**Kaunas University of Technology**

Faculty of Mathematics and Natural Sciences

# Development of Liquid Radioactive Waste Filtration System

Master's Final Degree Project

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**Kaunas, 2025**

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Medical Physics (6213GX001)

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Study field and area (study field group): Medical Technologies (Health Sciences)

Keywords: liquid radioactive waste, ion exchange filtration, iodide removal.

Kaunas, 2025. 57 p.

## Summary

This Master’s final degree project confronts an issue of radioactive iodine isotope contamination in urine following nuclear medicine treatments, particularly the excretion of iodine-131, which poses environmental and health risks. Traditional liquid radioactive waste disposal methods, such as storage or dilution are often impractical in clinical settings due to cost and logical constraints.

To address this issue, during the research a 3D-printed ion exchange filtration system was designed to remove iodide ions from the patient urine, using MP artificial urine and potassium iodide as a phantom of patient’s urine. The filter system was made of stackable ion exchange resin elements to capture iodide ions. Comparative filtration experiments demonstrated that while double-stacked filter (one anionic and one cationic element) significantly reduced dissolved solids and fully removed traceable amounts of iodide, as proven with silver nitrate test for halogens and starch test for iodine. Quadruple-stacked filter (alternating between anionic and cationic elements) showed even better results in completely removing total dissolved solids, as well as iodide ions, which were also confirmed with qualitative tests with starch and silver nitrate.

3D-printed ion exchange resin stackable filter is proposed as a sustainable and cost-effective alternative to conventional radioactive medical wastewater management solution. Its adaptability and low environmental footprint make it especially beneficial for smaller healthcare facilities.

Radzevičiūtė, Julija. Skystųjų radioaktyviųjų atliekų filtravimo sistemos Magistro baigiamasis projektas / vadovas doc. dr. Benas Gabrielis Urbonavičius; Kauno technologijos universitetas, matematikos ir gamtos mokslų fakultetas.

Studijų kryptis ir sritis (studijų krypčių grupė): Medicinos technologijos (Sveikatos mokslai)

Reikšminiai žodžiai: skystos radioaktyviosios medžiagos, jonų mainų filtravimas, jodo jonų šalinimas.

Kaunas, 2025. 57 p.

## Santrauka

Šiame magistro baigiamajame projekte nagrinėjama problema, susijusi su radioaktyvaus jodo izotopų šlapime po branduolinės medicinos procedūrų, ypač dėl jodo-131 išskyrimo, kuris kelia pavojų aplinkai ir žmonių sveikatai. Įprasti skystųjų radioaktyvių atliekų šalinimo būdai. Tokie kaip saugojimas kanalizacijos sistemoje ar praskiedimas iki neaptinkamųjų lygių dažnai yra nepraktiški klinikinėse sąlygose dėl didelių kaštų ir logistinių apribojimų.

Siekiant išspręsti šią problemą, tyrimo metu buvo sukurtas 3D spausdintas jonų mainų filtravimo įrenginys, skirtas jodidų jonams pašalinti iš paciento šlapimo. Eksperimento metu buvo naudojamas MP sintetinis šlapimas ir kalio jodidas kaip paciento šlapimo pakaitalas. Filtras buvo sudarytas iš sudedamų jonų mainų dervos elementų, skirtų jodido jonų surinkimui. Lyginamieji filtravimo eksperimentai parodė, kad dvigubas filtras (vienas anijoninis ir vienas katijoninis elementas) ženkliai sumažino ištirpusių medžiagų kiekį ir visiškai pašalino aptinkamą kiekį jodo anijonų, tai patvirtinant sidabro nitrato testu halogenams ir krakmolo testu jodui. Keturgubas filtras (kaitaliojant anijoninius ir katijoninius elementus) parodė dar geresnius rezultatus – visiškai pašalino ištirpusias medžiagas, taip pat ir jodido jonus, įrodant tai kokybiniais krakmolo ir sidabro nitrato testais.

3D spausdintas jonų mainų dervos surenkamas filtras siūlomas kaip tvari ir ekonomiška alternatyva įprastiems radioaktyvių medicininių nuotekų tvarkymo sprendimas. Dėl savo pritaikomumo ir mažo poveikio aplinkai šis sprendimas yra ypač naudingas mažesnėms sveikatos priežiūros įstaigoms.

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## List of abbreviations

**[$^{18}$F]FDG –** 18-fluorodeoxyglucose

**DNA** – deoxyribonucleic acid.

**MP-AU** – MP artificial urine.

**MRI** – magnetic resonance imaging.

**PET –** positron emission tomography.

**PET-CT –** positron emission tomography combined with computed tomography.

**PUREX** – plutonium-uranium redox extraction.

**RIA** – radioimmunoassay.

**SAPPIER –** Strategic Action Plan for Implementation of European Regional Repositories.

**SPECT –** single photon emission computed tomography.

**TDS** – total dissolved solids.

## Introduction

The management of radioactive iodine isotopes in patient urine remains a critical challenge in nuclear medicine, particularly following therapeutic administrations for thyroid disorders. Over 80 % of administered radioiodine is excreted via urine within 72 hours [1], creating waste streams that risk environmental contamination through water systems. Radiopharmaceutical treatments generate waste containing isotopes like iodine-131, which persists in aquatic ecosystems and bioaccumulates in terrestrial food chain [2], demanding filtration solutions that do not introduce additional radioactivity to the environment. Conventional methods, such as decay in storage or dilute and discharge face limitations in clinical setting due to variety of radionuclides used, low alteration possibility and high installation and management costs.

Nuclear facilities like nuclear power plants already employ ion exchange resin systems to collect the radionuclides from reactor coolant water and spent fuel pool water [3]. Therefore, it shows that ion exchange technology is suitable for radionuclide removal, although it is not being widely researched and used in the sphere of nuclear medicine.

This study focuses on developing a 3D-printed ion exchange filtration system that is optimized for clinical iodide removal from patient urine. The proposed system aims to reduce storage demands, operational costs and less hazardous secondary waste in hopes to advance sustainable radioactive waste management practices.

**Objective:** to develop and evaluate a 3D-printed ion exchange filtration system for the efficient removal of iodide ions from liquids.

**Tasks:**

1. Design and 3D-print a filter frame with interlocking stackable elements.
2. Conduct a comparative filtration using double (one anionic and one cationic) and quadruple (alternating layers of anionic and cationic elements) filter array.
3. Perform iodide detection tests (using silver nitrate and starch) on artificial urine samples with iodide ions before and after filtration.
4. Comparative analysis of the developed system and currently employed systems.

# 1. Literature review

The management of radionuclide-contaminated liquid waste represents one of the most significant environmental and safety challenges in industries and sectors that utilize radioactive materials. As nuclear technologies continue to advance, so too does the volume and complexity of liquid waste generated, demanding great understanding and effective treatment methods. This literature review outlines the nature, sources and treatment strategies of radionuclide-contaminated liquid waste.

This overview first defines radionuclide-contaminated liquid waste, detailing the regulatory context (chapter 1.1), followed by a review of waste generated in medical applications due to the use of radiopharmaceuticals and diagnostic isotopes (chapter 1.2). It then addresses industrial sources (chapter 1.3), including waste from reactor coolant systems, spent fuel pools, reprocessing operations and uranium mining and milling. Chapter 1.4 outlines treatment technologies, covering main methods with a focus on their mechanisms, efficiency and applicability.

## 1.1. Radionuclide-contaminated liquid waste

Radionuclide-contaminated liquid waste is generally a cause of anthropogenic activity, commonly produced via industrial processes (for example nuclear power plants, nuclear weapons production or mining), medical applications of radioactive substances, and research laboratories. This waste, containing radioactive isotopes, can pose environmental and health risks if it is not properly managed. Due to the potential hazards associated with radionuclide-contaminated liquid waste, strict regulatory measures are enforced to ensure proper treatment, containment, and disposal, preventing radiation exposure and long-term ecological damage. To implement it properly, radionuclide contaminated waste is categorized by the radioactivity it emits, radionuclide concentration, longevity of the present radionuclides and the acting guidelines. The groups are as follows: exempt waste, very short-lived waste, very low-level waste, low-level waste, intermediate-level waste and high-level waste [4]. However, this nomenclature applies only to solid waste, as liquid waste is solidified by vaporizing, filtering and/or mixing with the cement [5, 6], because it is easier to manage, transport and store than liquids or gases.

Radionuclide-contaminated liquid waste is categorized as exempt waste when it does not reach the threshold of maximum activity and concentration of the waste fitting for clearance and exemption [4], depending on the acting guidelines. The values followed in Lithuania for some of the artificial radionuclides found in waste are shown in Table 1.

**Table 1.** Activity concentration and surface activity values for some radionuclides to qualify as exempt waste [7].

| Radionuclide | Activity concentration ($kBq \cdot kg^{-1}$) | Surface activity ($Bq/cm^2$) |
|---|---|---|
| $^{3}H$ | 100 | 100 |
| $^{60}Co$ | 0.1 | 0.4 |
| $^{90}Sr$ | 1 | 1 |
| $^{99}Tc$ | 1 | 70 |
| $^{137}Cs$ | 0.1 | 0.4 |
| $^{241}Pu$ | 0.1 | 10 |
| $^{241}Am$ | 0.1 | 0.1 |

If the materials or waste contain multiple types of artificial radionuclides, these materials can be reused or waste disposed of as general waste if they satisfy the condition [7]:

$$\sum_{i}^{n} \frac{C_i}{C_{Li}} \leq 1 \quad (1)$$

here: Ci – activity concentration (in kBq/kg) or surface activity (in Bq/cm$^2$) of the i-th radionuclide in the material or waste; $C_{Li}$ – maximum allowed activity concentration or surface activity of the i-th radionuclide present in the material or waste for it to be eligible for clearance; n – number of different radionuclides in the waste or material [7].

As very short-lived waste qualifies the materials and waste that are contaminated with short-lived radionuclides [4], usually the radionuclides that have a half-life of a 100 days or less, although it may vary depending on the regulations of the vicinity. For example, in Lithuania, for the waste to fit the criteria it should have the surface activity up to 0.2 mS/h, but to exceed the requirements for the exempt waste. This type of waste is generally placed into containers (metal or concrete [5]) and kept in designated landfills with artificial barriers, such as trench type or barrow-shaped surface repositories [5]. They would be kept there until they are deemed safe (from a few to a hundred years).

Very low-level waste does not meet the criteria for exempt waste but contains limited amounts of low activity longer lived radionuclides [4] and its activity concentration generally remains below 100 time the clearance levels specified for each relevant radionuclide [8]. This type of waste or materials are deemed suitable for disposal in near surface facilities, similarly to very short-lived waste, although the time it would be kept there would be longer in comparison.

Very short-lived and very low-level waste is kept in very low-level radioactive waste storage facility (in Lithuanian: *labai mažai radioaktyvių atliekų saugykla*), from where it will be transferred into the very low radioactive waste repository (in Lithuanian: *labai mažai radioaktyvių atliekų atliekynas*) which is being prepared for industrial exploitation [9]. Construction of the repository can be divided into three main parts – upper level, storage space and lower level. The lower level of the repository is made of (from the bottom up) pressed existing soil layer, geotextiles, gravel and rubble, concrete, bentonite, high density polyethylene, concrete and reinforced concrete plate layers [9] (Fig.1). Above it is a space designed to store radioactive waste, and the upper level is made of sorbent, sand and gravel, bentonite, high density polyethylene, gravel, geotextiles, existing soil and vegetation layers [9] (Fig.1). Although the mentioned two classes of waste usually would not be considered safe (0.2 mSv/h is more than a thousand times higher than normal background radiation), the shielding that these layers provide mostly relies on thickness of the layers and the specific stopping power characteristics of the shielding. For example, reinforced concrete plate layers attenuate γ radiation, while high density polyethylene is good for attenuating β and neutron particles [10], so the materials complete one another to provide adequate shielding.

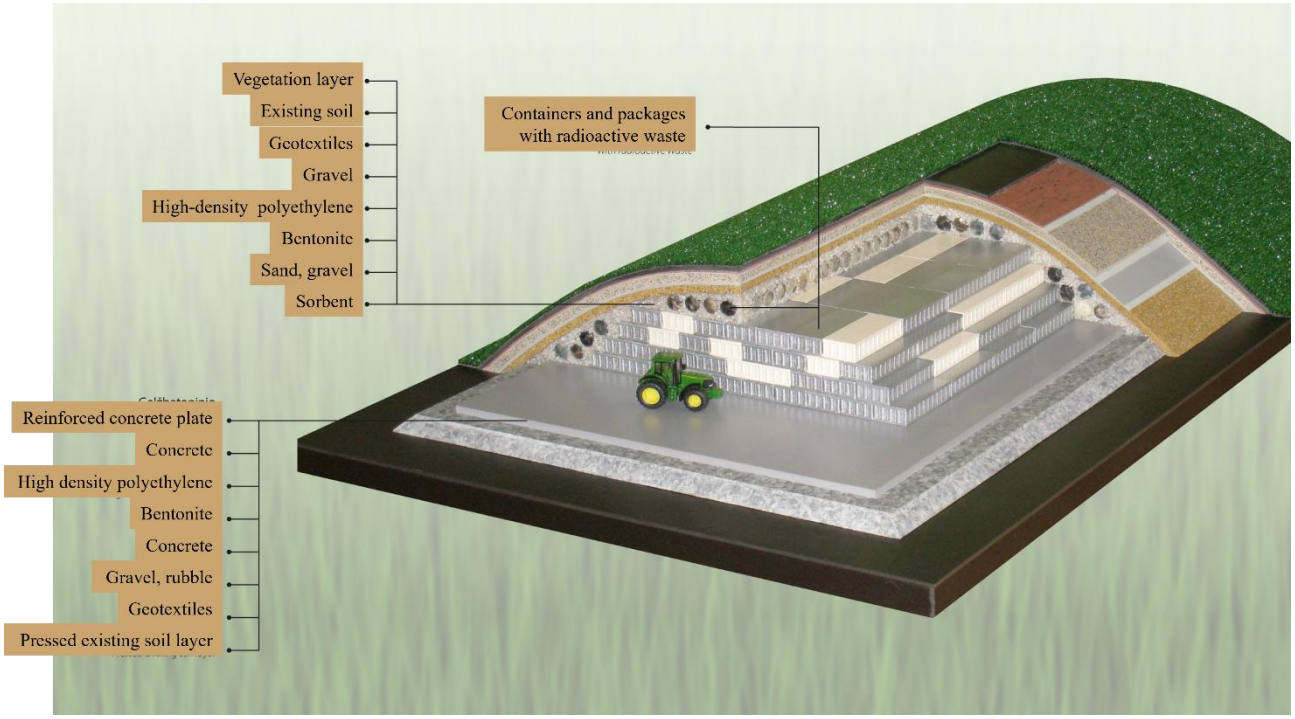



**Fig. 1.** Very-low level waste repository [9]

Low-level waste exceeds clearance levels and contains limited amount of long-lived radionuclides or short-lived radionuclides with higher levels of activity concentration [4], which requires secure isolation and containment for up to several hundred years and is suitable for disposal in specifically engineered near-surface facilities [4], similarly to very short lived and very low level waste repositories, with adjustments to certain layers to provide sufficient shielding. This type of waste, according to Euratom commission, should contain radionuclides having half-lives less than or equal to those of Cs-137 and Sr-90 (around 30 years) [11]. Alpha emitters are limited to 4000 Bq/g in individual packages and 400 Bq/g on average [11].

Intermediate-level waste contains long lived radionuclides that generate little to no heat [4]. It does not require extensive heat dissipation measures for storage and disposal but the α-emitting radionuclides, which remain active for more than a thousand years, the period during which institutional controls can be relied upon, necessitates specific handling and storage [4]. This type of waste would generally be buried several tens of meters below ground [12, 13], which can be artificially dug or just naturally occurring caverns to minimize the human interference [13]. This type of facilities mostly rely on the naturally occurring rocks to detain the radiation spread to the environment, although constructing an additional barrier system within is also necessary [13]. Sweden [13] and Finland [14] operate intermediate level facilities for disposal, such as tunnel based and silo-based designs.

High-level waste has a high concentration of high activity radionuclides, producing substantial heat through radioactive decay [4, 14]. Due to its high radioactivity and heat generation, high-level waste requires long-term isolation from the biosphere [14]. The widely accepted method for high-level waste disposal involves placement in deep, stable geological formations, typically several hundred meters or more below the surface [4]. Such repositories rely on naturally occurring rocks for heat dissipation, but the engineered barriers of the repository are expected to contain most of the ionizing

radiation from waste. The first long-term disposal facility for this kind of waste is expected to be opened in Finland, Onkalo spent nuclear fuel repository (in Finnish: *Onkalo ydinjätteen loppusijoitustila*) [15].

It is important to note that underground repositories for radioactive waste cost a lot, as there are many components to consider – site selection, geological stability, engineered barriers, ventilations systems, long-term monitoring, regulatory compliance, and safety measures to prevent radiation leakage and human intrusion. Moreover, the construction should be big in size so that there would not be any need to build another complex repository. Therefore, it is advised and strategized to create regional repositories due to the high cost of individual geological disposal projects. Some of the organizations from European countries (Austria, Belgium, Bulgaria, Croatia, Czech Republic, Hungary, Italy, Latvia, Lithuania, Netherlands, Romania, Slovakia, Slovenia and Switzerland) are making plans for such repositories, the project is called SAPIERR II (Strategic Action Plan for Implementation of European Regional Repositories), although the project is still in early stages [16].

Overall, the class of radionuclide-contaminated waste depends mostly on its activity content and the half-life of radionuclides present. Higher radionuclide concentrations and higher half-life radionuclides, that do not fall within the range for exempt waste regulations, require stricter policies applied to handling, storage and disposal of waste. The relationship between the two criteria and their according class is presented in Fig.2.

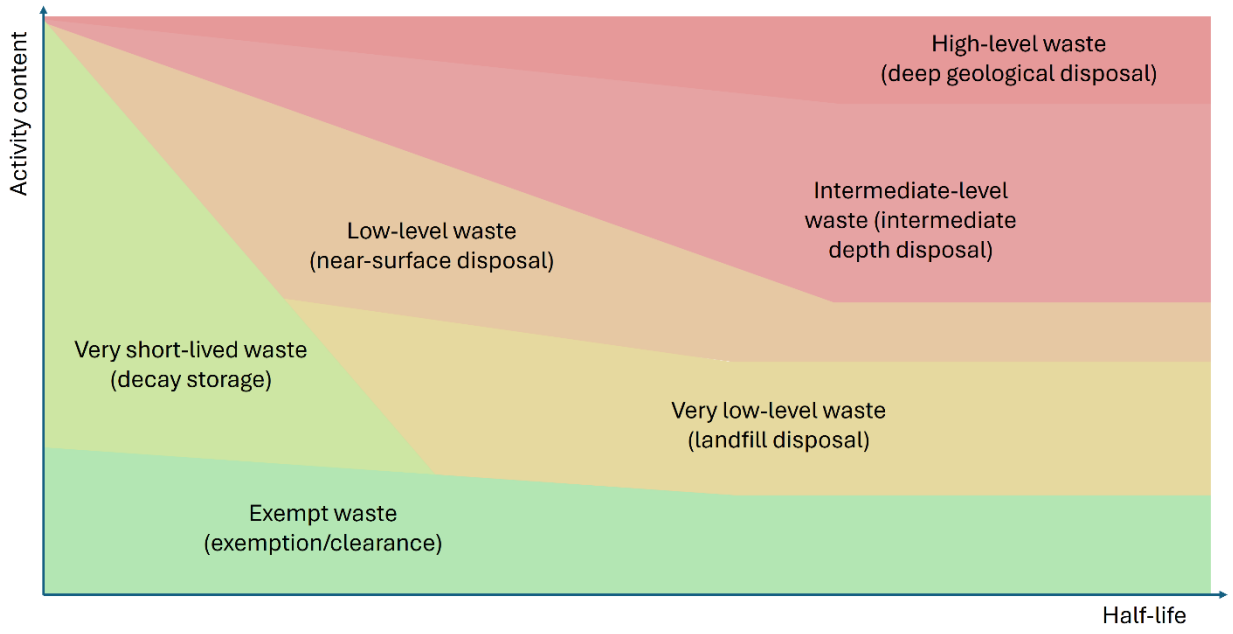



**Fig. 2.** Conceptual illustration of the waste classification scheme [4]

### 1.2. Radionuclide-contaminated liquid waste in medicine

Radionuclide-contaminated liquid waste in medicine comes from bodily fluids, certain pharmaceuticals and agents used for testing. This waste is commonly generated in diagnostic imaging, radiation therapy and laboratory procedures involving radioactive tracers. Urine, blood and other excreta from patients who have undergone nuclear medicine treatments often contain residual isotopes. Additionally, liquid waste may result from the preparation and administration of

radiopharmaceuticals, as well as from the cleaning of equipment and disposal of expired or unused radioactive substances. Proper handling and disposal are essential to minimize environmental contamination and radiation exposure.

The human body may contain radionuclides after various diagnostic procedures for certain medical conditions. In diagnostics the radionuclides are used in radiotracers for positron emission tomography (PET), positron emission tomography – computed tomography (PET-CT) and single photon emission computed tomography (SPECT). Radiotracers are compounds with molecules containing radioactive atoms. They accumulate in the target areas, where it decays, radiating positrons or photons into the detectors. Undecayed radiotracer and its metabolites are excreted via bodily fluids, primarily through urine [17, 18], although, depending on the radiotracer used, small parts may be found in saliva and feces. The blood and flesh of the patient after the procedure is also contaminated with radionuclides, so if the blood is drawn or the wound is being cleaned, the waste liquids that accumulate (for example the blood testing solutions or saline solution for cleaning the wound) are also contaminated. Therefore, the radionuclides used for diagnostic procedures procure contaminated liquid waste like urine, blood and the solutions that were in contact with urine and/or blood.

The most used radioisotope in diagnostics is metastable technetium-99 (Tc-99m) [19]. Because of short half-life (around 6 hours) and no high-energy beta emissions, it minimizes the exposure to the patient [19]. Also, it is versatile when it comes to incorporating it into other molecules - that is important for creating radiotracers which reach the region of diagnostic interest [19]. For example, scans $^{99m}$Tc-diethylenetriaminepentaacetic acid or $^{99m}$Tc-mercaptoacetyltriglycine are typically used to diagnose impaired or delayed cortical uptake [20], whereas with $^{99}$mTc chemically bound to exametazime ($^{99}$mTc HMPAO), the radiotracer is capable of crossing the blood-brain barrier, therefore is used in SPECT imaging to assess cerebral blood flow, aiding the diagnosis and monitoring of neurological conditions such as stroke, dementia and other neurodegenerative diseases [21, 22].

Fluorine-18 (half-life of 110 minutes, decays by $\beta^+$ emissions) is also very common in the form of fluorodeoxyglucose ([$^{18}$F]FDG) since it mimics glucose, which allows to detect changes in metabolic activity [23]. Because of that, [$^{18}$F]FDG is used to detect primary and extra metastases, that sometimes are undetected by other imaging techniques such as MRI or CT [24]. Continuing to integrate CT and [$^{18}$F]FDG-PET, the specificity is enhanced since both anatomical and metabolic information improves diagnostic accuracy and guiding treatment decisions [23].

Overall, radionuclides are used in diagnostics mostly in imaging, they allow the imaging of areas of interest by accumulating in significant organs or identifying unusual metabolic activity. However, the short half-life of these radionuclides and relatively low injection dose results in waste that after a short period of time can be considered as exempt waste and disposed of as general waste.

In therapy, various radiopharmaceuticals are used for the treatment of cancer such as thyroid cancer, lymphomas, bone metastases and many others. They can target specific tumor cells or tumor environments, delivering radiation that damages the DNA of cancer cells, leading to cell death [25]. Certain radionuclides are incorporated in molecules that have necessary metabolic characteristics, allowing them to accumulate in the region of interest and deliver localized radiation [25]. The localization of the radiopharmaceutical also allows for the $\alpha$ radiation to take effect, which is not achievable with external radiation therapy as it has very shallow penetration depth.

Iodine-131, a beta-emitter with some low-energy gamma emissions [26], is the most frequently used radionuclide in therapy, with a half-life of around 8 days. Iodine-131 is versatile in its use. Administered as sodium/potassium iodide, it targets the thyroid gland for the purpose of treating thyroid cancer and hyperthyroidism [27]. In the form of iodine-131 lipiodol it can be used to treat primary hepatocellular carcinoma [27], whereas iodine-131 metaiodobenzylguanidine (MIBG) is used for treating various neuroendocrine tumors, such as pheochromocytomas or neuroblastomas [27]. Regardless of iodine-131 molecular form, predominantly it is excreted from the body via urine, although, because of different molecular subtleties, the time frame during which it will be out of body and the portion eliminated from the dose delivered, is different for each radiopharmaceutical.

The iodine isotope iodine-125 (electron capture decay, half-life around 60 days) is also utilized in biodistribution studies involving iodine-125 labeled drugs, peptides and antibodies, it is used in cell-targeted therapies employing iodine-125 labeled nucleic acid precursors [28]. Furthermore, iodine seeds are permanently implanted into the prostate for therapeutic purposes [28]. Other radioisotopes of iodine, such as iodine-123 (electron capture decay with low energy x-rays, half-life of 13.27 hours), and iodine-124 (mostly x-rays and some positron emissions, half-life of around 4 days) are also used in medical conditions, but it is applied for diagnostic imaging of thyroid with PET and SPECT [28].

Another radioisotope, commonly used in radiotherapy, is yttrium-90 (half-life of 2.7 days, $\beta^-$ emissions). It is used to treat non-Hodgkin's lymphoma, as ibritumomab tiuxetan links it to the protein CD20, where yttrium-90 radioactive characteristics target malignant B-cells to destroy tumors [29]. It may also be used to cure liver cancer – in the form of microspheres, yttrium-90 blocks the tumor's blood supply while delivering localized beta radiation to destroy it [30]. It proved to be effective for hepatocellular carcinoma, metastatic liver tumors and neuroendocrine tumors [30]. Moreover, it showed better overall survival with a dose of 150 Gy or above [31], despite the risks that the ionizing radiation poses.

The radionuclides introduced into the body for therapeutic purposes also leave it mostly via urine. The concentration of radionuclides in urine depends on the kidney function of the patient and the specificity of the treatment. For example, the patient who is well hydrated, and uses diuretics or gelofusine (a plasma expander that inhibits renal reabsorption of radioligands [32]) will likely excrete the radionuclides via urine faster, resulting in higher doses in the volume of urine [32, 33]. Also, when using [$^{18}$F]FDG, around 22.7 % of the effective radiation dose is excreted within the first hour [33], meanwhile only 0.041 (±0.033) % of the activity of yttrium-90 resin microspheres are excreted via urine during the first 48 hours [34]. Furthermore, urine and blood may get contaminated with heavy metals as a byproduct of radionuclide decay, such as rhenium-188 decay into stable osmium-188 [35].

Overall, the radioisotopes used in therapy tend to have longer half-lives (days to weeks) to deliver sustained radiation to target tissues, whereas diagnostic radionuclides are effective for lower exposure and higher resolution due to the relatively fast decay rate, a result of the shorter half-life. The longer half-life, higher administered doses and higher number of administered doses increases the demand for the correct handling of the bodily fluids after the procedure to minimize exposure.

Some radionuclides are also used as agents for in vitro medical testing, producing radioactive liquid waste in the process. Radioimmunoassay (RIA) is one of these methods, which is used to measure the levels of specific substances, such as hormones, drugs or antigens in biological samples [36]. RIA

works by using specific antibodies that bind to a target antigen, along with a radioactive version of the antigen [36]. When mixed, they form complexes and adding an unlabeled antigen from a sample causes competition for binding sites [36]. The more unlabeled antigen present, the less radioactive antigen binds, so after separating bound from unbound antigens, the radioactivity is measured and compared to standards to determine the antigen concentration [36]. Radionuclides iodine-125 and tritium are commonly used to label the antigens, in medical conditions often used to assess thyroid hormone levels, cortisol and other hormones [36]. Moreover, radioactive tracers like chromium-51 or iodine-131 are used to label red blood cells or plasma proteins, and their dilution can be measured to calculate total blood volume [37]. Iron-59 may be used as well to analyze iron metabolism by tracing iron absorption, transport and utilization in blood samples [37].

The characteristics of the mentioned radionuclides are shown in Table 2. The waste produced by these radionuclides is likely to be in liquid form, which allows the use of a filter for radionuclide collection from waste samples. The doses administered for a patient or a test are likely to be less than the quantities mentioned in Table 2, however, it still shows that radionuclides administered with high activity content and longer half-lives may pose an environmental issue as it comes to disposal.

**Table 2.** Characteristics of common radionuclides in medicine [38]

| Radionuclide | Half-life | Typical quantity per application | Waste characteristics |
|---|---|---|---|
| $^{3}H$ | 12.33 y | Up to 50 GBq | Solvents, solid, liquid |
| $^{18}F$ | 109.8 min | Up to 500 MBq | Solid, liquid |
| $^{51}Cr$ | 27.7 d | Up to 5 MBq for clinical measurements; up to 100 MBq for biological research | Solid, mainly liquid effluent |
| $^{59}Fe$ | 44.5 d | Up to 50 MBq | Solid, mainly liquid effluent |
| $^{90}Y$ | 2.7 d | Up to 300 MBq | Solid, liquid |
| $^{99m}Tc$ | 6 h | Up to 100 GBq | Solid, liquid |
| $^{123}I$ | 13.2 h | Up to 500 MBq | Solid, liquid |
| $^{125}I$ | 60.1 d | Up to 500 MBq | Solid, liquid |
| $^{131}I$ | 8.0 d | Up to 11.1 GBq | Solid, liquid |

Altogether, radionuclides not only aid in diagnosing various conditions, but also provide critical insights into physiological processes. This reliance on in vitro diagnostics with radioactive substances highlights the ongoing need for proper handling and disposal of radioactive liquid waste generated from these tests.

### 1.3. Radionuclide-contaminated liquid waste from industrial processes

Various non-medical industrial processes generate liquid waste with radionuclide contamination. These may result from various nuclear power plant processes, such as nuclear power generation, fuel reprocessing and uranium mining and milling, also it may come from nuclear weapon development and production processes or other research laboratory activities [39, 40]. Depending on the specificity of the industrial process, the type of radionuclides accumulated in the liquid waste will differ. Moreover, a high distinction between the processes and the variety of radionuclides present in the waste necessitates the need for waste handling systems to be adapted for each case separately.

The radioactive liquid waste, generated during nuclear power plant processes and containing radionuclides comes from several sources, mostly from reactor coolant system [41] but also can be generated from spent fuel pool [42], chemical decontamination processes, condensate and steam generator blowdown or laundry and personal decontamination [43].

### 1.3.1. Radioactive liquid waste from reactor coolant system

The reactor coolant system usually uses high-purity water or heavy water (deuterium) as a cooling agent [44]. Liquid metals or molten salts, such as sodium, may also be used, especially because it allows the reactor to operate safer, faster and at higher temperatures [45], however, chemical properties of the liquid metal make it inefficient to use with an ionic resin filter. Other nuclear reactors use gasses, such as helium or carbon dioxide [46], however, to apply the ionic resin to the gas filtration it would be a challenge on its own due to the additional complexity it poses – designing a specific structure of the ionic resin and wetting or ionizing the gas to create ions, since neutral atoms and molecules will not bond to the ionic resin structure.

Tritium is found in nuclear reactor cooling system water or deuterium – as boron, lithium or deuterium absorbs the neutron, it produces a radioactive tritium radionuclide [43].

$$^{6}_{3}\mathrm{L} + \mathrm{n} \rightarrow ^{4}_{2}\mathrm{He} + ^{3}_{1}\mathrm{H} \quad (2)$$

$$^{7}_{3}\mathrm{L} + \mathrm{n} \rightarrow ^{4}_{2}\mathrm{He} + ^{3}_{1}\mathrm{H} + \mathrm{n} \quad (3)$$

$$^{10}_{5}\mathrm{B} + \mathrm{n} \rightarrow 2^{4}_{2}\mathrm{He} + ^{3}_{1}\mathrm{H} \quad (4)$$

Boron is frequently present in the reactor cooling system liquid as an additive to control the fission process [43] ant the stable lithium isotope compounds are used as a pH stabilizer, which mitigates the acidic effect of boric compounds and therefore reduces corrosion of the mechanical parts of the system into the water [47]. Stopping corrosion is also important because of the radionuclides that the corroded splinters may produce, for example, iron-59 [44], manganese-52 and manganese-54 [48]. Compared to the products of corrosion, tritium emits relatively low energy $\beta^-$ particles and decays into inert helium, whereas radioactive iron or manganese isotopes, despite shorter half-life, are heavy metals by nature, have high-energy emissions and decay into other heavy metals, which are hazardous.

Various heavy metals may also occur in the reactor cooling water, even when using corrosion stopping agents, as completely avoiding the corrosion is impossible. Previously mentioned iron-59, manganese-52, manganese-54 may form within the reactor cooling water media, but, depending on the material of the reactor cooling water container, iron-55, cobalt-60 [49] or other radioactive isotopes can be produced via neutron capture or proton bombardment.

$$^{52}_{24}\mathrm{Cr} + \mathrm{p} \rightarrow ^{52}_{25}\mathrm{Mn} + \mathrm{n} \quad (5)$$

$$^{54}_{26}\mathrm{Fe} + \mathrm{n} \rightarrow ^{52}_{25}\mathrm{Mn} + ^{1}_{1}\mathrm{H} \quad (6)$$

$$^{54}_{26}\mathrm{Fe} + \mathrm{n} \rightarrow ^{55}_{26}\mathrm{Fe} + \gamma \quad (7)$$

$$^{58}_{26}\mathrm{Fe} + \mathrm{n} \rightarrow ^{59}_{26}\mathrm{Fe} + \gamma \quad (8)$$

$$^{59}_{27}\mathrm{Co} + \mathrm{n} \rightarrow ^{60}_{27}\mathrm{Co} + \gamma \quad (9)$$

Carbon-14 is also typically found in water cooling agent [50]. It may be generated through the activation of nitrogen-14 or oxygen-17 and rarely carbon-13 in the coolant or structural materials [50].

$$^{14}_{7}\mathrm{N} + \mathrm{n} \rightarrow ^{14}_{6}\mathrm{C} + \mathrm{p} \quad (10)$$

$$^{17}_{8}\mathrm{O} + \mathrm{n} \rightarrow ^{14}_{6}\mathrm{C} + ^{4}_{2}\mathrm{He} \quad (11)$$

$$^{13}_{6}\mathrm{C} + \mathrm{n} \rightarrow ^{14}_{6}\mathrm{C} \quad (12)$$

Uncertain carbon-14 speciation and multiple release pathways in gas and liquid phases create major uncertainty in its dose contribution from radioactive waste disposal [50], which is why it is important to control the concentration of carbon-14 and keep it to the minimum not only in the environment, but also near the nuclear reactors.

The reactor coolant water in functioning facilities is already being filtered, reportedly with ion exchange resins. However, the research shows that high temperatures in low oxygen environment leads to the detachment of sulfo- groups and the formation of polystyrene-like structures [51]. Moreover, under heating and gamma irradiation combination conditions, amorphous carbon-like structures may form [51]. This kind of chemical change affects the composition of the reactor coolant water and may add to the issue instead of solving it. To solve this problem, a 3D-printed ceramic porous structures are being made [52], which eliminates some of the problems that polymeric ion exchange resins pose, however, it is not yet applied in nuclear reactor facilities.

Such spent ion exchange resins are being solidified for the disposal in the radioactive waste disposal repositories, since it contains high activity and long-lived radionuclides. The solidification currently is mostly performed by binding the resin beads with cement-based materials [53], or including them into polymer matrixes by using polymer binders [54]. Overall, using the polymer binders to solidify ion exchange resins is preferred because it adds less to the volume of the material [54]. These types of radioactive waste processing adhere to the conditions for solidified waste, such as mechanical strength, water or radiation resistance, however, they pose a challenge when it comes to the recycling of said waste [54]. Radioactive waste solidified via cement-based binding materials is not likely to be recycled at all [53], whereas polymer-based binding materials may be recycled, but it is hard to choose an optimal method and technology for such a task [54], which also discourages the recycling of these materials.

### 1.3.2. Radioactive liquid waste from spent fuel pool

A spent fuel pool is a specially designed storage pool used to safely store used nuclear fuel rods after they've been removed from a nuclear reactor. These rods are still highly radioactive and have high temperatures, so they need cooling and shielding, which is provided by the spent fuel pool [55]. Spent fuel pool is a deep, reinforced pool, filled with demineralized light water [56] (borated and/or reinforced with boral panels [57]), which blocks radiation and draws heat away from rods [58] (Fig. 3). As a result of high temperatures, ionizing radiation and direct contact of nuclear fuel cladding with this water, a radioactive nuclear waste is produced.

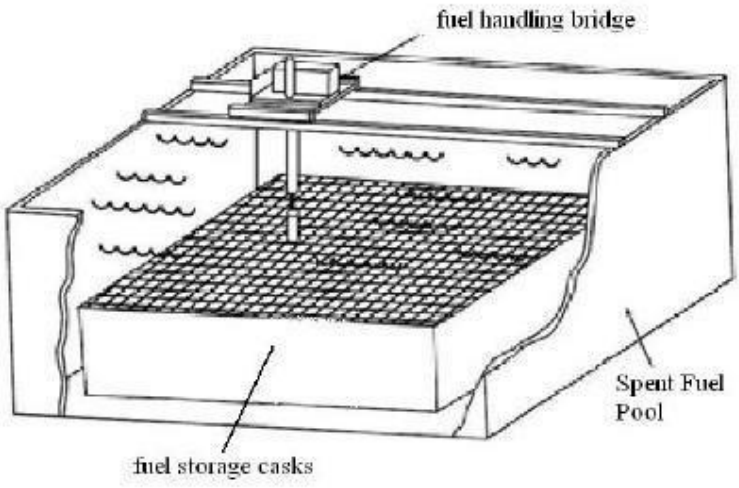



**Fig. 3.** Schematic diagram of spent fuel pool [59]

Overall, the radionuclides found in spent fuel pools are the same as the ones in the reactor coolant water, since they use similar additives to mitigate the corrosion, such as boron additives or structural elements and incorporate a lot of the same elements for the reinforcements. However, the overall quantity of radionuclides is typically smaller than in reactor coolant water [60], because the neutron flux is much smaller – the fuel in the pool is no longer undergoing active fission reactions, and the rate of neutron emission decreases over time as the fuel cools and the fission products decay [60]. Moreover, the spent fuel pool water may contain radioisotopes as a result of damaged or corroded nuclear fuel cladding. As the fuel gets in contact with water, some radionuclides, especially those water soluble, enrich the surrounding water with radionuclides. Due to the damaged or defective fuel cladding, uranium fuel is released into the water (uranium-238 and uranium-235), as well as fission products, such as ceasium-137 or strontium-90 [61]. Moreover, less soluble radionuclides, for example plutonium, may also get into the water, especially if the damage to the fuel cladding is severe.

Spent fuel pool water, similarly to reactor coolant water, gets filtered before the disposal to remove radionuclides and accumulated debris. The mechanisms, needed for the radioactive wastewater treatment, are equipped based on the need and the composition of reactor coolant water or spent fuel pool water.

### 1.3.3. Radioactive liquid waste from fuel reprocessing

Radioactive liquid waste from nuclear fuel reprocessing is a highly hazardous byproduct generated during the chemical separation of usable materials (primarily uranium and plutonium) from spent nuclear fuel. The most common reprocessing technique, such as the PUREX (plutonium-uranium redox extraction) process, involves dissolving spent fuel in nitric acid, resulting in a solution containing uranium, plutonium isotopes, transuranic elements and a complex mixture of fission products [62, 63]. Liquid waste acquired thorough PUREX process is typically categorized as high-level waste due to its intense radioactivity and heat generation [62].

Management of radioactive liquid waste from this process involves concentrating and immobilizing the radioactive components into stable solid forms, such as vitrified glass [64], suitable for long-term geological disposal. This method of management reduces the risk of environmental contamination

and ensures safer storage due to compositional flexibility, sufficient immobilization of hazardous materials, high corrosion resistance, mechanical and radiation durability [64].

Advanced reprocessing technologies aim to improve waste minimization and enhance separation efficiency, including the recovery of minor actinides like americium and curium, which are challenging to extract with conventional solvents [65]. These efforts support fuel cycle strategies that recycle actinides to reduce the volume and radiotoxicity to create waste suitable for disposal under regulatory authorization [65]. Alternative pyrochemical processes, involving high-temperature molten salt dissolution and electrochemical separation, also produce radioactive liquid waste streams [66, 67] that require specific handling and treatment protocols.

In summary, the effective management of radioactive liquid waste generated during fuel reprocessing is essential, whether it is produced via conventional methods like PUREX or more novel pyrochemical techniques. As in any field with radioactive liquid waste, continued research and innovations are necessary. However, fuel reprocessing waste needs additional focus because of the high radioactivity content.

### 1.3.4. Radioactive liquid waste from uranium mining and milling

Radioactive liquid waste from uranium mining and milling primarily originates from the processing of uranium ore [68]. During uranium ore processing, the ore is crushed and chemically treated (commonly with acid or carbonate solutions) to extract uranium, resulting in radioactive mill tailings [68]. These tailings contain most of the radioactivity originally present in the ore, including uranium decay products such as radium and thorium isotopes, as well as toxic metals like arsenic and vanadium [68]. Dissolved radionuclides and heavy metals have increased potential for environmental contamination if not properly managed.

The liquid waste is typically stored in tailings ponds or impoundments (Fig. 4), but these can leak or overflow, releasing radioactive contaminants into surrounding soil and water bodies. Such releases have been documented to cause elevated radioactivity in surface waters and groundwater near mining sites, posing risks to ecosystems and human health [68, 69]. Radionuclides, released into the environment in such cases include long-lived isotopes from uranium-238 and thorium-232 decay series, like radium-226, lead-210 and thorium-232 [70]. Monitoring studies have shown that while most releases meet radiological safety limits, chemical toxicity from associated metals may still restrict water use [70].

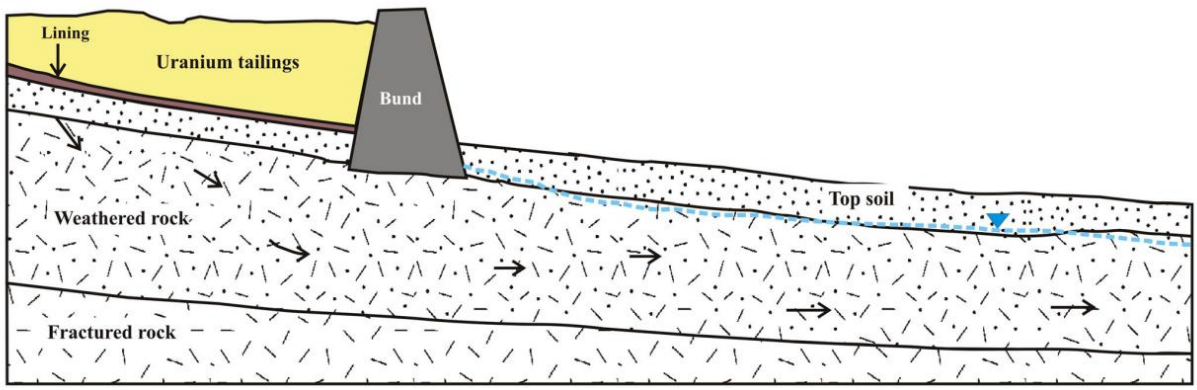



**Fig. 4.** Conceptual model of uranium tailings [71]

The treatment of radioactive liquid waste from uranium mining and milling involves processes to reduce radionuclide concentrations and chemical toxicity before discharge, including precipitation, ion exchange and pH adjustment [72]. However, the complex chemistry of uranium and its decay products, combined with the large volumes of waste, makes long-term containment and remediation challenging. Uncovered tailings and tailings water can generate acid mine drainage, further mobilizing radionuclides and metals [68]. Consequently, strict waste management, continuous monitoring, and remediation efforts are essential to mitigate the environmental impact of radioactive liquid waste from uranium mining and milling [68, 73].

## 1.4. Radioactive wastewater purification methods

This chapter provides an insight into the key technologies used for the treatment of radioactive wastewater, focusing on the effective removal of radionuclides to meet environmental and safety standards. The covered treatment technologies include chemical precipitation, physical filtration with mechanical and membrane filters, evaporate concentration technology, adsorption method, microorganisms utilizing biotechnology, photocatalysis and ion exchange technologies via resins or inorganic materials [74] (Fig. 5).

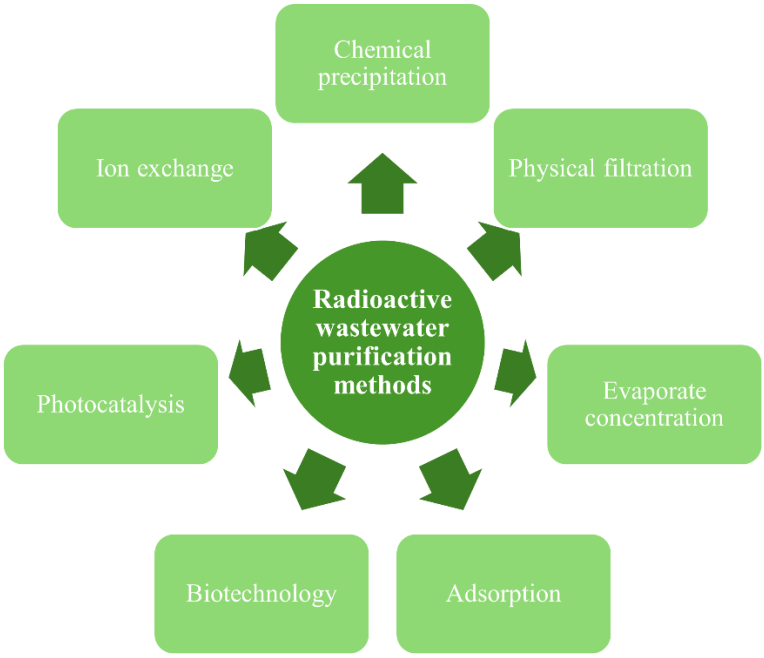



**Fig. 5.** Summary of radioactive wastewater purification methods

### 1.4.1. Chemical precipitation

Chemical precipitation is a foundational method for treating radioactive wastewater, relying on insoluble compound formation to remove radionuclides. This process involves adding precipitants like aluminum salts, phosphates, iron salts or sodium bicarbonate [74] to the wastewater, which bound contaminants into larger, physically filterable particles. While cost effective and simple, traditional agents struggle to coagulate cesium, iodine and ruthenium [74], although they are likely to be present in the radionuclide contaminated liquid waste, especially in nuclear power production nuclear waste. There is research on how to achieve precipitation of these resistant radionuclides, but the methods

require additional, more expensive compounds and additives to attain the desired result [75]. Moreover, factors such as pH value, mixing duration and dosage play a crucial role in chemical precipitation [76], which might impact how efficiently the radioactive liquid waste contaminants and the chemical agent interact. Such treatment is fruitful for the low-level waste, but for higher radioactivity and longevity contents chemical precipitation is combined with other radioactive wastewater purification methods, for example evaporation or ion exchange materials [74].

### 1.4.2. Physical filtration

Physical filtration involves creating a physical barrier with adjusted pore sizes and structure that only lets through the solvent or non-toxic particles of the filtration solutions. The physical filtration barriers may be a simple metal or fabric mesh, but for smaller particles - membranes may be used. For instance, membranes made from amyloid fibers filters out technecium-99m, iodine-123 and gallium-68 from water, as well as iodine-131 and lutetium-177 from clinical wastewater [77]. These membranes work by adsorbing radioactive ions onto their extensive surface area, effectively reducing radioactivity to below detectable levels and converting large wastewater volumes into manageable solid waste [77]. Ultrafiltration systems integrated with low-cost sorption agents, such as activated carbon or aluminosilicates, enhance efficiency of radionuclide capture while maintaining cost-effectiveness [78], although this type of filtration usually requires additional processing to optimize the process and collect low concentration, standalone isotopes [79, 80].

### 1.4.3. Evaporate concentration technology

Evaporate concentration technology remains a cornerstone for radioactive wastewater treatment, relying on the non-volatile nature of most radionuclides to concentrate contaminants into residual solids. For example, mechanical vapor recompression systems achieve decontamination factors of $7 \cdot 10^5$ for strontium, cobalt and cesium [74], by using heat-pump based method of recycling the latent heat, enabling efficient treatment of high-salinity wastewater without chemical additives [74, 81]. Furthermore, to reduce the cost and lessen the environmental impact of this technology, there is research and application of solar power [82]. However, the evaporate concentration method is not suitable for volatile compounds, such as iodine [83], which is one of the main radionuclides in clinical wastewater, and the process overall is energy consuming [74], especially when advanced heat recovery or renewable integration is not employed.

### 1.4.4. Adsorption

Adsorption method is based on the physical adherence or bonding of ions and molecules onto the surface of another molecule [84], offering efficient and cost-effective radioactive wastewater purification from contaminants like uranium and cesium isotopes. For example, nanocomposite aerogels, composed of graphene oxide and chitosan, show enhanced uranium adsorption capacity due to the high surface area, functional groups of graphene oxide and biocompatibility of chitosan [85]. Besides, environmentally friendly materials, like biochar – recycled general waste, are explored to clean radionuclide-contaminated liquid waste [86].

Additionally, cesium may be collected by adsorption with various inorganic, organic and biological adsorbents [87]. The inorganic adsorbents used for cesium removal are zeolite composites, metal hexacyanoferrates [87], attapulgite clay [88] and activated carbon derivatives [89]. Organic counterparts for cesium removal are some of the functionalized polymers, such as electroactive

polypyrrole [90], or biological adsorbents, like tea leaves crosslinked with concentrated sulfuric acid [91]. Overall, adsorption technology offers a wide range of materials to choose from, some of which are relatively cheap. However, the materials are often combined to achieve the complete removal of wastewater contaminated with several types of radioisotopes, which may reduce the cost-effectiveness and simplicity of this method.

### 1.4.5. Biotechnology

Biotechnology uses microbial and fungal mechanisms to immobilize or detoxify radionuclides, like plutonium, uranium, neptunium thorium, cobalt, technetium and strontium isotopes. For example, microbial biosorption (BioS in Table 3) utilizes cell surface functional groups, like phosphates or sulfides, to biologically adsorb radioactive ions, *Pseudomonas* microbial species effectively separating uranium from the environment contaminated with radionuclides [92, 93].

Bioreduction (BioR in Table 3) advantages from bacteria that are capable of living in extreme environments to convert soluble radionuclides into insoluble forms. For example, sulphate- and iron-reducing bacteria at United States Department of Energy (U.S. DOE) sites enzymatically reduce $U^{6+}$ cation into $U^{4+}$ which forms less mobile and less soluble compounds, preventing uranium of getting into the groundwaters [94].

Bioaccumulation (BioA in Table 3) exploits structural mimicry, where microbes like *Arthrobacter nicotianae* and *Bacillus megaterium* uptake thorium and uranium ions, often metabolizing them as phosphates [92, 93]. However, as microorganisms take up the radionuclides, they are continuously exposed to the ionizing radiation, which can damage microbial DNA and cellular components, limiting the survival and metabolic activity of bioaccumulating microbes [95, 96]. Even radiotolerant species like *Deinococcus radiodurans* have limits to radiation exposure [96], and high radiation doses can reduce biomass and bioaccumulation efficiency. Moreover, as the radionuclides are confined within the cell barriers as pseudonutrients, as the cell membrane gets damaged, (under stress or cell death), radionuclides and contaminants may be released back into the environment [97], posing secondary contamination risks, which is not the case with other bioremediation processes.

Biomineralization and bioprecipitation (BioM in Table 3) harness the metabolic activity of microorganisms to convert soluble radionuclides into stable, insoluble mineral forms [92], which makes their removal from the environment easier, especially in the liquid medium. One of those instances is genetically engineered *Deinococcus radiodurans* which, as it expresses the phoN gene from Salmonella enterica, liberates inorganic phosphate on the cell surface, leading to precipitation of over 90 % of uranium from solution within 6 hours [98]. Similarly, naturally occurring bacteria like Serratia species have shown an ability to promote the formation of uranium phosphate precipitates through inherent phosphatase activity [99].

**Table 3.** Summary of bacteria species and genera mechanisms of radionuclide bioremediation [92]

| Genus | BioR | BioM | BioS | BioA | Element |
|---|---|---|---|---|---|
| Geobacter metallireducens GS 15 | ✓ | | | | Plutonium |
| Shewanella oneidensis | ✓ | | | | Plutonium, uranium |
| Bacillus mycoides | ✓ | | | | Plutonium |
| Serratia marcescens | ✓ | | | | Plutonium |
| Shewanella putrefaciens | ✓ | | | | Plutonium, neptunium |
| Citrobacter sp. | | ✓ | | ✓ | Neptunium, thorium, plutonium |
| Pseudomonas sp. | | | ✓ | ✓ | Uranium |
| Deinococcus radiodurans | ✓ | ✓ | | ✓ | Uranium, cobalt |
| Desulfobvibrio desulfuricans G20 | | ✓ | | | Uranium, technetium |
| Anaeromyxobacter dehalogenans | ✓ | | | ✓ | Uranium, technetium |
| Halomonas sp. | | ✓ | | | Strontium |
| Bacillus subtilis | | | | ✓ | Thorium |
| Arthrobacter nicotianae | | | | ✓ | Thorium |
| Micrococcus luteus | | | | ✓ | Strontium |
| Bacillus megaterium | | | | ✓ | Thorium |

Mycoremediation (fungal) and phycoremediation (algal) systems complement bacterial methods. Fungi exhibit high biosorption capacities, while algae, like Chlorella, adsorbs radionuclides via surface polysaccharides [94, 100]. Moreover, fungi naturally tend to accumulate more radioactive isotopes, especially cesium, than the surrounding flora [101], which shows their resilience to radioactivity. Research by Davies et al. (2015) [102] showed that arbuscular mycorrhizal fungi play particularly crucial roles in radionuclide bioaccumulation, with studies demonstrating their ability to enhance the translocation of uranium-238 into plant root tissues while simultaneously preventing further mobilization into shoot tissues [102]. This selective transport mechanism makes arbuscular mycorrhizal fungi especially valuable for containment strategies, as approximately 80 – 90 % of plant species maintain symbiotic relationships with these fungi [102]. These complementary processes allow fungi to function both as biological barriers that limit radionuclide spread and as active removal agents [102].

Generally, bioremediation is a sustainable and eco-friendly option for contaminant removal from radionuclide contaminated wastewater. However, bioremediation of wastewater faces several obstacles that limit its field application despite the promising in vitro results. For one, it is difficult to predict their spread and ensure their containment within wastewater, raising issues of cross-contamination and equipment damage. To solve this problem the following microorganism immobilization technologies may be implemented:

- Entrapment - embedding microorganisms in polymeric matrices, like agar, alginate gelatin [93] or polyvinyl alcohol [103].
- Encapsulation – confining bioremediatory cells within semi-permeable capsules, which enables substrate and nutrient exchange [104].

- Physical adsorption – relies on weak intermolecular forces to attach biocatalysts to solid support [104].
- Chemical attachment – forms strong covalent bonds to bind microorganisms onto the chemically modified solid support using crosslinkers [104]

Different immobilization techniques introduce other advantages and limitations. For example, physical adsorption is a reversible method, but it is limited by low binding strength and reduced efficiency under dynamic or harsh conditions [104]. Chemical attachment offers high stability and resistance to harsh conditions, but its application is limited by challenges in preserving cell viability and reproducibility [104]. Entrapment and encapsulation provide strong protection against environmental stress and mechanical damage, but the barrier limits substrate diffusion thus affecting contaminant filtration and microbial survivability [104].

On the other hand, research shows that immobilized microorganisms may outperform free cells in some cases. For example, Nisar et al. (2024) [105] found that immobilized *Pseudomonas oleovorans* biodegraded 98 to 99 % of used engine oil in all investigated concentrations after seven days, whereas their free counterparts were well below 85 % mark under the same conditions and below 70 % in highest used engine oil solution concentration (7 % solution) [105]. This result was explained by this research to be caused by higher stability and greater mechanical strength of immobilized cells compared to the free *Pseudomonas oleovorans* [105].

Another drawback of using microorganisms for bioremediation is that they are a complex formation – not only the efficiency of the bioremediation, but also the survivability of the cells heavily depends on temperature, alkalinity, acidity and pressure [93, 104, 105]. These sensitivities necessitate precise control of environmental parameters in bioremediation systems, especially when applied in situ where external conditions can be highly variable. To overcome these challenges, researchers have explored strategies such as genetic engineering of robust microbial strains [106], optimization of immobilization matrices to buffer environmental fluctuations [107] and integration of real-time monitoring systems to maintain optimal conditions [108]. However, the solutions for keeping the environmental factors stable introduce additional cost and pollution, which takes away from the initial biotechnology appeal that it promises.

### 1.4.6. Photocatalysis

Photocatalysis process involves the use of photocatalysts (typically semiconductor materials such as titanium oxide ($TiO_2$) or novel porphyrin-based frameworks) that, upon light irradiation, generate electron-hole pairs [109]. These charge carriers induce redox reactions that can mineralize or transform radioactive contaminants [109]. Specifically, in radioactive wastewater treatment, photocatalysis enables the reduction of soluble radioactive ions to insoluble forms (like uranium $U^{6+}$ to uranium $U^{4+}$), enabling their removal from water. For example, a study by Wu et al. (2024) [110] has demonstrated that using Ni-metallized porphyrin-based hydrogen-bonded organic framework (Ni-pHOF) it is possible to remove over 86 % of $U^{6+}$ from real low-level radioactive wastewater within two hours under visible light, with high selectivity and reusability [110]. The underlying mechanism involves enhanced charge separation and participation of reactive species such as superoxide radicals and electrons in reduction reactions. Photocatalysis also generates highly reactive hydroxyl radicals that oxidize organic and inorganic pollutants, further purifying the water [111]. Despite challenges like the need for optimal light conditions and photocatalyst stability, advances in

material design and reactor engineering continue to improve the efficiency and practical applicability of photocatalytic treatment for radioactive wastewater [112]. This approach is a cost-effective and sustainable alternative to conventional methods, capable of degrading complex contaminants while minimizing secondary pollution.

### 1.4.7. Ion exchange technology

Ion exchange technology is a widely used and highly effective method for purifying radioactive wastewater, especially in nuclear power plants and fuel reprocessing facilities [113, 114]. Ion exchange technology can be divided into two main sections based on the materials used – organic (resins) and inorganic ion exchangers.

Organic ion exchange resins are insoluble, cross-linked polymer matrices containing functional groups that facilitate reversible ion exchange with surrounding solutions. These synthetic polymers can be divided into two main groups: cation and anion exchangers. Cation exchange resins may also be divided into strong acid cation and weak acid cation resins. Weak acid cation resin is typically a polymer bound with carboxylic acid (-COOH) functional groups, more suitable to remove ions which contribute to the alkalinity of the solution [115]. However, weak acid cation resin is not suitable for high acidity solutions and removal of strong mineral salts, that are likely to be present in urine, such as sodium chloride (NaCl) or sodium sulfate ($Na_2SO_4$) [115].

On the other hand, strong acid cation resin, usually made of styrene-divinylbenzene copolymers, with functional groups of sulphonic acid (-$SO_3^-H$) attached (Fig. 6) function well thorough whole pH range and withstands temperatures up to 120 °C [116]. Sulfonated polymer ion exchange resin is able to remove all cations from the solutions, but there is a selectivity for ions – resin is more likely to bind the ions of the higher valence and higher atomic number [116].

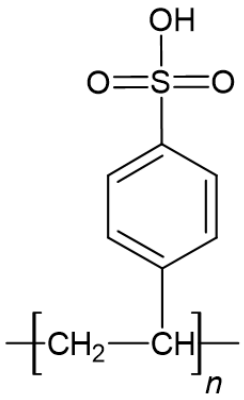



**Fig. 6.** Sulfonated polystyrene monomer

Anion exchange resins are categorized into weak base and strong base resins. Weak base anion exchange resins are polymer matrices attached to primary, secondary or tertiary amines [117]. This type of anionic ion exchange resin lacks ion exchange capability, especially under alkaline setting, because weakly alkaline functional group does not dissociate under alkaline conditions [117]. Weak base anion resin is more effective at removing anions of strong acids (for example chloride or bromide) and is easily recuperated [117], but utilization of such ion exchange resin is disadvantageous towards some products of urinary output, like strong mineral salts.

Strong base anionic ion exchange resin features quaternary ammonium ($-N^+R_3$) groups embedded into the styrene structure [118] (Fig. 7). These, depending on the radicals attached to nitrogen, can also be attributed to type I or type II strong base anionic resin and have distinct characteristics. Type I strong base resin has methyl ($-CH_3$) radicals attached to the nitrogen [118]. Type I resin is overall more stable compared to type II – it withstands higher temperatures, works in a wider pH range and has a higher resistance to organic fouling (buildup of carbon-based materials on the compound surface). Type II resin has two methyl ($-CH_3$) and one ethanol ($-CH_2CH_2OH$) groups bonded with nitrogen [118], which results in lower efficiency compared to type I resin, but the regeneration of this resin is easier and requires less recuperation solution. Both type I and type II anionic exchange resin are capable of removing ions from aqueous strong mineral salts [118].

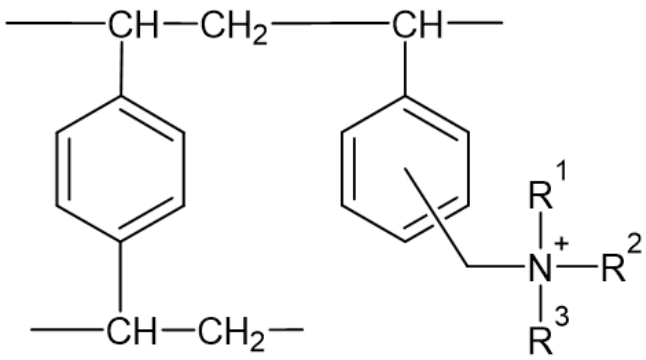



**Fig. 7.** Structure of strongly basic anion exchange resin; R1, R2 and R3 represent radicals of organic nature

Also, to get the effect of both basic and acidic organic resins, mixed-bed resins are composed of a combination of cation and anion exchange resins, typically in a 1:1 ratio and are used to achieve high levels of deionization. These resins consist of a mixture of strong acid cation exchange and strong base anion exchange components, which allows for the simultaneous removal of both positively and negatively charged ions from a solution. Due to their high purity output, mixed-bed resins are commonly employed as a final step in water purification systems. However, their use can be limited by regeneration complexity, as the resins must be separated and regenerated individually, making the process more labor-intensive and less cost-effective compared to single-bed systems [119].

Traditional inorganic ion exchangers include a diverse array of materials, including zeolites, titanium and zirconium phosphates, silicotitanates and titanosilicates, hydrous oxides and various other materials [120, 121]. These materials are characterized by their firm and adaptable framework structures, which confer high selectivity for specific ions, particularly trace radionuclides and rare earth elements [120, 122]. The selectivity arises from the precise control over pore size and charge distribution within the frameworks, allowing only ions of matching size and charge to be effectively exchanged.

In addition to their selectivity, inorganic ion exchangers exhibit superior chemical, thermal and radiation stability compared to their organic counterparts [120]. This robustness makes them particularly suitable for applications in harsh environments. For instance, materials like zeolites and titanosilicates have been effectively employed in the removal of cesium ions from radioactive wastewater [122, 123], demonstrating their resilience and efficacy under challenging conditions.

Moreover, the inorganic nature of these materials renders them more resistant to microbial growth, thereby reducing contamination risks during use [122]. This property is advantageous in applications where sterility and longevity of the ion exchange material are critical.

Conversely, organic ion exchange resins, typically composed of porous polymer matrices, are favored for general water treatment and application under milder conditions. Their porous structure results in faster ion exchange kinetics, and their lightweight, cost-effective nature makes them suitable for both small- and large-scale purification processes. Additionally, the plasticity of organic resin allows them to be easily molded into various shapes, enhancing their versatility in different system designs.

In summary, while inorganic ion exchangers are indispensable in specialized scenarios that demand high selectivity and stability under extreme conditions, organic ion exchangers are preferred for general applications due to their favorable kinetics and adaptability.

## 2. Methods and materials

This chapter outlines the experimental procedures and materials employed to assess the effectiveness and reusability of resin-based filter in processing artificial urine and artificial urine with iodide additives.

### 2.1. Filter construction

The filter was designed to be printed with Zortrax M200 plus 3D-printer. The frame of the main filter elements was constructed out of 3 cylindrical shapes, giving the element a shape which allows filter elements to stack on top of each other (Fig. 8).

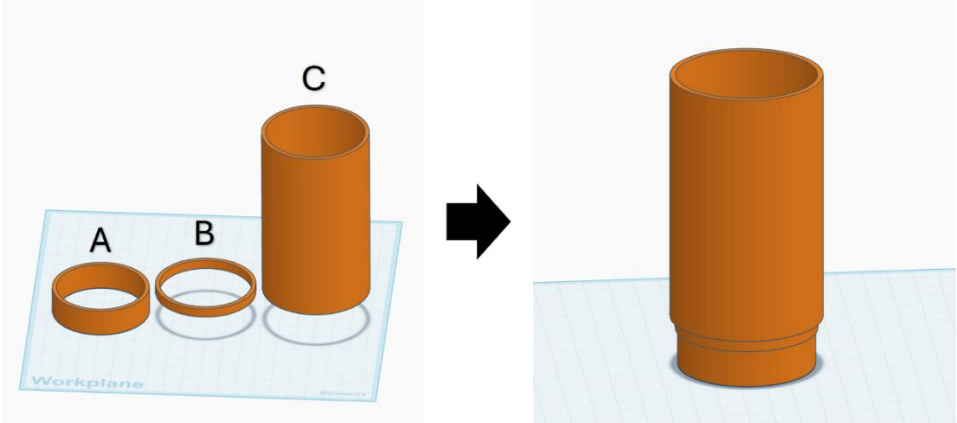



**Fig. 8.** Filter frame design

To make the collection of the filtrate easy, a collection container (Fig. 9) was also designed – a cylinder with closed off bottom. It also becomes a stable bottom part for the filter.

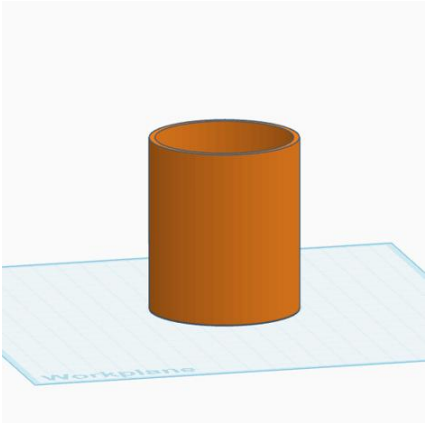



**Fig. 9.** Frame of filtrate container

The filter was designed as a slim column to ensure that the solutions poured through the filter cover larger surface area of the ion exchange resin in the filter, enhancing the efficiency of the filtration process, but is still wide enough to easily pour in the solution (Table 4).

**Table 4.** Parameters of the filter frame

| Cylinder | Outer diameter, mm | Inner diameter, mm | Height, mm | Wall thickness, mm |
|---|---|---|---|---|
| A | 56 | 52 | 15 | 2 |
| B | 58 | 54 | 5 | |
| C | 60 | 56 | 100 | |
| Element | 56 – 60 | 52 – 56 | 120 | |
| Container | 60 | 56 | 70 | |

The bottom part of the filter element was closed off to retain the resin beads within the filter element. It was done by fixing the double layer of cotton bandage with thermoplastic elastomer at a height of around 20 mm from the bottom of the filter element (Fig. 10). Also, to ensure that stacked filter elements are connected stably, to minimize the possibility of the spill and to differentiate the filter elements the colored rubber bands were attached to the outer part of filter elements, at around 15 mm height. Purple and green marked filters were designated to filtrate the anions in the solution, while red and yellow marked filters were for cation filtration.



**Fig. 10.** The 3D-printed filter elements and filtrate container and the inside of the filter element without resin present

The filter elements were filled with ion exchange resin beads (Fig. 11) – sulfonated polystyrene as cation-exchange resin (in yellow and red marked filter elements) and quaternary ammonium beads as anion-exchange resin (in purple and green marked filter elements). Every element contains around 200 ml of ion exchange resin beads.

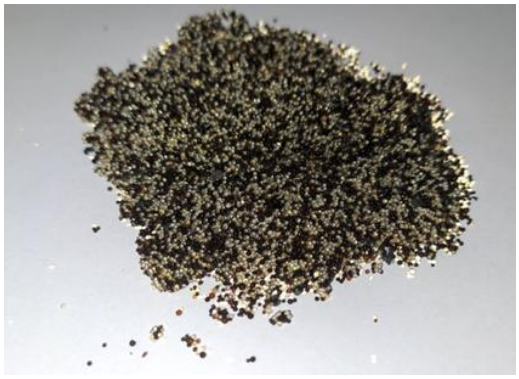


**Fig. 11.** Ion exchange resin beads

## 2.2. MP artificial urine preparation

The artificial urine (Fig. 13) for testing was prepared as described by Neslihan et al. (2019) [124]. The components used for the 500 g MP-AU are shown in Table 5.

**Table 5.** Components used for MP-AU [124]

| Component | Molarity (mM) | Quantity (g/100ml) | Quantity for (g/500ml) |
|---|---|---|---|
| Sodium sulfate | 11.97 | 0.1700 | 0.8586 |
| Uric acid | 1.49 | 0.0250 | 0.1263 |
| Sodium citrate dihydrate | 2.45 | 0.0720 | 0.3636 |
| Creatinine | 7.79 | 0.0881 | 0.4495 |
| Urea | 249.75 | 1.5000 | 7.5377 |
| Potassium chloride | 30.95 | 0.2308 | 1.1657 |
| Sodium chloride | 30.05 | 0.1756 | 0.8824 |
| Calcium chloride | 1.66 | 0.0185 | 0.0944 |
| Ammonium chloride | 23.67 | 0.1266 | 0.6362 |
| Potassium oxalate monohydrate | 0.19 | 0.0035 | 0.0179 |
| Magnesium sulfate heptahydrate | 4.39 | 0.1082 | 0.5465 |
| Sodium dihydrogen phosphate dihydrate | 18.67 | 0.2912 | 1.4857 |
| Disodium hydrogen phosphate dihydrate | 4.67 | 0.0831 | 0.4197 |

Sodium sulfate (≥ 99 % purity), uric acid (≥ 99 %), urea (≥ 99.5 %), calcium chloride (≥ 98 %), ammonium chloride (≥ 99.5 %) potassium oxalate monohydrate (≥ 98 %), magnesium sulfate heptahydrate (≥ 99 %), sodium dihydrogen phosphate dihydrate (≥ 98 %), disodium hydrogen phosphate dihydrate (≥ 99 %) used to make the artificial urine were from Carl Roth. Potassium chloride (99.0 – 100.5 %) and sodium chloride (≥ 99.5 %) from Sigma-Aldrich. Sodium citrate dihydrate (≥ 99 %) from Reachem Chemicals and creatinine (98 %) from Thermo Scientific. All components used for the artificial urine were in powder or crystalline powder form (Fig. 12).

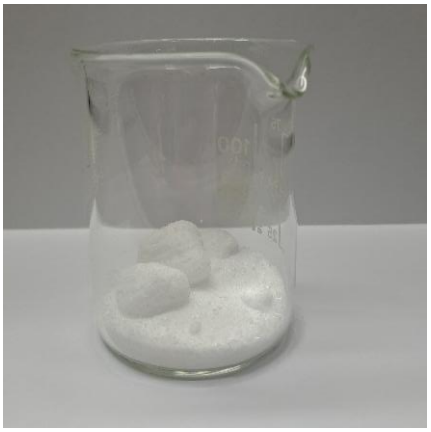


**Fig. 12.** Powder materials for the MP artificial urine

The components to the MP artificial urine were added into the water as shown in Table 3 and mixed at 700 rpm, keeping the temperature around 37 °C for 24 hours [124]. For the mixing and heating Heidolph MR 3001 K hot plate with magnetic stirrer (Fig. 14. A) was used. MP-AU was prepared in a 500 ml conical flask, covered with parafilm to negate the effects of evaporation of water and keeping the precise volume of artificial urine.

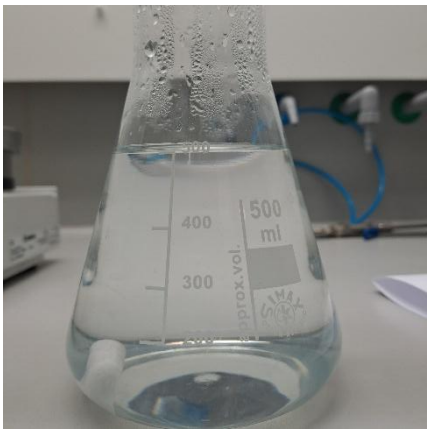



**Fig. 13.** MP artificial urine solution

Total dissolved solids after the filtration in filtrate were measured with the ZeroWater total dissolved solids meter (Fig. 14. B) and the pH of the artificial urine before the filtration, artificial urine with additives before the filtration and the solutions after the filtration were measured with Bigstren digital water tester (Fig. 14. C).

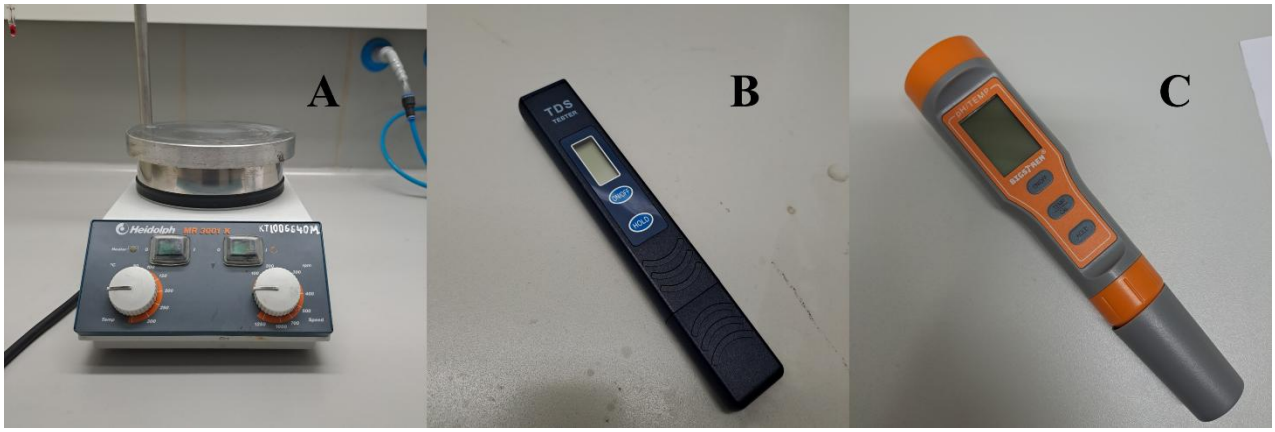



**Fig. 14.** A – Heidolph MR 3001 K hot plate; B – ZeroWater total dissolved solids meter; C – Bigstren digital water tester

Since the materials added were measured with 0.001 g precision, the total dissolved solids (TDS) in the starting artificial urine solution were calculated using this formula:

$$TDS\ (ppm) = \frac{Total\ mass\ of\ dissolved\ solids\ (mg)}{Volume\ of\ solution\ (L)} \quad (13)$$

The precise quantities of the materials added to the certain batches of the artificial urine are as shown in Table 6.

**Table 6.** Factual mass of components used for each artificial urine batch

| Component | 1st batch, g | 2nd batch, g | 3rd batch, g |
|---|---|---|---|
| Sodium sulfate | 0.8602 | 0.8565 | 0.8586 |
| Uric acid | 0.1374 | 0.1263 | 0.1263 |
| Sodium citrate dihydrate | 0.3606 | 0.3621 | 0.3636 |
| Creatinine | 0.4475 | 0.4509 | 0.4495 |
| Urea | 7.5130 | 7.5343 | 7.5377 |
| Potassium chloride | 1.1713 | 1.1665 | 1.1657 |
| Sodium chloride | 0.8736 | 0.881 | 0.8824 |
| Calcium chloride | 0.0945 | 0.0944 | 0.0944 |
| Ammonium chloride | 0.6805 | 0.6725 | 0.6362 |
| Potassium oxalate monohydrate | 0.0207 | 0.0195 | 0.0179 |
| Magnesium sulfate heptahydrate | 0.5487 | 0.5461 | 0.5465 |
| Sodium dihydrogen phosphate dihydrate | 1.4790 | 1.4934 | 1.4857 |
| Disodium hydrogen phosphate dihydrate | 0.4131 | 0.4158 | 0.4197 |
| Total mass of dissolved solids | 14.6001 | 14.6193 | 14.5840 |

The specifications of the prepared MP-AU batches were measured and calculated as described and the results are shown in Table 7.

**Table 7.** Specifications of artificial urine batches

| No. | Uses | pH | TDS, ppm |
|---|---|---|---|
| 1 | Testing the MP-AU, performing filtrations | 5.89 | 29200 |
| 2 | Performing filtrations with potassium iodide | 5.58 | 29238 |
| 3 | Performing filtrations with potassium iodide | 5.60 | 29486 |

The pH of the MP-AU in the paper is described to be between 5.5 and 6.5 (slightly acidic) [124], which was achieved after following the preparation procedure. The pH of the artificial urine were in the range from 5.58 to 5.89 and TDS calculations were from 29200 to 29486 ppm.

### 2.3. Filtration and the recovery of resin beads for reuse

To ensure proper filtration, these steps were taken for the filtrations of artificial urine and artificial urine with iodide ions:

- Washing filter with 200 ml of distilled water to remove previous filtrate fully.
- Washing filter with 200 ml of the artificial urine or artificial urine with iodide (based on the solution filtered).
- Pouring solution into the filter.

Following these steps ensures that the filtrate gotten after the last step accurately reflects the efficacy of the filter on the specific solution that is being filtered.

Depending on the surface area of the filter ion exchange beads and the nature of ions (oxidation state), 200 ml of resin beads should clear around 2 to 4 grams of dissolved solids from the solutions. However, after the resin is spent, it is possible to recover ion exchange resin. To recover sulfonate polystyrene (cationic) resin, an acid should be used and for the quaternary amine – alkaline agent. In this project, 5 % sulfuric acid ($H_2SO_4$) and 5 % sodium hydroxide (NaOH) solutions were prepared to recover the filter elements for repeated use.

### 2.4. Qualitative iodide ($I^-$) analysis

The general reactivity of the chemical element relies on the number of protons, electrons and the arrangement of electrons in the orbitals. Since the parameters mentioned correspond in the case of radioactive and non-radioactive isotopes (in this case - iodide ion), only the atomic mass is different (number of neutrons in the nucleus), it is inferred that normally sold stable iodine will behave identically to pharmaceutically used iodine-123, iodine-125 or iodine-131 in chemical reactions. Ion exchange-based filter works by creating chemical bonds, so to test the efficacy of the filter non-radioactive iodine isotopes potassium iodide aqueous solution (Fig. 15) was used. 0.84 g of potassium iodide salt from Eurochemicals (≥ 99.0 % purity) was added to 50 ml of distilled water to get 0.1 M potassium iodide solution.

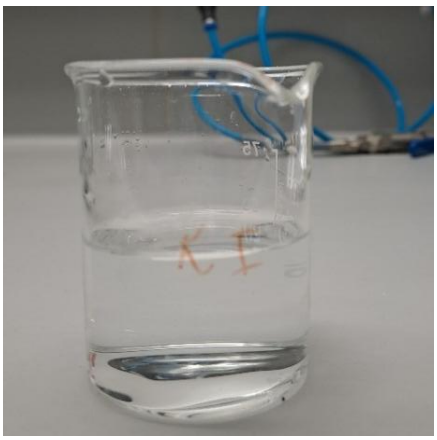


**Fig. 15.** 0.1 M potassium iodide solution

In the study by Lee et al. (2015) [125] maximum iodide ion concentration in urine after iodine procedure was registered at 0.157 g/L [125]. To achieve that, 5 ml of 0.1 M potassium iodine was added to 395 ml artificial urine solutions, making the concentration of iodide ions in the artificial urine around 0.159 g/L.

Further in this chapter are described qualitative analysis methods – silver nitrate test for halogens and starch test for iodide, used in this project to detect the presence of iodide ions.

#### 2.4.1. Silver nitrate test for halogens

The silver nitrate ($AgNO_3$) test is a classic qualitative test for identifying halide ions ($Cl^-$, $Br^-$ and $I^-$) in a solution. In the case of iodide, it is as described in the following formula:

$$AgNO_3(aq) + I^-(aq) \rightarrow AgI(s) + NO_3^-(aq) \quad (14)$$

To carry out the test, firstly the solution should be acidified with the dilute nitric acid ($HNO_3$) to remove interfering anions like carbonates and hydroxides, as silver ions form other solids with these reagents. Carbonate and silver ions produce insoluble silver carbonate ($Ag_2CO_3$), which appears as white crystals. Silver and hydroxide forms silver hydroxide, which rapidly split into silver oxide ($Ag_2O$), giving the solution a rusty color.

$$AgNO_3(aq) + CO_3^{2-}(aq) \rightarrow Ag_2CO_3(s) + NO_3^-(aq) \quad (15)$$

$$AgNO_3(aq) + OH^- \rightarrow AgOH(s) + NO_3^-(aq) \quad (16)$$

$$2AgOH \rightarrow Ag_2O + H_2O \quad (17)$$

Halide ions with silver nitrate form solid silver chloride (white crystals), silver bromide (pale yellow crystals) and silver iodide (bright yellow crystals). As silver carbonate and silver hydroxide will not be forming if nitric acid is added beforehand and bromide ions should not be present in the prepared solutions, that leaves two solids forming to consider – silver iodide and silver chloride (because of chloride from the artificial urine components). However, dilute ammonia dissolves the silver chloride and does not impact silver iodide crystals, which allows to distinct the presence of iodide ions in the starting solution.

Therefore, the steps of the procedure are as follows:

- Adding around 0.3 ml of diluted nitric acid into the test tube.
- Adding around 0.3 ml of silver nitrate solution to form crystals.
- Adding 0.1 ml dilute ammonia, dissolving silver chloride (required if the crystals have formed in the test tube).
- If the crystals are present, the test for iodide ions is positive. If there are no crystals in the test tube, the test for iodide ions is negative.

Dilute ammonia (14.0 – 15.0 %) was prepared by mixing 25 ml of ammonium hydroxide aqueous solution from Sigma-Aldrich (28.0 – 30.0 % $NH_3$ basis) with 25 ml of distilled water. Diluted nitric acid (7 %) has been composed by adding 5 ml of nitric acid from Sigma-Aldrich (70 % nitric acid basis) into 45 ml of water. 0.1 M silver nitrate solution (Fig. 16) was prepared by adding 0.85 g of silver nitrate from Labochema (99.99 % purity) to 50 ml of distilled water. All these solutions are clear colorless liquids.

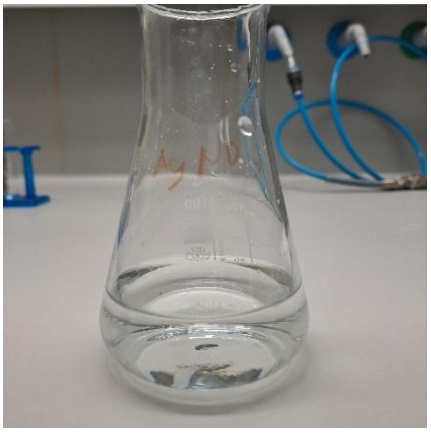


**Fig. 16.** 0.1 M silver nitrate solution

### 2.4.2. Starch test for iodide

This method is a reversed test of iodine test for starch. The basis of this test is the elemental iodine ($I_2$) positioning into the helical structure of amylose, which influences the light absorption of the complex to change and appear as deep blue color complex. To convert iodide ions in solution into elemental iodine, hydrogen peroxide can be used in acidic conditions to initiate oxidation-reduction reaction, resulting in the following reaction:

$$2I^-(aq) + H_2O_2 + 2H^+(aq) \rightarrow I_2 + H_2O \quad (18)$$

The steps for the procedure are:

- Adding around 1 ml of starch solution to the test tube.
- Adding 0.1 ml of dilute nitric acid.
- Adding around 0.2 ml of hydrogen peroxide.
- If the solution turns into dark blue color, the test for iodide ions is positive. If the solution is dyed in another color or stays the same, the test for iodide ions is negative.

For the test Valentis hydrogen peroxide (3 %) was used. Starch solution (Fig. 17) was prepared by mixing 1 g of potato starch with 20 ml of cold water and adding it into 80 ml boiling water. After stirring it for 3 – 5 minutes, the solution turns clear, taken off the hot plate and cooled to the room temperature.

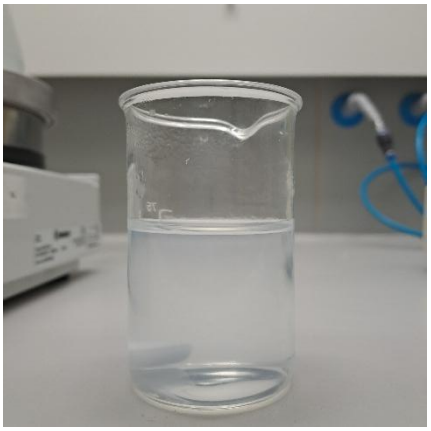


**Fig. 17.** Starch solution

## 3. Results and discussion

The results of this study highlight the performance of an ion exchange filtration system in removing iodide ions from artificial urine samples, both in the absence and presence of added potassium iodide. The filter's capacity to filtrate iodide ions specifically was also researched and discussed. Additionally, the cost-effectiveness of employing ion exchange filters for iodide removal was evaluated in the context of various potential applications, such as medical waste treatment.

### 3.1. Filtration of artificial urine and artificial urine with potassium iodide solution

The filter was prepared for filtrations by stacking two (Fig. 18. B) (cationic and anionic) and four (Fig. 18. C) (alternating between anionic and cationic) filter elements, to figure out whether the path that the solution travels through the filter is sufficient to remove total dissolved solids.

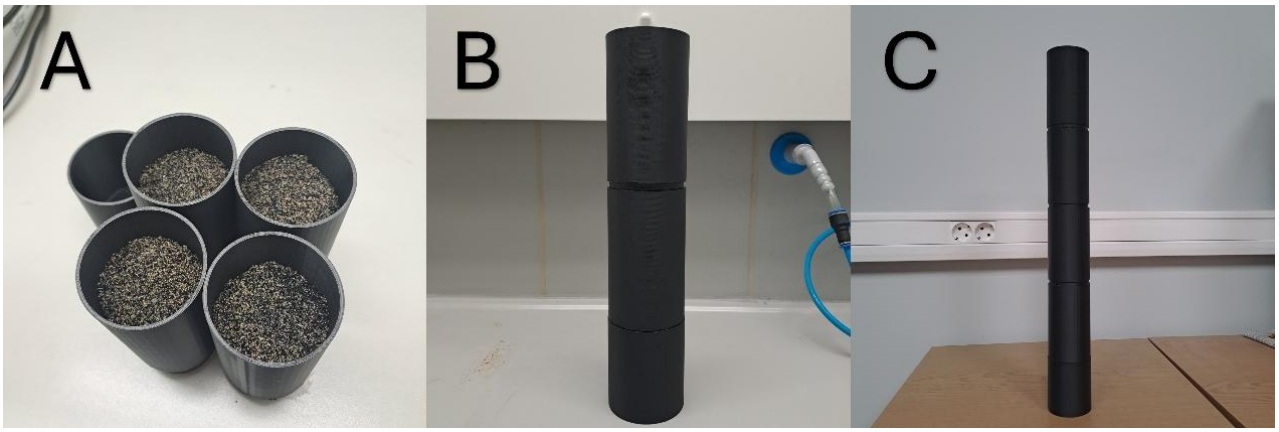



**Fig. 18.** A – completed filter elements; B – double stacked filter; C – four stacked filter elements

Ion exchange resins operate based on reversible chemical reactions in which mobile ions from the solution are exchanged with functional groups of the resin (-$SO_3H$ of cationic resin and -$NR_3OH$ of anionic resin). During the ion exchange process, these resins do not retain hydrogen ($H^+$) or hydroxide ($OH^-$) ions, because of selectivity for higher charge and atomic mass ions. Instead, when the resin exchanges a cation or an anion from the solution, it releases an equivalent charge of hydrogen ($H^+$) and hydroxide ($OH^-$) ions into the surrounding medium to maintain electrochemical balance.

Forexample, when a sodium ion ($Na^+$) in solution is captured by strongly acidic cation exchange resin in its hydrogen form (-$SO_3H$), a hydrogen ion is released into the solution:

$$R_1 - SO_3H + Na^+(aq) \rightarrow R_1 - SO_3Na + H^+(aq) \quad (19)$$

Here: $R_1$ – the polymeric part of strongly acidic ion exchange resin.

Similarly, when a chloride ion ($Cl^-$) is removed from solution by a strongly basic anion exchange resin in its hydroxide form (-$NR_3OH$), a hydroxide ion is released:

$$R_2 - N \cdot OH + Cl^-(aq) \rightarrow R_2 - N \cdot Cl + OH^-(aq) \quad (20)$$

Here: $R_2$ – the polymeric part of strongly basic ion exchange resin.

Hydrogen ($H^+$) and hydroxide ($OH^-$) released into the filtrate can recombine in a neutralization reaction to form water, as shown below:

$$H^+(aq) + OH^-(aq) \rightarrow H_2O \quad (21)$$

This neutralization process contributes to the formation of additional water molecules in the filtrate, effectively diluting the ionic content. Moreover, the lack of ions with acidic and basic properties in the solution results in reduced dissociation of water molecules, raising the pH value even higher. Therefore, the increased pH level in both solutions is a qualitative device to show that ion exchange was responsible for the capture of the ions in the solution.

Total dissolved solids (TDS) refer to the total concentration of inorganic and organic substances that are dissolved in a liquid, expressed in parts per million. TDS reflects the combined content of all ionic and molecular species present in a solution. A decrease in TDS after filtration indicates that the filtration system has effectively removed a portion of these dissolved substances from the unfiltered solution. In the context of ion exchange, a lower TDS means that cations and anions originally present in the solution have been replaced with hydrogen and hydroxide ions. This exchange reduces the overall ionic strength and conductivity of filtrate solution. Therefore, a reduction in TDS after treatment is generally associated with reduced concentrations of ionic substances.

Artificial urine filtrations through double element filter system demonstrate an increase in the pH of the resulting filtrate, rising from an initial pH value of 5.89 to around 6.5 (Table 8). It suggests a reduction in the concentration of acidic species in the resulting solution. In addition, a significant reduction in total dissolved solids was observed, the value of 29200 ppm decreasing down to the average of 1.67 ppm (Table 8), indicating that the majority of compounds initially present in the artificial urine solution were effectively removed. However, the presence of residual TDS in the filtrate indicates that the filtration process does not achieve complete deionization. This residual conductivity may be attributed to ions with lower binding affinities to the resin or insufficient surface area of ion exchange resin in contact with the solution.

The artificial urine filtrations through the quadruple filter also show an increase in pH value (from 5.89 to around 6.5, Table 8), indicating the reaction with ion exchange resin. Furthermore, the total dissolved solids measurement of the filtrate following quadruple element filter filtration reads as 0 ppm (Table 8), indicating a complete removal of dissolved solids. This absence of detectable TDS signifies that quadruple filter system is suitable for iodide removal from urine.

**Table 8.** Results of the artificial urine filtration

| | **Before filtration** | | **After filtration** | | |
|---|---|---|---|---|---|
| | **pH** | **TDS, ppm** | **pH** | **TDS, ppm** | **TDS average** |
| Double filter | 5.89 | 29200 | 6.45 | 2 | 1.67 |
| | | | 6.86 | 0 | |
| | | | 6.50 | 3 | |
| Quadruple filter | 5.89 | 29200 | 6.44 | 0 | 0 |
| | | | 6.51 | 0 | |
| | | | 6.54 | 0 | |

Filtration of artificial urine supplemented with potassium iodide through both double and quadruple filtration systems results in an increase in pH (from 5.93 to around 6.5 for both double and quadruple filter systems, Table 9). These pH shifts are indicative of the exchange of ionic species between the urine solution and the resin.

In terms of ionic content, total dissolved solids measurements confirm differences in filtration efficiency between the two systems. As in control experiments without potassium iodide, the quadruple filter consistently reduced TDS levels to undetectable levels (0 ppm, Table 9), demonstrating complete removal of dissolved ions, including iodide. In contrast, the double filtration setup resulted in an average TDS of 2 ppm (Table 9) in the filtrate, suggesting incomplete ion removal. The residual dissolved solids are likely to include iodide, which may not have been fully exchanged or adsorbed due to limitations in the resin capacity or less filtration stages.

**Table 9.** Results of the artificial urine and potassium iodide solutions filtration

| | Before filtration | | After filtration | | |
|---|---|---|---|---|---|
| | pH | TDS, ppm | pH | TDS, ppm | TDS average |
| Double filter | 5.93 | 29080 | 6.62 | 1 | 2 |
| | | | 6.39 | 2 | |
| | | | 6.52 | 3 | |
| Quadruple filter | 5.93 | 29080 | 6.51 | 0 | 0 |
| | | | 6.43 | 0 | |
| | | | 6.43 | 0 | |

### 3.2. Filter efficiency at removing iodide ions

To evaluate the silver nitrate test for halides, control tests were done with (from left to right accordingly (Fig. 19)) distilled water, potassium iodide solution and artificial urine solution.

Distilled water (Fig. 19. A, 1st test tube from the left) did not experience a change in color after adding the silver nitrate. Potassium iodide sample (Fig. 19. A, 2nd test tube from the left) formed yellow solutions, indicating the presence of halide ions. Artificial urine sample (Fig. 19. A, 3rd test tube from the left) acquired milky white color, also indicating the presence of halide ions, specifically chloride, which was expected due to the components of artificial urine, such as potassium chloride, sodium chloride, calcium chloride and ammonium chloride dissolved salts.

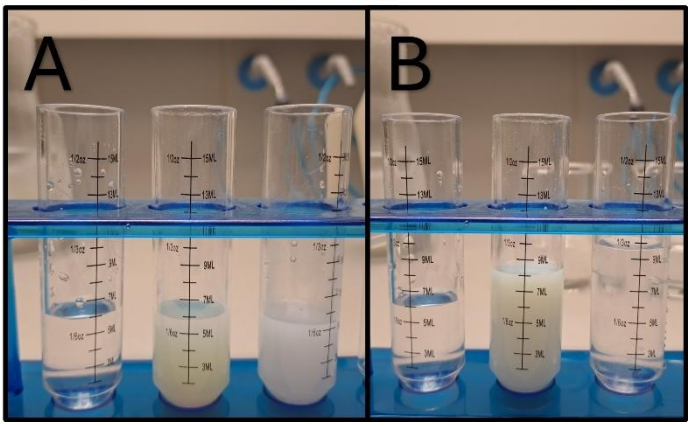



**Fig. 19.** Silver nitrate test for halides control samples. A – after silver nitrate was added; B – after dilute ammonium hydroxide was added

After adding the ammonium hydroxide solution, the change in distilled water test tube (Fig. 19. B, 1st test tube from the left) did not occur. Potassium iodide (Fig. 19. B, 2nd test tube from the left) yellow solution with silver nitrate slightly lightened due to dilution, but the yellow color remained. However, diluted ammonium hydroxide solution, added to the artificial urine sample (Fig. 19. B, 3rd test tube from the left), completely dissolved white compound in the solution, appearing as transparent colorless liquid. These results show that distilled water does not contain any halide ions, while potassium iodide solution has iodide ions and artificial urine contains chloride ions. Results of the control test were as expected, which shows that chemical agents and methodology used to identify halide ions are suitable for this project.

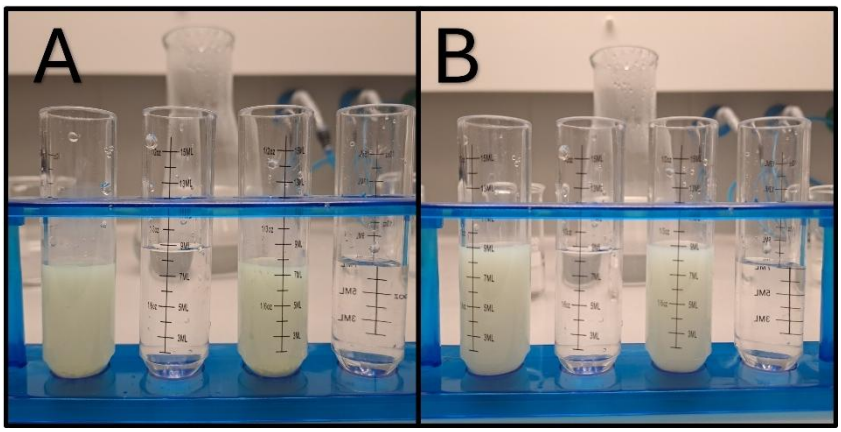



**Fig. 20.** Silver nitrate test for halide ions. A – after silver nitrate was added; B – after dilute ammonium hydroxide was added

Artificial urine with potassium iodide samples (Fig. 20. A, 1st and 3rd test tubes from the left) formed yellow solution with silver nitrate added, which confirms the presence of iodide ions in the samples. However, filtrate from the quadruple filter (Fig. 20. A, 2nd test tube from the left) and filtrate from the double filter (Fig. 20. A, 4th test tube from the right) did not form any crystals. Ammonium hydroxide, added to the artificial urine with potassium iodide samples (Fig. 20. B, 1st and 3rd test tubes from the left), did not visibly change the color of the samples although diluted – likely due to the fact that silver chloride crystals dissolved, and removed the white crystals from the solution.

Qualitative analysis for iodide with silver nitrate is viable to detect iodine concentrations that are above 10 μM (corresponds roughly to 1.27 μg per liter of iodide in solution), because of the low solubility of the silver iodide crystals. Therefore, this test shows that the concentration of iodide ions, if there are any in the filtrate, is below 1.27 μg/L.

To evaluate the starch test for iodide, control tests were done with (from left to right accordingly (Fig. 21)) distilled water, potassium iodide solution and artificial urine solution.

Distilled water (Fig. 21. A, 1st test tube from the left), potassium iodide solution (Fig. 21. A, 2nd test tube from the left) and artificial urine sample (Fig. 21. A, 3rd test tube from the left) appeared as clear solutions after adding the starch solution into it. Distilled water (Fig. 19. B, 1st test tube from the left) and artificial urine samples (Fig. 21. B, 3rd test tube from the left) remained as clear liquids after adding hydrogen peroxide. However, the solution with potassium iodide (Fig. 21. B, 2nd test tube from the left) turned dark blue color with the introduction of the oxidizing agent. These results show that distilled water and artificial urine did not have iodide ions present, while the potassium iodide

solution did. Results of the control test were as expected, which shows that chemical agents and methodology used to identify iodide ions are suitable for this project.

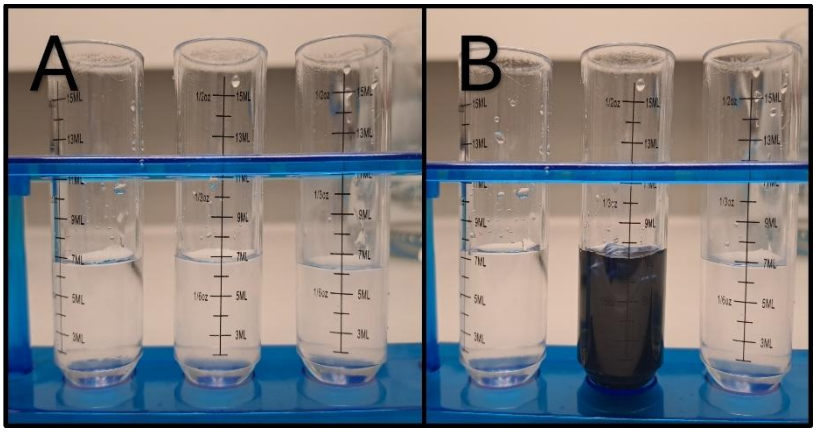



**Fig. 21.** Starch test for iodide control. A - samples with starch solution; B - samples after adding hydrogen peroxide

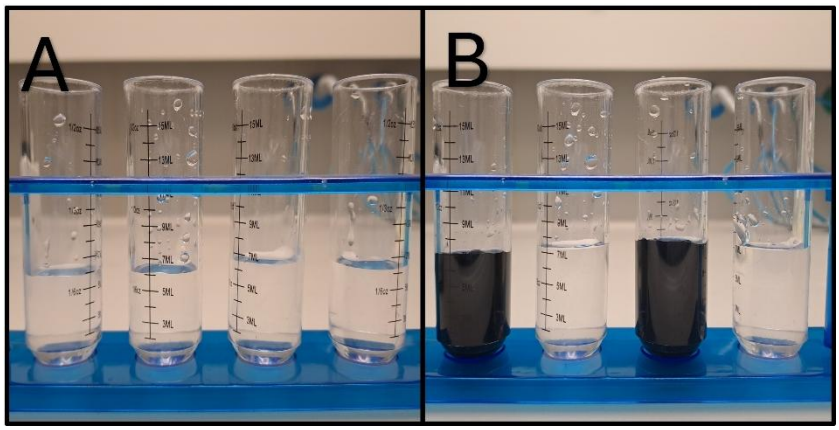



**Fig. 22.** Starch test for iodide. A - samples with starch solution; B - samples after adding hydrogen peroxide

Artificial urine with potassium iodide samples (Fig. 22. A, 1st and 3rd test tubes from the left), quadruple filter filtrate (Fig. 22. A, 2nd test tube from the left) and double filter filtrate (Fig. 22. A, 4th test tube from the left) remained as clear liquids with the starch solution. After adding the hydrogen peroxide to the test tubes with samples, only the artificial urine with potassium iodide samples (Fig. 22. B, 1st and 3rd test tubes from the left) turned dark blue, meanwhile both quadruple filter filtrate (Fig. 22. B, 2nd test tube from the left) and double filter filtrate (Fig. 22. B, 4th test tube from the left) remained clear, signifying an absence of iodide ions in the solution.

At room temperature, visually blue complex can be seen at concentrations as low as 20 μM (2.48 μg per liter), because of the intense color of the iodine-amylose complex. It shows that the iodide contents in the solution are below this threshold.

Overall, the results (Table 10) show that although the double filter failed to completely remove dissolved solids in the urine, the iodide ions were prioritized due to the selectivity for higher atomic number ions in anionic ion exchange resin and ions of artificial urine with potassium iodide solution reaction. It led to total removal of iodide ions both in double and quadruple filter systems, which shows that the efficiency of this filter is 99.99 % when it comes to removing iodide ions.

**Table 10.** Results of the qualitative iodide analysis

| | Outcome of tests for iodide ions (+ positive; - negative) | |
|---|---|---|
| | **Silver nitrite** | **Starch** |
| **Artificial urine with potassium iodide** | + | + |
| **Double filter filtrate** | - | - |
| **Quadruple filter filtrate** | - | - |

### 3.3. The benefit and environmental implications of using an ion exchange filter to remove radionuclides from urine

Evaluating the benefit of using an ion exchange filter to remove radionuclides from urine involves a combination of technical, economic and health risk assessment. The filter has shown that it is capable of completely removing iodide ions from the urine, which is a major radionuclide in medicine. Iodine-131 radionuclide is widely used therapeutically for treatment of thyroid cancer and hyperthyroidism diseases, iodine-123 in SPECT (single photon emission computed tomography) for imaging thyroid, parathyroid and endocrine tumors, iodine-125 may also be used for SPECT or for localized treatment such as brachytherapy, as well as iodine-124 in PET (positron emission tomography), although it is more frequently used in research. All the pharmaceuticals, containing radioactive iodine isotopes, are metabolized into iodide ions and discharged from the body in its aqueous form via urine.

Iodine-125 and iodine-123 administered doses are usually up to 500 MBq and 11.1 GBq for iodine-131 [38], which correspond to micrograms or less of the radioactive isotope administered to a patient. It shows that even a small amount of radioactive substance may be enough for the treatment; however, if it is not effectively managed, it may be introduced into drinking waters, groundwater and bioaccumulate in plants and animals.

**Table 11.** Specifications of iodine isotopes [38]

| Iodine isotope mass | Half-life | 10 half-lives | Emissions | Specific activity, Bq/μg |
|---|---|---|---|---|
| 123 | 13.2 h | 5.5 d | Gamma rays | $7.1 \cdot 10^7$ |
| 124 | 4.18 d | 41.8 d | Beta positrons, gamma rays | $9.3 \cdot 10^9$ |
| 125 | 60.1 d | 1.69 y | Gamma rays, Auger electrons | $6.4 \cdot 10^8$ |
| 131 | 8.02 d | 80.2 d | Beta electrons, gamma rays | $3.7 \cdot 10^7$ |

Generally, urine containing radionuclides would be disposed of by these methods:

- Urine stored in delay tanks, which allows radioactivity to naturally decay (usually for around 10 half-lives, which reduces radioactivity to 0.01 % of its primary radioactivity). After the storage, urine is discharged into the sewer system as non-radioactive waste.
- Urine diluted by additional water or other non-radioactive liquid waste and discharged directly into the sewer system.

Decay in storage method requires extensive space to hold the radioactive waste. Furthermore, the patients are advised to drink more water after the procedures with radionuclides to assist their removal from the body, which may also increase the volume of the waste. However, urine with radionuclides would be held for around ten half-lives despite its content concentration. Moreover, if the delay tanks are installed as part of the sewage system, placing a single delay tank to collect radioactive urine

raises a problem due to widely varying isotope half-life – some can be disposed of after just a few days, while others might be kept for months. Additionally, if multiple delay tanks are an option to collect the radioactive waste, it introduces more expenses both for initial installation and management.

Dilution and discharge methods are more appealing, especially in big hospitals with a significant amount of general liquid waste generated. However, if the amount of waste is relatively small, the additional mass would be compensated by introducing water to effectively dilute the concentration of radionuclides. Moreover, despite the low concentration of radionuclides, the emissions still might have an impact on the environment, especially microorganisms and plants that tend to accumulate toxicants. Under most regulatory bodies, it is allowed to discharge radioactive liquid waste into the sewer system if the concentration is below the established limits and the facilities operating this type of liquid waste have a valid license for these operations. Therefore, although the method is cost-effective and simple, it introduces environmental risks as it may impact the flora and fauna and lower the water reserves.

In comparison, the 3D-printing of the filter is easily achievable as it is a low-cost solution and easily modified to the occurring settings. First, the filter, especially if it is divided into elements, is easy to store and takes up less space compared to the primary urine discharge. Also, the system with separate smaller, detachable parts allow to follow the storage time and activity contents more precisely.

Moreover, the ion exchange resin can be recuperated as it reaches the capacity for captured ions. Running the 5 % sulfuric acid through the cationic resin beads and 5 % sodium hydroxide through the anionic ion exchange beads restored its ability to capture the ions in this project. Furthermore, this required around 200 ml of solution for each corresponding element. The resulting solution contained artificial urine ions and additional potassium and iodide ions, since the isotope used in this project was non-radioactive and did not decay into other products. Also, since acidic and alkaline recuperation solutions are used, mixing them together can be used to neutralize the pH level, which makes it safer for the environment. On top of that, the iodide ions were bound to anionic ion exchange resin, it also proposes that only anionic filter element should be stored, which also minimizes the space it occupies.

Overall, using an ion exchange filter to avoid contamination and reduce the cost of the process is a plausible idea, especially for smaller hospitals or research centers. However, careful monitoring of resin saturation and proper afterward waste processing protocols are essential to ensure consistent performance and compliance with radiological safety standards.

## Conclusions

1. The 3D-printed filter was designed for practical assembly and efficient filtration. Anionic filter elements were made with strongly basic anion exchange resin with quaternary ammonium functional groups and cationic elements – of strongly acidic cation exchange resin with sulfite functional group. The stackable construction of elements allows for the filter to increase or decrease ion exchange resin volume (200 ml per element) as needed.
2. The filtration experiments demonstrated that increasing the number of filter elements improves the removal efficiency of total dissolved solids (TDS) from artificial urine solutions. While the double-stacked filter (one cationic and one anionic element) was partially effective (reducing TDS significantly from around 29000 ppm to 1.67 – 2 ppm), the quadruple-stacked filter (alternating layers of anionic and cationic elements) proved to be fully effective, yielding filtrates with zero TDS.
3. The qualitative analyses using silver nitrate and starch test confirmed the presence of iodide ions in artificial urine mixed with potassium iodide and the absence of iodide ions in the filtrate of urine from both double-stacked and quadruple-stacked filter. Starch test for iodine shows that the concentration of iodide in the filtrate is lower than 20 μM and silver nitrate test for halogens demonstrated that the concentration of iodide ions in the filtrate is lower than 10 μM. These findings indicate that while the double filter did not eliminate all total dissolved solids, it was still fully effective in removing of iodide ions to the untraceable amount.
4. Ion exchange filtration proved to be an effective, low-cost and space-efficient method for removing iodide ions from urine. Unlike traditional disposal methods such as delay tanks or dilution, this approach minimizes environmental risks and infrastructure demands. The filters can be regenerated and tailored for specific needs, making them especially suitable for small hospitals and research centers.

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