Evaluating the consequences of household hazardous waste diversion on public health and ecological risks of leachate exposure

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Abstract
Household hazardous waste (HHW) includes approximately 1% of municipal solid waste (MSW), which is related to the lack of efficient separate collection schemes co-disposed with MSW in many countries worldwide. In this research, the effect of HHW diversion on leachate quality and consequent reduction in risks to the environment and human health was investigated. In this regard, physicochemical parameters of MSW leachate samples with and without hazardous fraction were characterized. Next, leachate pollution index (LPI) and associated risks were assessed. The results showed that the source separation of HHW decreases the LPI value by 13.48% and improves the potential of leachate for biological treatment, leading to a reduction in leachate treatment costs and complexity. Based on risk assessment results, HHW diversion could lower cancer risks of organic compounds and heavy metals to acceptable levels for adults and children. Moreover, by the source-separating of HHW, hazard indices for aquatic and terrestrial bioassays reduced by 41.5 and 36%, respectively. Furthermore, despite the comparatively low generated amount of HHW, this fraction should be emphasized due to its unintended effects for public health and the environment.

Keywords Household hazardous waste · Leachate · Source separation · LPI · Risk assessment · Xenobiotics

Introduction
Globally, over 2.01 billion tons of municipal solid waste (MSW) are generated every year; which is expected to increase by 70% and hit 3.40 billion tones in 2050 (Kaza et al. 2018). Rapid growth of MSW generation, especially in developing countries, has been accompanied by several environmental effects.

Leachate is a major negative consequence of mismanagement of MSW which could pose various risks to the environment and human health (Ferronato and Torretta 2019). The findings of previous studies indicate that the leachate quality can be affected by waste composition, and other factors such as moisture content, presence of adsorbents in MSW stream, landfilled method, and landfill age (Fabris et al. 2010). Leachate, as an unpleasant and concentrated liquid, is a combination of various organic compounds, heavy metals, ammonium, total solid, and the like (Iravanian and Ravari 2020). More precisely, the previous studies reported a wide range of pollutants in the quality analysis of MSW leachate such as the contaminants of emerging concerns (CECs) (Masoner et al. 2014), 35 organic compounds listed as priority pollutants, as well as heavy metals with carcinogenic and non-carcinogenic effects (Slack 2007). A number of researchers believe that the above-mentioned contaminants are originated from household hazardous wastes (HHW) (Ozkaya 2005; Parvin and Tareq 2021; Slack et al. 2005; Wang et al. 2020; Yu et al. 2020).

Hazardous wastes are characterized by their flammability, reactivity, explosiveness, and toxicity properties (United States Federal Code, 1976). Waste such as fertilizers, paints, old solvent, pharmaceuticals, wood preservatives, indoors and outdoors pesticides, heavy metal-containing products (e.g., fluorescent lamps), home and vehicle maintenance, personal care, household cleaning products, biological-infectious waste, and batteries, are listed as HHW subcategory (Delgado et al. 2007; Gholampour Arbastan and Gitipour 2020).
By considering waste electrical and electronic equipment (WEEE) as HHW, the percentage of this fraction in MSW stream will be about 0 to 4%; otherwise, it would be 0 to 1% by weight (Slack and Letcher 2011). HHW is commonly collected and managed with non-hazardous MSW streams due to the lack of a proper source-separation program in many countries like Iran (Rupani et al. 2019). Therefore, due to the high percentage of landfilling in developing countries, a significant amount of HHW is buried in disposal sites (Kaza et al. 2018). Accordingly, the potential risks of emitted leachate to water bodies significantly increase due to the high concentration of toxic substances in such waste, meaning threatening to public health, as well as aquatic and terrestrial bioassays (Baderna et al. 2011; Ghosh et al. 2015). Also, a few studies indicated that the contaminants leaked or discharged due to the improper management of industrial and hazardous waste can pose serious threats to human health (especially children), ecosystem, and the environment which could play a key role in the occurrence and prevalence of acute and chronic diseases and reproductive disorders among residents living in the vicinity of landfill sites (Fazzo et al. 2017).

Although some studies have been conducted on human health and ecological risks assessment of leachate (Baderna et al. 2011; Ghosh et al. 2015; Parvin and Tareq 2021; Torkashvand et al. 2021), no study has focused on the effect of HHW diversion on leachate quality, public health, and ecological risks. Therefore, this research aims to evaluate HHW diversion effects on the leachate quality, pollution index, as well as human health and ecological risk of leaked leachate to the groundwater.

Materials and methods

Study area and sample collection

Tehran, as the capital of Iran, has a population of 8,693,706 people, constitutes around 11% of the population of Iran. The results of waste management master plan (WMMP) in Tehran denote that the average generation rate of MSW is 0.764 kg/person/day. Official reports indicated that 8% of total weight of MSW is roughly allocated to leachate (Islamic City Council of Tehran, 2021). Curbside collection service was provided to collect and transfer generated MSW from 22 districts and 123 neighborhoods of Tehran to 11 waste transfer stations (WTSs) every day. In the next step, the mixed MSW from the WTSs was transported to Aradkouh processing and disposal complex (APDC) by semi-trailers. After separating recyclable solid waste in APDC manually, residuals are disposed of into the composting plant, incinerator, and landfill (Rupani et al. 2019). Among the 11 defined Tehran WTSs, Darabad, Zanjan, Kuhak, Hakimiyeh, Shahr-e-Rey, and Azadegan were selected as sampling points since they cover approximately 5.6 million people including 1.87 million households (more than half of total households in Tehran) who are living in 10 different districts. On average, these stations receive 536, 546, 829, 817, 321, and 792 tons of MSW per day from different districts of Tehran, respectively (Islamic City Council of Tehran, 2021). The main inclusion criteria were the distribution in all over the city and the higher peak capacity in comparison with other stations. Figure 1 illustrates the allocated waste to WTSs.

Like the previous studies (Delgado et al. 2007; Ojeda-Benítez et al. 2013), the sample size of households living in each area covered by the WTSs was determined using Eq. 1 (Cochran 1977).

\[
n = \frac{Z_{\alpha/2}^2 \times N \times p \times q}{e^2 \times (N - 1) + Z_{\alpha/2}^2 \times p \times q}
\]

where \(n\) shows the sample size, \(Z_{\alpha/2}\) indicates the required level of statistical significance, \(p\) is the estimated proportion of a population attribute, \(q = p - 1\), \(N\) refers to the population size, and \(e\) is considered as the desired level of precision with a maximum variability of 50% (\(p = 0.5\)) and a confidence level of 95% with \(\pm 5\%\) precision. Table 1 indicates the required sample size for 1,875,295 households living in each district covered by a WTS.

As shown in Table 1, in the stations with less than 40 samples (i.e., Darabad and Shahr-e-Rey), 40 were considered as the minimum. Therefore, a sample size of 408 was regarded for the households living in the areas covered by the target WTSs. These households were randomly selected and trained face-to-face by Tehran waste management organization (TWMO) trainers. On the first day, the sample households were asked to dispose of their waste in a plastic bag without any source-separation as usual. The generated municipal solid waste with the hazardous fraction (MSWH) was collected by a door-to-door collection system and delivered to the WTSs. On the next day, they were asked to dispose of non-hazardous municipal solid waste and HHW (according to the list presented to households (Gholampour Arbastan and Gitipour 2020)), each separately in two plastic bags. The municipal solid waste source-separated (MSWS) and HHW were collected door-to-door and transported to the WTSs by open garbage trucks without compactor. At the WTSs, the collected MSWS bags were emptied, and misplaced HHW was separated manually. In the next step, MSWS and MSWH samples were compacted and stored in two separate containers. Leachate samples...
were collected from the mixture of derived leachate from MSWS and MSWH of the 6 WTSs. Next, the collected samples were transferred to the laboratory instantly and stored at 4 °C until the next steps. Due to the small amount and a variety of waste types and chemical composition of HHW, the above-mentioned procedures were repeated three times and the average was reported.

### Chemicals and materials

All chemicals and solvents were purchased from Sigma-Aldrich (Steinheim, Germany) and Merck (Darmstadt, Germany). High purity grade (> 99%) pyrethroids pesticides were purchased from Dr. Ehrenstorfer (Augsburg, Germany). Poly-Sery HLB (6 cc, 200 mg) and Poly-Sery

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**Table 1** Household sample size

| Transfer station name | Related district number | Number of households | Sample size of district | Sample size of transfer station |
|-----------------------|-------------------------|----------------------|-------------------------|---------------------------------|
| Darabad               | 1                       | 166,881              | 34                      | 34                              |
| Zanjan                | 2                       | 239,742              | 50                      | 74                              |
|                       | 10                      | 117,405              | 24                      |                                 |
| Hakimiyeh             | 4                       | 304,169              | 63                      | 93                              |
|                       | 8                       | 148,582              | 30                      |                                 |
| Kuhak                 | 5                       | 292,257              | 60                      | 84                              |
|                       | 21                      | 60,602               | 12                      |                                 |
|                       | 22                      | 55,111               | 12                      |                                 |
| Shahr-e-Rey           | 20                      | 115,719              | 24                      | 24                              |
| Azadegan              | 14                      | 171,121              | 35                      | 77                              |
|                       | 15                      | 204,306              | 42                      |                                 |
| Total                 |                         | 1,875,295            | 386                     | 386                             |

Bold represents the sum of the above cell values

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**Fig. 1** The details of sampling points
PSD SPE cartridges were supplied by (ANPEL, China). Membrane filters (0.45 μm) were supplied by Merck Millipore (Darmstadt, Germany).

**Analytical methods**

**Physicochemical analysis** Physicochemical parameters of MSWS and MSWH leachate samples were analyzed in accordance with the standard methods for evaluating water and wastewater (APHA 2013).

**Heavy metals analysis** An atomic absorption spectrometer (AAS) was used to determine the concentrations of lead (Pb), nickel (Ni), chromium (Cr), zinc (Zn), iron (Fe), cadmium (Cd), and copper (Cu) in the filtered samples (AANALYST 200, Perkin Elmer, USA). Cd, Fe, Cr, Pb, Zn, Ni, and Cu hollow cathode lamps were used as light source at wavelengths of 228.8, 248.3, 357.9, 283.3, 213.9, 232.0, and 324.8 nm, respectively. In addition, a linear regression method was used for the calibration curve. An atomic absorption spectrometry with the manual cold vapor (Varian-AA240FS), a vapor generation accessory (VGA-232.0, and 324.8 nm, respectively. In addition, a linear regression method was used for the calibration curve. An atomic absorption spectrometry with the manual cold vapor (Varian-AA240FS), a vapor generation accessory (VGA-276), and a GBC-932 atomic absorption spectrometer couple with hydride generation system (HG-AAS, 3000, AUS) were used to measure the concentrations of mercury (Hg) and arsenic (As) in leachate samples, respectively.

**BTEX determination in leachate samples** Extraction for analyzing benzene, toluene, ethylbenzene, and xylene (BTEX) in both MSWS and MSWH leachate samples proceeded as described before (Fakhari et al. 2012). In the practical procedures, 15 mL of a filtered leachate sample was placed in 18 mL headspace vial. A PTFE-faced silicone septum was used for capping the vial. The temperature of the sample was held constant at 80 °C and stirred at 750 rpm for 15 min via a water bath and magnetic stirrer.

The enriched headspace vapors with BTEX compounds were manually injected to the GC injector through a flexible HP-5 fused silica capillary column (12 cm × 0.32 mm × 0.25 μm). Gas chromatographic analyses were performed using an Agilent 4890 gas chromatograph (Wilmington, DE, USA) equipped with a split/splitless injector and flame ionization detector (FID). A 30 m × 0.25 mm (internal diameter) fused silica capillary column coated with a 0.25 μm film thickness of HP-5 and ultra-pure nitrogen (99.9999% purity) at a constant flow rate of 1 mL/min were used as the analytical GC column and carrier gas, respectively. The GC oven temperature program has set as follows: the oven temperature was held at 55 °C for 10 min, then increased to 120 °C at a rate of 15 °C/min and held for 1 min at this temperature. The temperature of the detector (FID) was 300 °C. Finally, the injection port was held at 250 °C.

**Extracting and analyzing phenolic compounds** Like the previous studies, solid phase extraction (SPE) (Reitzel and Ledin 2002) and derivatization (Padilla-Sánchez et al. 2011) procedures were conducted to determine the trace concentrations of phenolic compounds in leachate samples. In brief, pH of analytes (leachate samples and standard solutions) was adjusted to 2 with concentrated hydrochloric acid (2 M). Solid phase extraction was performed using Poly-Sery PSD cartridges with 200 g styrene–divinylbenzene copolymer sorbent material (ANPEL Corp., China). The Poly-Sery PSD cartridges were conditioned with 2 mL ethyl acetate and 2 mL MeOH followed by 5 mL of acidified ultrapure water (pH = 2). Then, the analyte was placed onto the conditioned cartridges, and the SPE cartridges were dried using nitrogen for 30 min. Subsequently, cartridges were eluted by 4 mL ethyl acetate. The eluate was gently concentrated in the nitrogen stream to 860 μL and added to 20 μL of pyridine (Py) and 100 μL anhydride acetic acid (AAA) to perform the derivatization reaction. The mixture was being stirred for 2 min in a rotary agitator and injected directly to the GC later. Phenolic compounds in the prepared samples were detected using an Agilent 7890 gas chromatograph equipped with an Agilent 5975 mass spectrometer with a quadrupole mass analyzer. A HP-5 MS (30 m × 0.25 mm i.d. 0.25 μm) and ultrapure helium (99.9999% purity) at the constant flow of 1 mL/min were used as capillary column and carrier gas, respectively. Mass spectrum was operated in the selected ion monitoring (SIM) mode to quantify phenolic compounds in leachate samples. The samples were injected in splitless mode. Gas chromatography (GC) oven temperature program was developed as follows: Initial equilibrium temperature of 60 °C for 2 min and 8 °C/min to 290 °C held for 5 min.

**Preconcentration of pyrethroid pesticides and GC–MS analysis** SPE method was employed in the manner described in previous study for the preconcentration of target pyrethroids pesticides (Rocha et al. 2012). In other words, Poly-Sery HLB cartridges (6 cc, 200 mg, ANPEL Corp., China) were preconditioned with 5 mL of ethyl acetate, followed by the same volume of methanol and ultrapure water. The filtered MSWS and MSWH leachate samples were passed through the HLB cartridges at a constant flow rate of 2 mL/min. Next, the cartridges were eluted using 5 mL of ethyl acetate. Finally, the extract reduced gently using nitrogen streams. About 1 g anhydrous sodium sulfate was added to the sample bottle, and the bottle was washed 3 times with 4 mL dichloromethane to remove any excess water. Subsequently, the washing solution was concentrated to 1000 μL via gentle nitrogen
streams and added to extract analyte and injected to GS–MS. The quadrupole mass spectrum was adjusted in the SIM mode, and 1 μL of sample was injected in the splitless mode. In addition, perfluorotributylamine (PFTBA) was employed to calibrate the mass axes of mass spectrometer. Target pyrethroids were determined using an Agilent 7890 gas chromatograph coupled to a 5975A MS. This apparatus was equipped with a HP-5MS ((5%phenyl)-95% Polydimethylsiloxane) (30 m × 0.25 mm id) capillary column with a 0.25 μm film thickness, coated with fused silica. The data were analyzed by MSD ChemStation software (E.02.01.1177). The GC–MS interface, quadruple, MS source, and GC injection port temperatures were kept at 290, 150, 230, and 290 °C, respectively. The GC oven was maintained for 2 min at 90 °C, ramp 7 °C/min to 290 °C and held for 5 min at this temperature.

Statistical analysis

The experimental results were presented as means ± standard deviation of triplicates. Shapiro–Wilk test was used to assess the normality of distributions of obtained data. In accordance with the results of the Shapiro Wilk test, a t-test at p-value < 0.05 and a Wilcoxon rank-sum test were used to compare statistical differences between the mean concentrations of heavy metals and organic contaminants in MSWS and MSWH leachate samples for normally and non-normally distributed data, respectively. IBM SPSS version 25 was used for data analysis.

Leachate pollution index (LPI)

Kumar and Alappat (2005) proposed a quantification method for leachate contamination potential of different landfills on a comparative scale with reference to leachate pollution index (LPI). LPI is widely used in ranking of landfill sites, assigning resources, conducting technical analysis, implementing standards, as well as systematic evaluation and public information. The LPI formulation process consists of choosing variables, obtaining weights for the selected pollutant variables, formulating their sub-indices curves, and accumulating the pollutant variables. In this study, chloride, cyanide, and total coliform were not analyzed based on LPI calculation parameters. Therefore, the modified equation as described in (Kumar and Alappat 2005) was used for calculating the LPI.

\[
LPI = \sum_{i=1}^{m} \frac{W_i \times p_i}{W_i}
\]

where LPI shows the weighted additive LPI, \( W_i \) indicates the weight of the \( i \)th pollutant variable, \( p_i \) represents the sub-index score of the \( i \)th leachate pollutant variable, and \( m \) is the number of leachate pollutant parameters for which data are available.

Human health risk assessment

Chronic exposure to heavy metals and other toxicants in the environment through potential exposure pathways is considered as the potentially detrimental factors for human health (Tay et al. 2019). Based on a function of hazard and exposure, risk assessment is defined as the process of evaluating the probability of occurring any possible magnitude of adverse health impacts over a specific period of time (Bempah and Ewusi 2016). The quantification of the risk level usually gives the health risk assessment for each potentially toxic organics and inorganics, which is expressed in terms of a carcinogenic or a non-carcinogenic health risk. This study separately assessed both carcinogenic and non-carcinogenic risks created by exposing to leachate samples (Wongsasuluk et al. 2014).

Based on the hypothesis of previous studies (Baderna et al. 2011; Ghosh et al. 2015), dilution ratio was assumed 1:100 and ingestion was considered as the main exposure route in order to examine the adverse effects of leachate leakage of the MSWS and MSWH to water on human health and the environment. The diluted concentrations of each contaminant were used to determine the human health and ecological risks. For heavy metals and organic compounds, the dose received (chronic daily intake CDI; average daily dose, ADD) through ingestion by adults and children was calculated via Eq. 3 (Baderna et al. 2011; USEPA 2004).

\[
CDI = \frac{(C_{\text{water}} \times \text{WI} \times \text{ED} \times \text{EF})}{\text{BW} \times \text{AT}}
\]

where CDI is the chronic daily intake of pollutants through ingestion pathway (CDI_{mg}) (mg/kg/ day), \( C \) refers to the concentration of the pollutants in (μg/L), and WI shows the water ingestion rate (L/day). Its average consumption rates for children and adults are 1 and 21 per day, respectively. The body weight (BW) of child and adult groups is 15 and 70 kg, respectively. ED shows the durations of human exposures for children and adults were 6 years and 30 years, respectively. EF represents the exposure frequency of 350 (days/year), and AT is the averaging time of human exposure: AT = 70×365 for carcinogenic risk and AT = ED×365 days for non-carcinogenic risk.

Non-carcinogenic risks

The hazard quotient (HQ), as given in Eq. 4, was used to assess the potential for non-carcinogenic risks of exposure to each pollutant. HQ signifies the
ratio between the reference dose (RfD) and average daily dose (ADD) of each pollutant.

\[
HQ = \frac{ADD}{RfD}
\]

(4)

where HQ shows the hazard quotient, and RfD indicates the reference dose for the selected compound (mg/kg/day).

To estimate the total potential non-carcinogenic health risks triggered by exposure to a mixture of pollutants in contaminated water by leachate, the computed HQs for each element were integrated and expressed as a hazard index (HI) as defined by Eq. 5:

\[
HI = \sum HQ
\]

(5)

According to the (USEPA 2004) where HQ/HI < 1, the exposed population is improbable to experience adverse health effects. If HQ/HI > 1 then there may be concern for potential non-carcinogenic effects.

**Carcinogenic risks** Carcinogenic risks were calculated for any compound with a slope factor (SF) value (mg/kg/day), namely As, Pb, Cr, benzene, ethylbenzene, 2,4,6 TCP, and PCP by using the following equation:

\[
CR = CDI \times SF
\]

(6)

where CR shows the cancer risk, SF is the slope factor (mg/kg/day) as an upper-bound estimate of risk per dose increment which can be used to estimate risk probabilities for different exposure levels. Cancer risk was divided into negligible risk (CR < 10^{-6}), acceptable low risk (1 \times 10^{-6} \leq CR < 5.1 \times 10^{-5}), acceptable high risk (5.1 \times 10^{-5} \leq CR < 10^{-4}), and unacceptable risk (CR \geq 10^{-4}) (Legay et al. 2011).

**Ecological risk assessment**

Ecological risk assessment (ERA) is used for evaluating and comparing the impacts of stressors, often chemical high-risk pollutants (HRPs), on the local environment. In ERA, an undesired event often relies upon the chemical HRP of interest and on the risk assessment scenario, which may adversely affect creatures, populations, or ecological communities (He and Huang 2019). ERAs provide technical support in management decisions such as allowing a discharge to a body of water, cleaning up a spill, or harvesting limits to set for a natural resource (Suter and Norton 2018).

Based on the study of (Baderna et al. 2011), the present research estimated the ecological risks to the environment through their hazard quotients (HQs) defined by the following equation:

\[
HQ = \frac{PEC}{PNEC}
\]

(7)

where PEC shows the predicted environmental concentration resulting from the chemical analysis of both MSWS and MSWH leachate at 1:100 dilution, and PNEC represents the predicted no-effect concentration.

The risk assessments for terrestrial (small rodents such as mice, rats, or rabbits) and aquatic (Oncorhynchus mykiss or Pimephales promelas) environments were performed by assuming a water daily intake of 20 ml, an average body weight of 100 g, and a ratio of water intake to body weight of 1, respectively. The following equation yields PEC:

\[
PEC = \frac{C \times WDI}{BW}
\]

(8)

where C indicates the concentration of pollutant in the contaminated water by MSWS and MSWH leachates, WDI shows water daily intake (L/day), and BW represents the body weight (kg).

To calculate PNEC concentrations, ecological parameters and assessment factors (AF) for studying aquatic species (Oncorhynchus mykiss or Pimephales promelas) and small rodents (rats) were obtained from the previously selected (Baderna et al. 2011), peer-reviewed databases such as ECO-TOX (USEPA 2021), and advanced ChemIDPlus (2021). The potential ecological risk was assessed according to a widely used risk ranking criterion. HQ values were divided into low or negligible (0.1 < HQ < 1), medium (1 \leq HQ < 10), and high ecological risk (HQ > 10) (Mendoza et al. 2015).

**Results and discussion**

**Household hazardous waste composition**

In the previous study, the effect of seasonal variations on the composition and generation rate of HHW in Tehran was investigated. The results indicated that the generation rate of HHW was within the range of 6.26–8.25 g/person/day with a mean value of 7.27 g/person/day, representing 0.95% of the municipal solid waste stream in the city. Figure 2 displays a detailed composition of HHW in Tehran (Gholampour Arbastan and Gitipour 2020).

**Target pollutant analysis**

Table 2 indicates the average concentration of target pollutants in MSWS and MSWH leachate samples. Based on the results, source separation of HHW mitigates heavy metals,
pesticides, BTEX, and phenolic compounds concentrations by 59.6, 69.9, 64.6, and 81.2%, respectively.

As presented in Table 2, though the separation of HHW at the source caused the reduction in more than half of concentration of metals with carcinogenic effects (i.e., arsenic, chromium, and lead), but still these metals concentrations even in MSWS leachate are above their safe limits of world health organization (WHO) in wastewater. In addition to metal-containing non-hazardous MSW including textiles, food packaging, and leather, crop contamination with Tehran Oil Refinery leakage could be considered as the main sources of the mentioned contaminants (Pourang and Noori 2014).

Phenolic compounds including chlorinated phenolic compounds (CPs) and methylated phenols (cresols) are considered as hazardous materials due to their properties such as toxicity, flammability, carcinogenicity, and mutagenicity (Duan et al. 2018). The average concentration of cresols in the MSWH samples is in good agreement with the reported average concentration of cresols in the fresh engineering landfill leachate (48 μg/L) (Ozkaya 2005),

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**Table 2** Characteristics of MSWS and MSWH leachate samples

| Heavy metals/Xenobiotic organic compounds (XOCs) | Elements/Compounds | Concentration (Mean ± SD) | Unit |
|-----------------------------------------------|--------------------|---------------------------|------|
| Heavy metals                                  | As                 | 30.3 ± 29.5               | µg/L |
|                                               | Cd                 | 41 ± 18.9                 | µg/L |
|                                               | Cr                 | 302.3 ± 72.8              | µg/L |
|                                               | Cu                 | 442 ± 183.7               | µg/L |
|                                               | Hg                 | 0.52 ± 0.46               | µg/L |
|                                               | Ni                 | 508.2 ± 170.8             | µg/L |
|                                               | Pb                 | 249.8 ± 106.6             | µg/L |
| Phenolic Compounds                             | Phenol             | 7.85 ± 3.8                | µg/L |
|                                               | Cresols            | 5.7 ± 4.1                 | µg/L |
|                                               | 2-CP               | 8.1 ± 5.8                 | µg/L |
|                                               | 2,4-DCP            | 6.14 ± 4.6                | µg/L |
|                                               | 2,4,6-TCP          | 1.9 ± 1.6                 | µg/L |
|                                               | PCP                | 0.32 ± 0.5                | µg/L |
| BTEX                                          | B                  | 4.2 ± 3.2                 | µg/L |
|                                               | T                  | 8.7 ± 4.3                 | µg/L |
|                                               | E                  | 0.82 ± 0.78               | µg/L |
|                                               | X                  | 8.2 ± 4.5                 | µg/L |
| Pesticides                                    | CYPE               | 61.9 ± 21.4               | ng/L |
|                                               | DELT               | 32.5 ± 15.2               | ng/L |
|                                               | PERM               | 48.8 ± 15.4               | ng/L |
and is dramatically lower than the concentration value (17.3 mg/L) in hazardous landfill leachate, which indicated the effect of hazardous waste on cresols concentration in leachate properly. Separating HHW from MSW stream drops cresols concentration to 5.7 μg/L, which is consistent with the result of the previous study conducted in Japan, a pioneer on HHW management, to examine municipal landfill leachate (Kurata et al. 2008). The results of the previous studies indicated that increasing chlorine bonds can enhance both the octanol–water partition coefficients ($K_{ow}$) and dissociation constant ($K_a$), and reduce water solubility of CPs and their concentrations in leachate which are in accordance with the present experimental data (Duan et al. 2018). The 81.2% reduction in total concentrations of CPs in MSWS could be attributed to separate containers of chlorine-releasing agents such as household bleach and phenolic disinfectants from MSW stream which are widely consumed during COVID-19 pandemic in Iran (Yari et al. 2020).

Regarding BTEX compounds, the lowest and the highest reduction were observed in benzene (57.7%) and ethylbenzene (72.1%), respectively. Previous studies reported that benzene is found in HHW category and non-hazardous waste fraction such as cigarette filter in high concentration. Contrarily, HHW is among the main sources of ethylbenzene in leachate. Compared to toluene, xylene is typically found in HHW in higher concentration. In addition, it is more compliant against the interaction of detergents (Fabris et al. 2010), which can intensify MSWH and MSWS variations.

As indicated in Table 2, the average concentrations of permethrin, cypermethrin, and deltamethrin are reduced by 76.7, 65.8, and 61.6% in MSWS samples, respectively. The highest reduction was observed for permethrin contaminant since it is used for medicinal purposes in addition to household pesticides. The remaining pyrethroids in MSWS results from food waste such as rice and animal origin food (Dallegrove et al. 2018) are contaminated by pesticides.

### Statistical analysis

Based on the statistical analysis, significant differences ($\rho < 0.05$) were observed between the average concentrations of As, Cd, Cr, Hg, Ni, Pb, cresols, 2CP, 2,4DCP, 2,4,6TCP, phenol, BTEX, and pesticides in MSWS and MSWH samples. Despite higher concentrations of Cu and PCP in MSWH than MSWS samples, the lack of significant differences ($\rho > 0.05$) between average concentrations

| Index     | Parameter                  | Analytical result | Pollutant weight ($w_i$) | Individual pollution rating ($p_i$) | Overall pollution rating ($w_i*p_i$) | Percentage of reduction |
|-----------|----------------------------|-------------------|--------------------------|-----------------------------------|-------------------------------------|-------------------------|
| LPI Heavy metal | Cr                       | 0.3 0.83          | 0.064                    | 5 6                                | 0.32 0.384                          | 30.04                   |
|           | Pb                        | 0.25 1.17         | 0.063                    | 6 10                               | 0.378 0.63                          |                         |
|           | Hg                        | 0.0005 0.007      | 0.062                    | 5 14                               | 0.31 0.868                          |                         |
|           | As                        | 0.003 0.34        | 0.061                    | 5 5                                | 0.305 0.305                         |                         |
|           | CN⁻                      | – –               | 0.058                    | – –                                | – –                                 |                         |
|           | Zn                        | 2.76 3.04         | 0.056                    | 5 5                                | 0.28 0.28                           |                         |
|           | Ni                        | 0.51 0.97         | 0.052                    | 5 6                                | 0.26 0.312                          |                         |
|           | Cu                        | 0.44 0.89         | 0.05                     | 6 7                                | 0.3 0.35                           |                         |
|           | Fe                        | 15.84 17.41       | 0.045                    | 5 6                                | 0.225 0.27                          |                         |
| LPI Inorganic | pH                       | 5.37 5.53         | 0.055                    | 5 5                                | 0.275 0.275                         | 18.85                   |
|           | TKN                      | 1463.3 2076.6     | 0.053                    | 49 75                              | 2.597 3.975                         |                         |
|           | Ammonia nitrogen          | 931.1 1404.9      | 0.051                    | 95 100                             | 4.845 5.1                           |                         |
|           | TDS                      | 10,990 13,860     | 0.05                     | 25 34                              | 1.25 1.7                           |                         |
|           | Cl⁻                      | – –               | 0.048                    | – –                                | – –                                 |                         |
| LPI Organic | COD                      | 27,100 31,583.3   | 0.062                    | 84 87                              | 5.208 5.394                         | 1.87                    |
|           | BOD5                     | 16,033.3 16,566.7 | 0.061                    | 70 70                              | 4.27 4.27                          |                         |
|           | Phenolic compounds        | 0.030 0.160       | 0.057                    | 5 5                                | 0.285 0.285                         |                         |
|           | Total coliform            | – –               | 0.052                    | – –                                | – –                                 |                         |
| LPI       |                           |                   |                          |                                    | 25.07 28.98                         | 13.48                   |

Bold represents the sum of the above cell values

*All average values in mg/L except pH
of these pollutants may be related to the presence of copper and pentachlorophenol in the composition of non-hazardous wastes such as non-ferrous and wood waste, respectively.

**Leachate pollution index (LPI)**

LPI is extensively used as an appropriate quantitative tool for diagnosing hazardous leachate and predicting adverse effects on water body (Kumar and Alappat 2005). The present study compared the LPI values for both MSWH and MSWS leachate samples.

Table 3 indicates the LPI values of 25.07 and 28.98 for both MSWS and MSWH leachate, respectively. Kumar and Alappat (2005) showed that the LPI values of leachate from hazardous industrial waste landfills are significantly higher than domestic waste landfills. The result is consistent with the finding of current study. Although source separation of HHW reduced the LPI value by 13.48%, both the LPI values were dramatically higher than the standard limit (7.378) needed to release into the environment, which requires leachate treatment.

As shown in Table 3, COD and ammonia nitrogen play important roles in exceeding the LPI standard limit, which have been highlighted in previous studies (Naveen et al. 2017). In the present study, ammonia nitrogen and COD values in MSWS decreased by approximately 33.7 and 14.2% compared to MSWH, respectively. A reduction by one third of ammonia nitrogen value indicates that HHW such as home cleaners, oil-based solvents, and fertilizers may affect ammonia concentration in leachate although the protein fraction of MSW (Yirong et al. 2017) is a main source of nitrogen. As presented in Table 3, the highest reduction in three sub-indices is related to LPI$_{hm}$ (30.04%), followed by LPI$_{inorganic}$ (18.85%), and LPI$_{organic}$ (1.87%).

**Human health risk assessment**

Table 4 presents average daily dose (ADD), chronic daily intake (CDI), hazard quotient (HQ), hazard index (HI), and carcinogenic risk values of heavy metals and organic compounds for adults and children through ingesting contaminated water by MSWS and MSWH leachates.

**Non-carcinogenic risks**

Based on the results in Table 4, the HI value of contaminated water by MSWS leachate decreased by 38.56% for all population groups in comparison to MSWH. However, HI values were higher than the threshold (HI > 1) for both adults and sensitive groups. Like the results of previous studies, ammonia allocated 89.58 and 96.63% of the hazard index values in MSWS and MSWH, respectively (Baderna et al. 2011; Naveen et al. 2017). Although the HQ values of heavy metals and other studied contaminants for both MSWS and MSWH are lower than the threshold (HQ < 1) for children and adults, the ingestion of leachate-contaminated water may pose non-carcinogenic risks to human health due to high concentrations of ammonia.

**Carcinogenic risks**

Carcinogenic risks of heavy metals (i.e., As, Cr, and Pb), and organic compounds such as benzene, 2,4,6-trichlorophenol, ethylbenzene, and pentachlorophenol, for adults and children were evaluated through ingesting the contaminated water by MSWS and MSWH leachates. As shown in Table 4, the calculated CRs of organic compounds in all leachate samples were negligible for adults and children, and CRs of the contaminated water by MSWH leachate among adults was acceptably low for chromium and lead, and acceptably high for arsenic. After HHW segregation, cancer risk levels for arsenic and lead decreased to the acceptable low and negligible levels, respectively. However, As and Cr for MSWH samples exceeded the threshold of CRs ($10^{-4}$) for children. HHW diversion can properly reduce the cancer risks of drinking leachate-contaminated water to an acceptable level for sensitive groups. The results are confirmed by previous studies, which indicated that CRs of heavy metals for leachate-contaminated water were at the unacceptable risk levels for adults and children in Pakistan and Italy (Baderna et al. 2011; Iqbal et al. 2019). In other words, cancer risks of drinking water from nearby wells for children is the unacceptable and acceptable low risk for As, and Cr, respectively (Baghvand and Mashhadi 2009). The present study proved that HHW diversion from MSW stream could significantly reduce the cancer risks of leachate leakage.

**Ecological risk assessment**

The ecological risk assessment was performed on fish species and small mammals due to the leakage of MSWS and MSWH leachates to the water environment.

As indicated in Table 5, remarkable induced-risk reductions in target bioassays were noticed due to the HHW separation. It should be mentioned that a great portion of the ecological risk due to ingestion of contaminated water by MSWS and MSWH leachates was attributed to their high ammonia concentrations, which in previous studies were discussed. (Baderna et al. 2011; Ghosh et al. 2017; Naveen et al. 2017).
Table 4  Results of human risk assessment through ingesting contaminated water by MSWS and MSWH samples

| Parameter  | RID (mg/kg/day) | SF (mg/kg/day) | MSWS | MSWS |
|------------|----------------|----------------|------|------|
|            | ADD (mg/kg/day) | Hazard Quotient (HQ) | CDI (mg/kg/day) | Carcinogenic risk (CR) | ADD (mg/kg/day) | Hazard Quotient (HQ) | CDI (mg/kg/day) | Carcinogenic risk (CR) |
|            | Adults | Children | Adults | Children | Adults | Children | Adults | Children |
| Arsenic (As) | 3.00E−04 | 1.5 | 9.28E−05 | 2.16E−04 | 3.09E−01 | 7.22E−01 | 3.98E−05 | 2.16E−04 | 5.96E−05 | 3.25E−04 |
| Cadmium (Cd) | 5.00E−04 | 2.58E−05 | 6.02E−05 | 5.16E−02 | 1.20E−01 | 2.16E−04 | 1.12E−07 | 2.62E−05 | 2.25E−02 | 5.24E−02 |
| Chromium (Cr) | 3.00E−03 | 2.27E−04 | 5.30E−04 | 7.57E−02 | 1.77E−01 | 9.73E−05 | 5.30E−04 | 4.87E−05 | 2.65E−04 | 8.28E−05 |
| Copper (Cu) | 4.00E−02 | 1.34E−04 | 3.13E−04 | 3.35E−03 | 7.82E−03 | 3.98E−05 | 1.94E−05 | 5.96E−05 | 3.25E−04 | 8.31E−06 |
| Mercury (Hg) | 3.00E−04 | 1.87E−06 | 4.37E−06 | 6.24E−03 | 1.46E−02 | 2.24E−04 | 1.42E−07 | 3.30E−07 | 4.72E−04 | 1.93E−05 |
| Nickel (Ni) | 2.00E−02 | 2.66E−04 | 6.22E−04 | 1.33E−02 | 3.11E−02 | 2.16E−04 | 1.34E−07 | 3.10E−07 | 4.72E−04 | 1.93E−05 |
| Lead (Pb) | 3.50E−03 | 8.50E−03 | 3.19E−04 | 7.45E−04 | 9.12E−01 | 2.13E−01 | 6.84E−05 | 1.60E−01 | 1.96E−02 | 4.56E−02 |
| Benzene | 4.00E−03 | 5.60E−03 | 2.69E−06 | 7.45E−06 | 9.12E−01 | 2.13E−01 | 6.84E−05 | 1.60E−01 | 1.96E−02 | 4.56E−02 |
| Toluene | 8.00E−02 | 6.35E−06 | 1.48E−05 | 7.94E−05 | 1.85E−04 | 4.88E−04 | 1.14E−06 | 2.66E−04 | 2.25E−02 | 5.24E−02 |
| Ethylbenzene | 1.00E−01 | 8.05E−07 | 1.88E−06 | 8.05E−06 | 2.97E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 | 2.97E−05 | 6.92E−05 |
| Xylene | 2.00E−01 | 7.08E−06 | 1.65E−05 | 8.75E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 | 2.97E−05 | 6.92E−05 |
| 2CP | 5.00E−03 | 9.36E−06 | 2.18E−05 | 7.94E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 | 2.97E−05 | 6.92E−05 |
| 2,4 DCP | 3.00E−03 | 9.36E−06 | 2.18E−05 | 7.94E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 | 2.97E−05 | 6.92E−05 |
| 2,4,6 TCP | 1.00E−03 | 1.10E−02 | 2.38E−05 | 9.36E−06 | 2.18E−05 | 7.94E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 |
| PCP | 5.00E−01 | 1.01E−00 | 2.35E−07 | 9.36E−06 | 2.18E−05 | 7.94E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 |
| Phenol | 3.00E−01 | 1.13E−05 | 2.63E−05 | 8.75E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 | 2.97E−05 | 6.92E−05 |
| Cypermethrin | 1.00E−02 | 4.96E−05 | 1.16E−04 | 9.36E−06 | 2.18E−05 | 7.94E−05 | 2.30E−05 | 1.14E−06 | 2.24E−06 | 5.54E−05 |
| Deltamethrin | 1.00E−02 | 2.32E−05 | 5.41E−05 | 2.32E−03 | 5.41E−03 | 2.32E−03 | 5.41E−03 | 2.32E−03 | 5.41E−03 | 8.91E−05 |
| Permethrin | 5.00E−02 | 5.75E−05 | 1.34E−04 | 3.13E−03 | 6.26E−03 | 3.13E−03 | 6.26E−03 | 3.13E−03 | 6.26E−03 | 8.91E−05 |
| Cresols | 1.00E−01 | 1.09E−05 | 2.55E−05 | 5.41E−03 | 1.09E−04 | 2.55E−05 | 5.41E−03 | 1.09E−04 | 2.55E−05 | 8.91E−05 |
| Ammonia | 7.90E−01 | 8.98E−01 | 4.87E−00 | 1.14E−01 | 8.91E−05 | 2.97E−05 | 1.14E−06 | 3.23E+00 | 7.53E+00 | 3.23E+00 |
| Hazard Index (HI) | 5.44E+00 | 1.27E+01 | 8.89E+00 | 2.30E+01 | 5.44E+00 | 1.27E+01 | 8.89E+00 | 2.30E+01 | 5.44E+00 | 1.27E+01 |

Bold indicates the values that exceed the threshold limit

aRID from USDOE (2021)
bAlidadi et al. (2019), Baderna et al. (2011)
cPress and Jones (2010)
dRID from IRIS database (USEPA 2020)
Table 5 Results of ecological risk assessment of contaminated water by MSWS and MSWH leachate on aquatic and terrestrial environment

| Parameter       | MSWH        | MSWS        |
|-----------------|-------------|-------------|
|                 | Rodent      | Aquatic     | Rodent      | Aquatic     |
|                 | PEC (mg/kg) | PNEC (mg/kg) | Hazard Quotient (HQ) | PEC (mg/kg) | PNEC (mg/kg) | Hazard Quotient (HQ) |
| Arsenic (As)    | 6.77E−04    | 1.04E−02   | 6.51E−02 | 3.39E−03 | 9.90E−03 | 3.42E−01 |
| Cadmium (Cd)    | 1.88E−04    | 7.70E−03   | 2.45E−02 | 9.42E−04 | 7.00E−06 | 1.35E+02 |
| Chromium (Cr)   | 1.66E−03    | 2.40E−02   | 6.91E−02 | 8.29E−03 | 3.70E−02 | 2.24E−01 |
| Copper (Cu)     | 9.78E−04    | 5.60E−02   | 1.75E−02 | 4.89E−03 | 1.20E−04 | 4.08E+01 |
| Mercury (Hg)    | 1.37E−04    | 3.17E−02   | 4.31E−03 | 6.83E−05 | 5.00E−06 | 1.37E+01 |
| Nickel (Ni)     | 1.95E−03    | 5.00E+00   | 3.89E−04 | 9.73E−03 | 5.00E−05 | 1.95E+02 |
| Lead (Pb)       | 2.33E−03    | 4.70E−02   | 4.96E−02 | 1.17E−02 | 7.20E−05 | 1.62E+02 |
| Benzene         | 1.96E−05    | 9.30E−01   | 2.11E−05 | 9.82E−05 | 5.30E−03 | 1.85E−02 |
| Toluene         | 4.64E−05    | 6.36E−01   | 7.29E−05 | 2.32E−04 | 3.61E−02 | 6.42E−03 |
| Ethylbenzene    | 5.87E−06    | 3.50E+00   | 1.68E−06 | 2.94E−05 | 4.20E−03 | 6.99E−03 |
| Xylene          | 5.17E−05    | 7.50E+00   | 6.89E−06 | 2.58E−04 | 3.30E−03 | 7.83E−02 |
| 2CP             | 7.92E−05    | 4.00E−02   | 1.98E−03 | 3.96E−04 | 6.29E−03 | 6.30E−02 |
| 2,4 DCP         | 6.84E−05    | 4.70E−02   | 1.45E−03 | 3.42E−04 | 7.00E−05 | 4.88E+00 |
| 2,4,6 TCP       | 9.29E−06    | 8.20E−01   | 1.13E−05 | 4.65E−05 | 5.73E−04 | 8.11E−02 |
| PCP             | 7.37E−07    | 2.70E−01   | 2.73E−05 | 3.68E−06 | 1.80E−05 | 2.05E−01 |
| Phenol          | 8.21E−05    | 3.17E−01   | 2.59E−04 | 4.11E−04 | 8.00E−05 | 5.13E+00 |
| Cypermethrin    | 3.62E−07    | 5.75E−02   | 6.29E−06 | 1.81E−06 | 8.20E−07 | 2.21E+00 |
| Deltamethrin    | 1.69E−07    | 9.36E−03   | 1.81E−05 | 8.46E−07 | 7.00E−07 | 1.21E+00 |
| Permethrin      | 4.19E−07    | 3.83E−01   | 1.10E−06 | 2.10E−06 | 1.00E−06 | 2.10E+00 |
| o-Cresols       | 3.39E−05    | 1.21E−01   | 2.80E−04 | 1.69E−06 | 8.40E−03 | 2.02E−02 |
| m-Cresol        | 1.95E−05    | 2.42E−01   | 8.07E−05 | 9.77E−05 | 5.00E−03 | 1.95E−02 |
| p-Cresol        | 2.63E−05    | 2.07E+01   | 1.27E−06 | 1.31E−04 | 3.88E−03 | 3.39E−02 |
| Ammonia         | 2.81E+00    | 7.90E−01   | 3.56E+00 | 1.40E+01 | 1.19E−02 | 1.18E+03 |
| Total Risk      | 3.79E+00    | 1.74E+03   | 2.43E+00 | 1.86E+00 | 2.36E+00 | 9.31E+00 |

Bold indicates the values that exceed the threshold limit.
Several studies reported that leachate-contaminated water exposure changes the ratio of internal organs such as brain, heart, kidney, and liver to the total body weight of small rodents. Likewise, the exposure increases the frequency of chromosomal aberrations (CA) and micronucleus (MN) formation in bone marrow, protein oxidation content (PCO) in viscera, number of sperms with morphological head defects, as well as DNA–protein crosslinks (DPC) formation in the brain, spleen, liver, and kidney of mice in a concentration-dependent manner. Thus, as it was already mentioned, the higher concentration of the contaminants in the contaminated water by MSWH leachate samples poses greater potential ecological risks to small mammals (Ghosh et al. 2017).

Based on the results of the studies conducted on evaluating leachate toxicity on fish species, the range of toxicity in MSW leachate samples varies from being confirmed (40 > EC50 > 10) to severe toxic (EC50 > 9), which is mainly related to the constituents of the MSW leachate samples such as ammonium ion, heavy metals, NaCl, and pyrethroid pesticides (Ghosh et al. 2017).

The results of the present study indicated that HQ of the contaminated water by MSWH is higher than the permissible limit (HQ ≥ 1) due to the existence of heavy metals (i.e., cadmium, mercury, nickel, copper, and lead), ammonia, pesticides, and phenolic compounds. In addition, the hazard quotients of phenolic compounds and pesticides are in the medium risk range, while heavy metals for aquatics display a greater risk. However, ammonia represents the greatest risk, which is predicted to be more than 1000 times than the permissible limit.

The HQ of MSWS leachate decreased by 36 and 41.5% for aquatic and terrestrial bioassays, respectively. The highest reduction was observed in pesticides, followed by ammonia, and heavy metals. After HHW diversion, the ecological risk of mercury and phenolic compounds for aquatic environment reduced to moderate and low risk, respectively. However, the risk of drinking contaminated water by MSWS leachate is still high for aquatic animals due to the presence of ammonia and some heavy metals, and accordingly a treatment process is required before discharging leachate to the environment.

Regarding the results of previous studies, MSW leachates increased mortality and caused a significant increase in the number of immature red blood cells, frequency of micronucleus (MN), and formation of binucleated cells in the fishes exposed to contaminated water in a dose-dependent manner (Ghosh et al. 2017). Therefore, implementing the source separation of HHW and decreasing the concentration of pollutants result in reducing the above-mentioned negative effects and toxicity of leachate.

The results of the present study confirmed that HHW diversion in the MSW stream can reduce the ecological risk, which is consistent with the results of Bakare et al. (2005).

**Conclusion**

The present study sought to evaluate the effect of HHW diversion on leachate quality and consequent reduction in risks to the environment and human health. Physicochemical analysis of leachate samples with (MSWH) and without (MSWS) household hazardous waste, were conducted to evaluate leachate pollution index (LPI) and associated risks.

Based on physicochemical analysis, a statistically significant difference (p-value < 0.05) in average concentrations of target pollutants was observed, except for Cu and PCP in MSWS and MSWH. Although the results indicated that both the LPI values were above the leachate discharge standard due to the higher concentration of pollutants, especially ammonia, the source separation of HHW fraction was able to considerably reduce the LPI value (13.48%), and improve the potential of leachate for biological treatment leading to lower treatment costs.

Considering the results of the human health risk assessment, HHW diversion reduced the HI values for non-carcinogenic health risks by 38.56% for all population groups. Furthermore, the separation of HHW from MSW stream decreases the carcinogenic risks of leachate-contaminated water consumption for arsenic and chromium for sensitive groups below the acceptable levels. Based on the results of the present study, the risk caused by leachate exposure in aquatic and terrestrial environments by separating HHW at the source can lead to a reduction of 41.5 and 36%, respectively.

In spite of the low percentage of HHW, a reduction in pollutant concentration up to 94% indicated that the effects of this kind of waste on leachate quality are inevitable. Therefore, an appropriate management system including HHW source separation, collection, and disposal should be implemented for improving leachate quality and decreasing risks to environment and human health.

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**Declarations**

**Conflict of interest** The authors declare that they have no conflict of interest.
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