

Attachment 5

Fate and Transport Evaluation of Potential Leaching Risks from Cadmium Telluride Photovoltaics

FATE AND TRANSPORT EVALUATION OF POTENTIAL LEACHING RISKS FROM CADMIUM TELLURIDE PHOTOVOLTAICS

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Abstract—Fate and transport analysis has been performed to evaluate potential exposures to cadmium (Cd) from cadmium telluride (CdTe) photovoltaics (PV) for rainwater leaching from broken modules in a commercial building scenario. Leaching from broken modules is modeled using the worst case scenario of total release of Cd, and residential screening levels are used to evaluate potential health impacts to on-site workers and off-site residents. A rooftop installation was considered rather than a ground-mount installation because rainwater runoff is concentrated via building downspouts in a rooftop installation rather than being dispersed across large areas in a ground-mount installation. Fate and transport of Cd from leachate to soil are modeled using equilibrium soil/soil-water partitioning. Subsequent migration to ambient air as wind-blown dust is evaluated with a screening Gaussian plume dispersion model, and migration to groundwater is evaluated with a diffusion-attenuation factor approach. Exposure point concentrations in soil, air, and groundwater are one to six orders of magnitude below conservative residential soil, residential air, drinking water, human health screening levels in both a California and southern Germany (Baden-Württemberg) exposure scenario. Potential exposures to Cd from rainwater leaching of broken modules in a commercial building scenario are highly unlikely to pose a potential health risk to on-site workers or off-site residents. *Environ Toxicol Chem* 2012;31:1670–1675. © 2012 SETAC

Keywords—Cadmium telluride; Leaching; Risk assessment; Fate and transport; Cadmium telluride photovoltaics

INTRODUCTION

Solar energy is an important technology for climate change mitigation and development of a low carbon economy because it offers the highest global technical potential for electricity generation among renewable energy sources [1]. In particular, cadmium telluride (CdTe) thin film photovoltaic (PV) modules have the lowest life cycle carbon footprint and fastest energy payback time of current PV technologies [2]. Although CdTe has been shown to be significantly less toxic than elemental cadmium (Cd) on an acute basis [3], the primary health and safety concern for CdTe PV is the potential introduction of Cd compounds into the environment. When considered on a life cycle basis from raw material acquisition through product end-of-life, CdTe PV has been found to produce environmental Cd emissions to air that are no higher than those from conventional silicon PV technologies [4,5]. Moreover, because Cd is an unavoidable by-product of Zn mining, large-scale deployment of CdTe PV sequesters waste Cd that would otherwise be disposed of [6]. Pretended end-of-life takeback and recycling programs also significantly reduce the overall environmental impact of CdTe PV modules [7].

Under normal operation, CdTe PV modules do not pose a threat to human health or the environment, because during the manufacturing process, the CdTe semiconductor layer is bound under high temperature to one sheet of glass, coated with an industrial laminate material, and then encapsulated between a second sheet of glass. However, some stakeholders have raised

concerns about the potential exposure to CdTe from leaching of broken modules, defined as modules with cracked glass or broken pieces. Breakage results from extreme weather or human factors. Although rare, breakage followed by precipitation may potentially result in leaching of CdTe from modules and subsequent exposure to Cd compounds in soil, air, or groundwater. This analysis uses fate and transport modeling to estimate potential exposures to Cd compounds resulting from leaching and then evaluates the potential health effects associated with these exposures.

Fate and transport scenarios were evaluated for two geographic locations, southern Germany and California. Germany is among the world's leading PV markets, having accounted for nearly half of global demand in 2010 [8]. This analysis focuses on the higher solar irradiance region of southern Germany (Federal State of Baden-Württemberg). California is a leading PV market in the United States, and in 2011, the California state legislature adopted a renewable portfolio standard of 33% by 2020 (<http://www.cpuc.ca.gov/PUC/energy/Renewables/index.htm>).

In the present analysis, a commercial building scenario was chosen rather than a residential building scenario because the larger PV array size for commercial buildings increases the probability that module breakage may occur in a given year. However, both nonresidential (on-site) and residential (off-site) exposure scenarios were considered and evaluated using residential screening values. A rooftop installation was considered rather than a ground-mount installation because rainwater runoff can be concentrated via building downspouts in a rooftop installation (impact via concentrated streams) rather than being dispersed across large areas in a ground-mount installation. The evaluation considers the worst-case scenario in which the total mass of Cd in each broken module is released.

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MATERIALS AND METHODS

The present analysis considers broken CdTe PV modules located on the rooftop of a commercial building. Potential receptors considered for analysis include on-site commercial/industrial workers and off-site residents. Under this exposure scenario, potential exposure to Cd is considered for commercial/industrial workers via inhalation of, dermal contact with, and ingestion of Cd leached into soil, as well as exposure to groundwater potentially impacted by leachate. Also under this exposure scenario, potential exposure to Cd is considered for off-site residents via inhalation of windblown dust from affected soil and exposure to groundwater potentially impacted by leachate.

To characterize these potential exposure scenarios, exposure point concentrations of Cd in soil, air, and groundwater are estimated using a fate and transport analysis. The estimated exposure point concentrations are the relevant concentrations to which on-site workers or off-site residents may potentially be exposed. The exposure point concentration for soil is only relevant to the on-site worker who may potentially have incidental contact with on-site surface soil during the workday. The exposure point concentration for air is relevant to both the on-site worker and off-site resident who may potentially inhale affected ambient air. The exposure point concentration for groundwater is relevant to both the on-site worker and off-site resident who may potentially use groundwater as drinking water.

To evaluate potential human health impacts, estimated exposure point concentrations are compared to human health screening levels. Nonresidential screening levels are applicable to the on-site worker, whereas residential screening levels are applicable to the off-site resident. In this evaluation, the residential screening levels are used in comparison with estimated exposure point concentrations to be protective of both on-site workers and off-site residents. Specifically, for California, residential screening levels for soil (1.7 mg/kg) and air ($1.4 \times 10^{-5} \mu\text{g}/\text{m}^3$) are used instead of commercial/industrial screening levels of 7.5 mg/kg and $6.8 \times 10^{-5} \mu\text{g}/\text{m}^3$, respectively. For Germany, a residential screening level for soil (2 mg/kg) is used instead of a commercial/industrial screening level of 60 mg/kg.

The fate and transport methodology used to estimate migration of Cd from the emission point (broken module) to the exposure point (soil, air, or groundwater) is summarized in Figure 1 and described with Equations 1 to 5 below. The concentration of Cd in leachate resulting from rainwater that falls upon and runs off broken modules is estimated based on a worst-case mass balance approach, where all the mass of Cd in each broken module is assumed to be transferred from the module into the volume of rainfall that falls upon the module during the exposure period. The subsequent concentration of Cd in rainwater runoff from the overall module array is calculated using a weighted average between impacted runoff from broken modules and nonimpacted runoff from unbroken modules. It should be noted that the assumption of total release of Cd from a

broken module was adopted for the purpose of conducting screening level risk assessment, but is unlikely in the light of low experimentally measured emissions from broken or burnt modules [4].

It is assumed that the rooftop runoff is conveyed via downspouts and discharged onto the ground surface over an area of 1 m^2 per downspout. Chemical concentrations in vadose (unsaturated) zone soil pore water at these discharge locations are assumed to be equal to the concentrations in the rooftop runoff discharge. The vadose zone soil pore water throughout the rest of the site is assumed to be nonimpacted. For the commercial building scenario, a roof with dimensions of $50 \times 50 \text{ m}$ is assumed to be completely covered by CdTe PV modules of dimensions $0.6 \times 1.2 \text{ m}$ each. Twenty-five downspouts are assumed for the building, based on the roof area being 25 times larger than a standard residential building ($10 \times 10 \text{ m}$) [9], where the latter would have one downspout.

The vadose zone soil pore water concentration in each 1 m^2 downspout ground surface area is estimated with the worst case mass balance approach in Equation 1, where the numerator represents the total annual release of Cd and the denominator represents the total annual column of rainfall.

$$CV = \frac{N \times M \times CF \times B}{P \times A} \tag{1}$$

where CV is the Cd concentration in vadose soil pore water (mg/L); *N* is the number of modules (unitless); *M* is the mass of Cd per module (g); *CF* is the conversion factor (mg/g); *B* is the module breakage rate (year^{-1}); *P* is the annual average precipitation ($\text{L}/\text{m}^2 \cdot \text{year}$), which is annual precipitation (m/year) falling over 1 m^2 converted to units of L from m^3 ; and *A* is the area of roof-top array (m^2).

The potential transport of Cd to soil is evaluated in accordance with the equilibrium-partitioning approach described in the U.S. Environmental Protection Agency (U.S. EPA) soil screening guidance [10,11]. It is assumed that the surface soil where rainwater runoff is discharged is instantaneously impacted with Cd, at the concentration predicted by equilibrium partitioning between the water and soil matrices, as expressed by the soil/soil-water partitioning coefficient (*K_d*) value for Cd (Eqn. 2).

$$CS_{eq} = CV \times \left(K_d + \frac{\theta_w}{\rho_b} \right) \tag{2}$$

where *CS_{eq}* is the equilibrium concentration of Cd in impacted soil (mg/kg); *CV* is the concentration of Cd in vadose zone soil pore water (mg/L); *K_d* is the soil/soil water partitioning coefficient (L/kg); θ_w is the soil water-filled porosity (unitless); and ρ_b is the soil dry bulk density (kg/L).

For this scenario, it is assumed that the entire area of the site evaluated here is uncovered by concrete or asphalt and is open bare soil to allow the runoff water to penetrate into site soils. In actuality, commercial sites are often completely covered by concrete or asphalt. On-site commercial/industrial workers are assumed on average to be exposed to site soils across the entire portion of the site that is not occupied by the building. Exposure point concentrations of chemicals in soil are therefore calculated as site-wide average concentrations, incorporating areas of impacted soils (at the worst-case concentrations predicted by equilibrium partitioning) and nonimpacted soils (Eqn. 3). The exposure area (*S_A:A*; Fig. 2) is assumed to be the same as that for a residential building [9], even though a commercial building property would likely be larger, therefore with larger

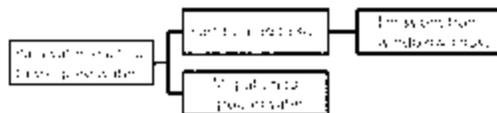


Fig. 1. Fate and transport schematic of migration from emission point (leachate leaching from broken module) to exposure point in soil, air, and groundwater.

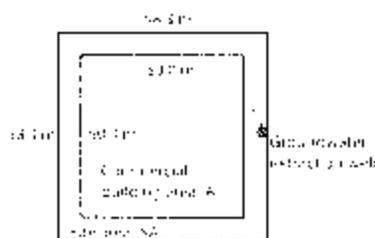


Fig. 2. Site schematic.

nonimpacted areas. This assumption accounts for the potential of at least part of the site to be covered.

$$CS = CS_{eq} \times \frac{IA}{(SA - A)} \quad (3)$$

where CS is the exposure point concentration of Cd in soil (mg/kg); CS_{eq} is the equilibrium concentration of Cd in impacted soil (mg/kg); IA is the impacted area (m^2), SA is the site area (m^2), and A is the area of building (m^2).

The potential transport of Cd from impacted soil to ambient air is estimated (Eqn. 4) using the U.S. EPA-recommended default windblown dust emissions flux for wind erosion ($1.38 \times 10^{-3} \text{ g/s-m}^2$) [14]. As noted above, the uncovered portion of the site is assumed to be bare earth for the purpose of this analysis, whereas commercial sites are frequently landscaped or covered by concrete or asphalt. It is assumed that Cd is present in this windblown dust at the soil concentration predicted by equilibrium partitioning (Eqn. 2). The U.S. EPA screening Gaussian plume dispersion model SCREEN3 [12] is used in conjunction with the emissions flux to estimate worst-case concentrations of dust and thus Cd in ambient air. The maximum hourly dust concentration from SCREEN3 was adjusted with a persistence factor of 0.08 [13] to derive the annual worst-case concentrations of dust.

$$CA = CS_{eq} \times CD \times CF_1 \times CF_2 \quad (4)$$

where CA is the exposure point concentration of Cd in air ($\mu\text{g}/m^3$); CS_{eq} is the equilibrium Cd concentration in soil (mg/kg); CD is the worst case dust concentration in air (mg/m^3), CF_1 is the conversion factor (kg/mg); and CF_2 is the conversion factor ($\mu\text{g}/\text{mg}$).

The potential transport of Cd to groundwater is evaluated in accordance with the dilution-attenuation factor (DAF) approach described in the U.S. EPA soil screening guidance [10,11]. It is assumed that vadose zone soil water, from the ground surface to the groundwater table, contains Cd at the module array-rimmed concentration discussed above in Equation 1 (i.e., it is assumed the soil column does not adsorb any Cd). The potential concentration of Cd in groundwater at the hypothetical point of usage, which is assumed by the model to be a groundwater extraction well located 25 ft from the edge of the impacted area, is calculated by applying an upper bound (95th percentile) DAF [14] to the vadose soil water concentration (Eqn. 5). Note that for DAF values, higher percentiles represent numerically lower values, indicating less dilution-attenuation, and therefore higher groundwater concentrations.

$$CW = \frac{CV}{DAF} \times CF \quad (5)$$

where CW is the exposure point concentration of Cd in groundwater ($\mu\text{g}/L$); CV is the concentration of Cd in vadose

zone soil pore water (mg/L); DAF is the dilution-attenuation factor (unitless); and CF is the conversion factor ($\mu\text{g}/\text{mg}$).

The specific fate and transport modeling parameters used in Equations 4 to 5 are summarized in Table 1. The parameters are the same for the two geographies evaluated, with the exception of higher average annual precipitation (37.32 inches/year; http://www2.lfw.baden-wuerttemberg.de/public/abt5/klimaatlas_bw/klima/aenderungen/ba-wue/niederschlag/index.html) for Baden-Wuerttemberg, relative to California (21.44 inches/year; <http://www.nationalatlas.gov/printable/precipitation.html>). In addition, the German dry soil bulk density (1.4 kg/L; average between settlement and grassland areas [15]) is slightly lower than that used for California (1.5 kg/L [11]).

It should also be noted that the German Federal Environment Ministry does not provide a default value for the soil/soil-water partitioning coefficient data (K_d) for cadmium, due to low mobility in groundwater [16]. In this evaluation, the K_d value used for the California exposure scenario is applied to the southern Germany exposure scenario.

RESULTS

Exposure point concentrations of Cd in soil, air, and groundwater derived in Equations 3 to 5, respectively, are summarized in Table 2, and compared to human health screening levels for each of these media. For the California case, the screening levels in soil, air, and groundwater are from the California Human Health Screening Levels, U.S. EPA Region 9 Regional Screening Levels, and U.S. National Primary Drinking Water Regulations, respectively. Residential soil and indoor air screening values are used, both of which are more protective than the commercial building soil and outdoor air exposure scenarios considered here.

In the southern Germany case, the soil screening level is from the residential trigger value in Annex 2 of the Federal Soil Protection and Contaminated Sites Ordinance (<http://www.umweltbundesamt.de/boden-und-airlasten/altlast/webthemen/inter/pdl/bb0d-schv-engl.pdf>). The standard residential trigger value in soil is 20 mg/kg, whereas for the special case of gardens in which children stay and food plants are grown, a residential trigger value of 2 mg/kg applies. Table 2 presents the latter more protective soil screening value, which is similar to the California Human Health Screening Levels value used for California. The German air screening level is based on World Health Organization air quality guidelines for Europe [17] and is slightly higher in magnitude than the California air screening level. The groundwater screening level is from the German regulation on drinking water (<http://www.umweltbundesamt.de/wasser-ethemen/trinkwasser/gesetz.html>) and is the same as the U.S. drinking water standard.

In the California and southern Germany cases, exposure point concentrations in soil, air, and groundwater are one to six orders of magnitude below human health screening levels, indicating that it is highly unlikely that exposures to these media would pose potential health risks to on-site workers or off-site residents. In particular, air concentrations are below screening levels by five to six orders of magnitude, indicating exposure to ambient air is a de minimis exposure pathway.

For reference, the average background Cd concentration in California surface soils is 0.36 mg/kg [18], whereas average background surface soil Cd concentrations in Baden-Wuerttemberg range from 0.2 to 0.3 mg/kg [19]. Therefore, modeled impacts to soil are over an order of magnitude below both human health screening levels and regional background levels.

Table 1. Fate and transport modeling parameters used in conjunction with Equations 1 to 5 for California (CA) and Baden-Württemberg (B-W) exposure scenarios^a

Equation 1 parameters ^b	Equation 2 parameters ^c	Equation 3 parameters ^d	Equation 4 parameters ^e	Equation 5 parameters ^f
N: 3472	CV (CA/B-W): 0.00612/0.00352 mg/L	CS _{eq} (CA/B-W): 0.49/0.265 mg/kg	CS _{eq} (CA/B-W): 0.46/0.267 mg/kg	CV (CA/B-W): 0.00612/0.00352 mg/L
M: 6 modules	K _d : 751 L/kg	IA: 25 m ²	CD: 5.5 × 10 ⁻³	DAF: 7.82
CF: 0.00 mg/g	ρ _w : 1.0 ^g	SA: 3.10E10 m ²	CF _g : 0.000001 kg/mg	CF: 0.001 mg/mg
B: 0.01/year	ρ _s (CA/B-W): ^g 5/1.4 kg/L	A: 2500 m ²	CF _g : 0.001 μg/mg	
P (CA/B-W): 545/947 L/m ² /year				
A: 2500 m ²				

^a When two values are provided for a given parameter, first value is for CA and second value is for B-W.

^b Parameters in Equation 1 are N (number of modules), M (mass of Cd per module), CF (conversion factor), B (module breakage rate), P (annual average precipitation), and A (area of building).

^c Parameters in Equation 2 are CV (concentration of Cd in vadose zone soil pore water), K_d (soil-water partitioning coefficient), ρ_w (soil water-filled porosity), and ρ_s (soil dry bulk density).

^d Parameters in Equation 3 are CS_{eq} (equilibrium concentration of Cd in impacted soil), IA (impacted area), SA (soil area), and A (area of building).

^e Parameters in Equation 4 are CS_{eq} (equilibrium Cd concentration in soil), CD (worst case dust concentration in air), CF_g (conversion factor), and CF (conversion factor).

^f Parameters in Equation 5 are CV (concentration of Cd in vadose zone soil pore water), DAF (dilution-attenuation factor), and CF (conversion factor).

^g Precipitation parameter (P) is based on annual average precipitation of 21.44 and 37.32 inches for California and Baden-Württemberg respectively.

For further perspective on soil impacts, Cd is commonly found in agricultural fertilizers. California is among the top users of agricultural fertilizer in the United States and analysis of metals in fertilizer samples has been performed by the California Department of Food and Agriculture, with median Cd concentrations of 89 mg/kg in phosphate fertilizer and 37 mg/kg in nitrogen/phosphorus/potassium (NPK) fertilizer [20]. Similarly, average Cd concentrations in phosphate and NPK fertilizer in Germany are 60 and 18 mg/kg, respectively (http://www.bfr.bund.de/cm/343/cadmiumaustrag_ueber_duengemittel.pdf). These values are over three orders of magnitude higher than the estimated exposure point concentration in soil in California and southern Germany (Table 2).

For reference, average background Cd (total suspended particulate) concentrations in California ambient air monitoring stations ranged from 0.0008 to 0.001 μg/m³ in 2008 (<http://www.epa.gov/air/data/geosel.html>). Similarly, average background Cd concentrations in Europe range from 0.0001 to 0.0004 μg/m³ in rural areas and 0.0002 to 0.0025 μg/m³ in urban areas [21]. Therefore, modeled impacts to air are five orders of magnitude below both health screening levels and background levels.

For reference, the average background Cd concentration in groundwater from 1984 to 2004 in California Air Force bases ranged from <0.001 mg/L (50th percentile) to 0.006 mg/L (95th percentile; http://www.dhsa.ca.gov/assessingrisk/upload/metal_handout.pdf). In Baden-Württemberg, average background Cd concentrations in groundwater range from 0.0052 to 0.0039 mg/L [22]. Therefore, modeled impacts to

groundwater in California and southern Germany are below both human health screening levels and background levels.

In addition to soil, air, and groundwater, another route of potential concern is direct discharge of rooftop runoff to storm-water catch basins. In combined sewer systems, stormwater and wastewater are collected together and treated at a publicly owned treatment works (POTW). The worst-case rooftop runoff Cd concentration (assuming total release of Cd from broken modules) is equivalent to the estimated Cd concentration in vadose soil pore water (CV: 0.001-0.006 mg/L; Table 1). Because this concentration is approximately consistent with drinking water standards, impacts to POTW's from rooftop runoff are expected to be minimal.

DISCUSSION

The fate and transport analysis conducted here represents a worst-case scenario of total Cd release from broken modules. An implicit assumption for this scenario is that a broken module would remain undetected and in the field over the exposure duration. This is a screening level assumption that would likely not occur given routine inspections of modules or power output monitoring. For example, the latter may include diagnostic comparison of actual to expected performance or comparison of co-located arrays to identify low performance areas and modules that are nonfunctioning potentially due to breakage.

Another implicit assumption is that emissions of CdTe from rainwater leaching of broken modules can be modeled as emissions of Cd, a "read-across" approach. This is a screening

Table 2. Estimated exposure point concentration (EPC) and corresponding human health screening level in soil, air, and groundwater.

	Soil EPC ^a (mg/kg)	Soil screening level (mg/kg)	Air EPC ^b (μg/m ³)	Air screening level (μg/m ³)	Ground water EPC (mg/L)	Ground water screening level (mg/L)
California	1.28 × 10 ⁻³	1.7	2.53 × 10 ⁻³	1.4 × 10 ⁻³	7.82 × 10 ⁻³	5 × 10 ⁻³
Baden-Württemberg ^c	7.15 × 10 ⁻³	2	1.46 × 10 ⁻³	5 × 10 ⁻³	1.50 × 10 ⁻³	5 × 10 ⁻³

^a California screening levels are from the California Human Health Screening Levels (<http://www.calepa.ca.gov/brownfields/lorr/narrs/2005/CHHSLS/Guide.pdf>) for soil, U.S. EPA Region 9 (<http://www.epa.gov/region9/superfund/prg>) for air, and U.S. National Primary Drinking Water Regulations (<http://water.epa.gov/dw/contaminants/index.cfm>) for groundwater.

^b German screening levels are from Annex 2 of the Federal Soil Protection and Contaminated Sites Ordinance (http://www.umweltbundesamt.de/bundes-und-landschaftsrecht/wgbl_berichte/pdb/baubschv-engl.pdf) for soil, World Health Organization air quality guidelines for Europe [27] for air, and German regulation on drinking water (<http://www.umweltbundesamt.de/wasser-erhaltung/nik/wasser-gesetze.html>) for groundwater.

level assumption because CdTe is relatively insoluble [3,23]. For example, transformation and dissolution testing is designed to determine the rate and extent to which sparingly soluble metal compounds can produce soluble available ionic species in aqueous media under a set of standard laboratory conditions representative of those generally occurring in the environment. Based on long-term transformation and dissolution testing of CdTe, a 1 mg/l. loading showed a concentration of 15 μg of Cd per l. after 28 d, indicating approximately 1.5% solubility [24]. This is also consistent with the very low solubility product ($K_{sp} = 9.5 \times 10^{-35}$) for CdTe [25]. In addition to low solubility, CdTe can be contrasted with elemental Cd and other Cd compounds based on limited bioavailability and low acute toxicity, which result in an overall margin of safety of two orders of magnitude likely inherent to CdTe screening assessments developed using the real across approach from Cd [25].

Because of the low solubility of CdTe, aggressive extraction methods are required to leach CdTe from a module. Such methods are used, for example, in the recycling process for CdTe modules. They involve crushing the module into mill-scale pieces and agitating it in an acidic solution [7]. These extraction methods in no way mimic actual broken or cracked module exposure to rainwater. Therefore, the assumption of total Cd release from broken modules is highly unlikely.

In addition to this worst-case assumption, other upper bound assumptions are used in the analysis. Migration from vadose zone soil pore water to soil is modeled with equilibrium partitioning, which represents the theoretical maximum concentration possible in the solid phase, for a given concentration in soil pore water. Subsequent migration from soil to air is modeled using the SCREEN3 U.S. EPA Gaussian plume dispersion model to estimate worst-case concentrations of windblown dust.

The approach used to estimate groundwater impacts is also upperbound because it does not account for the loss of chemical mass from the pore water during soil-water partitioning, instead assuming that the pore water is instantaneously in equilibrium with the solid soil phase. Accordingly, no mass in pore water is lost to the solid soil phase during partitioning, when in actuality some of this mass partitions into the solid soil phase, with a subsequent reduction in the concentration of Cd in the pore water with depth, until equilibrium is reached. Accounting for the loss of chemical mass from the pore water to the solid phase would lower chemical concentrations in soil water that are assumed to penetrate to groundwater and so reduce predicted groundwater exposures. In addition, the DAF assumes that there is an infinite source of mass available for release. Conserving mass would likely reduce the average long-term groundwater concentration estimated using the DAF approach and so result in lower groundwater exposures. Moreover, the dilution-attenuation factor used was a 95th percentile DAF where the higher percentiles represent numerically lower DAF values, indicating less dilution-attenuation and therefore higher groundwater concentrations. All of these factors contribute to the likelihood that impacts to groundwater are overestimated. Also as described earlier, under German groundwater assessment methodology, a default soil/soil-water partitioning coefficient (K_d) is not provided, due to low mobility of Cd in groundwater [16] implying that using the DAF approach will result in an overestimate of groundwater concentration.

The soil/soil-water partitioning coefficient used in Equation 2 is pH-dependent. In the absence of site-specific soil pH, the default recommended soil pH of 6.8 was used in this analysis, corresponding to a Cd soil/soil-water partitioning coefficient of

75 L/kg. The latter coefficient ranges from 17 L/kg at soil pH of 5 to 4,300 L/kg at soil pH of 8 [11]. The equilibrium concentration of Cd in impacted soil is proportional to the soil/soil-water partitioning coefficient (Eqn. 2). Therefore, under acidic soils, the exposure point concentration in soil may be up to a factor of 4.4 lower than the concentration estimated in Table 2. For alkaline soils, the exposure point concentration in soil may be up to a factor of 57 higher than the concentration estimated in Table 2. However, because the soil exposure point concentrations in Table 2 are over two orders of magnitude below screening levels, potential health risks from exposure to soil are highly unlikely under varying soil pH.

The number of building downspouts (25) is based on the commercial building roof area being 25 times larger than a standard residential building with one downspout. The number of downspouts affects the impacted soil area (parameter IA in Eqn. 3), with each downspout discharging onto 1 m² of ground surface area. With additional downspouts, the soil exposure point concentration estimated with Equation 3 would increase proportionally. However, because the soil exposure point concentrations in Table 2 are over two orders of magnitude below screening levels, potential health risks from exposure to soil are highly unlikely under variations in the number of building downspouts.

Another screening level assumption is the module breakage rate. Product return statistics have been obtained in the 2011 fourth quarter from First Solar's warranty manager evaluating global warranty trends (J. Sokol, First Solar, Perrysburg, Ohio, USA, personal communication), including five years of actual performance data with extrapolations to later years of product life, based on an observed decline in breakage rate after the installation and initial operating period. Module breakage is rare, occurring in approximately 1% of modules over the 25 year warranty operating life, including the shipping and installation period. Of these breakages, over one-third occurs during shipping and installation and are removed for takeback and recycling. In addition, a proportion of broken modules have only chipped glass that does not affect the CdTe semiconductor layer. These two considerations considerably reduce the relevant breakage rate for modules that may be subject to leaching by rainfall. Nevertheless in this analysis, a conservative breakage rate of 1% over a 25 year life (0.01%/year) is applied.

The screening level approach used in this evaluation considers each exposure medium (soil, air, groundwater) separately. If an exposure point concentration for a chemical exceeds a screening level, the chemical is of potential concern to human health and requires further risk assessment. Conversely, if a screening level is not exceeded, it is highly unlikely that the chemical may pose a potential health risk in that exposure media. In addition to screening health risks for each exposure medium, cumulative risks across exposure media were considered using the exposure point concentrations in Table 2 in conjunction with U.S. EPA exposure assessment methodology [26,27] and the inhalation unit risk and oral reference dose for Cd (<http://www.epa.gov/nis/subst0141.htm>). Across the exposure media of soil, air, and groundwater, cumulative risks and hazards are below one in one million and the hazard index of 1, respectively, as expected given that the media-specific exposure point concentrations are orders of magnitude below human health screening levels.

Overall, a worst case leaching scenario with screening level fate and transport modeling yields impacts to soil, air, and groundwater that are one to five orders of magnitude below human health screening levels in a California and southern

Germany exposure scenario. Potential exposures to Cd from rainwater leaching of broken modules in a commercial building scenario are highly unlikely to pose a potential health risk to on-site workers or off-site residents.

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