

Microbiological Treatment of Oil-Containing Radioactive Waste Prior to Cementation

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Abstract—Microbiological oxidation of liquid organic radioactive waste (ORW), spent vacuum and transformer oils, is described. The physicochemical properties of oils and their structural-group composition before and after using in the process cycle were studied. Changes in the structural-group composition and physicochemical parameters of oils upon microbiological treatment were determined. The distribution of radionuclides between the organic and aqueous phases upon microbiological treatment was studied by the example of model ORW containing ¹³⁷Cs, ⁹⁰Sr, ²³⁸Pu, and ²⁴¹Am. The compatibility of oils subjected to microbiological treatment with the cement compound intended for ORW disposal was evaluated. The biological step of the radioactive oil processing leads to a decrease in the weight and volume of the organic radioactive waste and increases its amount that can be incorporated into the cement compound (to 30 vol %) without deterioration of the strength characteristics of the cement.

Keywords: oil-degrading microorganisms, liquid organic radioactive waste, transformer oil, vacuum oil, tributyl phosphate, cement compound, biotechnology, radionuclides

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Nuclear fuel cycle enterprises face the problem of management of organic radioactive waste. Its chemical and radiochemical composition is specific for each enterprise. The most complex components of liquid organic radioactive waste are vacuum and transformer oils containing small amounts of β, γ -emitting fission products, activation products, ¹⁴C, and ³H [1]. In addition, the oils can contain U and Pu [2]. The complexity of management of such waste is caused by hydrophobic properties of oils and their multicomponent variable composition (more than a hundred of components belonging to classes of linear and cyclic alkanes, alkenes, arenes, etc.). Along with hydrocarbon components, oils can also contain antioxidant and emulsifying additives and other microimpurities [3]. Modern methods for processing such oils are based on degradation, purification, and immobilization (or solidification) [4].

ORW incineration is performed at temperatures from 850 to 1200°C with a large excess of oxygen (up

to 75%). It involves oxidation of the waste with oxygen to carbon dioxide, water, and ash residue containing radionuclides [5, 6]. Fakel incineration installation is now under operation at the Radon Federal State Unitary Enterprise. The drawbacks of the waste incineration are complex gas treatment system and high cost of plasma torch operation.

Another method for ORW treatment is ozonation involving breakdown of organic compounds under the action of a mixture of ozone [7] and hydrogen peroxide [8]. The reaction mixture can be additionally subjected to UV irradiation and heating to 100–110°C. The advantages of this method are moderate operation temperatures and relatively low cost. The drawbacks are harsh conditions and toxic highly reactive agents causing the equipment corrosion and making necessary its frequent repair, which makes the process more expensive.

One of the ways of ORW processing is treatment with concentrated acids [9]. This treatment results in

the formation of a multiphase system consisting of insoluble salts, acid tar, and purified organic phase. The radionuclides are mostly concentrated in the tar, which can be incorporated into cement and organic host materials in an amount of up to 42%. The main drawbacks of this method are the use of aggressive concentrated acid solutions and large volume of the secondary waste.

Sorption treatment is mainly applied to aqueous radioactive waste containing organic components and involves treatment of the waste with sorbents (bentonite, titanium phosphate), salts tending to coagulation (mainly ferrocyanides) [10, 11], or coarsely porous adsorbents (silica gel, alumina gel) [12]. Sorption treatment is highly efficient for wastes with constant chemical composition and can be complicated in the presence of organic components because of their effect on physicochemical properties of the sorbents.

The main modern approach to ORW management, suggested by IAEA, is immobilization in a solid host material. Modern developments in this field consist in improvement of procedures for ORW pretreatment prior to cementation to reduce their volume, make them less hydrophobic, and increase the percentage of incorporation into the cement host material. Cementation can be performed both without preliminary treatment via direct mixing of the oil with a cement mortar or with preliminary adsorption of organic components onto porous materials. In the first case, the incorporation of hydrophobic components usually does not exceed 7–10%, and in the second case it can be increased to 45 vol % [13, 14].

The procedure for saturation of a cement host material by impregnation with organic waste, suggested at Radon and the Mayak Production Association, allows 50% incorporation to be reached [15]. The advantage of such approach is conversion of the dangerous substance into an immobile solid form, ensuring its long-term safe storage. The procedure is relatively cheap. Its most significant drawback is a considerable increase in the final waste volume (by a factor of at least 2). Furthermore, cement with the incorporated organic waste can undergo biological degradation making it less reliable as host material for immobilization [16].

This study deals with the behavior of organic components and radionuclides in radioactive oils upon microbiological treatment as a new approach to ORW solidification in a cement host material, followed by its storage/disposal.

The microbiological approach is based on the ability of microorganisms to oxidize petroleum products to carbon dioxide with cell enzymatic systems in the course of respiration, effecting emulsification of hydrophobic systems due to exopolymers and thus making them more available for a cell. Microbiological methods are widely used in treatment of oil spills, in technologies for enhancing the crude oil output, and in treatment of wastewaters contaminated with organic compounds. The microbiological methods are attractive because of their low cost and relative simplicity and of no need for using expensive aggressive agents. The drawbacks of the methods are relatively low process rates, possible implementation problems associated with toxic effects of waste components, including radioactive substances, and need for diluting the waste with an aqueous medium to ensure the microorganism growth [17].

Among studies on biological oxidation of ORW, papers on breakdown of organic complexones such as EDTA and DTPA, of their complexes with various metals [18], and of irradiated surfactants [19] have been reported. The biological consumption of a complex organic mixture (TBP, ethyl acetate, hexane, acetone, ethanol) containing ^{241}Am , ^{137}Cs , and U was studied by Ferreira [20]. The author reported that microbial communities assimilated within 20 days up to 80% of TBP and up to 75% of ethyl acetate and took up Am, U, and Cs from the solution [20].

Microbiological treatment of radioactive waste containing acetonitrile in the form of a 10% aqueous solution was studied at the Lawrence Berkeley National Laboratory (the United States). Based on the physicochemical and kinetic parameters of the process, Stringfellow and Komada developed an efficient laboratory installation for biological treatment of concentrated toxic and radioactive mixed wastes [21].

The procedure for treatment of oil-containing radioactive waste, studied by Filatov and Altunina [22], involves its mixing with tap water in 1 : 1 ratio. As stated in that paper, the microorganisms present in the oil ensured 80% consumption of the oil-containing waste. On the other hand, in our opinion, some methodical features of the procedure in that study led to low parameters of the biosystem reproduction, which significantly limited the results obtained. For example, according to [22], 1 cm³ of a solution of hydrocarbon-oxidizing microbial association with the biological material concentration of $(8-9) \times 10^3$ CFU dm⁻³ was

introduced into a 1.5-L vessel. Counting on the final solution volume, the concentration is about 15 CFU dm^{-3} , which is approximately 100 times lower than the amount of the biological material required for efficient and rapid microbial growth [23, 24].

For biological treatment of spent cutting fluid contaminated with radionuclides, Taylor and Freestone [25] suggested a procedure involving biodegradation, ultrafiltration of the cells and organic phase, and incorporation of the organic phase in a cement host material [25].

Thus, microbiological methods for radioactive waste treatment appeared to be successful in some cases, but are today at the early stage of their development.

EXPERIMENTAL

In the course of the study, we performed model experiments with nonradioactive oils, both fresh and discharged after completion of the working cycles (Table 1). To simulate the installation operation under industrial conditions, we used as an organic phase a real waste containing TBP in kerosene, contaminated with ^{238}Pu ($5.33 \mu\text{M}$) and ^{241}Am ($0.104 \mu\text{M}$), to which scintillation liquid and spent GK oil were added. The waste was additionally spiked with ^{137}Cs by adding 1 cm^3 of its neutral aqueous solution. The volume ratio of the waste, scintillator, oil, and water was 6 : 1 : 2 : 1, respectively. The mixture was vigorously stirred for 2 days. After the stirring completion, the resulting organic phase was a dark orange liquid undergoing phase separation in more than 4 h.

The structural-group composition of oils and its change upon biodegradation were determined using IR and UV-visible spectrometry. The IR spectra were recorded using a KRS-5 cell with the layer thickness of 0.1 mm in the range $4000\text{--}600 \text{ cm}^{-1}$ with a Specord M80 device (Carl Zeiss). The UV-visible spectra were taken with a Cary 100 device (Agilent Technologies) in the range 200–510 nm using a quartz cell with the layer thickness of 1.0 cm. Differential measurements were used for interpreting the IR and UV spectra. The samples were diluted to the required concentration with hexane (ultrapure grade) for taking the UV spectra and with chloroform (chemically pure grade) for taking the IR spectra. When processing the results, the diluent spectrum was subtracted from the solution spectrum with the Excel program using the calibration

Table 1. Model organic radioactive wastes

| Oil | Producer (source) |
|-------------------------|--|
| VM4 (I, S) ^a | Ryazan Pilot Plant of Petroleum Products |
| VM1 (I, S) | Temporary repository of radioactive waste treatment station, Frumkin Institute of Physical Chemistry and Electrochemistry, Russian Academy of Sciences |
| GK (I) I5a (I) | OOO Nekton-Sia |
| GK (S) | Nizhny Novgorod Enterprise of Magistral Electrical Networks, Sechenovo-1 200-kV substation, T-2 |

^a (I) Initial oil before using in the process cycle and (S) spent oil discharged after using in the process cycle.

based on the reference bands at 1700, 1600, 1460, 1380, 1030, and 720 cm^{-1} . For processing the spectra, we also used the methods for determining the structural-group composition and coefficient of branching of linear hydrocarbon chains, described in [26, 27].

The structural-group composition of the model radioactive waste was also studied by the $n\text{--}d\text{--}M$ method [28] (based on determination of the refractive index, density, and mean molecular mass) and by IR spectrometry. From the data obtained, we calculated the gravimetric content of aromatic (C_a , wt %) and paraffin structures (C_p , wt %). The content of naphthene structures was calculated as $100\% - C_a - C_p$ [28]. The difference between the results obtained by the three methods in determination of the structural-group composition of the oils did not exceed 7%.

Sorption of radionuclides on the biomass was performed by adding 1 cm^3 of the aqueous phase of the metal to a suspension of cells (concentration 0.01 g cm^{-3}) and vigorously (120 rpm) stirring for 24 h with an Elmi 105 orbital shaker. After that, the biomass was separated by centrifugation on an Eppendorf centrifuge at 6000g, and the radionuclide content in the supernatant was determined. From the data obtained, we calculated the distribution coefficient K_d ($\text{cm}^3 \text{ g}^{-1}$) using the formula $K_d = [(A_0 - A_{eq})/A_{eq}](V/m)$ and the degree of decontamination of the aqueous phase from radionuclides, $DD = [(A_0 - A_{eq})/A_0] \times 100\%$.

The specific activity of the radionuclides was measured with a TriCarb 3180 TR/SL low-background liquid scintillation spectrometer (for α - and β -emitting radionuclides) and with a γ -ray spectrometer equipped with an Ortec GMX25P4 semiconductor HPGc detector.

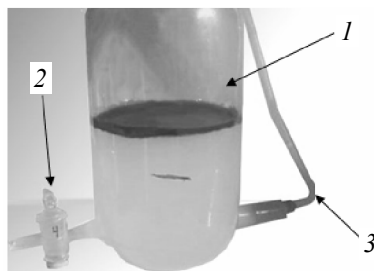


Fig. 1. Laboratory installation for ORW biodegradation. $V = 200 \text{ cm}^3$. (1) Main vessel, (2) discharge stopcock, and (3) air supply tube.

A mixture of the aqueous and organic phases was subjected to solidification. As control samples, we used samples of the composition cement + distilled water + spent oil. PTs M-500 D0 Portland cement was used as the host material. Distilled water or aqueous phase after the cultivation (cultural fluid without separation of microorganisms) was used as the aqueous phase. The initial oils, spent oils, and oils after biological oxidation were used as the organic phase.

A concentrated CaCl_2 solution was added to the compound to enhance the strength and hardening rate of the samples. The volume of this solution was taken into account when calculating the water-to-cement ratio.

To prepare a compound, the aqueous phase and excess organic phase were added to a weighed portion of cement. The components were thoroughly mixed, and the mixture was allowed to stand for 30 min. After that, the excess organic phase was separated and weighed. The remaining compound was placed in a mold to obtain three 8 cm^3 specimens. Then, the mold was placed in a sealed container with water to ensure hydrostatic conditions (water was present in the container but was not in contact with the specimens). The specimens were kept in the container (with visual monitoring of the solidification), and the strength was measured on the 28th day (by Radon staff members)

Table 2. Structural-group composition of initial oils, content of main components, wt %^a

| Oil | Aromatic structures | | | | Paraffins | Naphthenes |
|-----|---------------------|------|------|----------|-----------|------------|
| | B | N | P | Σ | | |
| VM4 | 8.56 | 7.81 | 1.14 | 17.51 | 77.68 | 4.80 |
| VM1 | 1.21 | 0.15 | 0.21 | 1.56 | 85.77 | 12.67 |
| GK | 1.82 | 1.40 | 0.16 | 3.37 | 46.83 | 49.80 |
| I5a | 1.86 | 1.79 | 0.38 | 4.04 | 40.43 | 55.53 |

^a (B) Benzene rings, (N) naphthalene structures, and (P) phenanthrene structures; the same for Table 3.

with a Testing Cybertronic 500/50 kH machine. The mean measurement uncertainty was $\pm 1.2 \text{ MPa}$. A total of 15 compound formulations were examined, and three specimens were prepared for each formulation.

The microorganisms were cultivated on a mineral medium of the following composition, g dm^{-3} : NaNO_3 2.0, K_2SO_4 0.04, NH_4NO_3 0.26, CaCl_2 0.1, Na_2SO_4 0.06, Na_2CO_3 0.025, and KH_2PO_4 0.5 with the addition of oils to the water/oil ratios of 10 : 1, 50 : 1, and 100 : 1. The microbial suspension was introduced in an amount of 1–2% relative to the final volume of the experimental sample. OD11 was used as the main culture. It included pure microbial cultures isolated from oil strata: *Pseudomonas putida* K12, *Taurea aminoaromatica* K8, *Tepidiphilus margaritifera* K13, *Brachymonas petroleovorans* K14, and also *Saccharomyces cerevisiae* yeast produced by Saf-Neva (Russia).

The consumed oil concentration was determined as the difference between the initial and final concentrations measured gravimetrically. The oil after microbial treatment was extracted from the medium with chloroform and weighed after the complete solvent evaporation.

Experiments on studying the influence of temperature and pH on the oil consumption were performed in wider ranges of parameters. The samples were cultivated in thermostats at a temperature step of 5°C . The required pH was adjusted with 5% HCl or 5% Na_2CO_3 solution. The influence of nitrogen and phosphorus sources was studied by varying separate components and their concentrations in the aqueous phase.

Biodegradation experiments were performed in 10–100 cm^3 flasks with stirring using an MS-01 Elmi magnetic stirrer or an ES-3800 top drive stirrer and with air bubbling using an SK-9858 HETO air pump with variable air supply. After studying the main biodegradation parameters, the experiment was scaled using a 200 cm^3 laboratory installation (Fig. 1) and then a pilot laboratory installation with the liquid phase volume of 20 dm^3 and a top drive stirrer (Fig. 2).

RESULTS AND DISCUSSION

Component Composition of Oils

Because the chemical composition is one of the most important parameters for evaluating the possibility of biological treatment of petroleum products, we first studied the composition changes caused by the operation conditions. Data on the structural-group composition of the initial oils are given in Table 2.

Table 3. Structural-group composition of spent oils, wt %

| Oil | Aromatic structures | | | | Paraffins | Naphthenes | Tars |
|-----|---------------------|-------|------|----------|-----------|------------|------|
| | B | N | P | Σ | | | |
| VM4 | 7.91 | 11.14 | 0.40 | 19.46 | 72.73 | 5.06 | 2.75 |
| VM1 | 3.48 | 2.88 | 1.13 | 7.48 | 63.44 | 27.56 | 1.52 |
| GK | 0.21 | 0.48 | 1.04 | 1.73 | 80.41 | 17.16 | 0.69 |

Chain structures prevailed in the oils studied. In industrial and transformer oils, more than half of structural components were cyclic (total content of naphthene components 52–59%). The total content of aromatic structural components did not exceed 5% on the average. Increased content of aromatic structures (up to 20%) was found in VM4 vacuum oil, which accounts for its high boiling start point. In the spent oils (Table 3), resinous compounds formed by polymerization of the oxidized components appear, and the content of unbranched structures increases. For example, in a sample of VM4 oil the content of phenanthrene structures decreased, and that of naphthalene structures increased by a factor of almost 1.5. Analysis of the IR spectra revealed an increase in the absorption in the region of 2000 cm^{-1} , which can be attributed to dispersed impurities of carbon black, to polycarboxylic acids, and to glycols (absorption bands at 1760 and 1080 cm^{-1}).

In VM1 oil, the content of aromatic and naphthene structures considerably increased, whereas the content of paraffin structures decreased. In addition, absorption bands characteristic of nitro and carboxy groups appeared.

In GK oil, the content of aromatic and naphthene structures decreased, whereas the content of paraffin structures increased.

Biological Oxidation

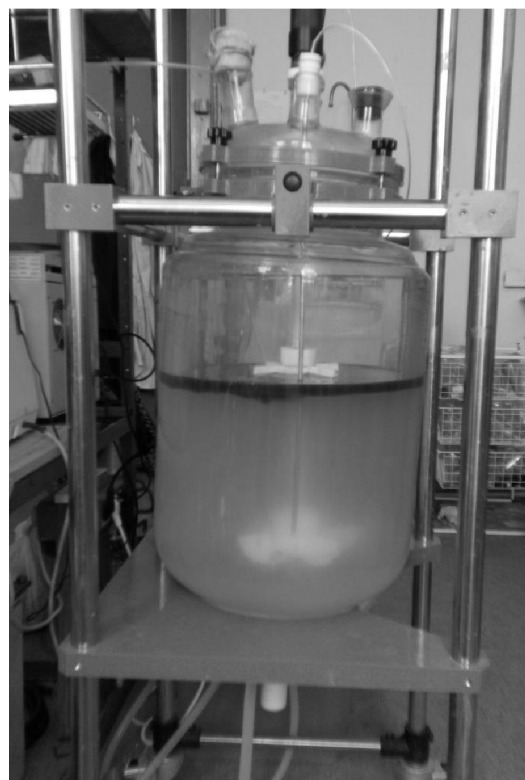
Because biodegradation involves incorporation of an oxygen molecule into a hydrocarbon molecule, followed by decomposition of the hydrocarbon molecule by microbial enzymatic systems, particular attention in the experiments on the oil biodegradation was paid to changes in the structural-group composition (Fig. 3). We found that the common trend for all the types of oils is a decrease in the fraction of paraffin and aromatic components except spent GK oil for which only paraffin components were consumed.

Figure 4 shows the data on the consumption of various organic compounds. Along with oils, we took sev-

eral model substances: kerosene, scintillation liquid, and an organic mixture of water-soluble compounds of the composition, g dm^{-3} : sodium acetate 2, sodium oxalate 1.5, and EDTA 0.5.

As we found, spent oils and kerosene were oxidized with the chosen culture most efficiently, with the consumption of TBP and scintillation liquid also detected. The mixture of EDTA with oxalate was oxidized insignificantly.

We examined the influence of the physicochemical factors of the medium (temperature range $5\text{--}50^\circ\text{C}$, pH $5\text{--}12$) on the growth of the cultures. The optimum temperature parameters for the operation of the chosen culture were as follows: temperature range from 25 to 35°C , pH $4.5\text{--}5.5$ for yeast cultures and $6.5\text{--}8.0$ for microbial cultures.

**Fig. 2.** Pilot installation of 20 dm^3 volume.

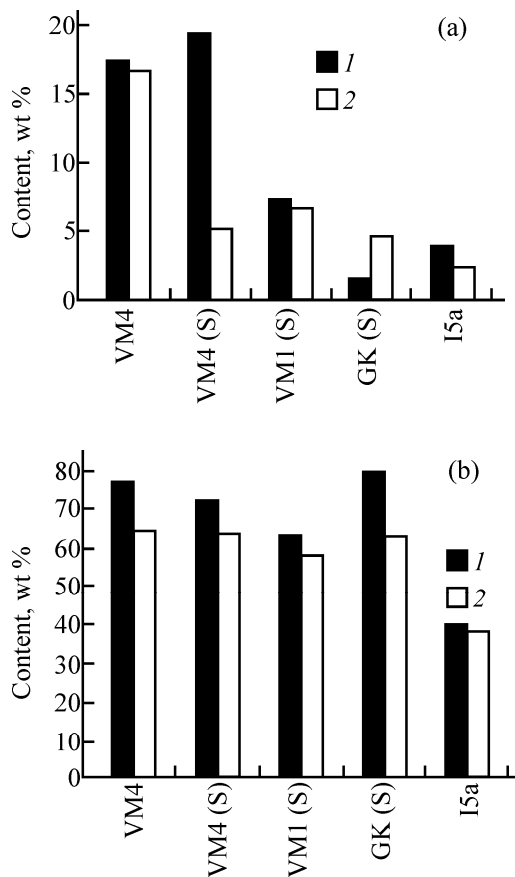


Fig. 3. Structural-group composition of oils (1) before and (2) after biodegradation: (a) total content of aromatic components and (b) total content of paraffin components.

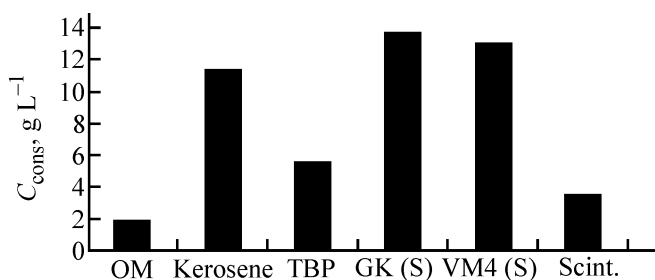


Fig. 4. Concentration of liquid radioactive waste components, consumed by OD11 culture. (OM) Organic mixture of water-soluble organic compounds (sodium acetate 2, ammonium oxalate 1.5, EDTA 0.5, total 4 g dm⁻³) and (Scint.) scintillation liquid.

Along with the main physicochemical parameters of the medium, the microbial growth is significantly influenced by the concentrations of nitrogen and phosphorus, which are essential elements for supporting the microbial activity. Organic protein nitrogen, urea, ammonium chloride, sodium nitrate, ammonium nitrate,

and sodium glutamate were used as nitrogen sources. As we found, both reduced (ammonium ions) and oxidized (nitrate ions) forms of nitrogen could be sources of nitrogen for microorganisms of the OD11 culture.

Tributyl phosphate could act as a source of phosphorus, as its addition increased the total consumption of oil components by a factor of 1.2–1.7.

Thus, addition of other kinds of waste, e.g., TBP, to oil-containing waste can increase the oil biodegradation. In addition, we examined the possibility of introducing low-salinity low-level liquid waste containing nitrates and sulfates for replacing the culture medium components. We used the simulated low-level waste solution of the following composition, g dm⁻³: NaNO₃ 2.0, K₂SO₄ 0.04, and NH₄NO₃ 0.26. This replacement did not decrease the oil oxidation efficiency.

Trials of the Laboratory Installation with Model Radioactive Waste

To perform the experiment, we used mixed organic waste containing spent GK oil, TBP, and kerosene (1 : 2 : 7 ratio) with the following radionuclide content, Bq cm⁻³: ²³⁸Pu 400, ²⁴¹Am 152, ¹³⁷Cs 21, and $\sum\beta$ 580. Cultivation was performed for 14 days at 20°C and 1/100 ratio of the organic and aqueous phases (O/W) with aeration and stirring. The emulsion from the installation was taken for analysis immediately after the cultivation completion.

The experiments showed that the organic phase was consumed with 68.4% efficiency. The degree of decontamination of the liquid phase from the radionuclides was as follows, %: ²³⁸Pu 87.25, $\sum\beta$ 90.51, ²⁴¹Am 99.9, and ¹³⁷Cs 2.43.

Thus, OD11 mixed culture is capable to grow under the conditions of real low-level waste and efficiently oxidizes the organic phase.

The liquid phase was decontaminated from radionuclides owing to their coprecipitation with components of the medium and interaction with the surface of microbial cells.

To evaluate the contribution of different mechanisms to decontamination of the aqueous phase from the radionuclides, the organic waste used in the above experiment was mixed with the aqueous phase (medium for cultivating the microorganisms) in the ratio O/W = 3 : 5. The resulting emulsion was stirred for 24 h to reach the equilibrium in the system. After that,

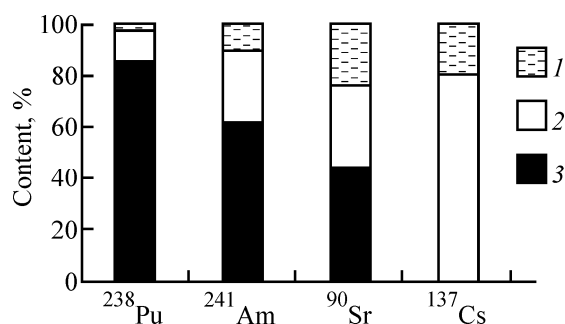


Fig. 5. Phase distribution of radionuclides at pH 4.0. Phases: (1) organic, (2) aqueous, and (3) solid; the same for Figs. 6 and 7.

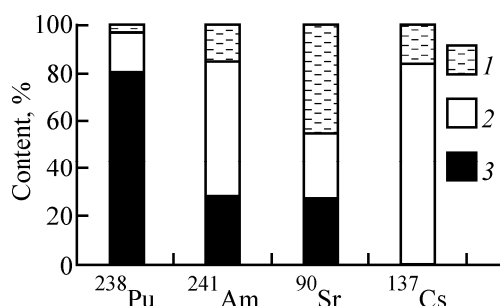


Fig. 7. Phase distribution of radionuclides at pH 9.0.

a solid phase was formed in the system. It was separated by centrifugation (6000 rpm, 15 min), the organic and mineral liquid phases were separated, and the radionuclide content of the three phases was determined. The radionuclide distribution between the phases is shown in Figs. 5–7. This experiment was performed at pH 4.0, 7.0, and 9.0.

The radionuclide distribution between the phases suggests the occurrence of the two main processes: radionuclide transfer from the organic phase to the aqueous phase (stripping) and precipitation. The radionuclide precipitation is associated with two chemical processes that occurred in the system: hydrolysis characteristic of $^{241}\text{Am}(\text{III})$, $^{238}\text{Pu}(\text{IV})$, and some products of their degradation and coprecipitation with phosphates present in the aqueous phase. Cesium-137 in this system was quantitatively stripped into the aqueous phase irrespective of pH and was not precipitated. Also, the pH value did not appreciably influence the behavior of ^{238}Pu . The residual content of ^{238}Pu in the organic phase did not exceed 1–3%, and the residual content in the aqueous phase varied in the range 5–16%. The residual content of ^{238}Pu was minimal at neutral pH values, which were optimum for the growth of the majority of microorganisms. The phase distribu-

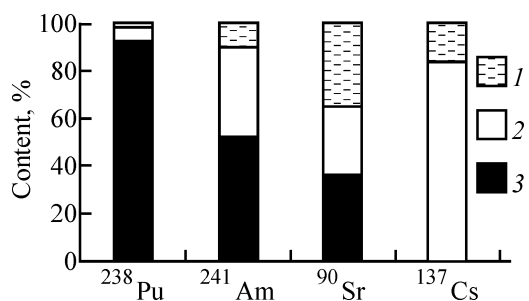


Fig. 6. Phase distribution of radionuclides at pH 7.0.

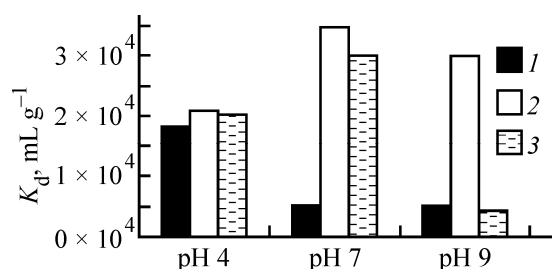


Fig. 8. Coprecipitation of ^{241}Am on biomass of various organisms: (1) *S. cerevisiae*, (2) *Ps. putida*, and (3) *Hm. mono*; the same for Figs. 9–11.

tion of ^{90}Sr noticeably depended on pH of the medium. The residual content of Sr in the aqueous phase was virtually the same at all the pH values studied, ~30% of the total amount. At pH 9, the residual content of Sr in the organic phase was two times higher than that at pH 4. Such behavior was caused by the formation of the extractable complex.

The residual content of ^{241}Am in the organic phase was stable, about 10% of the introduced amount, at pH in the range from 4 to 9. The ^{241}Am concentration in the aqueous phase appreciably increased at alkaline pH values, which can be attributed to the formation of soluble Am hydroxo complexes.

Thus, the contact of the organic and aqueous phases of the given composition led to the formation of a precipitate and thus to decontamination of the liquid as a whole. The overall degree of decontamination was 89.8%.

We studied the behavior of radionuclides in contact with the *Ps. putida* and *S. cerevisiae* biomass. These species were components of the communities studied in this work. The results are shown in Figs. 8–11.

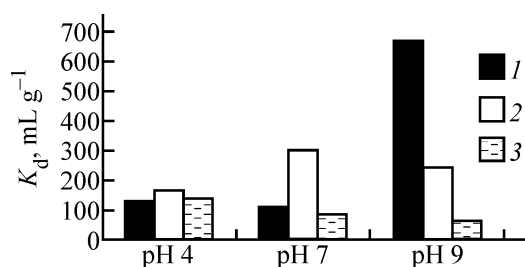
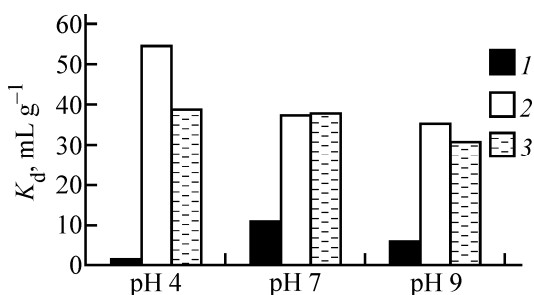
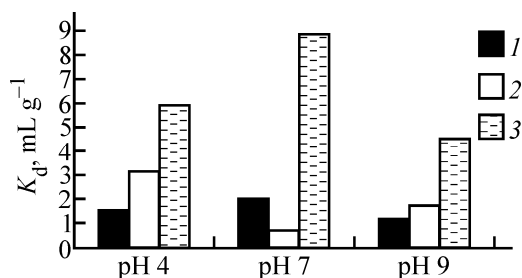
The data analysis allowed the following conclusions. The efficiency of the aqueous phase decontami-

Table 4. Efficiency of various decontamination mechanisms, %^a

| Radionuclide | pH 4 | | | pH 7 | | | pH 9 | | |
|-------------------|-------|------|-------|-------|------|-------|-------|------|-------|
| | C | B | T | C | B | T | C | B | T |
| ²³⁸ Pu | 87.45 | 64.0 | 95.48 | 94.91 | 75.6 | 98.76 | 82.34 | 44.1 | 90.13 |
| ²⁴¹ Am | 68.17 | 99.6 | 99.87 | 58.56 | 99.7 | 99.88 | 33.1 | 98.2 | 98.80 |
| ⁹⁰ Sr | 58.13 | 2.9 | 59.34 | 46.64 | 37.4 | 66.60 | 37.14 | 34.2 | 58.64 |
| ¹³⁷ Cs | 0 | 2.2 | 2.20 | 0 | 0.6 | 0.60 | 0 | 5.3 | 5.30 |

^a (C) Chemical, (B) biological, and (T) total.

nation from the radionuclides using the biomass, irrespective of pH of the medium, decreased in the series ²⁴¹Am > ²³⁸Pu > ⁹⁰Sr > ¹³⁷Cs. The following sorbents showed the highest performance: for ²⁴¹Am, *Ps. putida* biomass at pH 7; for ²³⁸Pu, *S. cerevisiae* biomass at pH 9; and for ⁹⁰Sr, *Ps. putida* biomass at pH 4.

**Fig. 9.** Coprecipitation of ²³⁸Pu on biomass of various organisms.**Fig. 10.** Coprecipitation of ⁹⁰Sr on biomass of various organisms.**Fig. 11.** Accumulation of ¹³⁷Cs with biomass of various organisms.

To compare the efficiency of the metal uptake, we also present in Figs. 8–11 the data for *Halomonas mono* halophilic bacteria living in strongly alkaline water bodies.

The calculated contributions of physicochemical (Figs. 5–7) and biological (Figs. 8–11) mechanisms to decontamination of the liquid phase from radionuclides are compared in Table 4. The total degree of decontamination Σ (%) was calculated using the formula

$$\Sigma_{\text{tot}} = \Sigma_{\text{chem}} + (100 - \Sigma_{\text{chem}})\Sigma_{\text{biol}}/100.$$

As seen from Table 4, the chemical precipitation occurring on contact with components of the medium made a considerably larger contribution to the aqueous phase decontamination from radionuclides than the presence of the biomass did. ²⁴¹Am was an exception, as for this nuclide the aqueous phase decontamination via interaction with cells was the main decontamination pathway.

We considered the possibility of incorporation of biodegradation products into a cement compound. One of the main requirements to such compounds is their compression strength on the 28th day, which should be no less than 5 MPa. The main results of this study are given in Table 5.

Among the formulations tested, only compound no. 3 had the required strength characteristics. With respect to the degree of filling with the organic waste, the results obtained with this compound were comparable with the results reported by other authors (Table 6).

Thus, we have studied the structural-group composition of the main grades of oils used in nuclear industry. Chain structures prevailed in all the oils, with the main differences observed in the amounts of aromatic and naphthene structures. The use of oils in process cycles leads to partial polymerization and oxidation of chain structures. The OD11 microbial community is

Table 5. Parameters of cementation of various oil treatment products^a

| Compound no. | Organic phase | W/C | Additives | Incorporation percentage | σ_{28} , MPa |
|--------------|--|------|----------------------------------|-------------------------------|---------------------|
| 1 | VM4 (I) | 0.5 | – | 12.3 | 2.84 |
| 2 | VM4 (M) | 0.5 | – | 22.3 | 2.21 |
| 3 | VM4 (M) | 0.9 | CaCl ₂ , 5% of cement | 33.3 | 11.6 |
| 4 | Water–oil emulsion (40 vol % H ₂ O) | 0.8 | " | 64.9 (38.9 for organic phase) | 1.40 |
| 5 | Biomass | 1.16 | " | 7.8 | 0.6 |

^a (I) Initial oil, (M) oil treated with microorganisms, and (W/C) water-to-cement ratio.

Table 6. Comparison of the characteristics of the cement compound obtained with published data

| References | Incorporation percent | σ_{28} , MPa |
|------------|-----------------------|---------------------|
| This work | 33.3 | 11.28 |
| [21] | 39.9 | 9.12 |
| [22] | 50 | – |
| [23] | 20 | 6.95 |
| [24] | 64 | 5.5 |

capable to oxidize aliphatic and aromatic components of oil responsible for its highly hydrophobic properties. Introduction of TBP and simulated nitrate-containing low-level waste as sources of phosphorus and nitrogen positively influenced the consumption of the organic substrates. Experiments with model organic waste containing radionuclides revealed no appreciable decrease in the microbial activity. Significant fraction of radionuclides passed into the aqueous phase and then into the solid phase formed by phosphate precipitates and the biomass, which allowed decontamination of the aqueous-organic system from ²³⁸Pu to 87.25, from ²⁴¹Am to 99.9, and from ¹³⁷Cs to 2.43%. The solution obtained after microbiological treatment can be incorporated into a cement host material. The strength characteristics of the cement are preserved at the organic phase content of up to 33%.

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