

Greenhouse-gas Emissions from Solar Electric- and Nuclear Power: A Life-cycle Study

Vasilis M. Fthenakis^{1,2,*} and Hyung Chul Kim¹

¹Energy Sciences and Technology Department, Bldg 475B, Brookhaven National Laboratory, Upton, NY 11973, U.S.A.; ²and Center for Life Cycle Analysis, Department of Earth and Environmental Engineering, Columbia University, New York, NY 11027

* Corresponding author. Tel.: +1-631-344-2830; fax: +1-631-344-7650. Email address: vmf@bnl.gov

ABSTRACT

Solar- and nuclear-electricity-generation technologies often are deemed “carbon-free” because their operation does not generate any carbon dioxide. However, this is not so when considering their entire lifecycle of energy production; carbon dioxide and other gases are emitted during the extraction, processing, and disposal of associated materials. We determined the greenhouse-gas emissions (GHG), namely, CO₂, CH₄, N₂O, and chlorofluorocarbons due to materials and energy flows throughout all stages of the life of commercial technologies for solar-electric- and nuclear-power-generation, based on data from 12 photovoltaic (PV) companies, and reviews of nuclear-fuel life cycles in the United States, Europe, and Japan. Previous GHG estimates vary widely, from 40 to 180 CO₂-eq./kWh for PV, and 3.5 to 100 CO₂-eq./kWh for nuclear power. Country-specific parameters account for many of these differences, which are exacerbated by outdated information. We conclude, instead, that lifetime GHG emissions from solar- and nuclear-fuel-cycles in the United States are comparable under actual production conditions and average solar irradiation, viz., 22-49 g CO₂-eq./kWh (average U.S.), 17-39 g CO₂-eq./kWh (South West) for solar electric, and 16-55 g CO₂-eq./kWh for nuclear energy. However, several factors may significantly change this picture within the next five years, and there are unanswered questions about the nuclear fuel cycle that warrant further analyses.

Keywords: greenhouse gas, photovoltaic, nuclear, life cycle

Introduction

The production of energy by burning fossil fuels generates pollutants and carbon dioxide. Since current electricity production heavily relies on fossil fuels, it is envisioned that expanding generation technologies based on nuclear power and renewable energy sources would dramatically reduce future greenhouse gas (GHG) emissions (Hoffert et al., 2002; Service, 2005; Rohatgi et al., 2002). However, all anthropogenic means of energy production, including solar and nuclear, generate pollutants when their entire life cycle is accounted for. Quantifying the environmental and health damage due to such emissions led to the concept of “external” costs (ORNL, 1995; European Commission, 2003). The best-known effort to quantify the external costs of different options is the European Union’s series of ExternE (External Costs of Energy) projects. The methodology starts from emissions generated at specific sources, and follows their impact on receptors through atmospheric dispersion and dose-response functions. These emissions differ for different countries with the particular mixture of electricity used, and the varying methods of material/fuel processing. The latest (2003) ExternE report to the European Commission (European Commission, 2003) showed photovoltaic installations in Germany as emitting 180 g CO₂-eq. /kWh, i.e., ten times higher than the GHG emissions of the nuclear-fuel cycle, and 45% of those of combined cycle (CC) natural-gas power generation in the same country. Further, this document represented PV as having two times higher toxic-gas emissions than the CC cycle. This publication, with its official status, is intended to influence energy-policy decisions, as its foreword says: “The ExternE methodology has been applied in a large number of European and national studies to give advice for environmental, energy and transport policies”. We note that these findings, based on technologies from the late 80s, are far from representing today’s systems and efficiencies. More recent data (i.e., 2000) from the Ecoinvent database (Dones and Heck, 2005) are expected to be used in future ExternE reports. However, even these 2000 data already are outdated. Comparing solar- with nuclear-generated energy although the first is an intermittent and the second is a base load power generation technology, is pertinent especially in evaluating the use of these technologies for production of hydrogen fuel. Our analysis, reported here, is based on the latest (2004-

2005) material- and energy-inventories for solar technologies gathered from twelve U.S. and European PV companies, reviews of LCA studies on the nuclear-fuel cycle, and our original analysis of GHG emissions during the permanent storage of high-level radioactive waste (HLW) in the Yucca Mountain repository. The differences in estimates from different investigators/countries vary by a factor of 33. We assessed the main reasons for such deviations, and determined that, in fact, the minimum and maximum estimates of the life cycle GHG emissions from current U.S. nuclear-fuel cycles, differ only by a factor of 3.

The Solar Electric Cycle

The life-cycle of solar electric-energy generation (i.e., photovoltaic) covers material production (e.g, mining, smelting, refining, purification), solar cell- and PV module-production, balance of system (BOS) production (e.g., inverters, transformers, wiring, structural supports), system operation and maintenance, system decommissioning, and disposal or recycling (Figure 1).

Typically, the life cycles of PV modules and the BOS are studied separately since the former entails more options and more evolution than the latter. The raw materials used for PV modules include glass, semiconductors, electrical conductors, and encapsulation materials. The semiconductors in today's commercial PV systems are silicon (monocrystalline, multicrystalline, amorphous), copper indium diselenide, or cadmium telluride. Crystalline silicon modules constituted about 94% of the 1250 MW of PV system capacity installed in 2004. However, thin-film cells (a-Si, CuIn(Ga)Se₂ and CdTe) have a higher growth rate than c-Si and now comprise about 50% of U.S. production.

The various processes for producing crystalline silicon modules are discussed below and detailed elsewhere (Markvart and Castaner, 2003). Silicon is produced from silica (SiO₂), mined as quartz sand, that is reacted with carbon (wood, charcoal, and coal) in arc furnaces to yield metallurgical-grade Si, and then further purified to solar-grade Si. The Siemens process is the predominant purification method for generating polycrystalline

silicon; it entails reacting silicon with trichlorosilane at 1150 °C. The trichlorosilane itself is manufactured in a catalytic fluidized bed by treating pulverized metallurgical-grade Si with hydrogen fluoride, a process that is highly energy-intensive and wastes about 80% of the initial Si feedstock. Also, it produces semiconductor-grade Si with a purity much higher than that required for solar cells. For years, the PV manufacturing industry used the off-spec electronic scrap; however, with its continuing growth, only 5% is currently (2005) used. Recent LCA studies incorporated a “modified Siemens” process dedicated to producing solar-grade Si for multi-crystalline Si solar cells. To obtain monocrystalline silicon, large ingots of silicon crystals are grown by the Czochralski method from a bath of molten metallurgical-grade Si. Relatively little waste results, but considerable energy input is required. Both polycrystalline and monocrystalline products must be cut into wafers, the basis of the solar cells. Growing ribbons is more efficient in terms of energy and materials because they are produced at the thickness required for the cells. Thin film solar cells are produced by vapor deposition processes from solid (for CuIn(Ga)Se₂ and CdTe) or gaseous (for a-Si) feedstocks.

GHG Emissions from the Solar-electric Fuel Cycle

Alsema and de-Wild recently updated earlier LCAs of all commercial crystalline-silicon PV technologies (i.e., multi- and mono-crystalline wafers and ribbons) to reflect the status of technology in 2004 and early 2005 (Alsema and de Wild, 2006; de Wild and Alsema, 2006; Fthenakis and Alsema, 2006). They comprehensively assessed these PV systems, based on life-cycle inventories collected by the major European and U.S. photovoltaic companies. The primary energy requirement for modules with multicrystalline Si wafers is about 3940 MJ per m² of module area, 72% of which is used producing the wafers. The rated electrical-conversion efficiencies of the PV modules are 13.2% for multicrystalline Si, 14% for monocrystalline Si, and 11.5% for Si ribbons. The electricity produced by a PV system is directly proportional to its efficiency for converting photon energy to electricity, and the intensity of solar radiation at the site.

Fthenakis and Kim (2006) conducted an LCA of the CdTe solar modules currently manufactured by First Solar, Perrysburg, Ohio. These frameless, double-glass, CdTe

modules rated at 9% photon-to-electricity conversion efficiency, have total life-cycle GHG emissions of 18 g CO₂-eq./kWh, a performance ratio of 80% (i.e. 20% losses in inverters, wiring etc) under average U.S. insolation conditions (1800 kWh/m²/yr), and a 30-yr. lifetime. Raugei et al. (2006) estimated GHG emissions of 12 g CO₂-eq./kWh, based on Antec Solar's CdTe PV-module production data in Germany. This difference primarily reflects the "cleaner" average electricity mix in Europe than in the United States, and, to a lesser degree, to our including minor contributions, like overhead electricity, office supplies, transport of personnel, and a detailed inventory of over 400 consumable items. The total primary energy required in the life cycle of the U.S. CdTe PV production was 1200 MJ/m² of module, for which electricity demand in module manufacturing is the most significant contribution (i.e., 55%) (Fthenakis and Kim, 2006).

The BOS for residential rooftop installations was estimated from Alsema's data (Alsema and de Wild, 2006) as 6.1 kg CO₂-eq./m² for array support and cabling, and 125 kg CO₂-eq./kW for inverters. The BOS for utility installations is described in a detailed life-cycle analysis of the 3.5 MW_p multi-crystalline PV BOS installation at Tucson Electric Power's (TEP) Springerville, AZ, field PV plant (Mason et al., 2006). The total primary energy in the BOS life cycle is 542 MJ/m² of installed PV modules. The estimated life-cycle CO₂ emissions are 5.7 g CO₂ -eq./kWh for the BOS under average U.S. insolation. The total GHG emissions over the life-cycle of the whole system then becomes 24 g CO₂-eq./kWh for utility ground-mount installations, and 20 g CO₂-eq./kWh for residential rooftop installations under average U.S. solar irradiation, for modules produced in the United States, and 16 g CO₂-eq./kWh for rooftop applications involving European modules where the electricity mix is more carbon-free.

Perspective analysis

Alsema and deWild predicted clear trends towards 40-50% lower GHG emissions in the crystalline-Si PV cycle within the next 5 years (Alsema and de Wild, 2006) based on the reduction in the wafer's thickness, and the increase in electric-conversion efficiency

underway within the CrystalClear project¹. Corresponding to these targets, GHG emissions with the current Western European average electricity mixture (UTCE) will be 19 g CO₂-eq./kWh (higher in the U.S. electricity mix). On the other hand, about half of c-Si PV manufacturing facilities employ CF₄, a potent GHG for dry etching of Si wafers. We determined that the current use of this gas is 30 kg CO₂-eq./ m² of manufactured solar cells. If unabated, this usage could add up to 6 g CO₂-eq./kWh to the GHG emissions of the c-Si PV lifecycle; however, commercial point-of-use abatement systems with higher than 90% destruction efficiencies are available.

The US manufacturer of CdTe PV projects a linear increase in electrical-conversion efficiency from the current 9% to 12% by 2010; today's laboratory record is 16.5%. Also, tested optimization of the deposition processes in CdTe lines is expected to reduce electricity requirements by about 25% within a couple of years; the corresponding emissions of this cycle will be 11 g CO₂-eq./kWh by 2010 in the current average U.S. electricity supply. A further perspective scenario utilizes PV-derived electricity for manufacturing. Then, assuming that 20% of the manufacturing requirement will be satisfied via on-site solar-electric generation, the GHG emissions would fall by another gram of CO₂-eq./kWh, for a total life cycle of installed CdTe PV of 10 g CO₂-eq./kWh. For a plant manufacturing 25 MW/yr of 12% efficient PV modules, the latest scenario will require a 1.3 MW PV installation on 2.7 acres, which is area available on the roof and parking of the facility.

Further developments are expected in current technologies, and new, cheaper, or more efficient technologies are expected to emerge from ongoing R&D programs. Examples include the new fluidized bed reactor (FBR) for producing solar-grade Si which may cut electricity consumption in Si-purification by up to 90%, expected within 5 years. Also, thinner thin-films, dye-sensitized cells, and organic cells promise lower production costs by 2015 or shortly after, with correspondingly lower GHG emissions.

¹ The target wafer for multi-Si is 150 µm thick (down from the current 285 µm), and its target efficiency 16%, up from the current 13.2%. Efficiencies above 20% have been demonstrated in small size c-Si cells in the laboratory.

The Nuclear-fuel Cycle

The nuclear-fuel cycle consists of mining/milling uranium ores, fuel conversion, enrichment, and fabrication, along with nuclear power plant (NPP) construction/decommission, operation, reprocessing, and ultimately, waste disposal.

Uranium is mined either at the surface or underground. The ore is crushed, ground into fine slurry (milling), and then leached in sulfuric acid. Uranium then is recovered from solution and concentrated to solid uranium oxide (U_3O_8), often called “yellow cake.” Uranium (U_3O_8) is converted into hexafluoride (UF_6) and heated; the UF_6 vapor then is loaded into cylinders where it is cooled and condensed to a solid. ^{235}U is separated from ^{238}U and subsequently enriched either by gaseous diffusion or by gas centrifuge; the first uses about 40 times more electricity than the second. Currently, in the United States enrichment is done only by gaseous diffusion in a plant in Paducah, KY; however, modern plants in Europe and Russia use centrifugation. The global capacity for enrichment is shared roughly equally by the two technologies (Gautrot, 2002). Depending on the prices of uranium and the enrichment service, producing 1 kg of enriched uranium generates about 7 kg of depleted uranium, containing less than 0.35% ^{235}U . This latter could be used as a resource in future, but currently is stored in steel cylinders. Enriched UF_6 is converted to fuel (UO_2) powder and shaped into small pellets that are stacked inside thin fuel rods made of a zirconium alloy or stainless steel; they are then sealed and assembled into fuel assemblies.

In the reactor, the ^{235}U isotope fissions producing heat. Half of the plutonium that is converted from ^{238}U undergoes fission, generating about 30% of energy output from the reactor. One-third of the spent fuel is replaced with fresh fuel every 12 to 18 months. Constructing a nuclear power station requires a large amount of concrete, steel, copper, and aluminum, as well as diesel fuel for machinery. Decommissioning also entails significant amounts of energy. Spent fuels that are highly radioactive and continue to generate heat are stored in pools or dry storage casks at the reactor site for further reprocessing or disposal in the future (the United States does not reprocess fuel). Spent-

fuel rods for disposal are encapsulated in corrosion-resistant metals, such as copper or stainless steel. Most countries plan to use stable rock structures deep underground for permanent disposal. Since the 1950s, depleted UF₆ has been stored in steel cylinders for future disposal or recycling (ANL, 1999).

GHG Emissions from the Nuclear-fuel Cycle -Review

Widely varying estimates of life cycle greenhouse gas emissions were presented in recent LCA studies, ranging from 3.5 to 100 g CO₂-eq./kWh. The biggest differences occur during enrichment, production, and operation (Figure 4). These large differences in GHG emissions during enrichment reflect its type (diffusion vs. centrifuge), as well as country-specific background electricity mixtures. The energy requirement for gaseous centrifuge enrichment ranges from 40-100 kWh/SWU, whereas gaseous diffusion requires 2400-3000 kWh/SWU (DeLucchi, 1991; DeLucchi, 2003; Choppin et al., 2002). The U.S., Australian, Japanese, and Swiss (2) cases in Figure 4 include diffusion-enriched uranium fuel. Upstream electric energy sources also affect the GHG emissions during enrichment. Thus, electricity generation in Switzerland and Sweden is almost entirely fossil-fuel free (respectively, 60% hydroelectric, plus 37% nuclear, and 51% hydroelectric plus 43% nuclear). The best possible case studied that of Vattenfall, of a Swedish utility combining almost fossil-fuel-free electricity and a large fraction (i.e., 80%) of centrifugal enrichment. The worst published cases correspond to the United States where mainly diffusion is used for enrichment (DeLucchi, 1991) and a hypothetical Australian case (ACA, 2001). The dramatic difference in the GHG estimates during the construction and operation stages stem from employing different methodologies. For example, Storm and Smith's estimates of 17 g CO₂-eq./kWh for construction (with a range of 8.5-25.4) and 31 g for operation are based on Economic Input/Output (EIO) analysis (Storm and Smith, 2005; Storm, 2006), Hondo's estimates of 2.8 g CO₂-eq./kWh for construction is based on process data and of 3.2 g CO₂-eq./kWh for operation, is partially based on EIO analysis (Hondo, 2005a; Hondo, 2005b), whereas Dones's corresponding estimates of 0.7 and 0.5 g CO₂-eq./kWh, and Vattenfall's estimates of 0.3 and 0.25 g CO₂-eq./kWh result from process-based analyses (Dones, 2003; Vattenfall, 2004).

The GHG estimate of Storm and Smith is about 50% higher than that for the UK nuclear-fuel cycle obtained from a cost analysis of the Sizewell B nuclear power plant (NPP). Even greater uncertainty applies to decommissioning, a stage for which actual data do not exist. The Vattenfall report assigns only 0.5 g CO₂-eq./kWh for construction and decommissioning together (Vattenfall, 2004; Bodlund, 2006), whereas Storm and Smith assign a range of 17-34 g CO₂-eq./kWh (Storm and Smith, 2005; Storm, 2006).

The EIO's approach allows gleaning more emissions (sometimes beyond the system boundary) than the process-based LCA method (Suh et al., 2004). The latter, however, method may not fully capture the real impact of GHG emissions during construction. In studies by Dones (2003), and Hondo (2005), the GHG estimates are closely linked to the amount of steel and concrete, which account for over 95% of the total impact. In fact, in terms of mass and GHG emissions, they are the two major materials for both NPPs and the BOS of PV plants. Thus, we examined the deviation of the two methods in determining GHG emissions from steel and concrete for constructing both types of plants. The EIO-based LCA approach gave 10-20 times higher GHG emissions than the process-based one. It also gave a factor of 3 times higher GHG estimates for the PV BOS system than did a process-based approach. The gap between these two approaches is especially striking for the NPP's steel components, the most important material in terms of GHG emissions. Our analysis shows that, although the construction cost per kWh electricity produced is similar for both structures, the amount of steel used for the NPP is 3-10 times less than the PV BOS. There are two ways to explain this anomaly. First, the steel components used in NPPs may be more expensive, labor intensive, and perhaps, energy intensive. Second, the cost of constructing NPPs is relatively high due to expenses unrelated to steel and concrete components. For example, instrumentation- and control- (I&C) related energy is not included in any of the process-based or materials-based analyses, although a NPP is expected to have a much higher fraction of such costs than a PV plant. Also, the incremental energy required for manufacturing finite, high specification, components from materials listed in data-bases, is not included in the process-based analyses of the nuclear fuel cycle, but is expected to be captured with EIO

analysis. The degree of underestimating and overestimating in each approach deserves further analysis.

GHG Emissions from the U.S. Nuclear-fuel Cycle; an Update

We investigated the GHG emissions from the stages of the U.S. nuclear-fuel cycle, based on the most current statistics, databases, and literature. It includes some stages that DeLucchi's early work of the U.S. cycle omitted (DeLucchi, 1991), such as spent-fuel disposal and the de-conversion of depleted uranium.

The baseline GHG factor for mining and milling in the U.S. nuclear-fuel cycle is estimated to be 1.7 g CO₂-eq./kWh, calculated from the estimated energy consumption to explore for, mine, and mill uranium, and the 1987-1990 average uranium-ore grade of 0.2% U₃O₈ (DeLucchi, 1991). Energy usage for mining and milling ores increases considerably with low-grade ores (Chapman and Roberts, 1983). In Figure 5, we show GHG emissions under a "best-case" scenario with Canadian uranium (12.7% U), and "worst-case" GHG emissions with uranium concentrations typical of Australian ores (0.05% U).

Although U.S. nuclear reactors demand 10-12 million Separative Work Units² (SWUs) of enrichment annually, the Paducah plant currently produces 5 million SWUs (U.S. NRC, 2004), a fraction of which is exported. According to five-year average data (1998-2002) (EIA, 2003), local diffusion enrichment supplies about 34% of U.S. needs. A considerable amount of highly enriched weapon-grade uranium (>90% ²³⁵U), originally produced for warheads in Russia is diluted to fuel-grade (<5% ²³⁵U) and burned in U.S. nuclear reactors, accounting for 36% of the enrichment needed. The other 30% is imported from France, the Netherlands, the United Kingdom, and Germany. For weapon-grade fuel, we assumed that it was enriched totally by centrifuge, and, for the imports, we used the actual country-specific method and corresponding energy use. The

² The standard measure of enrichment services which decided by the amount of uranium processed and the increased level of concentration of the ²³⁵U isotope relative to remainder.

weighted average fuel burnup between 1998-2002 was 42 MW_{th}d/kgU, and the average enrichment concentration (product assay) was 3.8%, values derived by combining burnups and initial uranium contents of all U.S. PWRs and BWRs (EIA, 2002; EIA, 2004). The life-cycle GHG emission factor of upstream electricity is about 695 g CO₂-eq./kWh (Kim and Dale, 2005). The baseline scenario assumes that 20% of the electricity supply to the diffusion enrichment plant at Paducah, KY is directly from coal power plants, and 80% from the grid of the Southeastern region of the North American Electric Reliability Council (USEC, 2005a). The best-case scenario assumes that the electricity for enrichment is supplied from the average U.S. grid mix, with a requirement of 2400 kWh/SWU of energy (WNA, 2003). The worst-case scenario assumes that enrichment requires 3000 kWh/SWU of energy (DeLucchi, 2003) and that U.S. gas diffusion plants obtain their power primarily from coal-fired plants in the Ohio Valley (ORNL, 1995; DeLucchi, 1991). These conditions give GHGs of 11-, 13-, and 20-g CO₂-eq./kWh, respectively, for best, baseline, and worst-case scenarios for the enrichment (Figure 7).

The estimates of GHG emissions from constructing NPPs are based on the steel and concrete materials used, i.e., 36,000 and 480,000 metric tons, respectively, for an 1100-MW NPP (ORNL, 1995). Our best case is based on the Swedish (Vattenfall's) materials-based approach, while the worst case is estimated from a construction cost of \$ 3.3 billion (1989\$) by employing the EIO-LCA approach (ORNL, 1995). The GHG emissions from conversion/fabrication are taken from the fuel- and material- use reported in the U.S. DOE report (U.S. DOE, 1983). We selected the impacts of the Swedish and Japanese cases as the best and worst, respectively, for US conversion/fabrication.

We estimated the GHG emissions during plant operation from the amount of fuel used for start-up of the auxiliary steam generators and in-plant heating, along with the plant's annual running expenses. The published fuel usages range from 0.1 to 3 million gal/yr (DeLucchi, 1991; Rotty et al., 1975; Bowers et al., 1987), that is, 0.1-6 g CO₂-eq./kWh. Using a maintenance and operation cost guideline for a light-water reactor, the EIO-LCA based on annual costs allocates 2.4 g CO₂-eq./kWh of GHG emissions for miscellaneous

supplies, makeup materials and chemicals, and non-radioactive waste management combined (Bowers et al., 1987). We estimated GHG emissions of 2.5, 3.9, and 10.8 g CO₂-eq./kWh for the best, baseline, and worst cases, correspondingly. The worst case assumes that 10% of reactor materials are replaced every 10 years through regular maintenance; no replacement is assumed in the other two cases.

For the temporary storage of spent nuclear fuel we used ~1 g CO₂-eq. /kWh reported in the Australian Coal Association's (ACA's) research study (ACA, 2001). For permanent storage, we used the materials and energy data cited in the Environmental Impact Statement and other supporting literature for the Yucca Mountain project (U.S. DOE, 2002a; U.S. DOE, 2002b), and estimated the potential GHG emissions for constructing, transporting spent fuel, operating, monitoring, and closing a "permanent" repository to be in the range of 0.9-1.9 g CO₂-eq./kWh, depending on the mode of operating temperature³. For de-converting depleted uranium, we used as baseline impact the arithmetic average of that from the U.S. uranium-fuel fabrication and conversion stages, i.e., 0.5 g CO₂-eq./kWh.

In general, the baseline U.S. nuclear-fuel-cycle GHG emissions are comparable with those shown in Australian and the Japanese studies. However, our estimates are much higher than those from European studies. Such differences were expected due to the difference in America and Europe of enrichment methods and grid-emission factors. For our "worst case", we used the EIO method for estimating GHG emissions in construction, knowing that it likely overestimates them, while the material-based estimates shown in the "best case" and the "baseline case" probably underestimate the same.

These estimates can be used in comparing nuclear with other grid technologies. However, if the comparison is made with decentralized technologies that feed directly into the demand application (e.g., rooftop PV installations), then a transmission loss of 6% (EIA,

³ Higher temperature mode operation is defined as an operating mode in which the surface temperature of the waste package exceeds the boiling point of the water, while lower temperature operation mode scenarios are a series of options that keep the surface temperature below it.

2005) should be assigned to the nuclear fuel cycle; this would increase by 6% the associated GHG emissions. On the other hand, the impact of storage should be considered in stand-alone PV systems.

Perspective analysis of the nuclear-fuel cycle

Eventually, the U.S. enrichment services will shift to gas-centrifuge enrichment from the current gaseous diffusion method, although the transition could be delayed by opposition to building an enrichment plant in Piketon, OH. If built on schedule, the proposed centrifuge enrichment plant would reach to an annual capacity of 3.5 million SWU by 2010 (USEC, 2005b). Considering that the typical residence time of uranium fuel is 3-5 years, U.S. NPPs would use gaseous diffusion-enriched-fuel at least until 2015. When this technology is totally displaced by centrifugal enrichment, then the total life cycle GHG emissions from the U.S nuclear-fuel cycle would be ~ 12 g CO₂-eq./kWh, assuming today's upstream electricity mix.

Another important parameter is the uranium concentration in the ore. The energy for mining and milling uranium from ores increases with its decreasing concentration (Chapman, 1975; Chapman and Roberts, 1983; Storm and Smith, 2005). Chapman (1975) estimated that the concentration of uranium must be above 0.002% for the extracted fuel to produce more energy by fission than the energy required for its processing; however he did not account for the energy required in conversion, decommissioning, operation, and waste disposal. Recently, Storm and Smith (2005) showed that uranium with ore grade of 0.01% or lower could be valueless from a full life-cycle perspective. However, these authors did not account for improvements in the efficiency of mining/milling likely to occur with time, nor that an increase in price could lead to increased exploration for uranium, which, in turn, could delay the need for mining of low-grade ores. Nonetheless, it is evident that at some level, the usability of nuclear-fuel resources is limited by the second law of thermodynamics. The issue becomes more pertinent in scenarios assuming several new NPPs in the United States and elsewhere. In fact, based on current reserves, it is believed that the world's uranium supply would fall

short of meeting future demands. The IAEA's recent study concludes that, to meet the potential, gradual increase in uranium demands until 2050, more efforts are required to discover large, high-grade ores by 2010 so that new uranium production from them could begin by 2025 (IAEA, 2001). Currently, the United States produces less than 5% of the total uranium demand of its nuclear reactors, and relies on the free-trade agreement of 1988 with Canada to secure enough uranium sources for any emergencies (USGS, 2003).

CONCLUDING REMARKS

An evaluation of alternative energy technologies for their potential to decrease GHG emissions requires careful analyses of all the stages in the life of the fuels and devices. Quantifying such emissions in both present-day and prospective contexts is paramount for comparing the environmental profiles of different electricity-generation options. GHG emissions in the lifecycles of solar electric- and nuclear-fuel- technologies vary, depending on the efficiencies of upstream energy, local conditions, and other assumptions. Previous studies showing nuclear technology to have a clear GHG advantage over PV are greatly outdated; while both life cycles were described by old data, the problem is especially pertinent to the fast evolving PV industry. The emissions from the life-cycles of the two cycles are comparable under today's average U.S. conditions. Established trends in the PV cycle are expected to keep further reducing emissions in the solar-electric cycle, while planned centrifugal fuel enrichment will significantly reduce GHG emissions in the U.S. nuclear cycle after 2015. However, the estimates associated with the latter bear large parametrical- and methodological-uncertainties that can be reduced only with detailed and all-inclusive process-based analysis. In addition, estimates based on concepts (e.g., re-conversion) and untested stages (e.g., Yucca Mountain permanent repository) need to be examined in greater detail.

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Figure 1. The Life Cycle of PV Systems

Figure 2. GHG Emissions: a) European Production & Installation; average solar radiation ($1700 \text{ kWh/m}^2/\text{yr}$), rooftop mounted; b) Si European production, US module production and installation; average solar radiation ($1800 \text{ kWh/m}^2/\text{yr}$), rooftop mounted; c) Si European production, US module production and installation, SW solar radiation on latitude tilt surface ($2280 \text{ kWh/m}^2/\text{yr}$), rooftop mounted; d) Si European production, US module production and installation, solar radiation of Springerville, AZ ($2060 \text{ kWh/m}^2/\text{yr}$), ground mounted.

Figure 3. The Nuclear Fuel Life Cycle (without reprocessing)

Figure 4: Estimates of GHG emissions in the nuclear-fuel cycle

Figure 5: Breakdown of the GHG emissions from the current U.S. nuclear-fuel cycle.









