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Renaud Gueroult, David T. Hobbs,
and Nathaniel J. Fisch

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Economic feasibility of plasma filtering techniques for nuclear waste remediation

Renaud Gueroult¹, David T. Hobbs², Nathaniel J. Fisch¹

¹ *Princeton Plasma Physics Laboratory, Princeton, New Jersey, 08540, USA*

² *Savannah River National Laboratory, Aiken, South Carolina, 29808, USA*

Abstract

The economical viability of nuclear waste cleanup efforts could, in some cases, be put at risk due to the difficulties faced in handling unknown and complex feed stocks. Plasma filtering, which operates on dissociated elements, offers advantages over chemical techniques for the processing of such wastes. In this context, the economic feasibility of plasma mass filtering for nuclear waste pretreatment before ultimate disposal is analyzed. Results indicate similar costs for chemical and plasma solid-waste pretreatment per unit mass of waste, but suggest significant savings potential as a result of a superior waste mass minimization. This performance improvement is observed over a large range of waste chemical compositions, representative of legacy waste's heterogeneity. Although smaller, additional savings arise from the absence of a secondary liquid waste stream, as typically produced by chemical techniques.

Keywords: Nuclear waste, Separation, Plasma mass filter, Economic feasibility

1. Introduction

Beginning with the Manhattan project, and throughout the cold-war, nuclear weapons development generated large quantities of radioactive waste as a byproduct of uranium and plutonium production. Most of this waste originated from production reactors at Hanford and Savannah river sites, and from the enrichment plant at Oak Ridge [1]. Before the 1970s, the composition of this waste was poorly documented, and significant quantities of liquid waste were released directly to the environment [2]. Only the most highly radioactive fraction of the waste was piped to underground storage tanks.

At Savannah River, 36 million gallons of high level waste are stored in 45 underground tanks [3]. Processing and immobilization of high level waste in borosilicate glass started in 1996. A salt waste processing facility is currently under construction, with first operations scheduled in 2018. Completion of clean-up activities is scheduled for 2033 [4].

At Hanford, 54 million gallons of waste were stored in 177 underground tanks [2, 5]. The oldest, single shell, tanks were built between 1943 and 1964, with designed service lives of 10 to 20 years. Out of these 177 tanks, 67 have or are suspected to have leaked up to 1 million gallon into the environment [2], with first leaks confirmed in 1959. Double shell carbon-steel tanks were built starting in 1968 to provide better confinement, and waste has progressively been pumped from single shell to double shell tanks. Yet, 2.8 million gallons were still stored in single shell tanks in 2012 [6], and leaks have been discovered between shells of double shell tanks [7]. Construction of a facility to immobilize the high level waste using similar approaches to those used at Savannah River began in 2002. However, due to various unresolved technical problems and work stoppages [8], the estimated cost to construct this treatment and immobilization facility has tripled from 4.3 to 13.4 billion dollars, and its scheduled completion date slipped by nearly a decade to 2019 [9]. Completion of clean-up activities is not expected before 2050 [10]. Across the different waste storage sites, clean up efforts are projected to cost more than 280 billion dollars [11].

In its simplest form, tank clean-up operations boil down to separating small volumes of high activity waste from much larger volumes of low activity waste. The separated high activity waste is then immobilized as glass for ultimate disposal in an underground repository. The low activity waste is immobilized
35 in a less durable wasteform for onsite disposal. Non radioactive elements inside the high activity waste stream is highly unfavorable for various reasons. First, from an economical standpoint, vitrifying non radioactive material incurs non negligible additional cost since the production cost of glass canisters is a significant fraction of the total cost (of the order of a million dollar per
40 canister) [12, 13, 14, 15, 16]. In addition, a larger number of glass canisters requires a greater number of vitrification facilities, increasing the capital cost. Second, the glass formulation has specific weight loading tolerances for different elements [16, 17]. For example, chromium, ruthenium, rhodium and palladium present in the glass can precipitate and eventually short circuit the glass melter
45 electrodes. Furthermore, chromium, phosphorus oxide and sodium sulfate dissolve poorly in borosilicate glass, forming on occasions refractory crystalline phases that could compromise the durability of borosilicate glass wasteform. For these reasons, an efficient separation of high level radioactive elements from the low level waste is highly desirable [12, 14].

50 In this paper, we analyze the economic feasibility of plasma mass filtering for nuclear waste clean-up. In Section 2, we examine the main challenges faced by waste tank clean-up operations using Hanford's waste as a baseline, and point out the intrinsic limitations of chemical techniques in this context. In Section 3, we review the essential characteristics of plasma mass filtering techniques. In
55 Section 4, we compare the projected costs of plasma techniques and chemical techniques for the particular application of sludge pretreatment. In Section 5, we summarize the main results.

2. Tanks clean-up challenges

Although conceptually simple, separating non-radioactive material from radioactive elements can prove to be extremely challenging depending on the input stream composition. In case of legacy waste analyzed here, this is made particularly difficult by the heterogeneity of the input stream, both in terms of physical and chemical forms. Waste stored in tanks is in one of three forms [14, 18]. Due to the high pH, the bulk of the metals precipitate as insoluble metal oxides/hydroxides that gravity settles to form a thick layer referred to as *sludge*. Typical metals include Al, Bi, Cr, Fe, Mn, Si and U. The liquid fraction of the waste, referred to as *supernate*, contains water-soluble components, principally the sodium salts of oxyanions including hydroxyde, nitrate, nitrite, aluminate, sulfate and carbonate. Historically, the supernate has been evaporated to minimize the volume. Cooling of the hot, concentrated supernate produced crystalline salts, which accumulated in a layer referred to as *saltcake*.

Typical waste pretreatment operations can be summarized as follows [19]. The sludge is recovered and goes through a series of caustic leaching, oxydative leaching and washing steps to remove non-radioactive elements, in particular Na, Al and Cr [20]. Saltcake is dissolved in water and combined with supernates and liquids from sludge leaching and washing. Undissolved solids are removed and the clarified liquids are treated to remove certain radionuclides such as ^{137}Cs , ^{99}Tc and ^{90}Sr . The leached and washed sludge, together with the elements removed from the dissolved saltcake and supernate, are combined and vitrified. After the removal of the radionuclides, the decontaminated supernate is immobilized in either cementitious (Savannah River Site) or glass (Hanford site) wasteform.

In the case of Hanford, designing separation processes suitable to tackle this problem is complicated by the large waste compositional variations between tanks [21, 17], as illustrated in Fig. 1. The high level waste at Hanford can be divided into six sub-groups based on their chemistry and glass formulation limiting factors [22]: high alumina wastes, high iron wastes, high iron, chromium,

nickel and manganese wastes, high chromium and sulfur wastes, high phosphorus and calcium wastes, and high alkali wastes. Removal of non radioactive
 90 elements by means of chemical techniques is then extremely challenging since the elements to be removed vary widely from batch to batch, and are usually a combination of elements with various chemical properties. As a result, chemical separation (*e. g.* aluminium in the chemical form of boehmite [23], and chromium present as Cr(III) compounds [24, 25]) has proven to be particularly
 95 time consuming and intensive in this context [20].

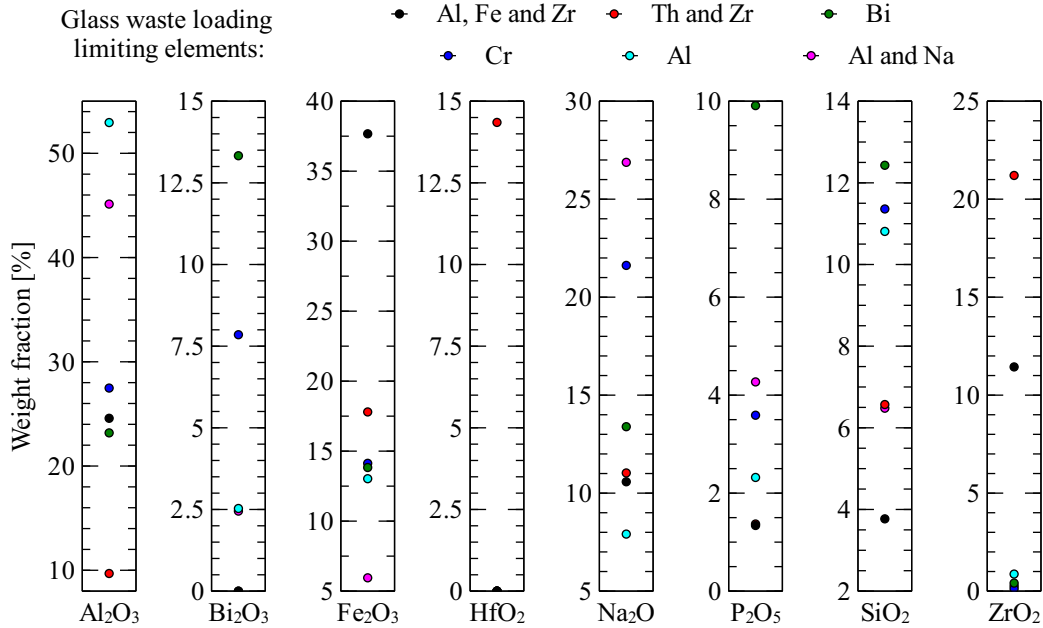


Figure 1: Oxide waste mass breakdown across the six different sub-groups of Hanford wastes presenting challenges in terms of canister waste loading and glass formulation, from [17, Table 2.2]. Only oxides with mass fraction over 10% are plotted here.

Due to these difficulties, recent years have seen new research effort directed towards glass formulation allowing higher aluminum and chromium fractions as well as higher waste loadings [26, 27]. Although this approach could in principle

alleviate the increase of canisters resulting from larger volumes, it would, how-
ever, come at the expense of greater constraints in terms of glass formulation.
100 The capability of these advanced glass formulation to respond to the typically
encountered waste composition variations is still to be demonstrated. In ad-
dition, higher aluminum content will most likely have detrimental side effects,
notably on achievable processing rates [28], and may call for different melter
105 technology solutions [29].

Recommendations from a Government Accountability Office review [30, p.
27] identified another path as an opportunity to reduce costs, namely the devel-
opment and testing of additional pretreatment technologies to ensure reliability
and efficiency of the pretreatment operations. In this respect, non-chemical sep-
110 aration techniques are attractive since they are in principle indifferent to waste
heterogeneity. One example of such non chemical technique is plasma mass
filtering.

3. Plasma mass filtering

The potential of plasma medium to separate elements based on their mass
115 has long been recognized [31]. An example of such a device is the plasma cen-
trifuge [32], which operates in a similar fashion to conventional gaseous or liquid
centrifuges, but offers higher separation factors due to its ability to operate at
much larger rotation speeds. Higher rotation speeds are in this case made pos-
sible by the absence of moving parts, with rotation produced in this device by
120 means of the combined effects of electric and magnetic fields [31]. However, the
main thrust for this research effort was originally isotope separation [33, 34].
As a result, most of the work was directed towards low mass differences and,
consequently, low throughput. Only recently has plasma mass filtering been
considered for nuclear waste remediation [35] and for nuclear spent fuel repro-
125 cessing [36]. The use of plasmas for these new applications was made possible by
the development of various new plasma filter concepts [37, 38, 39, 40] which offer
high-throughput processing granted sufficiently large mass differences between

species to be separated [41].

In these devices, material can be fed in the machine in different forms. Possible candidates include powder injection or laser evaporation. Although the
130 choice of a particular feeding technique has not been made yet, and will most likely depend on the specifics of the targeted process, general constraints can be obtained for this particular process. For example, in the case of powder injection, micron-size particles are likely to be required for the envisioned plasma
135 operating conditions [42]. Similarly, the desired throughput will dictate the required laser power.

Once ionized, charged particles respond to both electromagnetic and centrifugal fields. In plasma filters devices, these fields are generally designed such that there exists a mass threshold m_c for particle confinement. Elements heavier
140 than the mass threshold m_c are then directed one way, while elements lighter than this mass threshold are directed in another way. Fig. 2 illustrates the differential confinement properties of *light* and *heavy* elements for the three main filter concepts. It is worth noting here that variations on these concepts exist, such as the use of RF electric fields in place of DC electric fields controlling the
145 plasma rotation. This could in principle allow isolating a particular mass from the bulk [37], rather than discriminating elements based on a threshold mass.

The two separated streams can then be recovered individually. Depending on the selected filter concept, charged particles could be either deposited and neutralized on a surface, or neutralized in volume by locally tuning the plasma
150 parameters.

4. Economic feasibility of plasma filtering techniques for sludge pretreatment

Although different insertion points can be envisioned, the ability to separate a *light population* from a *heavy population* makes plasma filters attractive for
155 sludge pretreatment. For example, for the typical sludge composition introduced in Sec 2 and plotted in Fig. 3, one can imagine tuning the plasma filter in such

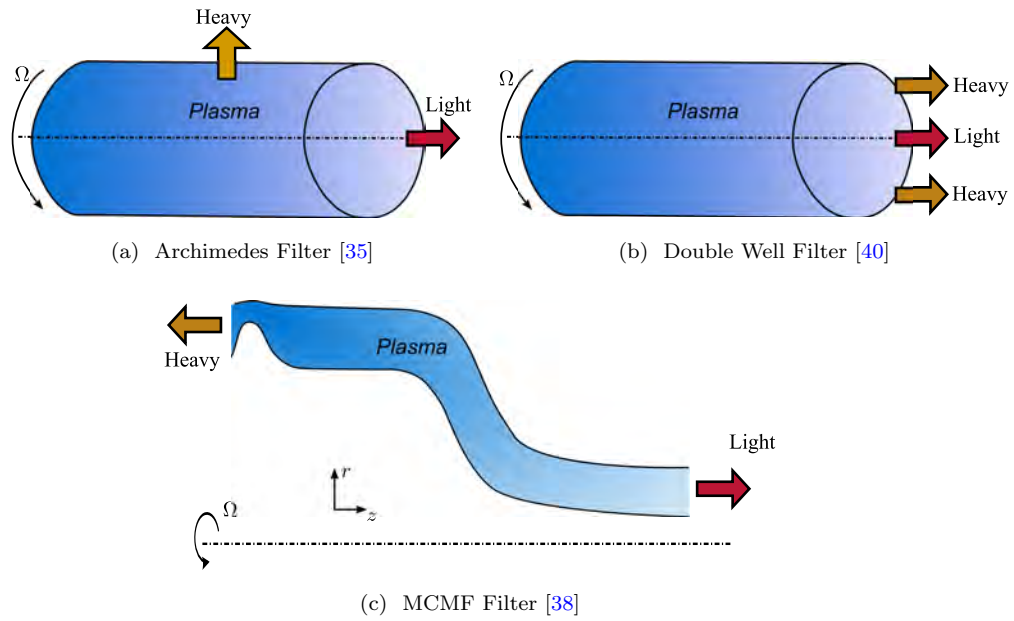


Figure 2: Different high throughput plasma filter concepts: axial/radial separation in the Archimedes filter (a), radial layering in the Double Well filter (b) and axial/axial separation in the MCMF filter (c). These three filter concepts feature axisymmetric rotating plasmas.

a way that Al, Cr, Fe, O, Na and Si are below the cutoff mass, while Sr, Tc, Cs, Bi, Th and U are heavier than the cutoff mass. As indicated in Fig. 4b, the low volume, heavy stream, could then be processed as high activity waste and vitrified together with radionuclides recovered from the liquid waste, while the larger volume, light stream will be processed as low activity waste. The total waste mass below this cutoff mass, as summarized in Tab. 1, gives an upper limit for the plasma treatment efficiency of 60% to 95%.

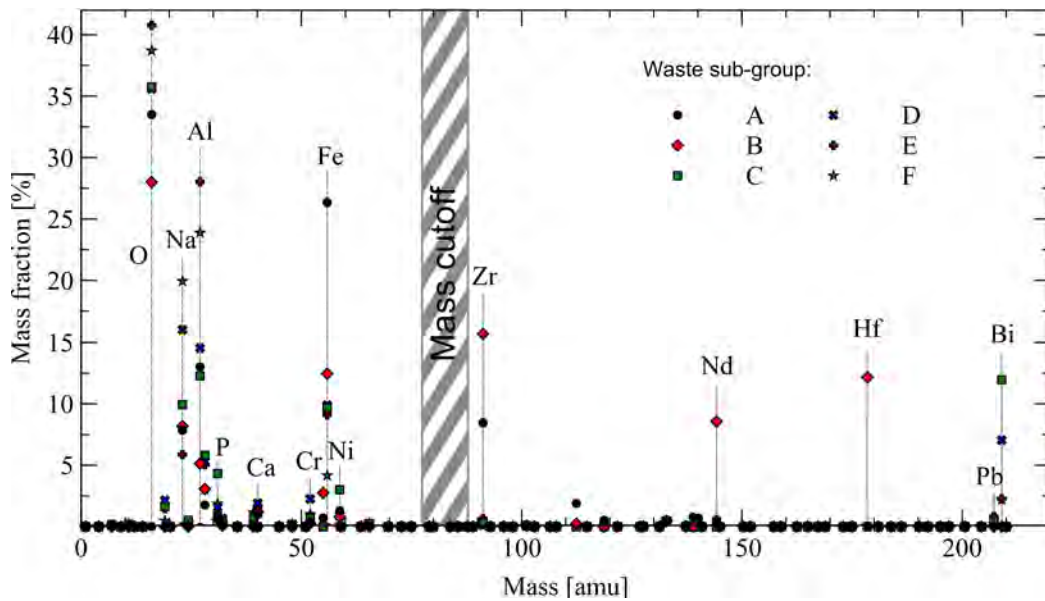


Figure 3: Waste weight composition by elements, from [17, Table 2.2]. The plasma mass filter cutoff mass is indicated by the shaded box. The different waste compositions correspond to the different sub-groups summarized in Tab. 1.

More generally, since the common pattern is to separate heavy radioactive elements from lighter non-radioactive elements, the plasma filters could be tuned to respond best to a given waste composition. It is worth noting here that, as opposed to chemical techniques, such a tuning could in principle be done on the fly as it would essentially consist in setting the rotation speed accordingly. Beyond the rotation speed control achieved through the transverse electric field,

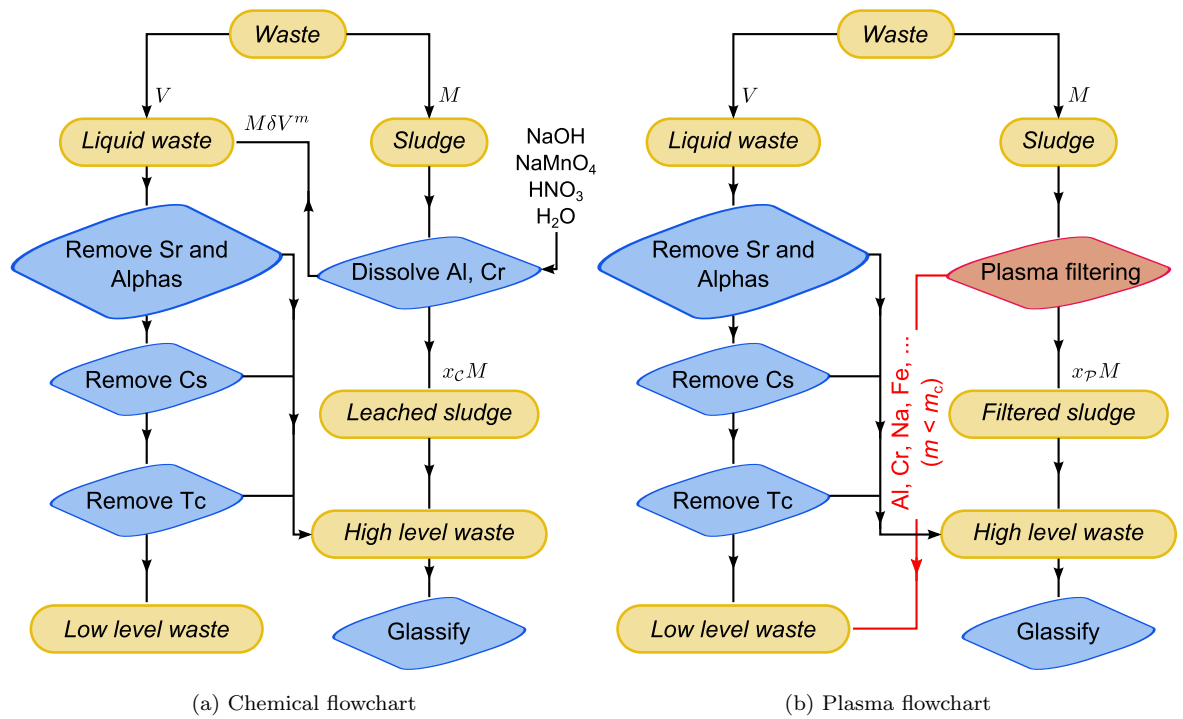


Figure 4: Chemical (a) and plasma (b) flowcharts for waste processing. Plasma approach could in principle suppress the additional liquid waste produced by sludge washing and leaching, as well as minimize the final volume of pretreated sludge to be vitrified.

Waste sub-group	Limiting glass factor	Mass fraction under 90 amu [%]
A	Al, Fe and Zr	86.5
B	Th and Zr	62.5
C	Bi	86.7
D	Cr	91.8
E	Al	95.5
F	Al and Na	96.8

Table 1: Waste subgroups by limiting factor for vitrification, from [17], and computed mass fraction under the mass filter threshold.

170 other plasma parameters, such as electron and ion temperatures and background
neutral pressure, can be modified to optimize the separation efficiency [39].

4.1. Cost of chemical sludge disposal

Looking at the chemical processing flowchart depicted in Fig. 4a, the cost
of sludge disposal per unit mass \mathcal{C}_c^m can be broken down to the sum of the
175 individual cost of three subprocesses,

$$\mathcal{C}_c^m = \mathcal{C}_s^m + x_C \mathcal{C}_v^m + \delta V^m \mathcal{C}_l^V, \quad (1)$$

where \mathcal{C}_s^m is the cost of sludge washing and leaching per unit mass, \mathcal{C}_v^m is
the vitrifying cost per unit mass of waste load, x_C is the mass of solid waste
after washing and leaching one kg of sludge and $\delta V \mathcal{C}_l^V$ is the cost of additional
liquid waste processing (\mathcal{C}_l^V is the liquid waste processing cost per unit volume,
180 and δV^m is the volume of liquid waste produced by washing and leaching of
a kilogram of sludge). To be exhaustive, one would also have to account for
the cost associated with the disposal of solid waste generated during the sludge
pretreatment, as well as during the removal of radionuclides in the additional
liquid waste. This additional solid waste will be combined with the sludge
185 for vitrification, and will consequently result in an incremental increase of x_C .

	Subprocess	Cost estimate
C_s^m	Sludge washing and leaching	\$45 per kg of sludge
C_v^m	Vitrification ¹	\$1200 – 2300 per kg of waste load
δV^m	Additional liquid waste produced	0.5 L per kg of sludge
C_l^V	Liquid waste treatment	\$50 per liter of liquid waste

Table 2: Breakdown of chemical processing costs.

However, since only limited amount are expected to derive directly from sludge washing and leaching, additional solid wastes are neglected in this study.

Pretreatment costs estimates can be inferred from previous studies, and are summarized in Tab. 2. Corrected for inflation, pretreatment costs are respectively \$24 and \$43 per kg of liquid and sludge waste [43]. Assuming that liquid waste is essentially made of sodium hydroxide ($\rho_{\text{NaOH}} = 2.13 \text{ g.cm}^{-3}$), this gives $C_l^V \sim \$50$ per liter of liquid waste and $C_s^m \sim \$45$ per kg of sludge. Moving to vitrification, the cost per unit mass can be estimated from the published incremental cost of producing one more, or one fewer, canister [44]. Corrected again for today’s dollar, this incremental cost is estimated between \$0.8 M and \$1.5 M, with about half of it resulting from storage. Using a standard 2 ft \times 10 ft glass canister [14, Appendix E], densities of 2.6 for the glass [15, p. 9] and 5 for the waste ($\rho_{\text{Al}} \sim 2.7 \text{ g.cm}^{-3}$, $\rho_{\text{Fe}} \sim 7.9 \text{ g.cm}^{-3}$), and a 25% waste weight loading gives a vitrification cost per unit mass C_v^m between \$1200 and \$2300. Finally, sludge washing is responsible for an additional 1.3 liter of salt waste per liter of sludge processed [45, p. 37]. Using here aluminum density as a baseline for sludge, this gives $\delta V^m \sim 0.5$ liter per kg of sludge.

Overall, the total chemical sludge disposal cost stems essentially from vitrification and glass canister storage costs, while pretreatment costs are negligible in comparison.

¹This includes the canisters storage cost.

4.2. Cost of plasma assisted sludge disposal

Looking back at the chemical and plasma flowcharts in Fig 4, one sees that a plasma approach would eliminate the secondary liquid waste stream ($\delta V^m \sim 0$). The cost of plasma sludge disposal $\mathcal{C}_{\mathcal{P}}^m$ per unit mass of sludge is thus the sum
 210 of only two subprocesses,

$$\mathcal{C}_{\mathcal{P}}^m = x_{\mathcal{P}}\mathcal{C}_v^m + \mathcal{C}_{pf}^m. \quad (2)$$

The first one is the vitrifying cost, where \mathcal{C}_v^m is the vitrification cost per unit mass of waste load, and $x_{\mathcal{C}}$ is the mass of solid waste after plasma filtering. \mathcal{C}_v^m is identical to the one obtained for chemical separation and listed in Tab. 2, and only $x_{\mathcal{P}}$ differs. The second one, \mathcal{C}_{pf}^m , is the cost of plasma filtering per
 215 unit mass. This plasma filtering cost, can itself be broken down into different processes.

Evaporation. First, the waste needs to be fed into the machine. As discussed in Sec. 3, one option consists in laser evaporation. Assuming that the latent heat of vaporization \mathcal{L}_v is dominant over both the latent heat of fusion and the
 220 enthalpy change due to the temperature increase, an firth order estimate for the evaporation cost \mathcal{C}_e^m is \mathcal{L}_v/χ , where χ is the laser absorptivity. For an aluminum rich waste, $\chi \sim 0.2$ [46] and $\mathcal{L}_v^{\text{Al}} \sim 10$ MJ/kg, $\mathcal{L}_v^{\text{Al}_2\text{O}_3} \sim 4.8$ MJ/kg [47, p. 115], gives $\mathcal{C}_e^m \sim 20 - 50$ MJ/kg.

Plasma production. Once the waste turned into a gas, the next step consists
 225 in ionizing this gas. Using once more aluminum as a baseline, this requires 21.4 MJ/kg. This figure is again an ideal value. In practice, one has to account for all energy dissipation channels. First, part of the electron energy will be dissipated through excitation of neutrals and ions. A measure of the deviation from the ideal case is the efficiency η , defined as the ratio of the energy required
 230 for one electron-ion pair creation over the atom ionization energy ε_i . For helicon discharges envisioned for this application, η is about 0.4 for an electron temperature $T_e \sim 4$ eV in pure Argon ($\varepsilon_i = 15.75$ eV) [48, p. 81]. However, the efficiency η is expected to be reduced for the more complex compositions

typically envisioned here. It is worth noting here that since excitation losses
235 scale with the square of the plasma density, η can in principle be maintained to
acceptable levels by limiting the plasma density.

In addition to electron losses, other energy dissipation channels might have
to be considered depending on the plasma parameters. These includes the en-
ergy transfer to ions in the form of rotational kinetic energy and temperature.
240 Quantitatively, a mass of 1 kg rotating at 3 km.s⁻¹ has a kinetic energy of
4.5 MJ, and increasing the temperature of 1 kg of aluminum gas by 1 eV re-
quires 3.3 MJ. These losses are therefore small in most cases compared to elec-
tron losses. Assuming a highly degraded $\eta \sim 0.02$, the cost of plasma formation
and maintenance is of the order of 1 GJ/kg.

245 *Total cost.* Summing up the costs of these two sub-processes, and assuming a
low laser electric efficiency of 0.1, gives a plasma filtering energy cost of the order
of 1.5 GJ/kg. Using a typical electricity cost of \$0.1 per kilowatt-hour (kWh),
i. e. 36 MJ/\$, this puts the cost of plasma pretreatment $C_{pf}^m \sim \$40$ per kg of
sludge. Interestingly, this cost is on par with C_s^m , the cost of sludge chemical
250 washing and leaching.

The absence of secondary liquid waste stream would represent a saving of
about \$25 per kg of sludge. However, the largest opportunity to reduce costs
lies in waste mass minimization. As a matter of fact, because of the significant
costs associated with vitrification, a higher mass minimization $x_P < x_C$ could
255 offer large savings.

4.3. Waste mass minimization: possible savings

Studies indicate high variations of chemical washing/leaching efficiency across
the various tank farms [49], with aluminum removal values ranging from 20% to
99%. Similar variations can be found for other elements of critical importance
260 such as chromium and iron [50], and studies targeted to one particular waste
type showed comparable results [51]. As a consequence, non-radioactive ele-
ments are still largely responsible of the large mass of the processed sludge.

Improvements in the chemical processes are challenging since further washing/leaching puts additional constraints on the liquid waste stream, and because
265 a given process might increase the performances with respect to one specific element while degrading the performances with respect to another. On the other hand, a plasma mass filter could in principle offer on average higher efficiencies since all elements lighter than the mass threshold would be removed.

Using the the typical waste subgroups compositions depicted in Fig. 3, one
270 can produce waste mass minimization estimates for various separation schemes. Three different separation processes are studied here. The first one consists simply in the removal of 80% of the aluminum oxide contained in the sludge. The second one corresponds to an optimized chemical separation process [50], with the removal of 86% of Al_2O_3 , 99% of Cr_2O_3 , 1.9% of Fe_2O_3 and 62% of
275 SiO_2 . The third one corresponds to a plasma filtering with a threshold mass $m_c = 90$ amu, and a given uniform separation efficiency for all oxides for which the non oxygen element is lighter than m_c . In all three cases, sodium is assumed to be totally removed during the process. The corresponding results are plotted in Fig. 5. As expected, there are large variations depending on waste
280 composition, and that both for plasma and chemical techniques. Performance is the worst for the high thorium and zirconium wastes (sub-group B), which makes sense since these elements are not recovered by either of these processes. Comparing chemical and plasma techniques, it appears that a moderate 60% plasma filter separation efficiency leads to comparable or better results than the
285 advanced chemical process. The advantage of plasma filtering decreases for high aluminum content wastes (sub-groups E and F), since in this case the higher aluminum separation efficiency offered by chemical techniques addresses better the problem. However, a 70% plasma filtering separation efficiency is sufficient to make the plasma approach more efficient than the chemical process modeled
290 across all waste sub-groups.

Waste mass minimization can be directly translated into cost savings using the typical vitrification cost per kg of processed sludge listed in Tab. 2. The cost difference per kg of sludge due to waste mass minimization is $(x_C - x_P)\mathcal{C}_v^m$.

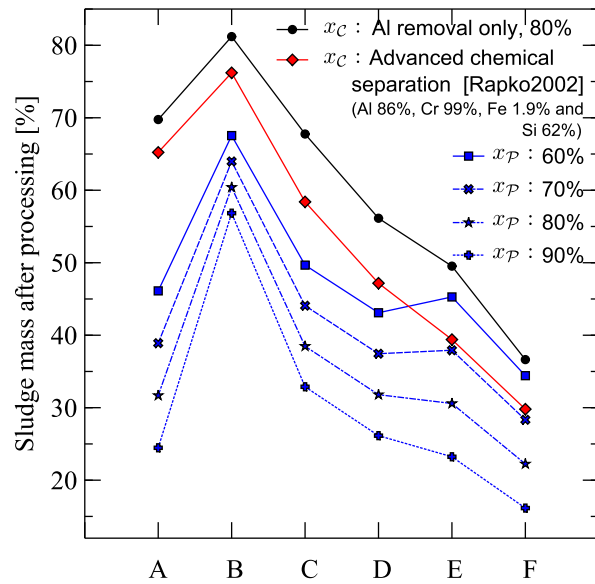


Figure 5: Sludge mass decrease per waste sub-group (A to F, see Tab. 1) as obtained by different pretreatment options and different plasma filtering efficiencies. x_P and x_C denote the mass fraction of sludge after respectively plasma and chemical processing.

The corresponding data is plotted in Fig. 6, and suggests that savings of \$80
295 and higher per kg of sludge are possible for two third of the waste types and
a 60% plasma mass filtering efficiency. Increasing the efficiency to 80% yields
savings of \$150 to \$650 per kg depending on the waste type. These values are
substantial, representing 7.5 to 32.5% of the total sludge processing cost.

It is worth noting here that a filtering efficiency of 70% appears to be well
300 within reach of the proposed plasma filter concepts [36]. However, if required,
higher values could be achieved by staging the filter, so that particles go through
multiple separation steps. Three passes, each at an efficiency of 70%, would
offer a 97% separation. This could ideally be achieved at negligible cost by
maintaining elements ionized throughout the full cycle. Even for the worst case
305 scenario where particles have to be re-ionized, the cost per kg would not exceed
two or three C_{pf}^m . This is about \$120 per kg, which would be lower than the
savings achieved as a result of an improved waste mass minimization.

To summarize, the cost of plasma filtering appears to be on par with chemical
washing and leaching for sludge pretreatment. However, thanks to no additional
310 liquid waste produced, and a greater ease to remove non-radioactive elements
critical to the vitrification process, plasma filtering has the potential to offer
significant savings as compared to chemical techniques when considering the
entire sludge pretreatment and vitrification process. This is especially true for
highly heterogeneous waste, such as legacy waste.

315 5. Summary

The generally accepted solution for ultimate disposal of nuclear waste con-
sists of immobilization of the radioactive components in glass for permanent
storage in geological repository. However, the high cost of vitrification per unit
mass of waste load generally prohibits vitrifying the waste as is. This is espe-
320 cially true for legacy waste, that is to say nuclear waste produced as a byproduct
of nuclear weapons development during the cold war era, which is typically made
of large volumes of non-radioactive material mixed with much smaller volumes

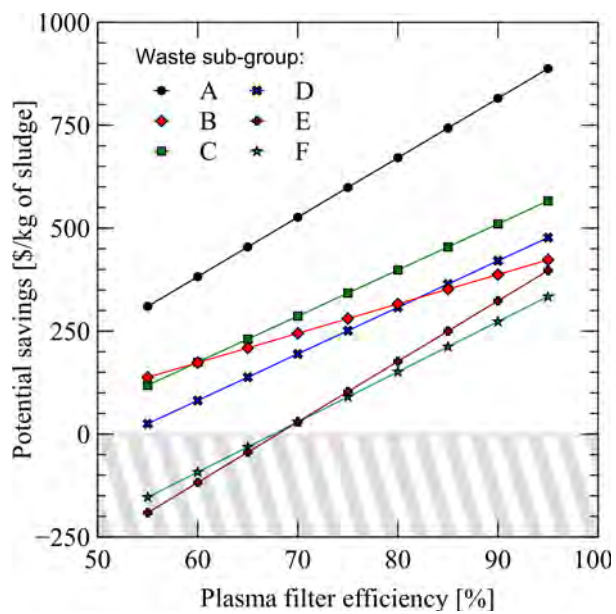


Figure 6: Projected savings by kg of sludge offered by plasma processing due to waste mass minimization, $(x_C - x_P)C_v^m$. A typical advanced chemical process [50] is used as the baseline for this comparison (removal of 86% Al, 99% Cr, 1.9% Fe and 62% Si). $C_v^m = 2000\$/\text{kg}$.

of highly radioactive elements. Disposal of this kind of waste hence requires efficient ways of separating radioactive and non-radioactive components, so that
325 the mass of the high-activity fraction is minimized.

The development of chemical processes to tackle this problem is particularly challenging due to the large number of chemical elements present in the waste, and because of the significant variations (both in terms of physical and chemical forms) observed across the different waste sub-groups. In contrast, plasma mass
330 filtering techniques appear promising because of their ability to discriminate elements irrespective of their chemical composition. For this application, plasma filtering takes advantage of the mass gap existing between most of the non-radioactive elements and the smaller fraction of radioactive elements.

Processing costs for chemical and plasma filtering techniques have been estimated for the processing of Hanford wastes. These estimates indicate that
335 chemical and plasma processing costs are on par with each other. However, preliminary calculations indicate that plasma processing could, in principle, provide for significantly higher reduction in the mass of the waste sent to the high-activity vitrification melter compared to chemical techniques. The reduction
340 in high-activity waste would yield significant savings. For example, a plasma filter offering 70% separation efficiency for the non-radioactive elements would decrease the overall cost by \$30 to \$530 per kg of sludge, which represents a 1.5 to 26.5% saving. In addition, as opposed to chemical washing and leaching, plasma sludge processing does not produce additional liquid waste that would
345 require additional treatment.

In conclusion, this preliminary evaluation confirms the economic feasibility of plasma filtering for sludge pretreatment. We recommend that more detailed cost comparisons be pursued that include capital, operation and maintenance costs. An interesting extension of this work would be to analyze how waste mass
350 minimization offered by plasma filtering can be combined with advanced glass formulations for increased waste loadings, reduced number of glass canisters, and, consequently, significant savings in the total clean-up costs.

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