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Task 12 PV Sustainability

PVPS

Human Health Risk Assessment Methods for PV
Part 3: Module Disposal Risks
2020

Report IEA-PVPS T12-16: 2020

What is IEA PVPS TCP?

The International Energy Agency (IEA), founded in 1974, is an autonomous body within the framework of the Organization for Economic Cooperation and Development (OECD). The Technology Collaboration Programme (TCP) was created with a belief that the future of energy security and sustainability starts with global collaboration. The programme is made up of 6.000 experts across government, academia, and industry dedicated to advancing common research and the application of specific energy technologies.

The IEA Photovoltaic Power Systems Programme (IEA PVPS) is one of the TCP's within the IEA and was established in 1993. The mission of the programme is to "enhance the international collaborative efforts which facilitate the role of photovoltaic solar energy as a cornerstone in the transition to sustainable energy systems." In order to achieve this, the Programme's participants have undertaken a variety of joint research projects in PV power systems applications. The overall programme is headed by an Executive Committee, comprised of one delegate from each country or organisation member, which designates distinct 'Tasks,' that may be research projects or activity areas.

The IEA PVPS participating countries are Australia, Austria, Belgium, Canada, Chile, China, Denmark, Finland, France, Germany, Israel, Italy, Japan, Korea, Malaysia, Mexico, Morocco, the Netherlands, Norway, Portugal, South Africa, Spain, Sweden, Switzerland, Thailand, Turkey, and the United States of America. The European Commission, Solar Power Europe, the Smart Electric Power Alliance (SEPA), the Solar Energy Industries Association and the Cop- per Alliance are also members.

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What is IEA PVPS Task 12?

Task 12 aims at fostering international collaboration in safety and sustainability that are crucial for assuring that PV grows to levels enabling it to make a major contribution to the needs of the member countries and the world. The overall objectives of Task 12 are to 1. Quantify the environmental profile of PV in comparison to other energy technologies; 2. Investigate end of life management options for PV systems as deployment increases and older systems are decommissioned; 3. Define and address environmental health & safety and other sustainability issues that are important for market growth. The first objective of this task is well served by life cycle assessments (LCAs) that describe the energy-, material-, and emission-flows in all the stages of the life of PV. The second objective is addressed through analysis of including recycling and other circular economy pathways. For the third objective, Task 12 develops methods to quantify risks and opportunities on topics of stakeholder interest. Task 12 is operated jointly by the National Renewable Energy Laboratory (NREL) and University of New South Wales (UNSW). Support from the United States Department of Energy (DOE) and UNSW are gratefully acknowledged.

This report addresses the third objective above by describing methods to assess potential health risks should a PV module be disposed in a non-sanitary landfill. This report is third of a three-part series on human health risk assessment methods for PV. Part 1 addressed potential health risks resulting from modules exposed to fire and Part 2 addressed potential health risks resulting from broken modules left in the field. These reports and further information on the activities and results of the Task can be found at: https://iea-pvps.org/research-tasks/pv-sustainability/.

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INTERNATIONAL ENERGY AGENCY
PHOTOVOLTAIC POWER SYSTEMS PROGRAMME

Human Health Risk Assessment Methods for PV
Part 3: Module Disposal Risks

IEA PVPS
Task 12: PV Sustainability

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cm – centimeter

days/yr – days per year

g/m^2 – grams per square meter

kg/m^2 – kilograms per square meter

kg/m^3 – kilograms per cubic meter

kg/acre/yr – kilograms per acre per year

kg-day/mg –kilogram-day per milligram

L/day – liters per day

L/yr – liters per year

m – meter

m/s – meters per second

m^2 – square meters

m^3 – cubic meters

m^3-year/kg-day – cubic meter-year per kilogram-day

mg/kg – milligrams per kilogram

mg/kg-day – milligrams per kilogram-day

mg/L – milligrams per liter

mg/m^3 – milligrams per cubic meter

mg/s – milligrams per second

MWac – megawatt alternating current

MWdc – megawatt direct current

W – watt

yr - year

ABSTRACT

End-of-life management is important for addressing large future photovoltaic (PV) waste volumes and conserving raw materials for use in new PV modules. In regions without regulatory mandates for PV recycling, end-of-life PV modules can be disposed in accordance with general waste laws. Given the use of various metals as raw materials in semiconductor compounds and electrical contacts in commercial PV modules, some stakeholders have raised concerns regarding potential environmental impacts if PV modules are subject to improper disposal instead of being recycled or disposed in sanitary landfills, as is required in most world regions. To evaluate these concerns, screening-level risk assessment methods are developed herein that evaluate potential human health risks from groundwater and surface (air, soil, surface water) exposure pathways. The methods estimate potential impacts from disposal of end-of-life (EOL) PV modules in non-sanitary landfills under the following worst-case conditions: no leachate collection or groundwater monitoring, no liner for preventing leachate migration, uncovered waste, and lack of stormwater management. Examining worst-case conditions allows the exploration of maximum potential risk to attempt to ensure disposal does not increase health risk above regulatory thresholds.

Specifically, this report presents an analysis of potential human health risks associated with non-sanitary landfill disposal for three PV technologies, focusing on release of the highest-prioritized chemical element for each: lead (Pb) in crystalline-silicon (c-Si) PV modules, cadmium (Cd) in thin film cadmium telluride (CdTe) PV modules, and selenium (Se) in thin film copper indium selenide (CIS) PV modules. The prioritization of these chemical elements for analysis is based on stakeholder interest. Because the methodology is chemical-specific, the risk assessment results for these chemicals cannot be directly generalized to other chemicals, although the risk assessment methodology can be applied to other chemicals. If the chemicals chosen are indeed the ones presenting greatest risk, then the results herein should represent the upper bound of health risk from exposure to a single constituent.

Under the layers of health-protective assumptions applied, for Pb for c-Si PV, Cd from CdTe PV and Se from CIS PV, cancer risks and non-cancer hazards are found to be at least one order of magnitude below the US regulatory screening thresholds of 1×10^{-6} cancer risk and non-cancer hazard quotient of 1. Health screening levels can differ by country or region; exposure-point concentrations in groundwater and surface water of Pb, Cd, and Se for c-Si, CdTe, and CIS PV, respectively, are also within water quality guidelines from the World Health Organization. The results presented herein do not represent a complete human health risk assessment for PV module disposal nor an assessment of cumulative risk, although the results are suggestive of low risk for the prioritized chemicals examined using best available regulatory methods. The screening-level methods employed in this report can be used in future work to assess potential health risks from other chemicals of potential concern and other PV technologies to establish a more complete set of results for chemicals of potential concern.

It is also important to note that examination of potential health risk from disposal of PV modules in landfills does not endorse this EOL management option. Indeed, recycling end-of-life PV modules would further mitigate environmental concerns, as found in several recent life cycle assessments.

EXECUTIVE SUMMARY

End-of-life management is important for addressing large future photovoltaic (PV) waste volumes and conserving raw materials for use in new PV modules. Recycling is expected to be a dominant strategy for sustainable end-of-life management and is both commercially underway and subject to further research and development activities globally. Recycling activities have been boosted by regulatory mandates such as the EU WEEE Directive, recycling standards such as CENELEC EN50625-2-4 and TS50625-3-5, project decommissioning requirements, and extended producer responsibility initiatives. In regions without regulatory mandates for PV recycling, end-of-life PV modules can be disposed in accordance with general waste laws.

Given the use of various metals (e.g., Ag, Al, Cd, Cu, Ga, In, Ni, Pb, Se, Sn, Te, Zn) as raw materials in semiconductor compounds and electrical contacts in commercial PV modules, some stakeholders have raised concerns regarding potential environmental impacts if PV modules are subject to improper disposal instead of being recycled or disposed in sanitary landfills, as is required in most world regions. To evaluate these concerns, screening-level risk assessment methods are developed herein that evaluate potential human health risks from groundwater and surface (air, soil, surface water) exposure pathways. The methods estimate potential impacts from disposal of end-of-life (EOL) PV modules in non-sanitary landfills under the following worst-case conditions: no leachate collection or groundwater monitoring, no liner for preventing leachate migration, uncovered waste, and lack of stormwater management.

It is important to note that these worst-case conditions are not legal in many world regions, and thus EOL PV modules are unlikely to be disposed of in this manner, especially in landfills where all these conditions simultaneously exist. Yet examining worst-case conditions allows the exploration of maximum potential risk to attempt to ensure disposal does not increase health risk above regulatory thresholds.

It is also important to note that examination of potential health risk from disposal of PV modules in landfills does not endorse this EOL management option. Where not already the law, there are emerging regulatory and voluntary efforts among manufacturers and others in the solar PV value chain to investigate and ensure that PV module and system components are recovered for valuable use. From a circular economy and resource efficiency perspective, disposal is the least preferred EOL option, particularly with respect to resource depletion as found in several recent life cycle assessments.

Fate and transport modelling can be conducted with the U.S. Environmental Protection Agency (USEPA) Delisting Risk Assessment Software (DRAS; V. 3.0) model to estimate chemical concentrations in soil, groundwater, surface water, and air. These exposure-point concentrations can be compared to health-protective screening levels based on 1×10^{-6} cancer risk and non-cancer hazard quotient of 1. The screening-level methods can be used to decide whether further evaluation of potential health risks is warranted. A few example scenarios demonstrate application of the methods.

Specifically, this report presents an analysis of potential human health risks associated with non-sanitary landfill disposal for three PV technologies, focusing on release of the highest-prioritized chemical element for each: lead (Pb) in crystalline-silicon (c-Si) PV modules, cadmium (Cd) in thin film cadmium telluride (CdTe) PV modules, and selenium (Se) in thin film copper indium selenide (CIS) PV modules. The prioritization of these chemical elements for analysis is based on stakeholder interest. Because the methodology is chemical-specific, the

risk assessment results for these chemicals cannot be directly generalized to other chemicals, although the risk assessment methodology can be applied to other chemicals. If the chemicals chosen are indeed the ones presenting greatest risk, then the results herein should represent the upper bound of health risk from exposure to a single constituent.

The proposed risk assessment method follows a screening-level approach, with the intent of developing order of magnitude-level estimates of potential risk after applying health-protective assumptions, consistent with general risk assessment approaches recommended by the U.S. National Academy of Sciences (NAS), U.S. Environmental Protection Agency (USEPA), and International Electrotechnical Commission (IEC).

The evaluation of Pb content in c-Si PV modules is based on current usage of Pb in metallization pastes and solders, which could be higher than future usage if Pb-free pastes and solders achieve greater market penetration. The evaluation of Cd content in CdTe PV modules and Se content in CIS PV modules is based on current usage in the semiconductor layer, which could be higher than future usage if semiconductor layer thickness is reduced. Dematerialization of these compounds is a trend observed in the marketplace today.

The release mechanisms considered in this evaluation are leaching from broken modules under acidic (fermentation) landfill conditions for the groundwater exposure pathway, and wind and stormwater erosion of uncovered landfill waste for the surface exposure pathway. The following potential transport pathways are quantitatively evaluated: leachate transport to groundwater and to a hypothetical downgradient receptor water well; particulate emissions to air from uncovered waste and subsequent deposition to soil; and rainwater erosion of uncovered waste and runoff to surface water.

Acidic (fermentation) conditions are required to be assumed over the life of the landfill according to the USEPA leachate testing procedure (USEPA Method 1311 Toxicity Characteristic Leaching Procedure (TCLP)) because acidic conditions can lead to a higher estimate of leaching potential which is therefore health protective. Yet it should be noted that landfills have predominantly neutral to alkaline (methanogenic) conditions over their lifetime, and such conditions can cause a lower leaching rate than that characterized by the TCLP test results. Leachate concentration as measured by application of the TCLP test procedure results from agitation of 1 cm samples in acidic solution. It should be noted that experiments show that crushing of PV modules by a landfill compactor leads to typical fragment size of larger than 1 cm, which would reduce leaching rate and thus leachate concentration as compared to that from fragments 1 cm in size (or smaller). In these ways, following the regulatory guidelines in this risk assessment leads to higher leaching potential than expected under real-world conditions.

The exposure scenario which is considered in this screening evaluation is a residential scenario in which adults and children who live in proximity to the non-sanitary landfill are assumed to experience potential exposures via surface pathways and groundwater pathways.

Potential Surface Pathways

- Direct inhalation of contaminated particles from wind erosion over the landfill surface
- Ingestion of fish living in water contaminated by landfill surface rainwater runoff
- Ingestion of drinking water from surface water sources contaminated by landfill surface rainwater runoff
- Ingestion of soils affected by airborne contaminated particles from wind erosion over the landfill surface

Potential Groundwater Pathways

- Ingestion of drinking water from groundwater sources contaminated by landfill leachate
- Dermal absorption via showering of groundwater sources contaminated by landfill leachate
- Inhalation of steam and mist via showering from groundwater sources contaminated by landfill leachate

The USEPA DRAS model follows a screening-level risk assessment approach, utilizing health-protective assumptions that likely overestimate risk. The DRAS model uses reasonable maximum exposure estimates for exposure factors for an offsite resident. The model uses Monte Carlo simulation which varies receptor groundwater well location to estimate a lower 90th percentile dilution-attenuation factor, after adjustment for landfill waste volume, for modeling the groundwater exposure pathway. For the surface exposure pathways, the DRAS model assumes the total release of chemicals in the waste. This assumption contrasts with PV module design wherein constituents are encapsulated in monolithic solid-state devices, thereby reducing potential emission rate compared to a total release assumption. For the surface exposure pathways, the DRAS model also assumes chemical release in elemental form. This assumption also contrasts with PV module design whereby the PV constituents modeled in this report are part of stable compounds and alloys. The surface exposure pathways also use conservative assumptions regarding the close proximity of exposure media to the landfill (100 m for rainwater runoff to surface water and 305 m for wind erosion to soil).

The following table summarizes health-protective assumptions in this methodology, including some examples of uncertainty quantification. In this report, “health-protective” refers to the use of conservative assumptions and methods in order to define a reasonable worst-case scenario.

Modeling assumption	Description	Example quantification of uncertainty
Non-sanitary disposal	Disposal of end-of-life PV modules in a non-sanitary landfill is assumed instead of recycling or disposal in a sanitary landfill. The non-sanitary landfill is assumed to have no leachate collection or groundwater monitoring, no liner for preventing leachate migration, uncovered waste, and lack of stormwater management.	Modern landfills are expected to be sanitary. For example, a survey of U.S. landfills found 97% had liners for preventing leachate migration.
Time duration of disposal	End-of-life PV modules are assumed to be disposed in 1 year into a single non-sanitary landfill, with the entire PV project decommissioned during that time period. This assumption leads to higher risks than if the same quantity of waste was disposed over a longer time period.	Increasing module efficiencies and declining cost per watt may lead to partial repowering of PV projects over the timeframe of their power purchase agreements, rather than a single decommissioning event.

Modeling assumption	Description	Example quantification of uncertainty
Waste volume	Waste volume is assumed to be equivalent to a 10MWac project, which is a 95% upper confidence limit on the mean capacity of operational large-scale solar facilities in the U.S.	The average capacity of operational large-scale solar facilities in the U.S is ~7.6 MWdc (or ~6 MWac). Large-scale facilities are typically owned by large investors and subject to decommissioning requirements, reducing the likelihood of non-sanitary disposal, yet we evaluate this worst case scenario to be health protective. The waste volumes associated with smaller PV installations (e.g., distributed rooftop solar facilities that are typically smaller than 1 MWac) are not considered here but would yield lower risk estimates than those used here.
Fermentation conditions	Acidic (fermentation) conditions are assumed over the life of the landfill, whereas landfills have predominantly neutral to alkaline (methanogenic) conditions over their lifetime. The fermentation conditions are characterized with use of the USEPA Method 1311 TCLP leaching procedure for estimating leachate concentrations, which requires agitation in acidic solution.	Leachate concentrations for Pb and Cd are lower under methanogenic conditions than for fermentation conditions, whereas leachate concentrations for Se are similar to or higher for methanogenic conditions than for fermentation conditions.
Particle size for leaching test	The TCLP leaching procedure requires 1 cm sample size, whereas experiments have found fragment size larger than 1 cm when modules are crushed using landfill equipment.	On average, three-quarters of fragments from experimental landfill crushing of PV modules are greater than 1 cm in size, and the front-back encapsulation is maintained.
Lower 90th percentile dilution-attenuation factor (DAF)	The USEPA DRAS model uses Monte Carlo simulation, varying receptor groundwater well location, to estimate a lower 90th percentile waste volume-adjusted dilution-attenuation factor (DAF) for modeling the groundwater exposure pathway.	The DAF varies exponentially with percentile, with the lower 90th percentile being health protective in comparison with the median.
Proximity to surface exposure media	For the surface exposure pathways, the USEPA DRAS model uses conservative assumptions regarding the close proximity of exposure media to the landfill: 100 m for rainwater runoff to surface water and 305 m for wind erosion to soil.	Based on a survey of U.S. landfills, only 3.6% are located within 1.6 km of a river or stream and the average distance from this subset of facilities to the closest river or stream is 586 m.

Modeling assumption	Description	Example quantification of uncertainty
Total chemical release for surface exposure pathways	Surface exposure pathways (surface water, soil, air) in the USEPA DRAS model assume total release of constituent chemicals, whereas PV module constituents are contained in a glass-encapsulant-glass or glass-encapsulant-backsheet structure. Furthermore, the surface exposure pathways assume chemical release in elemental form, whereas the PV constituents modeled in this report are part of stable compounds and alloys (Pb in SnPb solder for c-Si PV, Cd in CdTe for CdTe PV, Se in CIS for CIS PV) which are less likely to leach than elemental forms.	Because surface exposure pathways are based on soil erosion modeling, they overestimate dispersion of constituents from encapsulated solid-state devices, but have been utilized for upper bound risk screening.
Upper bound exposure factors	For all exposure pathways, the USEPA DRAS model uses reasonable maximum exposure (RME) estimates for exposure factors for an offsite resident.	The RME approach combines upper-bound and mid-range exposure factors so that the result represents an exposure scenario that is both health-protective and reasonable.

Potential health risks are evaluated in two ways. For Cd from CdTe PV and Se from CIS PV, cancer risks and non-cancer hazards are directly estimated by combining estimated exposure-point concentrations with human exposure factors and dose-response toxicological factors. For both of these cases under the layers of health-protective assumptions applied, cancer risks and non-cancer hazards are found to be several orders of magnitude below the screening thresholds of 1×10^{-6} cancer risk and non-cancer hazard quotient of 1 (Table 6). For Pb from c-Si PV, cancer risks and non-cancer hazards cannot be directly estimated with the USEPA DRAS model, because dose-response toxicological values are not available for Pb. Human health effects associated with potential exposure to Pb are typically based on toxicokinetic modeling of Pb concentrations in blood. As this is beyond the scope of the DRAS model, the potential for human health effects associated with exposure to Pb is assessed by back-calculating Pb concentrations in exposure media and comparing them to risk-based screening levels.

Under the health-protective assumptions applied herein, exposure point concentrations of Pb for c-Si PV are at least one order of magnitude below USEPA health screening values in soil, air, and water (Table 9, Figure B1). Exposure-point concentrations of Cd for CdTe PV and Se for CIS PV are several orders of magnitude below USEPA health screening values in soil, air, and water (Table B1, Figure B1). Health screening levels can differ by country or region; exposure-point concentrations in groundwater and surface water of Pb, Cd, and Se for c-Si, CdTe, and CIS PV, respectively, are also within water quality guidelines from the World Health Organization. The conclusions regarding risks and hazards in this study are consistent with results from previous studies which utilized even more aggressive but less realistic modeling assumptions (waste volumes, leaching rates, dilution-attenuation factors).

In this report, three chemicals (Pb, Cd, Se) have been used to demonstrate a human health risk assessment methodology for non-sanitary landfill disposal of three types of PV modules. Commercially available PV modules also use other environmentally sensitive elements (e.g., In, Ag, Cu, Se, Sn, Ni) and new PV technologies (e.g., perovskite) are emerging. Thus, the results presented here do not represent a complete human health risk assessment for PV

module disposal nor an assessment of cumulative risk, although the results are suggestive of low risk for the prioritized chemicals examined using best available regulatory methods.

Recycling end-of-life PV modules would further mitigate environmental concerns. Life cycle assessments of PV module recycling by IEA PVPS and Fraunhofer ISE indicate net environmental benefits from recycling, particularly with respect to resource depletion.

The screening-level methods employed in this report can be used in future work to assess potential health risks from other chemicals of potential concern and other PV technologies to establish a more complete set of results for chemicals of potential concern. Potential ecological risks have not been evaluated in this report. Screening-level methods such as used here are meant to identify potential health risk scenarios that are greater than defined thresholds and may warrant further analysis, though none are found in our preliminary application of the methods to selected constituents.

1 INTRODUCTION

Given past and projected large-scale global deployment of PV modules, product end-of-life management is important to managing large future waste volumes [1] and conserving raw materials for use in new PV modules [2]. Recycling is expected to be a dominant strategy for sustainable end-of-life management and is both commercially underway [3] and subject to further research and development activities globally [4]. Recycling activities have been boosted by regulatory mandates such as the EU WEEE Directive [5], recycling standards (CENELEC EN50625-2-4 and TS50625-3-5), project decommissioning requirements [6], and extended producer responsibility initiatives [7][8][9].

However, given the use of various metals (e.g., Ag, Al, Cd, Cu, Ga, In, Ni, Pb, Se, Sn, Te, Zn; [1]) as raw materials in semiconductor compounds and electrical contacts in commercial PV modules, some stakeholders have raised concerns regarding potential environmental impacts if PV modules are subject to improper disposal instead of being recycled or disposed in sanitary landfills. Use of engineering controls (daily cover, stormwater management, landfill liner, leachate collection, and groundwater monitoring) limits potential emissions from what are known as sanitary landfills [10].

The above constituents of potential concern are encapsulated in PV modules which are constructed as solid-state monolithic devices to achieve long-term field durability to withstand harsh environmental conditions for 25 years or more. Encapsulation of the module components is achieved with use of a polymer laminate material (e.g., ethylene vinyl acetate or polyolefin) in a glass-encapsulant-backsheet or glass-encapsulant-glass sandwich design (Figure 1). While crystalline silicon (c-Si) PV modules are primarily constructed with glass-backsheet design and thin film modules are constructed with glass-glass design, glass-glass c-Si PV modules are now also being deployed. The encapsulant bond strength is on the order of 5 megapascals (~50 kg/cm^2) making the modules very difficult to break open (i.e., to separate the front and back of the module). For example, this high encapsulant bond strength is the reason why efficient delamination is a core challenge for recyclers attempting to reverse engineer an end-of-life PV module into its raw materials [11].

The high encapsulant bond strength also limits the potential for end-of-life leaching from landfilled PV modules by maintaining the module metal constituents inside the glass-glass or glass-backsheet structure. For example, in a landfill experiment, PV modules were crushed with six passes by a landfill compactor with a contact load of 50 tons, and the crushed module pieces maintained the front-back encapsulation, with typical fragment size larger than 1 cm [12].

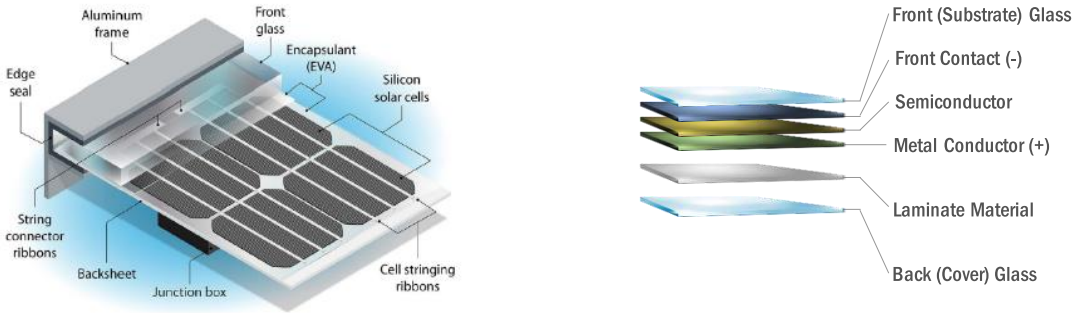

Figure 1: c-Si (left) [1] and thin film (right) [13] PV module components

With regard to evaluating human health risks from PV module disposal, a paradigm for human health risk assessment was first developed by the United States (U.S.) National Academy of Sciences [14] and consists of four main steps:

1. hazard identification,
2. dose-response assessment,
3. exposure analysis, and
4. risk characterization.

With regard to hazard identification, while a variety of chemicals (asphyxiant, corrosive, irritating, flammable or explosive, hazardous) are used in the manufacturing of PV modules [15], the highest-prioritized chemical element from three PV technologies are evaluated in this report as illustrative examples of the application of the methods developed herein (section 2.2). Dose-response assessment is incorporated in the use of toxicological and health screening values (section 5). Exposure analysis includes the characterization of chemical emissions (section 3) and environmental fate of emissions (section 4). Risk characterization and associated uncertainties are covered in sections 5 and 6, respectively.

Screening-level human health risk assessment methods, based on the above paradigm but specific to PV, have been outlined by the International Electrotechnical Commission [15]. These PV-specific methods are followed and expanded upon in this report using example cases of evaluating potential human health risks from a single, prioritized chemical from each of three PV technologies: Pb content in c-Si PV, Cd content in thin film CdTe PV, and Se content in thin film CIS PV.

The evaluation of Pb content in c-Si PV modules and Cd content in CdTe PV modules are based on current usage as described above. Increased market penetration of Pb-free pastes and solders has been forecasted [16] which would reduce risks estimated in the c-Si PV case study. Future reduction in semiconductor layer thickness has been identified as part of a resource efficiency strategy for thin film PV [13], which would reduce risks estimated in the CdTe and CIS PV case studies.

2 CONCEPTUAL SITE MODEL

2.1 Overview

Figure 2 summarizes the screening level methodology for evaluating potential human health risks from PV module disposal. The exposure scenario considered in this evaluation is:

- an off-site residence in proximity to a landfill

The exposure scenario is based on disposal of end-of-life PV modules in a non-sanitary landfill. Potential leachate formation in the landfill is then followed by potential chemical migration from leachate to groundwater. Potential erosion of uncovered waste by wind and rainwater is then followed by potential chemical migration to surface media (surface water, soil, and air). In this evaluation, the term non-sanitary specifically refers to the lack of specific (and typically required) landfill engineering controls: no leachate collection or groundwater monitoring, no liner for preventing leachate migration, waste that is uncovered at the surface, and lack of site stormwater management.

Potential exposure pathways and potentially exposed receptor populations are described below in sections 2.3 and 2.4. The screening methodology for estimating potential leachate concentrations and chemical fate and transport from point of emissions to point of exposure

are presented in sections 3 and 4. Cancer risks and non-cancer hazards are estimated in section 5 along with comparisons of exposure point concentrations to risk-based screening levels. Uncertainties in key parameters are evaluated in section 6.

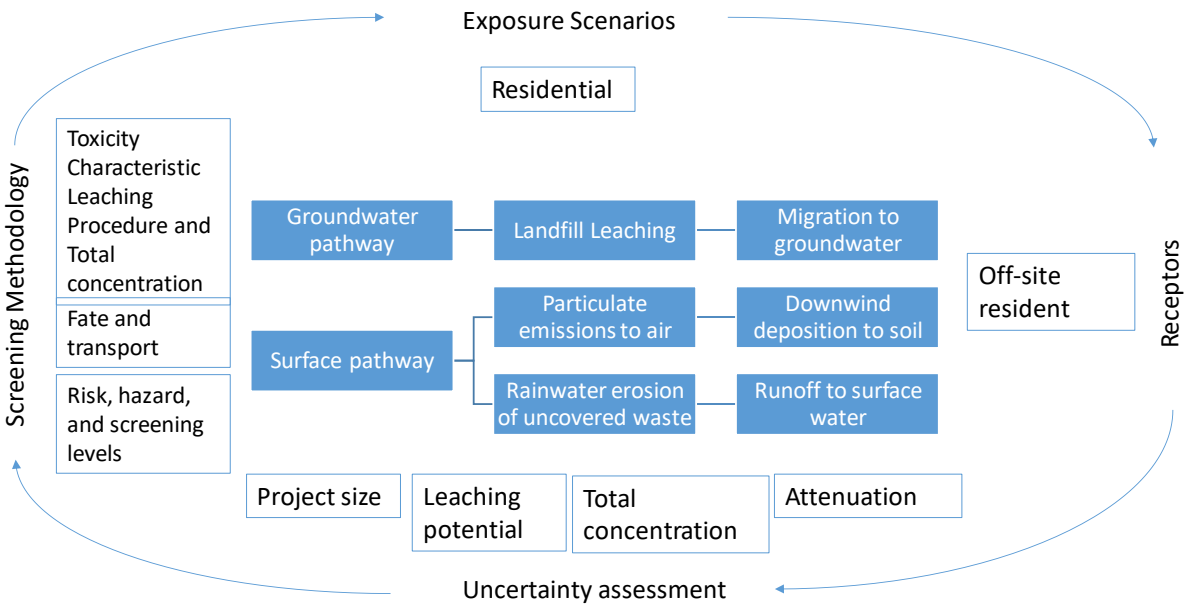

Figure 2: Conceptual site model for evaluating potential human health risks from non-sanitary landfill disposal of PV modules. Potential exposure pathways are shown in blue.

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2.2 Chemicals Included in Evaluation

The chemicals which are included in this evaluation are Pb content in c-Si PV, Cd content in CdTe PV, and Se content in CIS PV. In addition to these chemicals of potential concern, other metals used as raw materials for semiconductor compounds and electrical contacts in commercial PV modules include Ag, Al, Cu, Ga, In, Ni, Se, Sn, Te, and Zn [1]. While the methods developed in this report are not applied to these latter metals, they could be evaluated in future research. The metals chosen were selected by the authors as the ones with highest likelihood of presenting potential human health risk based on prior research and stakeholder interest.

2.3 Potential Release Mechanisms and Transport Pathways

The release mechanisms considered in this evaluation are leaching under acidic landfill conditions and wind and stormwater erosion of uncovered end-of-life PV modules disposed in a non-sanitary landfill. For these release mechanisms, it is assumed that chemicals are transported via groundwater pathways and surface pathways. For the groundwater pathway, it is assumed that PV modules are disposed in a non-sanitary landfill operating under acidic (fermentation) conditions for the life of the landfill. The surface pathways are based on assumed airborne particulate emissions resulting from wind erosion of uncovered landfill soil-

waste surfaces and from vehicular traffic over the waste as well as waste loading and unloading. Downwind deposition of airborne particulates is assumed to impact off-site soil. The surface pathways also consider rainwater erosion of uncovered waste and runoff to surface water.

Other potential end-of-life scenarios include incineration and disposal in a sanitary landfill. Incineration has not been considered here though has been evaluated by Raugei et al. [17], with higher potential emissions to air than landfill disposal. However, incineration may be a less likely disposal route due to the relatively low combustible content of PV modules (Figure 1). Sanitary landfills are engineered to prevent migration of landfill leachate outside of the landfill through liners, leachate collection systems, and groundwater monitoring as well as use of daily covers and stormwater management. With the use of these engineering controls, measured emissions to soil and to water from electronics in sanitary landfills are low [18], which is likely analogous to PV modules in sanitary landfills but needs to be verified.

2.4 Potentially Exposed Populations and Their Exposure Pathways

The human populations with potential to experience exposures to product-related chemicals, and that are considered in this quantitative evaluation, are described below in the residential scenario. We do not consider other potentially exposed populations such as adjacent commercial office building workers because they have shorter exposure frequency and duration and fewer exposure pathways than a resident. Therefore, the residential scenario should be health-protective of a commercial office building scenario.

Residential Scenario

Adult and child residents who live in proximity to the non-sanitary landfill (“off-site residents”) are assumed to experience exposures via potential surface pathways and groundwater pathways.

Potential Surface Pathways

- Direct inhalation of contaminated particles from wind erosion over the landfill surface
- Ingestion of fish living in water contaminated by landfill surface rainwater runoff
- Ingestion of drinking water from surface water sources contaminated by landfill surface rainwater runoff
- Ingestion of soils affected by airborne contaminated particles from wind erosion over the landfill surface

Potential Groundwater Pathways

- Ingestion of drinking water from groundwater sources contaminated by landfill leachate
- Dermal absorption via showering of groundwater sources contaminated by landfill leachate
- Inhalation of steam and mist via showering from groundwater sources contaminated by landfill leachate

3 LEACHATE CONCENTRATIONS

The determination of chemical leachate concentrations that may be released from PV modules in a landfill is based on the results of laboratory testing using regulatory methods. The average chemical concentrations measured in simulated landfill leachate, as summarized in Table 1,

are used to represent chemical concentrations in landfill leachate from end-of-life PV modules as an input for the fate and transport modeling described in Section 4.

As shown in Table 1, there are several regulatory leaching test methods which vary by geography. In the U.S., the USEPA Method 1311 Toxicity Characteristic Leaching Procedure (TCLP) measures the mobility of analytes in simulated landfill conditions (Table 1). While the TCLP test assumes the landfill is operating under acidic (fermentation) conditions, in actuality, landfills have predominantly neutral to alkaline (methanogenic) conditions over their lifetime [19]. Therefore, the waste characterization tests in Germany and Japan utilize distilled water instead of organic acid as the solvent. The results from the TCLP test are used in this report because the fate and transport methodology utilized in this study assumes the use of TCLP testing (fermentation conditions) to estimate leaching potential. As a result of using the TCLP leaching test results, the estimate of health risk from c-Si and CdTe PV modules will be higher than if the German or Japanese test results were used, whereas the results for Se from CIS modules will be lower through use of the TCLP results.

c-Si PV TCLP samples were prepared using water-jet cutting [20], a cutting method similar to coring that provides precision and field representativeness in sample preparation [24]. CdTe PV TCLP samples were obtained with both water-jet cutting [20] and from experimental landfill compactor crushing [21]. CIS PV TCLP samples were obtained with an unspecified cutting method [22]. Some previous non-standard leaching tests have utilized finely ground samples and/or extended extraction cycles, which can provide data on the total quantity of metals in a sample, but not their leaching potential under realistic landfill conditions [20]. Uncertainties related to leaching tests are further discussed in section 6.

Table 1: Summary of regulatory waste characterization leaching tests for c-Si, CdTe, and CIS PV modules

Geography	United States [20][21][22]	Germany [20][22]	Japan [23]
Leaching Test	U.S. EPA Method 1311 (TCLP)	DIN EN 12457-4:01-03	MOE Notice 13/JIS K 0102:2013 (JLT-13)
Sample size (cm)	1	1	0.5
Sample preparation	Water-jet cutting for c-Si PV; water-jet cutting and landfill experimental crushing for CdTe PV; unspecified cutting method for CIS PV	Water-jet cutting for CdTe PV; unspecified cutting method for CIS PV	Hammering
Solvent	Sodium acetate/ acetic acid (pH 2.88 for alkaline waste; pH 4.93 for neutral to acidic waste)	Distilled water	Distilled water
Liquid:Solid Ratio	20:1	10:1	10:1
Treatment Method	End-over-end agitation (30±2 rotations per minute)	End-over-end agitation (5 rotations per minute)	End-over-end agitation (200 rotations per minute)
Test Temperature	23±2˚C	20˚C	20˚C
Test Duration	18±2 hr	24 hr	6 hr
Leachate Pb Concentration for c-Si PV (mg/L)	5.5±2.8 (n=6)	-	Non-detect (<0.01) - 0.90 (n=63)
Leachate Cd Concentration for CdTe PV (mg/L)	0.15±0.09 (n=3)	0.0016 - 0.0040 (n=3)	0.10-0.13 (n=3)
Leachate Se Concentration for CIS PV (mg/L)	0.06±0.04 (n=4)	0.45±0.21 (n=5)	Non-detect (<0.01) - 1.1 (n=9)

4 EXPOSURE ASSESSMENT

4.1 Fate and Transport

Fate and transport modeling refers to the mathematical representation of the movement of chemicals through a medium and from one medium to another. In the context of exposure assessment, fate and transport modeling is used to estimate concentrations of chemicals in a medium to which humans are exposed, given concentrations of chemicals in an impacted medium.

The mechanisms by which chemicals are assumed to be released from PV modules are by leaching under acidic (fermentation) conditions and wind and stormwater erosion of uncovered waste for end-of-life PV modules disposed in a non-sanitary landfill. The chemical release mechanisms, and the subsequent chemical transport to groundwater, air, soil, and surface water, assume one calendar year as the time period of project decommissioning and disposal of PV modules.

Fate and transport modelling is conducted with U.S. Environmental Protection Agency (USEPA) Delisting Risk Assessment Software (DRAS V. 3.0) [25]. The DRAS software utilizes several constituent models, including the USEPA Composite Model for Leachate Migration with Transformation Products (CMTP), the Universal Soil Loss Equation (USLE), and the Ambient Air Dispersion Model (AADM). The DRAS model estimates potential human health risks associated with waste disposed in a non-sanitary landfill. Based on TCLP data, total constituent concentration in waste, and estimated waste volume entered by the user, DRAS assesses the carcinogenic risk and non-carcinogenic hazards associated with landfilled material.

In the previous two IEA PVPS human health risk assessment methodology reports for PV (Part 1: Fire Risks; Part 2: Breakage Risks), constituent fate and transport to exposure media (air, groundwater, soil) was explicitly derived in a series of report tables. In contrast, in this report, the fate and transport derivations are internal to the DRAS model and so are not shown explicitly. Section 2 of the DRAS technical support document [25] provides the full set of equations for constituent fate and transport, with the key equations summarized in Table 2.

Table 2. Key fate and transport equations used in the USEPA DRAS model [25]

	Fate and Transport to Ground Water		
DRAS Equation 2-1	$C_{gw} = \frac{C_L}{DAF}$	C_{gw}	concentration of waste constituent in groundwater (mg/L)
		C_L	leachate concentration (TCLP; mg/L)
		DAF	dilution-attenuation factor (unitless)
	Fate and Transport to Surface Water		
DRAS Equation 2-52	$C_{sw} = A * \frac{A_w}{Q_{stream}} * C_{total\ waste}$	C_{sw}	concentration of waste constituent in surface water (mg/L)
		A	area of the waste management unit (acres)
		A_w	rate of waste erosion from landfill (kg/acre/yr)
		Q_{stream}	volume of stream (L/yr)
		$C_{total\ waste}$	waste constituent concentration in landfilled waste (mg/kg)
	Fate and Transport to Ambient Air		
DRAS Equation 2-25	$C_{avg} = \frac{2.03 * Q_p}{\sum_z * U * L_v} * F$	C_{avg}	downwind concentration of waste constituent at point of exposure (mg/m^3)
		Q_p	emission rate of waste constituent particles (mg/s)
		$\sum_z$	vertical dispersion coefficient (m)
		U	mean annual wind speed (m/s)
		L_v	distance from the virtual point to the compliance point located 305 m downwind (m)
		F	frequency that wind blows from the sector of interest (unitless)
	Fate and Transport to Soil		
DRAS Equation 2-32	$C_{soil} = \left(\frac{q_d}{\rho_b * t}\right) * 3.154 * 10^7 \frac{s}{yr} * 1\ yr$	C_{soil}	concentration of constituent in soil at point of exposure (mg/kg)
		q_d	rate of deposition in mg/m/s
		ρ_b	soil bulk density (kg/m^3)
		t	soil thickness from which particles can be ingested (m)

The analysis in this report considers a one-time disposal of a waste volume equivalent to a 10MWac project, which is a 95% upper confidence limit (UCL) on the mean capacity of operational large-scale solar facilities in the U.S. [26]. The UCL was derived with USEPA ProUCL (V. 5) software [27], which estimates statistical confidence limits after assessing the data distribution type (normal, lognormal, gamma, or non-parametric). In this case, the UCL derivation was based on a non-parametric distribution using the Chebyshev statistical test [27].

The following specific potential transport pathways are quantitatively evaluated:

- transport to groundwater and to a hypothetical downgradient receptor well;
- particulate emissions to air from uncovered waste and subsequent deposition to soil
- rainwater erosion of uncovered waste and runoff to surface water

Releases to groundwater are modeled in DRAS by simulating one-dimensional, vertically downward flow and transport of contaminants in the unsaturated zone beneath a waste disposal unit, and multi-dimensional groundwater flow and contaminant transport in the underlying saturated zone.

Airborne releases to surface pathways are modeled in DRAS by simulating particulate emissions from uncovered waste due to wind erosion and vehicle travel for waste loading and unloading operations. Downwind dispersion modeling of particulate emissions to air is based on steady-state, Gaussian plume dispersion, accounting for deposition to soil at a hypothetical point of exposure located 305 m downwind of the landfill.

Waterborne releases to surface pathways are modeled in DRAS by simulating rainwater erosion of uncovered waste and runoff to a surface water body located 100 m from the landfill. Modeling of chemical concentrations in surface water accounts for partitioning between dissolved and suspended sediments. Bioaccumulation factors are used to model chemical concentrations in fish tissue from dissolved phase concentrations in surface water.

Major input parameters for the DRAS model runs are described in Table 3, including corresponding tables in this report where the input values are derived.

Table 3: Input parameters to the USEPA DRAS model

Input parameter	Description	Note
Waste Management Unit Type and Volume	The "landfill" option was selected for all model runs. Waste volume was set at 400 m^3 for a 10 MWac project.	See Table 4
Waste Management Unit Active Life	The "one year batch" option was selected for all model runs as it is assumed that the entire 10 MWac project will be decommissioned during one calendar year.	Disposal of all waste in a single year is the most conservative time duration, whereas disposal over 20 years is the default time duration in the DRAS model [25].
TCLP Concentration	The mean TCLP concentrations of 5.5 mg Pb/L for c-Si PV, 0.15 mg Cd/L for CdTe PV, and 0.06 mg Se/L were utilized.	See Table 1.
Total Concentration	The total Pb content in c-Si PV, Cd content in CdTe PV, and Se content in CIS PV (650, 500, and 300 mg/kg, respectively) were utilized.	See Table 5.
Chemical-specific properties	Default physical and chemical properties for Pb, Cd, and Se in the DRAS model were utilized for this evaluation.	[25]

Table 4: PV module waste volume derivation

	c-Si PV	CdTe PV	CIS PV	Notes
Project Capacity (MWac)	10	10	10	95% UCL of mean capacity of U.S. operational large-scale solar facilities [26][27]
DC:AC ratio	1.2	1.2	1.2	mode of DC:AC ratios [26]
Project Capacity (MWdc)	12	12	12	MWac × DC:AC ratio
Module efficiency	17.50%	17.50%	15%	[28-30]
Module area (m^2)	1.6	2.47	1.2	[29-31]
Module power (W)	280	432	180	Module area × Module efficiency × 1000 W/m2
# of modules	42,857	27,762	66,667	Project Capacity ÷ Module power
Module laminate thickness (m)	0.005	0.005	0.004	[30-32]
Frame weight (kg/m^2)	2.13	2.13	2.13	[31]
Module laminate waste volume (m^3)	343	343	320	Module area × Module laminate thickness
Frame composition	Aluminum	Aluminum	Aluminum	[31]
Frame density (kg/m^3)	2700	2700	2700	[33]
Frame waste volume (m^3)	54	54	63	Module area × Frame weight ÷ Frame density
Total waste volume (m^3)	397	397	383	Module laminate waste volume + Frame waste volume
Total waste volume (m^3)	400	400	400	Rounded

Table 5: Derivation of chemical concentration in waste

	Pb content in c-Si PV	Cd content in CdTe PV	Se content in CIS PV	Notes
Chemical content (g/m^2)	8.125	8.3 g	4.99	[34-36]
Module mass (kg/m^2)	12.5	16.6	16.6	[31][35-36]
Chemical concentration (mg/kg)	650	500	300	Chemical content ÷ Module mass

4.2 Off-site Resident Scenario

Table 6 describes exposure factors for an adult and child off-site resident in the DRAS model for the surface pathways and groundwater pathways. These factors are used to estimate chemical intake by the off-site resident. Chemical intake in conjunction with toxicological factors (section 5) are used to estimate cancer risk and non-cancer hazard from chemical exposure.

The exposure factors in Table 6 are USEPA reasonable maximum exposure (RME) estimates for each applicable exposure route. The RME approach is intended by USEPA to combine upper-bound and mid-range exposure factors so that the result represents an exposure scenario that is both protective and reasonable [25].

Child residents (1 to 6 years old) were evaluated for two exposure pathways: (1) the dermal absorption while showering with groundwater pathway and (2) the ingestion of soil contaminated with air particulate pathway. Child residents were not selected as receptors for the other exposure pathways because the adult resident receptor scenario has been determined by USEPA to be protective of children with regard to these pathways [25].

Table 6: Exposure factors for off-site resident [25]

Path-way	Exposure Route	Receptor	Fate and Transport Mechanism	Exposure Factors
Surface Pathway	Inhalation of particles	Adult Resident	Landfill → Total waste concentration → Air dispersion model → Inhalation	IFA_{adj} = 10.7 [m^3 -year]/[kg-days]; ED = 30 years; EF = 350 days/year; AT = 30 years
	Ingestion of fish	Adult Resident	Landfill → Total waste concentration → Uncovered waste erosion → Surface water → Fish tissue	CR_{fish} = 0.02 kg/day; EF = 350 days/yr; ED = 30 yrs; BW = 72 kg; AT = 30
	Ingestion of drinking water	Adult Resident	Landfill → Total waste concentration → Uncovered waste erosion → Surface water	CR_{water} = 2.0 L/day; EF = 350 days/yr; ED = 30 yrs; BW = 72 kg; AT = 30 yrs
	Ingestion of soil	Child Resident	Landfill → Total waste concentration → Air dispersion model → Ingestion	CR_{soil} = 200 mg/day; ED = 6 years; EF = 350 days/year; BW_{child} = 15 kg; AT = 6 years
Groundwater Pathway	Ingestion of drinking water	Adult Resident	Landfill → Results of TCLP test → Groundwater dispersion model → Downgradient well	CR_{water} = 2.0 L/day; EF = 350 days/yr; ED = 30 yrs; BW = 72 kg; AT = 30 yrs
	Dermal absorption via showering	Adult Resident	Landfill → Results of TCLP test → Groundwater dispersion model → Downgradient well → Shower model	EV = 1 events/day ; EF = 350 days/yr; ED = 30 yrs; A_{skin} = 20,000 $cm^{2;}$ AT = 30 yrs; BW = 72 kg
	Inhalation via showering	Adult Resident	Landfill → Results of TCLP test → Groundwater dispersion model → Downgradient well → Shower model	EV = 1 showers/day; EF = 350 days/yrs; ET_{shower} = 0.00792 day/shower; $ET_{bathroom}$ = 0.0338 days/shower; ET_{house} = 0.625 days/shower; ED = 30 yrs; IR = 20 m^3/day; AT = 30 yrs; BW = 72 kg
	Dermal absorption via showering	Child Resident	Landfill → Results of TCLP test → Groundwater dispersion model → Downgradient well → Shower model	EV = 1 events/day ; EF = 350 days/year; ED = 6 yrs; A_{skin} = 7,900 $cm^{2;}$ AT = 6 yrs; BW = 15 kg

Notes - A: area; AT: averaging time; BW: body weight; CR: consumption rate; ED: exposure duration; EF: exposure frequency; ET: exposure time; EV: event frequency; IFA_{adj}: inhalation factor, age-adjusted; IR: inhalation rate.

5 RISK, HAZARD, AND COMPARISON TO RISK-BASED SCREENING LEVELS

5.1 Dose-response

Table 7 summarizes the dose-response toxicological values used to estimate cancer risk and non-cancer hazard for the specific exposure scenarios in section 4. They are obtained from the USEPA Integrated Risk Information System (IRIS) database [37] and the USEPA health effects assessment summary tables (HEAST) [38] in that order of preference, and utilized in the DRAS model. Toxicological values from IRIS are preferred because they are more current than HEAST. Pb and Cd are characterized as probable human carcinogens and Se is not classifiable as to human carcinogenicity based on inadequate human data and inadequate evidence of carcinogenicity in animals [37].

Table 7: Dose-response toxicological values [37-38]

Chemical	Non-cancer oral reference dose (mg/kg-day)	Non-cancer inhalation reference dose (mg/m^3)	Oral Cancer slope factor (kg-day/mg)	Inhalation Cancer slope factor (kg-day/mg)[a]
Pb	Not available	Not available	Not available	Not available
Cd	5×10^{-4}	Not applicable	Not applicable	6.3
Se	5×10^{-3}	Not applicable	Not applicable	Not applicable

a – Inhalation cancer slope factor is only available for Cd so inhalation cancer risks are only estimated for Cd in Table 8.

In the case of Pb in Table 7, dose-response toxicological values are not available because human health effects associated with potential exposure to Pb are typically based on toxicokinetic modelling of Pb concentrations in blood [25]. As this is beyond the scope of the DRAS model, the potential for human health effects associated with exposure to Pb are assessed by comparing Pb concentrations in exposure media to risk-based screening levels (section 5.4). Also in Table 7, values listed as not applicable refer to exposure routes where there is insufficient toxicological evidence for quantifiable cancer risk or non-cancer hazard for a given chemical.

5.2 Risk and Hazard

Based on exposure assessment (section 4) and dose-response assessment (section 5.1), the DRAS model estimates potential cancer risks and non-cancer hazards for the off-site resident (Table 8). Total cancer risks and non-cancer hazards across the exposure scenarios are several orders of magnitude below regulatory screening thresholds (1×10^{-6} and 1, respectively) [39]. Therefore, the one-time disposal of PV modules from a 10 MWac project to a non-sanitary landfill is not likely to represent significant cancer risks or non-cancer hazards for the specific cases considered in this report (Cd from CdTe PV and Se from CIS PV) even under the

conservative assumptions and worst-case scenario evaluated. For Pb from c-Si PV, the potential for human health effects is assessed in section 5.4 due to the lack of dose-response toxicological values, as discussed in section 5.1.

For non-cancer hazards, the groundwater exposure pathway contributes to approximately 80-90% of the total hazard, primarily related to ingestion of groundwater as drinking water (Table 8). For cancer risks, the surface exposure pathway (inhalation of particles) is the only applicable pathway. Inhalation via showering in the groundwater exposure pathway is not applicable because the chemicals are non-volatile.

Table 8 Potential cancer risks and non-cancer hazards based on USEPA DRAS model output for an off-site resident exposed to PV module constituents from a 10 MWac project disposed in a non-sanitary landfill.

Path-way	Exposure Route	Receptor	Cancer Risk (unitless)			Non-cancer hazard (unitless)		
			Pb for c-Si PV*	Cd for CdTe PV	Se for CIS PV	Pb for c-Si PV*	Cd for CdTe PV	Se for CIS PV
Surface Pathway	Inhalation of particles	Adult Resident	---	2.45×10^{-9}	---	---	---	---
	Ingestion of fish	Adult Resident	---	---	---	---	4.38×10^{-4}	3.74×10^{-6}
	Ingestion of drinking water	Adult Resident	---	---	---	---	4.69×10^{-7}	2.82×10^{-8}
	Ingestion of soil	Child Resident	---	---	---	---	1.02×10^{-5}	6.14×10^{-7}
Groundwater Pathway	Ingestion of drinking water	Adult Resident	---	---	---	---	1.68×10^{-3}	6.80×10^{-5}
	Dermal absorption via showering	Adult Resident	---	---	---	---	4.20×10^{-6}	1.70×10^{-7}
	Inhalation via showering	Adult Resident	---	---	---	---	---	---
	Dermal absorption via showering	Child Resident	---	---	---	---	1.05×10^{-5}	4.25×10^{-7}
Total			---	2.45×10^{-9}	---	---	2.14×10^{-3}	7.30×10^{-5}
Human Health Screening Threshold			1×10^{-6}	1×10^{-6}	1×10^{-6}	1	1	1

* For Pb from c-Si PV, the potential for human health effects is assessed in section 5.4 due to the lack of dose-response toxicological values.

5.3 Comparison of Pb Exposure-Point Concentrations to Risk-based Screening Levels

Whereas cancer risks and non-cancer hazards could not be estimated for Pb using the DRAS model (see section 5.1), exposure point concentrations can be used for screening level human health risk assessment for Pb. The health risks of potential Pb exposure are evaluated through comparison of exposure point concentrations in air, groundwater, surface water, and soil to risk-based screening levels in Table 9. The exposure point concentrations for Pb are back-calculated from DRAS model outputs in Appendix A (Table A2). The screening levels are USEPA residential risk-based screening levels in soil, air, and water and USEPA maximum contaminant levels (MCLs), which are legal standards that apply to public water systems [40]. The screening levels are based on physical, chemical, and toxicological properties and default residential exposure assumptions as documented in USEPA risk assessment guidance [40-44]. The screening levels account for chronic exposure to chemicals and are intended to be protective of both cancer and non-cancer health endpoints.

Based on the scenarios assessed and numerous health protective assumptions made herein, we find that exposure-point concentrations of Pb for c-Si PV are at least one order of magnitude below risk-based screening levels and maximum contaminant levels, with groundwater having the highest exposure point concentration relative to risk-based screening levels.

For transparency and completeness, the screening-level risk assessment method used for Pb in this section is also applied to Cd and Se in Appendix B, with results summarized graphically for all three constituents (Figure B1). However, the direct estimation of cancer risk and non-cancer hazard for Cd and Se in section 5.2 is preferred over the screening-level risk assessment method in Appendix B, because it provides more specific quantification of potential risks and hazards and is a direct output of the regulatory-approved DRAS model.

Table 9. Comparison of Pb exposure point concentrations to residential risk-based screening levels and maximum contaminant levels [40] for PV modules from a 10 MWac project disposed in a non-sanitary landfill.

Groundwater			
Chemical	Exposure Point Concentration (mg/L)	Risk-based Screening Level (mg/L)	Maximum Contaminant Level (mg/L)
Pb	6.17×10^{-4}	1.50×10^{-2}	1.50×10^{-2}
Surface Water			
Chemical	Exposure Point Concentration (mg/L)	Risk-based Screening Level (mg/L)	Maximum Contaminant Level (mg/L)
Pb	1.14×10^{-8}	1.50×10^{-2}	1.50×10^{-2}
Ambient Air			
Chemical	Exposure Point Concentration (mg/m^3)	Risk-based Screening Level (mg/m^3)	
Pb	3.71×10^{-9}	1.50×10^{-4}	
Soil			
Chemical	Exposure Point Concentration mg/kg	Risk-based Screening Level mg/kg	
Pb	5.19×10^{-4}	4.00×10^{2}	

6 UNCERTAINTIES

Human health risk assessment of PV landfill disposal involves assumptions that are specific to a PV project, disposal site, PV technology, and chemical constituent. Uncertainties related to key modelling parameters are summarized in Table 10 and further described below. Note that the assumptions made to demonstrate the methods developed in this report have intentionally been made to lead to overestimates of true risk in order to be health protective.

Table 10. Uncertainty in key model parameters

Parameter	Best estimate	Approach in this evaluation
Waste volume	Project-specific	Waste volume equivalent to a 10MWac project, which is a 95% upper confidence limit on the mean capacity of operational large-scale solar facilities in the U.S.
Time duration of disposal	Project-specific	Project decommissioning is assumed to occur in 1 year with PV module disposal into a single non-sanitary landfill.
Chemical concentration	Technology-specific	Based on current commercial PV technologies (c-Si, CdTe, CIS). Could be reduced from future usage of Pb-free pastes and solders and reductions in thin film semiconductor layer thickness.
Leachate concentration	Chemical-specific	Based on acidic landfill conditions as characterized by USEPA Method 1311 TCLP.
Fate and transport and exposure modeling	Site-specific	Based on USEPA DRAS (V. 3.0) screening level risk assessment model.

Next, uncertainties in each exposure medium and pathway are discussed. In addressing known uncertainties (including variability), the approach developed herein (including the DRAS model) takes a health protective approach, i.e., when a particular input parameter or assumption has a range of variability or uncertainty, the value chosen for modelling is selected in the direction of potential overestimation of risk rather than potential underestimation of risk.

The DRAS model internally accounts for uncertainty using Monte Carlo simulation, varying receptor well location to estimate a lower 90th percentile waste volume-adjusted dilution-attenuation factor (DAF) for modelling the groundwater exposure pathway. Specifically, the DRAS model estimates the 90th percentile of all predicted groundwater concentrations resulting from the Monte Carlo simulation. The DAF is then calculated by dividing the TCLP leachate concentration by the predicted groundwater concentration at the 90th percentile. As a result, the 90th percentile DAF represents the lower 90th percentile of the waste volume-adjusted dilution-attenuation factors from the Monte Carlo simulation, with lower magnitude DAF's being health-protective due to less attenuation assumed.

For the surface exposure pathways, the DRAS model uses health protective assumptions regarding the proximity of exposure medium to the landfill (100 m for rainwater runoff to surface water and 305 m for wind erosion to soil).

For all exposure pathways, exposure factors for the offsite resident are reasonable maximum exposure estimates. In accordance with a screening-level risk assessment approach, use of upper-bound fate and transport and exposure assumptions is likely to overstate actual risk [25]. The input parameters to the DRAS model are listed in Section 4.1 and also discussed here in relation to health protective selection.

The waste volume is based on disposal of PV modules for a 10 MWac project, which is the 95% UCL on the mean capacity of U.S. operational large-scale solar facilities. Large-scale PV projects can range in sizes that are an order of magnitude smaller or larger than this assumption, distributed rooftop PV projects are typically less than 1 MWac, and waste from multiple PV projects could potentially be disposed in the same landfill. Therefore, the assumed waste volume is a source of variability that can be addressed with project-specific analysis. The evaluation assumes disposal instead of recycling and also assumes disposal in a non-sanitary landfill. With larger projects typically owned by large investors and subject to decommissioning requirements [2], these non-sanitary disposal assumptions may be less likely for larger projects. The actual waste volume per MWac of project capacity will also decrease with increasing PV module efficiency going forward as fewer modules are needed to achieve the target capacity. Thus, it can be seen that the selection of project size for analysis has been conducted using health protective assumptions.

The time duration of disposal is assumed to be 1 year, with the entire PV project decommissioned during that time period into a single landfill. Increasing module efficiencies and declining cost per watt may lead to partial repowering of PV projects over the timeframe of their power purchase agreements, rather than a single decommissioning event. The replaced modules may be reused rather than replaced, or recycled rather than disposed of. They may be disposed of in a sanitary landfill rather than a non-sanitary landfill, or they may disposed of in multiple landfills rather than a single landfill. As a result, the assumption that all of the project's PV modules are disposed of in one non-sanitary landfill in a single year is a worst-case health-protective assumption.

The variability in the TCLP leachate concentrations in Table 1 (±50-70%) is a source of uncertainty for the groundwater exposure pathway. A large compilation of PV leaching tests has been reviewed by Nain and Kumar [45] who found a wider range of variability than reported in Table 1. However, most of the tests compiled by Nain and Kumar follow non-standard waste characterization procedures that are either similar to recycling procedures (fine crushing and/or leaching with strong acids) or are taken from PV recycling studies, both of which will result in greater leaching than under landfill disposal conditions. The leachate concentrations in this evaluation are based exclusively on standard waste characterization testing (USEPA Method 1311 TCLP). Of the ~300 studies reviewed by Nain and Kumar, only ~10% (29 studies) had results for PV metal leaching and only 2 had results based on application of standard waste characterization tests [46][47]. The remaining studies covered PV recycling, life cycle assessment, end-of-life waste management, and emissions and environmental impact.

Of the 2 studies reviewed by Nain and Kumar that utilized standard waste characterization tests, standard TCLP test results from Collins and Anctil [46] for multi-c-Si and CIGS PV modules were higher (34.9 mg Pb/L) and lower (non-detect for Se), respectively, than those in Table 1. PV module samples were crushed to a particle size of less than 9.5 mm but the crushing procedure was not specified by Collins and Anctil. Standard TCLP test results from Brown et al. [47] for c-Si PV modules found leached Pb concentrations ranging from 0.0132 to 0.3444 mg per g module, or 0.66 to 17.22 mg/L in leachate based on the 20:1 liquid:solid ratio required by TCLP (Table 1). The mean Pb leachate concentration for c-Si PV modules in Brown et al. (5.6 mg/L) is similar to the mean in Table 1 (5.5 mg/L), but the standard deviation

in Brown et al. (6.9 mg/L) is larger than that in Table 1 (2.8 mg/L). The PV module samples in Brown et al. were crushed to a particle size of less than 9.5 mm by grinding with mortar and pestle.

In addition to Nain and Kumar, Kwak et al. also reviewed a compilation of about 85 studies with experimental data from leaching tests, ecotoxicity and toxicity analyses, and recycling research [48]. While there was overlap with the studies reviewed by Nain and Kumar, there was an additional standard waste characterization test result (USEPA Method 1311 TCLP) reported for CIS PV of 0.040 mg Se/L. Although the TCLP sample preparation method was unspecified, the results for Se were consistent with values in Table 1 used in this study (0.06±0.04 mg Se/L).

The above studies were not used in this evaluation because the TCLP sample preparation method was unspecified in Collins and Anctil and Kwak et al., and because the TCLP sample preparation method (grinding with mortar and pestle) was not field-representative in Brown et al. Because the TCLP test results in the above studies differ by less than an order of magnitude from Table 1, hypothetical use of those results would not change the conclusions of this evaluation.

The total concentrations of chemicals in commercial PV modules are subject to change as module design is varied to improve efficiency, durability, and cost. Variations in parameters such as semiconductor thickness, electrode composition, and number of junctions (single or multi-junction) affect the amounts of chemicals used in a module. The module packaging (polymer backsheet or back glass, frameless or framed) affects the overall weight of the module. The total concentrations of chemicals utilized in this study are based on current commercial PV technologies (c-Si, CdTe, CIS) and do not reflect potential future reductions from increased usage of Pb-free pastes and solders and reductions in thin film semiconductor layer thickness. In this way, the results presented herein are likely to be health protective as compared to the expected changes in PV module design in the future [13][16].

The total concentrations of chemicals in PV modules along with the waste volume are the major inputs to evaluating the surface exposure pathways in the DRAS model. These pathways are modelled based on the upper bound assumption that the chemicals are not bound to the PV panels and can fully migrate to the surface exposure pathways (air, soil, and surface water). However, as discussed in section 1, the chemicals in PV modules are encapsulated in a glass-encapsulant-glass or glass-encapsulant-backsheet monolithic structure, with the encapsulation observed to be maintained even after landfill crushing [12]. Furthermore, the surface exposure pathways assume the metals are released in elemental form, whereas they are part of stable compounds and alloys (Pb in SnPb solder for c-Si PV, Cd in CdTe for CdTe PV, Se in CIS for CIS PV). Therefore, the total chemical release assumption is a worst-case assumption and is not representative of observed field conditions. Worst-case assumptions are used in screening-level risk assessment when there is no standard basis for quantification. In the case of the groundwater exposure pathways, a non-representative total chemical release assumption is not needed as the chemical release can be experimentally measured through the TCLP leachate concentration.

The screening-level risk assessment results in this study can be compared to results from other studies which utilized more aggressive input assumptions (waste volumes, leaching rates, dilution-attenuation factors) [45][49]. The DRAS model has been previously used to assess potential human health risks from landfill disposal of PV modules by Cyrs et al. [49]. The study considered the specific case of CdTe PV modules, with similar conclusions as this evaluation, that landfill disposal of CdTe PV modules does not pose a human health hazard at the waste volumes considered. Yet in reaching this conclusion, Cyrs et al. considered annual waste

volumes that were an order of magnitude larger than this evaluation and assumed those volumes would be landfilled annually for twenty years. The resulting assumption was approximately 1 GWac or more of PV modules disposed in a single non-sanitary landfill, higher by two orders of magnitude compared with 10 MWac considered in this evaluation. Cyrs et al. also considered TCLP leachate concentrations up to the limit for non-hazardous waste disposal in the U.S. (1 mg Cd/L), approximately an order of magnitude higher than the values in Table 1. Given the use of model input assumptions that are collectively three orders of magnitude higher than assumed in this evaluation, the results in Cyrs et al. are consistent with the results in Table 8 that potential risks and hazards are several orders of magnitude below health screening levels. Thus, the conclusion that Cd risks from landfill disposal of CdTe PV modules do not pose a human health hazard at the waste volumes considered appears robust to even more aggressive assumptions about several key input factors.

Potential cancer risks from landfill disposal of PV modules were assessed by Nain and Kumar [45], considering exposure factors similar to those in Table 6 for the groundwater exposure pathway (ingestion of drinking water). A dilution-attenuation factor of 100 was assumed for potential Pb exposure from landfill disposal of c-Si PV modules, which is approximately two orders of magnitude smaller than the waste volume-adjusted dilution-attenuation factor for the groundwater exposure pathway (8920 for Pb for c-Si PV) in Table A2. As discussed above, use of a lower magnitude dilution-attenuation factors is health-protective because less attenuation results in higher exposure point concentrations. Although cancer risks and non-cancer hazards are not typically derived for Pb exposure due to the need for toxicokinetic modelling of Pb [25], Nain and Kumar utilized a Pb cancer slope factor and estimated potential cancer risks approximately one order of magnitude below the 1×10^{-6} screening threshold. Thus, the conclusion that Pb risks from landfill disposal of c-Si PV modules do not pose a human health hazard at the waste volumes considered appears robust to even more aggressive assumptions about several key input factors.

As shown in Table 10, modelling human health risks from landfill disposal of PV modules involves several case-specific assumptions, which can make it difficult to generically quantify uncertainty. However, some examples of quantifying uncertainty are shown in Table 11. Taking into consideration standard regulatory methods for landfill risk assessment and their associated uncertainties, health protective assumptions have been utilized wherever possible for model input parameters and assumptions. The results in this study are consistent with conclusions in other studies that utilized regulatory risk assessment methods with even more aggressive assumptions, concluding that Pb, Cd, and Se risks from landfill disposal of PV modules do not pose a human health hazard at the waste volumes considered.

Table 11. Example quantification of uncertainty

Modeling assumption	Description	Example quantification of uncertainty
Non-sanitary disposal	Disposal of end-of-life PV modules in a non-sanitary landfill is assumed instead of recycling or disposal in a sanitary landfill. The non-sanitary landfill is assumed to have no leachate collection or groundwater monitoring, no liner for preventing leachate migration, uncovered waste, and lack of stormwater management.	Modern landfills are expected to be sanitary. For example, a survey of U.S. landfills found 97% had liners for preventing leachate migration.

Time duration of disposal	End-of-life PV modules are assumed to be disposed in 1 year into a single non-sanitary landfill, with the entire PV project decommissioned during that time period. This assumption leads to higher risks than if the same quantity of waste was disposed over a longer time period.	Increasing module efficiencies and declining cost per watt may lead to partial repowering of PV projects over the timeframe of their power purchase agreements, rather than a single decommissioning event.
Waste volume	Waste volume is assumed to be equivalent to a 10MWac project, which is a 95% upper confidence limit on the mean capacity of operational large-scale solar facilities in the U.S.	The average capacity of operational large-scale solar facilities in the U.S is ~7.6 MWdc (or ~6 MWac). Large-scale facilities are typically owned by large investors and subject to decommissioning requirements, reducing the likelihood of non-sanitary disposal, yet we evaluate this worst case scenario to be health protective. The waste volumes associated with smaller PV installations (e.g., distributed rooftop solar facilities that are typically smaller than 1 MWac) are not considered here but would yield lower risk estimates than those used here.
Fermentation conditions	Acidic (fermentation) conditions are assumed over the life of the landfill, whereas landfills have predominantly neutral to alkaline (methanogenic) conditions over their lifetime. The fermentation conditions are characterized with use of the USEPA Method 1311 TCLP leaching procedure for estimating leachate concentrations, which requires agitation in acidic solution.	Leachate concentrations for Pb and Cd are lower under methanogenic conditions than for fermentation conditions, whereas leachate concentrations for Se are similar to or higher for methanogenic conditions than for fermentation conditions.
Particle size for leaching test	The TCLP leaching procedure requires 1 cm sample size, whereas experiments have found fragment size larger than 1 cm when modules are crushed using landfill equipment.	On average, three-quarters of fragments from experimental landfill crushing of PV modules are greater than 1 cm in size, and the front-back encapsulation is maintained.
Lower 90th percentile dilution-attenuation factor (DAF)	The USEPA DRAS model uses Monte Carlo simulation, varying receptor groundwater well location, to estimate a lower 90th percentile waste volume-adjusted dilution-attenuation factor (DAF) for modeling the groundwater exposure pathway.	The DAF varies exponentially with percentile, with the lower 90th percentile being health protective in comparison with the median.
Proximity to surface exposure media	For the surface exposure pathways, the USEPA DRAS model uses conservative assumptions regarding the close proximity of exposure media to the landfill: 100 m for rainwater runoff to surface water and 305 m for wind erosion to soil.	Based on a survey of U.S. landfills, only 3.6% are located within 1.6 km of a river or stream and the average distance from this subset of facilities to the closest river or stream is 586 m.

Total chemical release for surface exposure pathways	Surface exposure pathways (surface water, soil, air) in the USEPA DRAS model assume total release of constituent chemicals, whereas PV module constituents are contained in a glass-encapsulant-glass or glass-encapsulant-backsheet structure. Furthermore, the surface exposure pathways assume chemical release in elemental form, whereas the PV constituents modeled in this report are part of stable compounds and alloys (Pb in SnPb solder for c-Si PV, Cd in CdTe for CdTe PV, Se in CIS for CIS PV) which are less likely to leach than elemental forms.	Because surface exposure pathways are based on soil erosion modeling, they overestimate dispersion of constituents from encapsulated solid-state devices, but have been utilized for upper bound risk screening.
Upper bound exposure factors	For all exposure pathways, the USEPA DRAS model uses reasonable maximum exposure (RME) estimates for exposure factors for an offsite resident.	The RME approach combines upper-bound and mid-range exposure factors so that the result represents an exposure scenario that is both health-protective and reasonable.

7 SUMMARY

This report presents methods for analysis of potential health risks associated with non-sanitary landfill disposal of end-of-life PV modules. These methods are then demonstrated through analysis for the highest prioritized chemical in each of three commercial PV technologies: Pb content in c-Si PV, Cd content in thin film CdTe PV, and Se content in CIS PV. The analysis quantifies potential risks for an off-site resident using the USEPA DRAS (V. 3.0) fate and transport model. The analysis follows a conservative, screening-level approach, with the intent of developing order of magnitude-level estimates of potential risk after applying health-protective assumptions. Screening-level methods such as used here are meant to identify potential health risk scenarios that are greater than defined thresholds and may warrant further analysis.

Non-sanitary landfills do not conform to legal requirements in many world regions. They are characterized as having no leachate collection or groundwater monitoring, no liner for preventing leachate migration, leave the waste uncovered, and lack of stormwater management. Not only are EOL PV modules unlikely to be disposed in this manner, the combination of all of these practices is also unlikely. The conditions evaluated represent worst-case conditions for potential human health risk from PV module disposal. Yet examining worst-case conditions allows the exploration of maximum potential risk to attempt to ensure disposal does not increase health risk above regulatory thresholds.

Potential health risks are evaluated through direct estimate of cancer risk and non-cancer hazard and through comparison of predicted exposure-point concentrations in soil, air, groundwater, and surface water with risk-based screening levels published by USEPA. For Cd from CdTe PV and Se from CIS PV, cancer risks and non-cancer hazards are several orders of magnitude below screening thresholds (1×10^{-6} cancer risk and non-cancer hazard quotient of 1). For Pb from c-Si PV, exposure-point concentrations are at least an order of magnitude below USEPA health screening values in soil, air, and water.

It should be noted that commercially available PV modules also use other environmentally sensitive elements (e.g., In, Ag, Cu, Se, Sn, Ni) and new PV technologies (e.g., perovskite) are emerging. Thus, the results presented here do not represent a complete human health risk assessment for PV module disposal nor an assessment of cumulative risk, although the results are suggestive of low risk for the prioritized chemicals examined using best available regulatory methods.

The screening-level methods employed in this report can be used in future work to assess potential health risks from other chemicals of potential concern and other PV technologies to establish a more complete set of results for chemicals of potential concern. Potential ecological risks have not been evaluated in this report. For a more complete evaluation of the potential health risks from non-sanitary disposal of PV modules, the methods demonstrated here for Pb, Cd, and Se can be applied to other chemicals of potential concern for current or emerging PV technologies.

Finally, it is also important to note that examination of potential health risk from disposal of PV modules in landfills does not endorse this EOL management option. Where not already the law, there are emerging regulatory and voluntary efforts among manufacturers and others in the solar PV value chain investigate and ensure that PV module and system components are recovered for valuable use. From a circular economy and resource efficiency perspective, disposal is the least preferred EOL option. Recycling end-of-life PV modules would further mitigate environmental and material availability concerns, and life cycle assessment of PV

module recycling by IEA PVPS [50] and Fraunhofer ISE [51] indicates net environmental benefits from recycling, particularly with respect to resource depletion.

APPENDIX A. EXPOSURE POINT CONCENTRATIONS

Exposure point concentrations in air, groundwater, surface water, and soil are not explicitly provided in the DRAS model output, although they are needed for evaluating potential human health effects for Pb. The chemical concentrations in exposure media are back-calculated using the DRAS model equations in Table A1 and the exposure factors in Table 8. The back-calculated concentrations are summarized in Table A2. For consistency and transparency, the exposure point concentrations are back-calculated for all three constituents in this evaluation (Pb, Cd, Se).

The back-calculation of groundwater concentrations depends only on the TCLP leachate concentration (Table 1) and the waste-volume adjusted dilution-attenuation factor (DAF), which is an output of the DRAS model (see Table A2). The DAF is used to evaluate the migration of a chemical through soil to an underlying potable aquifer. Chemical migration through the unsaturated zone to the water table generally reduces the soil leachate concentration by attenuation processes such as adsorption and degradation, and groundwater transport in the saturated zone further reduces concentrations through attenuation and dilution. The reduction in concentration is represented by the DAF, defined as the ratio of original soil leachate concentration to the receptor point groundwater concentration. This approach assumes steady state flow, neglecting seasonal fluctuations in precipitation and groundwater flow [44].

The surface pathways (surface water, air, soil) are dependent on only two input variables: waste volume and total constituent concentration in waste [25]. Since waste volume was held constant for all DRAS model runs (at 400 m^3; Table 4), differences in surface pathway exposure point concentrations are attributable to only differences in total constituent concentration in waste (Eqs. A1- A6).

$$SW\_EPC_{Pb} = \frac{TC_{Pb}}{TC_{Cd}} * SW\_EPC_{Cd} \quad \text{Eq. A1}$$

$$SW\_EPC_{Se} = \frac{TC_{Se}}{TC_{Cd}} * SW\_EPC_{Cd} \quad \text{Eq. A2}$$

$$A\_EPC_{Pb} = \frac{TC_{Pb}}{TC_{Cd}} * A\_EPC_{Cd} \quad \text{Eq. A3}$$

$$A\_EPC_{Se} = \frac{TC_{Se}}{TC_{Cd}} * A\_EPC_{Cd} \quad \text{Eq. A4}$$

$$S\_EPC_{Pb} = \frac{TC_{Pb}}{TC_{Cd}} * S\_EPC_{Cd} \quad \text{Eq. A5}$$

$$S\_EPC_{Se} = \frac{TC_{Se}}{TC_{Cd}} * S\_EPC_{Cd} \quad \text{Eq. A6}$$

where:

SW_EPC is surface water exposure point concentration (Table A2),
A_EPC is ambient air exposure point concentration (Table A2),
S_EPC is soil exposure point concentration (Table A2), and
TC is total constituent concentration in waste (Table 5).

Table A1. DRAS model equations used to back-calculate chemical concentrations in exposure media [25]

	Chemical Concentration in Ground Water		
DRAS Equation 2-1	$C_{gw} = \frac{C_L}{DAF}$	C_{gw}	concentration of waste constituent in ground water
		DAF	dilution-attenuation factor (unitless)
		C_L	leachate concentration (TCLP; mg/L)
	Chemical Concentration in Surface Water		
DRAS Equation 4-95	$HQ = \frac{CR * C_{sw} * EF * ED}{BW * RfD * AT * 365day/year}$	HQ	hazard quotient for waste constituent (unitless)
		BW	body weight (kg)
		RfD	reference dose (mg/kg-day)
		AT	averaging time (yr)
		CR	water consumption rate (L/day)
		EF	exposure frequency (days/yr)
		ED	exposure duration (yr)
		C_{sw}	concentration of waste constituent in surface water (mg/L)
	Chemical Concentration in Particulate Air Emissions		
DRAS Equation 4-65	$Risk = \frac{C_{avg} * IFA_{adj} * EF * CSF}{AT * 365days/year}$	Risk	cancer risk for waste constituent (unitless)
		IFA_{adj}	inhalation factor, age-adjusted ([m^3 - year]/[kg-day])
		EF	exposure frequency (days/yr)
		CSF	cancer slope factor $(mg/kg\text{-}day)^{-1}$
		AT	averaging time (yr)
		C_{avg}	downwind concentration of waste constituent at point of exposure (mg/m^3)
	Chemical Concentration in Soil		
DRAS Equation 4-112	$HQ = \frac{CR * C_{soil} * EF * ED}{BW_c * RfD * AT * 365day/year * 1E+6mg/kg}$	HQ	hazard quotient for waste constituent (unitless)
		BW_c	body weight of child
		RfD	reference dose (mg/kg-day)
		AT	averaging time (yr)
		CR	soil consumption rate (mg/day)
		EF	exposure frequency (days/yr)
		ED	exposure duration (yr)
		C_{soil}	concentration of constituent in soil at point of exposure (mg/kg)

Table A2. Derivation of chemical concentrations in exposure media

Groundwater			
Module Type	Chemical	Media Concentration mg/L	Calculation Methodology
CdTe PV	Cd	3.16×10^{-5}	Back-calculated with DRAS Equation 2-1 (Table A1) based on DRAS model waste volume-adjusted dilution attenuation factor (DAF; 4750) and leachate concentration in Table 1.
CIS PV	Se	1.32×10^{-5}	Back-calculated with DRAS Equation 2-1 (Table A1) based on DRAS model waste volume-adjusted dilution attenuation factor (DAF; 4560) and leachate concentration in Table 1.
c-Si PV	Pb	6.17×10^{-4}	Back-calculated with DRAS Equation 2-1 (Table A1) based on DRAS model waste volume-adjusted dilution attenuation factor (DAF; 8920) and leachate concentration in Table 1.
Surface Water			
Module Type	Chemical	Media Concentration mg/L	Calculation Methodology
CdTe PV	Cd	8.80×10^{-9}	Back-calculated from DRAS hazard quotient (4.69×10^{-7}; Table 8), DRAS Equation 4-95 (Table A1), and DRAS exposure factors (Table 6)
CIS PV	Se	5.29×10^{-9}	Back-calculated from DRAS hazard quotient (2.82×10^{-8}; Table 8), DRAS Equation 4-95 (Table A1), and DRAS exposure factors (Table 6)
c-Si PV	Pb	1.14×10^{-8}	Eq. A1.

Ambient Air			
Module Type	Chemical	Media Concentration mg/m^3	Calculation Methodology
CdTe PV	Cd	2.85×10^{-9}	Back-calculated from DRAS hazard quotient (2.45×10^{-9}; Table 8), DRAS Equation 4-65 (Table A1), and DRAS exposure factors (Table 6).
CIS PV	Se	1.71×10^{-9}	Eq. A4.
c-Si PV	Pb	3.71×10^{-9}	Eq. A3.
Soil			
Module Type	Chemical	Media Concentration mg/kg	Calculation Methodology
CdTe PV	Cd	3.99×10^{-4}	Back-calculated from DRAS hazard quotient (1.02×10^{-5}; Table 8), DRAS Equation 4-112 (Table A1), and DRAS exposure factors (Table 6).
CIS PV	Se	2.40×10^{-4}	Back-calculated from DRAS hazard quotient (6.14×10^{-7}; Table 8), DRAS Equation 4-112 (Table A1), and DRAS exposure factors (Table 6).
c-Si PV	Pb	5.19×10^{-4}	Eq. A5.

APPENDIX B. COMPARISON OF Cd AND Se EXPOSURE POINT CONCENTRATIONS TO RISK-BASED SCREENING LEVELS

The USEPA DRAS model allows for direct estimation of cancer risk and non-cancer hazard for Cd and Se (section 5.2). Cancer risks and non-cancer hazards could not be directly estimated for Pb using the DRAS model due to a lack of dose-response toxicological values (section 5.1). Therefore for Pb, screening-level risk assessment methods were applied in which exposure point concentrations were compared with risk-based screening levels (section 5.3). For transparency and completeness, the screening-level risk assessment method used for Pb is also applied to Cd and Se in Appendix B. However, the direct estimation of cancer risk and non-cancer hazard for Cd and Se in section 5.2 is preferred over the screening-level risk assessment method in this appendix, because it provides more specific quantification of potential risks and hazards and is a direct output of the USEPA DRAS model.

The screening-level risk assessment method in section 5.3 is applied to Cd and Se through comparison of exposure point concentrations in air, groundwater, surface water, and soil from Table A2 to risk-based screening levels in Table B1. The exposure point concentrations for Cd and Se are compared to USEPA residential risk-based screening levels in soil, air, and water and to USEPA maximum contaminant levels, which are legal standards that apply to public water systems [40].

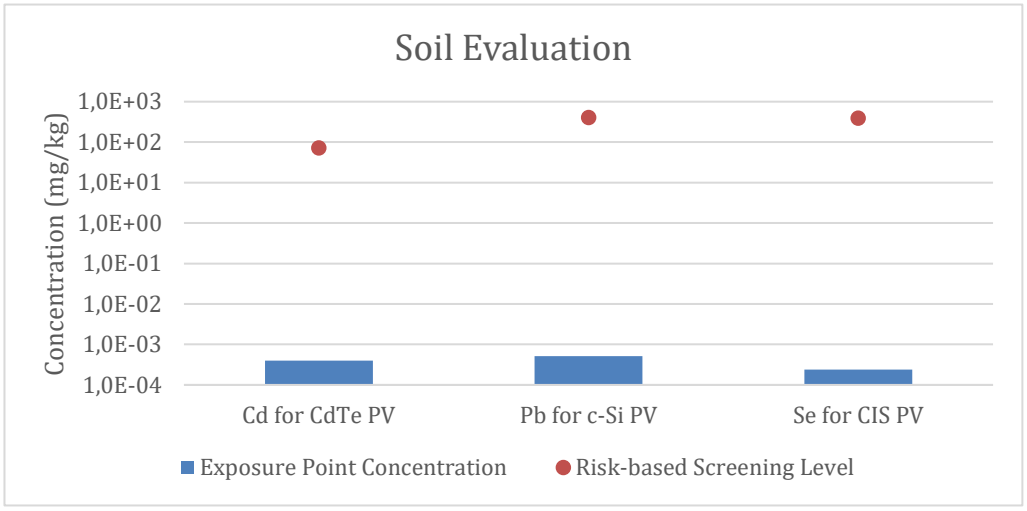
Table B1. Comparison of Cd and Se exposure point concentrations to residential risk-based screening levels and maximum contaminant levels [40] for PV modules from a 10 MWac project disposed in a non-sanitary landfill.

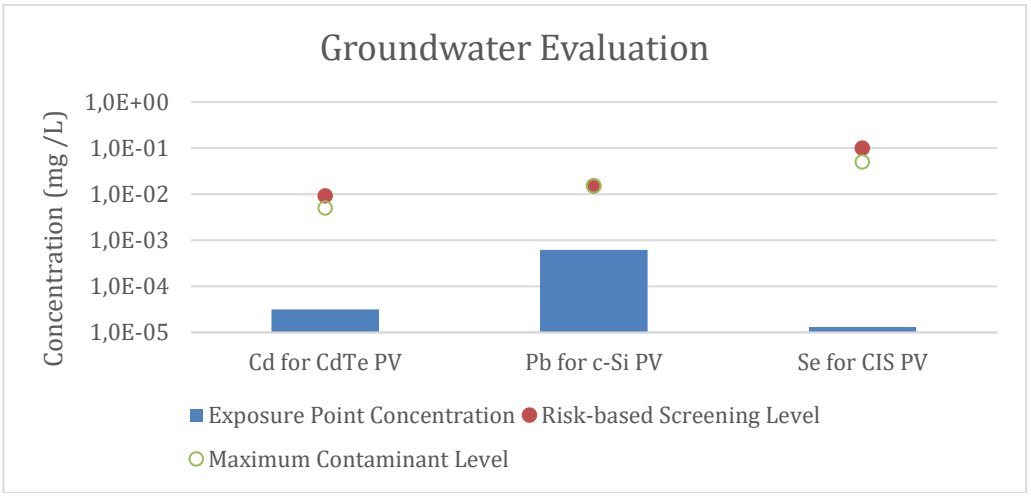
Groundwater				
Module Type	Chemical	Exposure Point Concentration (mg/L)	Risk-based Screening Level (mg/L)	Maximum Contaminant Level (mg/L)
CdTe PV	Cd	3.16×10^{-5}	9.20×10^{-3}	5.00×10^{-3}
CIS PV	Se	1.32×10^{-5}	1.00×10^{-1}	5.00×10^{-2}
Surface Water				
Module Type	Chemical	Exposure Point Concentration (mg/L)	Risk-based Screening Level (mg/L)	Maximum Contaminant Level (mg/L)
CdTe PV	Cd	8.80×10^{-9}	9.20×10^{-3}	5.00×10^{-3}
CIS PV	Se	5.29×10^{-9}	1.00×10^{-1}	5.00×10^{-2}
Ambient Air				
Module Type	Chemical	Exposure Point Concentration (mg/m^3)	Risk-based Screening Level (mg/m^3)	
CdTe PV	Cd	2.85×10^{-9}	1.60×10^{-6}	
CIS PV	Se	1.71×10^{-9}	2.10×10^{-2}	

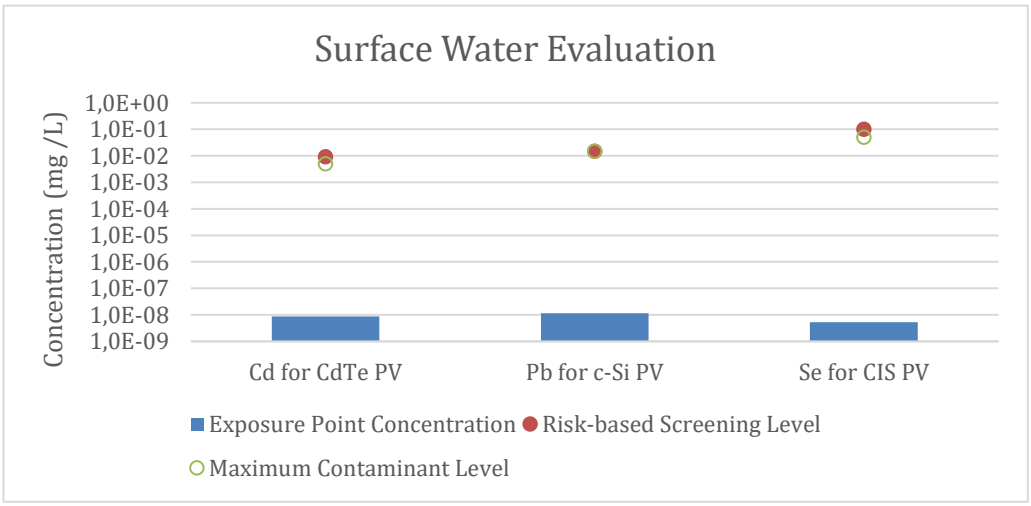
Soil				
Module Type	Chemical	Exposure Point Concentration mg/kg	Risk-based Screening Level mg/kg	
CdTe PV	Cd	3.99×10^{-4}	7.10×10^{1}	
CIS PV	Se	2.40×10^{-4}	3.90×10^{2}	

Similar to the risk and hazard results in Table 8, the exposure point concentrations of Cd for CdTe PV and Se for CIS PV are several orders of magnitude below risk-based screening levels and maximum contaminant levels. The comparisons in Table B1 for Cd and Se and the comparisons in Table 9 for Pb are summarized graphically in Figure B1, indicating that the one-time disposal of PV modules from a 10 MWac project in a non-sanitary landfill is not likely to pose significant health risks for the specific cases considered in this study (Cd from CdTe PV, Pb from c-Si PV, and Se from CIS PV).

Health screening levels can differ by region. For example, the World Health Organization (WHO) has established water screening levels for Pb (0.01 mg/L), Cd (0.003 mg/L), and Se (0.04 mg/L) [52]. The WHO screening levels are slightly lower than the USEPA screening levels for water in Table 9 and Table B1 (20-40% lower than the USEPA Maximum Contaminant Levels). Since the exposure point concentrations in the groundwater pathways and surface water pathways of Pb, Cd, and Se for c-Si, CdTe, and CIS PV, respectively, are at least an order of magnitude below USEPA health screening values, they are also below the WHO screening values for water.







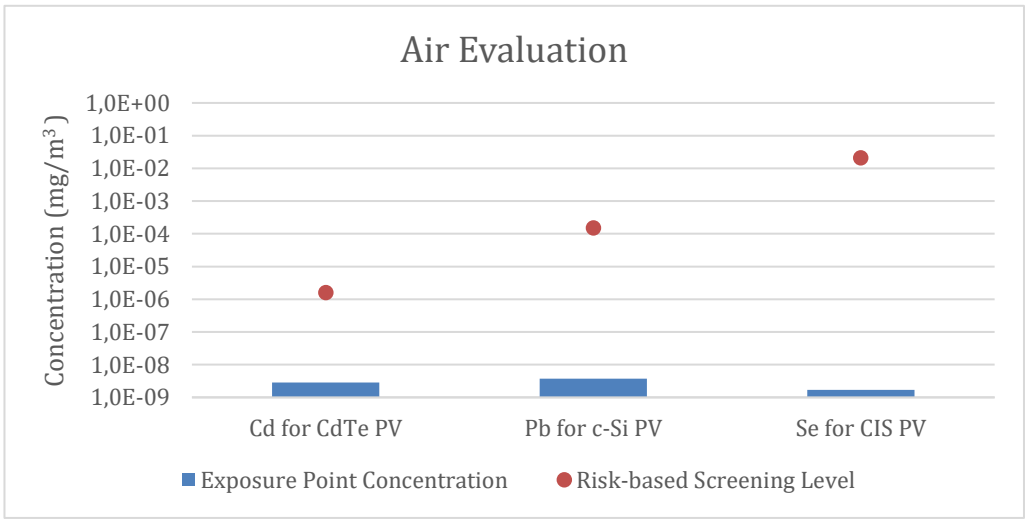


Figure B1. Comparison of exposure point concentrations in soil, groundwater, surface water, and air to residential risk-based screening levels and maximum contaminant levels [40] for PV modules from a 10 MWac project disposed in a non-sanitary landfill. Note the log scale, where the y-axis spans several orders of magnitude.

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