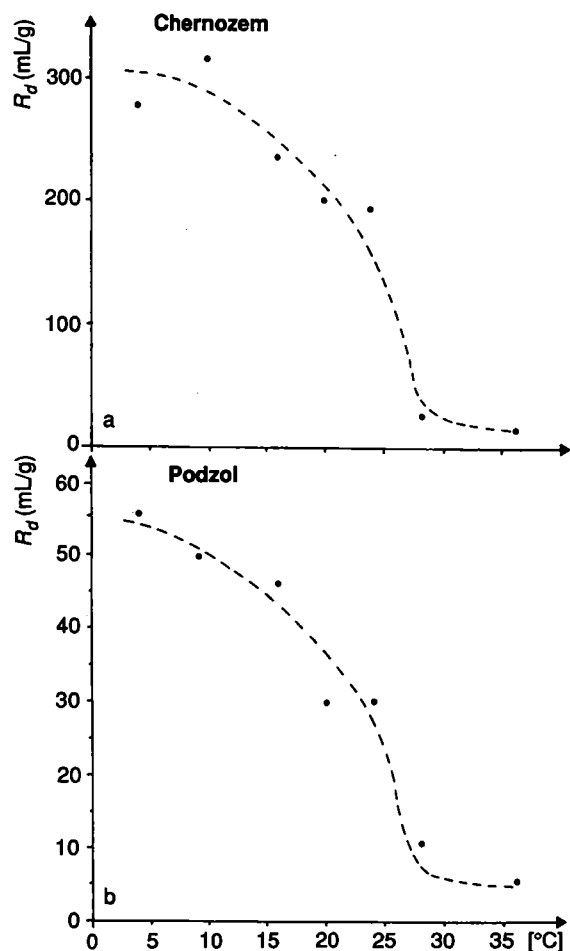As shown in Table 7 the sterilization of soils caused a considerable decrease of the R_d-values in both soil

Fig. 8. R_d-values of radioiodine in Chernozem (a) and Podzol (b) related to different incubation temperatures.

Fig. 9. R_d-values of radioiodine in Chernozem (a) and Podzol (b) resulting from different treatments before incubation.

types. The ^{60}Co gamma-irradiation was more effective than the $CHCl_3$-fumigation. Incubation of soil samples under anaerobic conditions (N_2-atmosphere) also caused a reduced iodine fixation, e.g. about 50% reduction of the R_d-values in the podzol and more than 90% reduction in the chernozem, indicating that iodine fixation may be mediated by enzyme systems (iodine oxidase) associated with microbial metabolism as was suggested elsewhere [9, 11]. In general, the treatments were more effective in the chernozem than in the podzol. These results support the validity of the assumption that soil biomass is participating in iodine immobilization mechanisms.

Another indication for biologically mediated processes is the observed dependence on the incubation temperature. Biologically catalyzed reactions principally show a more or less sharp temperature optimum and fall off when the temperature is increased beyond this point. There is an obvious dependence of iodine fixation on the incubation temperature as demonstrated in Fig. 1. However, the expected temperature optimum did not appear in this investigation. The highest iodine fixation was observed at relatively low temperatures (4°C and 10°C respectively) and it is remarkable that R_d-values decrease considerably above 28°C. Soil biomass is most stable in lower tem-
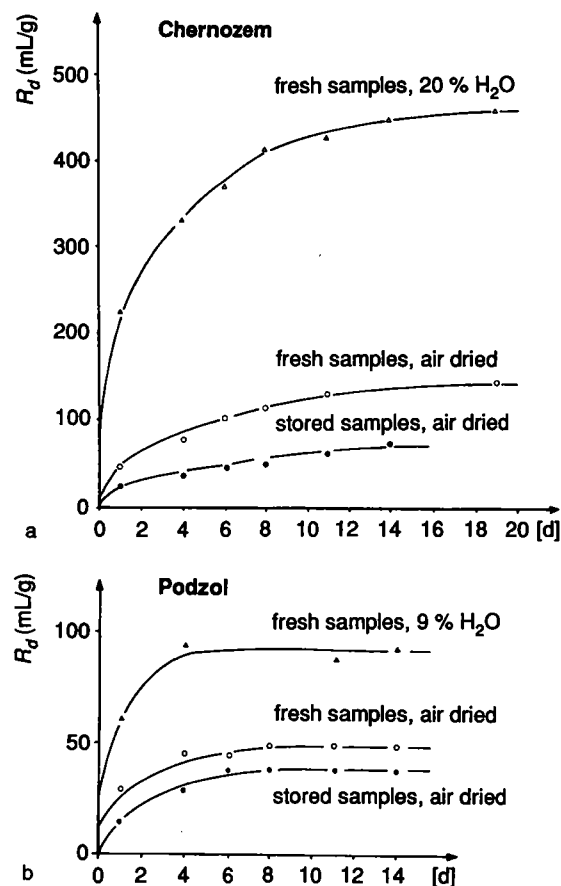
perature regions and will be slowly degraded when the temperature increases as was demonstrated by Anderson and Domsch [15]. Therefore it can be concluded, that resulting R_d-values are related to the temperature dependent survival of microorganisms. However, some influence of the temperature on the binding of iodine to soil itself must be considered also.
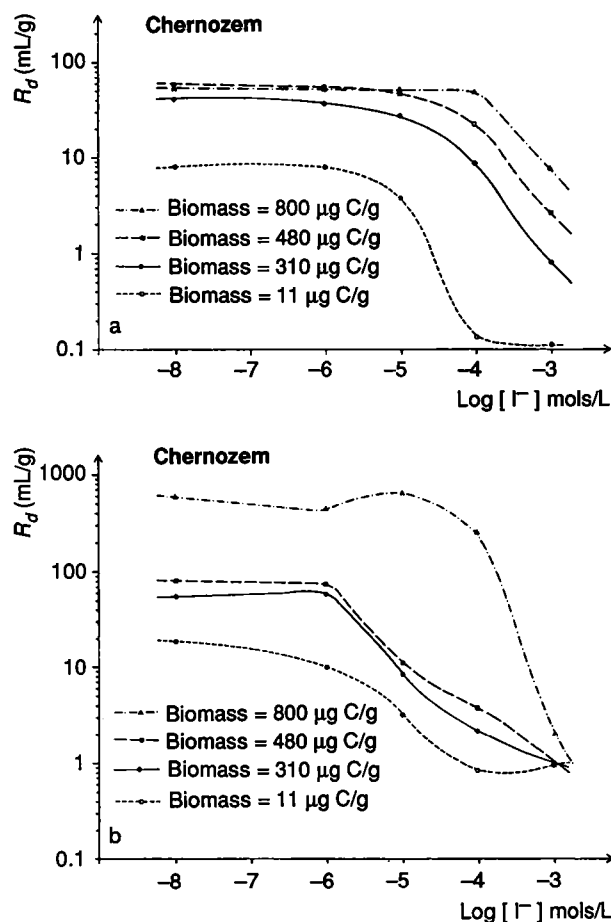
The experiments on the effects of soil moisture and storage conditions exhibited significant differences in the kinetics of iodine sorption (Fig. 9). Immobilization of radioiodine in soil samples of medium moisture content occurred at a much higher rate and intensity than in samples which had been previously air-dried. The lowest rate was observed in soil samples stored under room conditions for about one year. The natural process of drying is known to reduce microbial biomass content [16, 17]. Therefore the results support the assumption that iodine sorption is associated with the number of microorganisms and they emphasize the importance of the history and storage conditions of the soil material under investigation.

The effect of increased and reduced soil biomass on the R_d-values of chernozem and podzol is shown in Table 8. The data show that R_d-values are highly influenced by the quantity of soil biomass. The effects are more pronounced in chernozem (higher biomass content) than in the podzol. In chernozem, for example, the correlation between soil biomass and R_d-

Table 8. Effects of increased and reduced contents of soil biomass on the R_d-values in a chernozem and in a podzol. Values of the normal biomass and corresponding R_d-values are set in blod

Chernozem				Podzol			
Soil biomass		R_d-values		Soil biomass		R_d-values	
µgC/g d.w.	relative	absolute	relative	µgC/g d.w.	relative	absolute	relative
13	4	21±1 a*	8	3	2	9±1a	30
27	9	25±1 b	9	11	8	14±1e	45
317	100	276±12c	100	142	100	31±2c	100
431	136	375±22d	136	199	140	36±2d	115
546	172	385±19d	139	232	163	37±4d	118
716	226	376±30d	136	292	206	45±4d	144

* Values followed by the same letter are not significantly different (p = <5%).

Fig. 10. Interaction of soil biomass content and carrier I^- concentration of the equilibrium solution (a: bidistilled water, b: synthetic soil water) in radioiodine immobilization (R_d-values).

Table 9. Removal of ^{125}I from H_2O, $CaCl_2$ or NaCl solution by different soil bacteria and fungi

Microbial genus	% Removal of ^{125}I		
	H_2O	$CaCl_2$ (0.01 M)	NaCl (0.15 M)
Enterobacter	0	0	71
Bacillus	0	33	66
Pseudomonas	0	0	57
Thielaviopsis	0	35	41
Verticillium	0	5	16
Fusarium	35	28	21
Trichoderma	31	23	12
Penicillium	35	10	5

value is linear in the range from 4 to 136% of its original content. A further increase of soil biomass is ineffective, indicating depletion of the radioiodine applied.

To test this assumption experiments were carried out with the addition of carrier iodine in concentrations up to 10^{-3} mols $\cdot$ l^{-1} using distilled water and synthetic soil water as equilibrium solutions respectively. As indicated by Fig. 10a the highest level of

sorption was found when biomass was increased by 50% (480 µg $C \cdot g^{-1}$) and remained nearly constant over a concentration range up to 10^{-5} mol $\cdot$ l^{-1}. Also in these concentration range the highest biomass content (800 µg $C \cdot g^{-1}$) was ineffective.

At higher concentrations (10^{-4} to 10^{-3} mols $\cdot$ l^{-1}) R_d-values show a clear dependence on the biomass content; i.e. soil samples with a considerably increased biomass content have the capability to fix radioiodine with increased efficiency. These sorption characteristics are much more pronounced if soil samples are incubated in synthetic soil water (Fig. 10b). In this case the sorption of radioiodine depended clearly on the soil biomass content throughout the whole concentration range. Especially high iodine amounts were immobilized in soil samples with the highest biomass content (800 µg $C \cdot g^{-1}$).

On the basis of these observations the role of soil microorganisms in iodine immobilization is emphasized, but the ionic composition (physico-chemical soil properties) of the soil solution may be of additional importance. Cations for example influence the solubility of humic acids by forming "humates".

The incorporation experiments with single isolated soil microorganisms revealed that both bacteria and fungi were unable to remove radioiodine from the nutrient media, irrespective of the composition of the

media and irrespective whether iodine was applied from the beginning of the culture or to the already existing biomass (data not shown). This failure of iodine uptake might be due to the formation of nonavailable iodine complexes with the nutrients or/ and with substances exuded from the cells. However, a significant fraction of iodine was removed by the washed cells free of culture media (Table 9). The highest uptake was observed when bacterial cells were exposed to an isotonic NaCl-solution containing the radioiodine. After suspension of cells in H_2O-solution only Fusarium sp., Trichoderma sp. and Penicillium sp. were effective in removing radioiodine. Using $CaCl_2$ as an electrolyte fungal cells and Bacillus sp. incorporated radioiodine at a low to medium extent.

Though demonstrating, that radioiodine can be incorporated into soil biomass under certain conditions, these findings provide only limited evidence whether soil microorganisms participating in iodine immobilization act in a direct or an indirect way. It should be considered in this connection, that the experimental conditions used differed in may respects from natural soil conditions. Further, the microorganisms tested represented only a small part of the soil microflora and they will behave differently in pure cultures than in complex soil environment.

Acknowledgements

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