Life cycle freshwater ecotoxicity, human health cancer, and noncancer impacts of corn ethanol and gasoline in the U.S.

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1. Introduction

Recent global interests in biofuels have been driven by concerns about energy security and the negative environmental impacts of fossil fuels, particularly their impact on climate change. Supported by a variety of policies (Runge and Johnson, 2008), production of biofuels in the U.S. has increased significantly in the last decade, especially after 2005 (Fig. 1). Corn ethanol has remained the primary source of domestic biofuel, because cellulosic biofuels have yet to be commercialized (Schnoor, 2011). As a result, corn has been increasingly diverted from livestock feed to ethanol conversion. In 2010, ethanol overtook feed to become the largest use of corn in the U.S., accounting for about 44% of domestic corn production (USDA, 2013).

Despite rapid expansion, whether corn ethanol provides energy and carbon benefits remains unclear. Early life cycle studies reported negative results in both regards (Patzek, 2004; Pimentel and Patzek, 2005), but these studies failed to account for the benefit of ethanol co-products (Farrell et al., 2006). At the same time other studies found that corn ethanol could offer modest energy and carbon benefits (Hill et al., 2006; Kim and Dale, 2005; Wang et al., 2007). However, these findings have been further challenged by subsequent research on land use change (LUC), soil N2O emissions, and meta-error analysis of net energy (Crutzen et al., 2007; Fargione et al., 2008; Murphy et al., 2011; Searchinger et al., 2008).

In particular, the question of whether corn ethanol mitigates climate change seems to hinge on the magnitude of indirect LUC (Kim and Dale, 2011; O’Hare et al., 2011; Plevin et al., 2010; Wallington et al., 2012; Wang et al., 2011) and the rate of soil N2O emissions (Davidson, 2009; Smith et al., 2012; Smith and Searchinger, 2012).

Additionally, there has been a growing awareness that focusing narrowly on carbon and energy does not capture the full environmental impacts generated by transportation fuels (Guinee et al., 2011; Hill et al., 2009; Jacobson, 2009; Laurent et al., 2012; Von Blottnitz and Curran, 2007; Yang et al., 2012). Other impacts, such as smog formation, eutrophication, acidification, biodiversity, water use, and land use requirement, should also be considered in addressing the environmental sustainability of fuel substitution. Kim and Dale (2008), for example, found that corn ethanol results in higher eutrophication, smog formation, and acidification effects than gasoline. Ethanol was also shown to have a much higher water footprint than gasoline due to intensive use of irrigation water in...
corn growth (Chiu et al., 2009; Fingerman et al., 2010; Wu et al., 2009). Yang et al. (2012) explored the relative significance of a wide range of impacts when comparing corn ethanol and gasoline, suggesting that ethanol has a higher overall impact, and its replacement of gasoline results in burden-shifting from carbon and energy to mainly water quality and scarcity issues. In a recent study, Tessum et al. (2012) built a spatially-explicit life cycle inventory comparing the contributions of U.S. gasoline and ethanol to human health related air pollution.

In comparison, however, the life cycle health-related cancer, noncancer (e.g., reproductive, developmental, and nurotoxic), and ecotoxicity effects of corn ethanol and gasoline are relatively less well understood. These effects are caused by toxic pollutants released throughout each fuel’s life cycle, such as pesticide emissions from corn cultivation, heavy metals emissions from oil extraction, and volatile organic compounds (VOCs) from vehicle exhaust. Studies of the sort have centered either on a single process alone, rather than an entire life cycle, or on a limited number of air pollutants (Cook et al., 2011; Graham et al., 2008; Jacobson, 2007).

There have been only two life cycle studies that explicitly covered all the three impacts associated with corn ethanol and gasoline (Curran, 2007; Yang et al., 2012). However, it seems both studies failed to account for some important emissions. In a recent study, Curran (2007), for example, did not consider pesticides, and Yang et al. (2012) omitted heavy metals contained in fertilizers and VOCs from vehicle exhaust. In addition, for impact assessment both studies relied on the earlier version of the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) model (Bare et al., 2003), which seems to fall short of evaluating the toxic environmental impacts of corn ethanol. Over 100 types of pesticides, for example, were applied in corn growth in 2005 (USDA, 2006), of which only about 10 types are covered in TRACI1 (Bare et al., 2003).

This study seeks to improve our understanding of the life cycle cancer, noncancer, and freshwater ecotoxicity impacts of U.S. corn ethanol and gasoline. Compared with previous research, this study compiles data on toxic pollutants released at each life cycle stage of both fuel systems and evaluates their impacts based on the recently published USEtox model. USEtox was developed specifically for assessing chemical-related impacts (Hauschild et al., 2008) and covers >3000 environmental stressors. Specific objectives of the study include identification of key stressors in the corn ethanol system and opportunities for reducing ethanol’s toxic impacts, given its likely continuous dominance on the U.S. biofuels market.

2. Material and methods

2.1. Scope

The system boundary of the study is drawn to cover feedstock production, fuel conversion, transport, and vehicle operation for both fuel systems (see Fig. 2). The functional unit is set as 1 km driven, with an average passenger car (APC) chosen for gasoline and a flexible fuel vehicle (FFV) running on E85 (85% ethanol by volume) for corn ethanol (ANL, 2012). Corn production is studied at the state level, considering agriculture’s high spatial variation due to local topographic and climatic influences (Miller et al., 2006; Yang et al., 2012). Also, to capture technology change for ethanol conversion and possible temporal variation in corn production technology, this study evaluates corn ethanol produced in three years: 2001, 2005, and 2010. There are two reasons for selecting the three years: 1) they span the “ethanol decade” (Wallander et al., 2011), in which production of both corn and ethanol expanded significantly, and 2) data on on-farm agrichemical and fuel use at the state level are available only for these years.

2.2. Life Cycle Inventory (LCI): computational methods

The computation of the life cycle inventory in this study is based on the process method (Suh and Huppes, 2005), divided into foreground and background matrices. The foreground matrix includes inputs and outputs compiled for processes defined in the system boundary. The background matrix reflects the economy in which the studied system is embedded and is used to provide environmental information on inputs used in foreground matrix. Equation (1) shows the computational structure, following Strømman et al. (2009).

\[
E = \left[ \begin{array}{c} B_f \\ B_b \end{array} \right] \left[ \begin{array}{cc} A_f & 0 \\ A_b & A_b \end{array} \right]^{-1} \left[ \begin{array}{c} y \\ 0 \end{array} \right]
\]

In Equation (1), \(A_f\) and \(A_b\) denote foreground technological matrices, and \(B_f\) denotes foreground environmental matrix – direct

![Fig. 1. Ethanol production in the US from 1980 to 2011 (RFA, 2013).](image1)

![Fig. 2. Process flow diagram of corn ethanol (E85) and gasoline.](image2)
emissions from the productive activities included in $A_f$ and $A_{fb}$, $A_b$ and $B_b$ denote background technological and environmental matrices. In this study, $A_f$ is a simple $4 \times 4$ matrix made up of the four processes defined in the system boundary, while $A_{fb}$ complements $A_f$ with more inputs (e.g., fertilizers and pesticides) used to produce the outputs (e.g., corn) in $A_f$. $E$ is total environmental intervention induced by $y$, set as 1 km driven. Background matrices $A_b$ and $B_b$ are taken mostly from the Ecoinvent database (Frischknecht and Rebitzer, 2005) and partly from the USLCI database (NREL, 2010) (see Section 2.4).

2.3. Inventory construction for corn ethanol

This study focuses on compiling foreground process inputs and outputs ($A_f, A_{fb}$, and $B_f$) and relies on background matrices, $A_b$ and $B_b$, for indirect emissions embodied in inputs (i.e., $A_{fb}$). Given the scope of the study, only toxic emissions contributing directly to human health cancer, noncancer, and freshwater ecotoxicity effects are inventoried in $B_f$. For example, even though corn production causes an array of on-site environmental problems due partly to use of nitrogen fertilizers (Socolow, 1999), $B_f$ only includes emissions (e.g., pesticides) contributing directly to human health cancer, noncancer, and ecotoxicity effects (see 2.3.1). Table 1 summarizes the inputs for and emissions from the major foreground processes covered in this study.

2.3.1. Corn production

Data on state-level fertilizer and pesticide use in corn cultivation for all three years are from US Department of Agriculture (USDA) surveys (USDA, 2002, 2006, 2011). The surveys cover 19 program states: Colorado, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Michigan, Minnesota, Missouri, Nebraska, New York, North Carolina, North Dakota, Ohio, Pennsylvania, South Dakota, Texas, and Wisconsin. Details on state agrochemical use can be found in the Supporting Information (SI), which is published alongside this manuscript on the JCLP website. Figures and tables from the SI are designated “SX” here (e.g., Fig. S1). Data on state-level fuel use in corn cultivation and harvesting for the year 2001 are from another USDA survey (2004), which covers all the states above. Fuel data for 2005 are from a recent USDA report on the net energy balance of corn ethanol (Shapouri et al., 2010), which covers 9 of the states above. Similar fuel data for 2010, however, are unavailable thus far. Wherever data gaps exist, fuel data for the previous year is used, assuming the same fuel intensity (MJ per kg corn produced) between the two years (see Fig. S1).

Major on-farm sources of human and ecological toxicity are: 1) heavy metals contained in fertilizers, 2) toxic air compounds from farm equipment, and 3) pesticide emissions. Heavy metal contamination to farmland through fertilizer use and other pathways (e.g., atmospheric deposition) have long been recognized (Williams and David, 1973, 1976) and studied for many countries (Luo et al., 2009; McLaughlin et al., 1996; Nicholson et al., 2003; Nziguheba and Smolders, 2008). This study inventories only heavy metals contained in phosphate (see Table S11) (Mortvedt, 1996), given that it is the major inducer of metals to soils (Nziguheba and Smolders, 2008) and its metal content has been relatively well studied (EPA, 1999). Note that atmospheric deposition is another major source of heavy metals to agricultural soils (Luo et al., 2009; Nicholson et al., 2003) but is left out here because it is not directly attributable to agricultural activities.

Operation of farm equipment emits air pollutants, some of which are toxic and pose potential health effect. Of particular concern is diesel engine exhaust, which has been classified by the International Agency for Research on Cancer (IARC) as carcinogenic in humans based on Attfield et al. (2012) and Silverman et al. (2012). Debate continues over its nature, however, as subsequent research has called the two studies into question (Crump and Van Lindingham, 2012; Gamble et al., 2012; Morfeld, 2012). Nevertheless, air emissions from diesel and other fuels combusted in farm equipment are inventoried, with emission factors for various organic compounds derived from (EPA, 1995a).

The estimation of pesticide emissions hinges on impact assessment method specification. A detailed description is thus saved to Section 2.5, where the impact assessment model Usetox is introduced. Presented here is a background picture of pesticide use in different years. Over 80 different pesticides were recorded for 2001 corn growth, with usage data published for 52 of them in the form of active ingredients (USDA, 2002). Information on the rest was withheld to avoid disclosing data from individual farms. Likewise, over 100 pesticides were documented for 2005 and 2010 with usage data revealed for 55 and 36 of them, respectively. Despite the undisclosed data, the published pesticides amount to around 95% of the total applied in all three years. Herbicides have been applied most widely in all three years, covering up to 98 percent of planted areas, but the herbicide mix has shifted over time. In 2001, Atrazine was the most commonly used herbicide, followed by Acetochlor and S-Metolachlor. In 2005, Atrazine continued its lead, but Glyphosate iso, salt rose to the second most commonly used herbicide, eventually passing Atrazine in 2010 as the top herbicide in terms of both areas treated (%) and total applied (kg).

2.3.2. Ethanol conversion

The study selects dry-mill ethanol conversion for analysis because it has become the dominant technology in U.S. corn ethanol production (Mueller, 2010). Technology change from 2001 to 2010 is approximated based on literature (Shapouri et al., 2002; Wu, 2008; Mueller, 2010). Overall, the industry has achieved enhancements in yield and in energy and water efficiency, and has utilized an increasingly cleaner mix of process energy as the proportion originating from coal has declined. Data on process chemicals and enzymes are from MacLean and Spatari (2009).

Table 1

<table>
<thead>
<tr>
<th>Feedstock production</th>
<th>Fertilizers, N, P₂O₅, K, etc.</th>
<th>Pesticides atrazine, 2, 4-D, etc.</th>
<th>Energy diesel, gasoline, etc.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fuel conversion¹</td>
<td>Chemicals NH₃, enzymes, etc.</td>
<td>Energy natural gas, coal, etc.</td>
<td>Pesticides acetone, acrolein, etc.</td>
</tr>
<tr>
<td>Vehicle operation</td>
<td>Corn Ethanol, gasoline</td>
<td>VOCs 1,3-butadiene, etc.</td>
<td></td>
</tr>
</tbody>
</table>

¹ For ethanol conversion, 80% of environmental burden is allocated to ethanol, and 20% to its co-product, Distiller’s Grains with Solubles. For crude oil conversion, mass is used for allocation: about 0.41 kg gasoline, 0.21 kg diesel, 0.9 kg kerosene, and 0.026 kg liquefied petroleum gas produced per kg crude oil refined (NREL, 2010).
On-site activities affecting human and ecological health include energy combustion and wet grain drying that releases toxic VOCs like formaldehyde and acetic acid. Data on air emissions from combustion of coal and natural gas are from Ecoinvent (Frischknecht and Rebitzer, 2005), and on speciated VOC emissions are derived from emission tests conducted by the Minnesota Pollution Control Agency (Brady and Pratt, 2007). Details on ethanol conversion technology used in different years and speciated VOC emissions can be found in SI.

Another relevant issue is the allocation of environmental burdens to the co-products of ethanol conversion, primarily Distiller’s Grains with Solubles (DGS). Allocation in LCA is dealt with mainly through the partitioning method and system expansion (Ekvall and Finnveden, 2001; Guinee et al., 2002; Suh et al., 2010). The partitioning method splits process inputs and outputs between the multiple products based on their energy, mass, or market value, depending on the physical or economic relationships between the process and the multiple products (Suh et al., 2010). System expansion extends the original system to include equivalent products substitutable to co-products. Both methods have been applied in corn ethanol studies (Curran, 2007; Hill et al., 2006; Kim et al., 2008). Allocation in LCA is dealt with mainly depending on the physical or economic relationships between the process and the multiple products (Suh et al., 2010). System expansion extends the original system to include equivalent products substitutable to co-products. Both methods have been applied in corn ethanol studies (Curran, 2007; Hill et al., 2006; Kim et al., 2008). It appears that choosing either allocation methods (Yang et al., 2012), or any basis in the partitioning method (Curran, 2007), does not change the relative ranking of corn ethanol in relation to gasoline. Based on

2.3.3. Transport

Data on the logistics system are adopted from recent research (Yang et al., 2012), in which three transportation phases are differentiated. First, corn is shipped from farms to refinery plants by heavy-duty trucks over a distance of about 60 km. Ethanol is then shipped from plants to blend terminals by both rails and trucks.

### Table 2

<table>
<thead>
<tr>
<th>Composition of oilfield produced water</th>
<th>USLCI Crude oil emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>mg/l produced water</td>
<td>mg/kg crude oil</td>
</tr>
<tr>
<td>Min</td>
<td>Max</td>
</tr>
<tr>
<td>Chloride</td>
<td>80</td>
</tr>
<tr>
<td>Sulfate</td>
<td>1.2</td>
</tr>
<tr>
<td>Phenols</td>
<td>0.009</td>
</tr>
<tr>
<td>Calcium</td>
<td>13</td>
</tr>
<tr>
<td>Sodium</td>
<td>132</td>
</tr>
<tr>
<td>Magnesium</td>
<td>8</td>
</tr>
<tr>
<td>Iron</td>
<td>0.1</td>
</tr>
<tr>
<td>Alumium</td>
<td>310</td>
</tr>
<tr>
<td>Boron</td>
<td>5</td>
</tr>
<tr>
<td>Barium</td>
<td>1.3</td>
</tr>
<tr>
<td>Cadmium</td>
<td>0.005</td>
</tr>
<tr>
<td>Chromium</td>
<td>0.02</td>
</tr>
<tr>
<td>Copper</td>
<td>0.02</td>
</tr>
<tr>
<td>Lithium</td>
<td>3</td>
</tr>
<tr>
<td>Manganese</td>
<td>&lt;0.004</td>
</tr>
<tr>
<td>Lead</td>
<td>0.002</td>
</tr>
<tr>
<td>Strontium</td>
<td>0.02</td>
</tr>
<tr>
<td>Titanium</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Zine</td>
<td>0.01</td>
</tr>
<tr>
<td>Arsenic</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>Mercury</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>Silver</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Beryllium</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

a Data source: (Fakhru’Razi et al., 2009; Igunnu and Chen, 2012).
b Water-to-oil ratio in the US: 7.6 (Clark and Veil, 2009).
c Data name: Crude oil, at production (NREL, 2010).

Rairs assume 66% of the task over a distance of about 480 km, while trucks assume the remaining 34% task over a distance of about 130 km. Ethanol is distributed by trucks from blend terminals to refueling stations over a distance of about 16 km. Data on both direct and indirect emissions associated with these transport modes are from the Ecoinvent database.

2.3.4. Vehicle operation

Health effects of tail pipe emissions from the Flexible Fuel Vehicle (FFV) running on E85 have occasioned intensive research recently (Cook et al., 2011; Graham et al., 2008; Jacobson, 2007; Yanowitz and McCormick, 2009). Major pollutants studied include formaldehyde, acetaldehyde, benzene, 1,3-butadiene, and ethanol. To estimate these pollutants for FFVs, this study first uses the model Motor Vehicle Emission Simulator (EPA, 2012) to compute associated emissions from the Average Passenger Car (APC) and relies on a review by Yanowitz and McCormick (2009) which provides data on the percentage differences in these emissions between FFVs and APCs. Final emissions results are expressed per km driven (see Table S14 & S15). Note that environmental impacts of vehicle production are not considered in this study.

2.4. Inventory construction for gasoline

Tailpipe emissions from the Average Passenger Car (APC) have already been computed in deriving FFV emissions in Section 2.3.4. The pre-combustion gasoline LCI is taken from the USLCI database (NREL, 2010). Note that the crude oil data included in USLCI are specific to onshore oil technology, which accounts for three quarters of domestic oil production (Allen et al., 2012). Although it seems USLCI compiles data on a large number of water pollutants, it lacks detailed information on their fate after generated with crude oil.

A preliminary calculation, assuming that all pollutants enter freshwater, shows that crude oil extraction dominates all impacts in comparison to other life cycle stages within the gasoline system, due to barium and chromium emissions (see Fig. S2). Comparing the preliminary results with Ecoinvent crude oil data also shows that crude oil extraction in the U.S. seems much more toxic than crude oil extraction in other countries like Nigeria (see Fig. S3 & S4). These results lead to further investigation into US crude oil water pollution. Literature shows that the crude oil heavy metal emissions from USLCI are largely within the range of heavy metal concentrations in produced water, the major source of water pollution for oil extraction (see Table 2). This finding suggests that the original USLCI crude oil emissions data are likely about pollutants in produced water. Furthermore, most of produced water from onshore wells is injected underground to help maintain formation pressure (Clark and Veil, 2009); pollutants contained in the water are unlikely to come into direct contact with freshwater environments. These findings indicate that the initial assumption used in the preliminary calculation is invalid. Considering that 98% of onshore well produced water in the US is injected underground (Clark and Veil, 2009), it is assumed that the remaining 2% enters freshwater. Accordingly, USLCI crude oil water emissions are downscaled to 2% of their original values. Meanwhile, the literature data as shown in Table 2 are used to represent the uncertainty ranges for the new water emissions data. This adjustment brings U.S. crude oil toxic impacts into better agreement with corresponding estimates for other countries from Ecoinvent database (see Fig. S3 & S4).

2.5. Life Cycle Impact Assessment (LCIA): characterization model

USETox is a consensual and harmonized model developed especially to assess chemical-related impacts, and was built mainly
in response to 1) the low coverage of chemicals in previous characterization models and 2) the significant variations among different models (Hauschild et al., 2008). To compute characterization factors, USEtox identifies the cause-effect chain of a chemical emitted into certain environments and its toxicological effects, and quantifies this chain as is shown in Equation (2) (Rosenbaum et al., 2008).

\[
\text{CF} = \frac{\text{EF}}{\text{XF}} \times \text{FF}
\]

(2)

\( \text{FF} \) indexes fate factors in day, \( \text{XF} \) indexes exposure factors in cases/kg intake for human toxicity and the potentially affected fraction of species (PAF) in \( \text{m}^3/\text{kg} \) for ecotoxicity. \( \text{CF} \) for human toxicity, expressed in comparative toxic units (CTUs), provides estimated increase in morbidity in the total human population per unit mass of a chemical emitted (cases per kg). \( \text{CF} \) for freshwater ecotoxicity, expressed in comparative toxic units (CTUs), provides an estimate of the potentially affected fraction of species integrated over time and volume per unit mass of a chemical emitted (PAF \( \text{m}^3 \) day \( \text{kg}^{-1} \)).

Given space limitations, the reader is referred to Rosenbaum et al. (2008) for details regarding the USEtox model.

For most of the emissions compiled in this study, characterization factors from USEtox are directly applicable, but USEtox does not cover the fate of pesticides after application (Berthoud et al., 2011). Therefore, further effort is needed to approximate their fractions in different compartments before connecting them with USEtox characterization factors. Table 3 presents how pesticide emissions to different compartments are approximated.

3. Results and discussion

3.1. Freshwater ecotoxicity

Fig. 3 shows that the potential freshwater ecotoxicity impact induced by E85 appears to be much larger than that by gasoline, largely due to pesticide emissions. Key pesticides identified include atrazine, acetochlor, chlorpyrifos and cyfluethrin, mainly because of their leaching and runoff into aquatic systems. Second, large regional variation as represented by error bars in Fig. 3 is observed, mainly because pesticide use differs among states. States like Pennsylvania, for example, are found to have pesticide intensity (kg pesticide used/kg corn produced) 4–6 times that of states like North Dakota. Third, pesticide intensity does not explain the downward trend in the overall ecotoxicity impact of E85. In fact, the average amount of pesticides used to produce corn for ethanol in 2010 is slightly larger than that in 2005 (2.7E-4 vs. 2.6E-4 kg/kg corn). Instead, what drives the downward trend is the growing share of glyphosate and related compounds — which are much less toxic than the key pesticides identified — in the total pesticides applied (Section 2.3.1). Cyfluethrin, for example, has a freshwater ecotoxicity nearly one million times that of glyphosate iso. salt according to USEtox (Rosenbaum et al., 2008).

3.2. Human toxicity

Fig. 4 shows that E85’s potential cancer impact is roughly equivalent to that of gasoline. First, it is worth noting that the potential cancer effect associated with tailpipe emissions from E85 is twice that of gasoline, but their contribution in both systems is small compared with other processes. Second, the fuel conversion stage in both systems appears to be an important source of cancer impacts. For crude oil refining, it is found that emissions of chromium to water account for most of the impact. For ethanol conversion, process energy is the major contributor, which also explains its decreasing impact over time with increasing yield and energy efficiency (S12). Particularly, the declining use of coal (Mueller, 2010) plays a large part. For corn production, the embodied impacts of nitrogen and phosphate fertilizers, heavy metals contained in phosphate, and diesel combustion are responsible for most of the potential cancer impact. Finally, because the application intensities of fertilizers and diesel are not significantly different between states, regional variation in the overall cancer effect for E85 is small.

Fig. 5 shows that the potential noncancer impact generated by E85 appears much larger than that generated by gasoline. For E85 system, corn growth constitutes the major contributor largely because of the release of heavy metals from phosphate fertilizers, as reflected in the small pink columns in Fig. 5. On average, heavy metals — mainly cadmium — account for around 40% of the overall noncancer impact of E85, and nearly 80% of that of corn growth (see SI for detailed discussion on cadmium concentration in phosphate). Because state use of phosphate varies significantly, large regional variation is observed in Fig. 5. For example, phosphate intensity (kg/kg corn produced) in Kentucky is nearly 7 times that in Colorado in 2010. Compared with corn growth, ethanol conversion key data sources.

**Table 3**

<table>
<thead>
<tr>
<th>Fraction</th>
<th>Note</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Air&lt;sup&gt;a&lt;/sup&gt;</td>
<td>95%</td>
<td>When vapor pressure ( p &gt; 10 ) (mPa) (Berthoud et al., 2011)</td>
</tr>
<tr>
<td>50%</td>
<td>1 &lt; ( p &lt; 10 )</td>
<td></td>
</tr>
<tr>
<td>15%</td>
<td>0.1 &lt; ( p &lt; 1 )</td>
<td></td>
</tr>
<tr>
<td>5%</td>
<td>0.01 &lt; ( p &lt; 0.1 )</td>
<td></td>
</tr>
<tr>
<td>1%</td>
<td>( p &lt; 0.01 )</td>
<td></td>
</tr>
<tr>
<td>Water</td>
<td>5%</td>
<td>National runoff and leaching rate (USDA, 2000)</td>
</tr>
<tr>
<td>Soil</td>
<td>1-air-water</td>
<td>Capped at 85% (Audsley et al., 2003) (Berthoud et al., 2011)</td>
</tr>
</tbody>
</table>

<sup>a</sup> Data on pesticide vapor pressure are collected from various sources including the USDA pesticide properties database (Herner, 1992).
makes a moderate contribution, and its impact has declined over times, as in the case of cancer, on account of increasing yield and energy efficiency.

3.3. Improvement opportunities

This study shows that E85’s potential stress on freshwater ecotoxicity and human health noncancer effects is much higher than gasoline, while their potential cancer effects are comparable. Furthermore, corn production is found to be the major source of ecotoxicity and noncancer effects in the E85 system due to releases of pesticides and heavy metals from fertilizers, respectively. Considering the historical and potential corn expansion due partly to the ethanol mandate Renewable Fuel Standard (RFS) (Wallander et al., 2011; Wright and Wimberly, 2013), it is important to curb environmental impacts induced by corn cultivation. Otherwise, corn ethanol in its replacement of gasoline may likely further degrade water quality and exacerbate human health issues (Yang et al., 2012).

Considering that farmers tend to over fertilize and that nutrients have low use efficiency but high loss rates (Tilman et al., 2002), improved fertilizer management may be the best way to reduce the cancer and noncancer effects of corn and corn ethanol. This can be achieved through adding rotational complexity to cropping systems and developing tools for better predicting nutrient requirement and optimizing fertilizer timing and placement (Robertson and Vitousek, 2009). In addition, cover crops and reduced tillage can maintain or improve soil fertilizer; reduce leaching, runoff and volatilization; and increase nutrient-use efficiency (Tilman et al., 2002). The key, however, is not to apply all the tools that are available, but to provide proper incentives for farmers to adopt a combination that is suited for addressing site-specific challenges (Robertson and Vitousek, 2009).

Pesticides account for much of the freshwater ecotoxicity of corn ethanol. To reduce the impact in the short run, pesticides with low toxicity may be replaced with ones with lower toxicity. For practical reasons, however, substitutes should perform similar agronomic functions and be in the same economic order. Farmers may access such information from plant protection and agronomic experts (Berthoud et al., 2011). In the long run, however, farmers should go beyond this therapeutic mode of killing pests with toxics to adopting more sustainable pest management that builds up the inherent capacity of agricultural ecosystems to keep pests within acceptable limits (Bianchi et al., 2006; Lewis et al., 1997).

It is worth noting that the opportunities discussed above have a broader positive impact than reducing human and ecological toxicity of corn ethanol. In particular, since intensive use of nitrogen and phosphate fertilizers contributes to a range of other problems like global warming, eutrophication, and acidification (Socolow, 1999), reducing their use could also relieve these problems.

3.4. Policy implications

This study has evaluated the life cycle cancer, noncancer, and freshwater ecotoxicity of corn ethanol and gasoline, an area that was relatively poorly understood compared with other impacts like global warming, eutrophication, and acidification (Socolow, 1999), reducing their use could also relieve these problems. For environmental policies such as the RFS, this growing literature implies that it is relevant to extend assessment to other environmental areas than the targeted to explore the possible un-intended consequences of policy goals. In cases where burden-shifting is unpreventable, which is quite common in product substitution as products usually possess respective strengths and weaknesses, policies in favor of one product over another need to include supplementary plans to minimize potential burden-shifting. In addition, environmental policymaking needs to move beyond its isolated, narrow focus on only a couple of indicators...
(e.g., carbon) and embrace a more proper, encompassing matrix to address the complexity of environmental sustainability.

3.5. Limitations and further work

There are several areas where further research may be carried out to strengthen the analysis conducted in this study. First, the estimation of pesticide air emissions in the study is based solely on vapor pressure, neglecting other factors like spraying practices (EPA, 1995b) that may also affect the fate of pesticides after application. It has been found that discrepancies exist between calculated and field-measured air emission factors for some pesticides (Berthoud et al., 2011). Therefore, either more field experiments ought to be conducted – at least on the key pesticides identified in this study – or modeling tools should be improved to better simulate on-farm dynamics for more accurate estimation of pesticide air emissions. Furthermore, this study calculates pesticide water emissions based on an average national emission rate (USDA, 2000). However, the rate of pesticide leaching and runoff is likely to vary among states due to differences in local topographic, climatic, and soil conditions, as well as in management practices. Further research into pesticide leaching and runoff at finer scales will provide a more solid basis for estimating the regional variation in ecotoxicity impact of corn production.

Second, this study finds that, depending on how oilfield produced water is treated, pollutants present in the water may be an important contributor to the overall toxic impacts of gasoline. Although USLCI database includes a large number of pollutants, information on the fate of the pollutants is lacking. Considering 98% of onshore produced water is injected underground (Clark and Veil, 2009), this study assumes that only 2% of the pollutants recorded in USLCI crude oil data come into contact with freshwater environments. However, this 2% assumption is likely a worst-case scenario because it is reasonable to assume that some of this fraction is treated before being directly discharged into the environment. Further research is needed to better understand the fate of pollutants present in produced water, particularly regarding the coverage and effectiveness of water treatment technologies applied in domestic oil extraction and the rate of direct discharge of produced water into the environment.

Finally, the comparative results arrived at in this study are potential impacts, instead of absolute or actual risks, and should be interpreted with caution. Although covering a large number of environmental stressors, LCIA models lack the degree of modeling sophistication with respect to temporal and spatial details as opposed to, for example, risk assessment (RA) (Bare, 2006). Consequently, LCIA models do not offer absolute predictions of risks but rather potential impacts. To better evaluate the actual human and ecological health effects of E85 and gasoline at finer scales, one would need more sophisticated tools such as 3-D models that account for a higher level of details as to location of the sources, meteorology, etc. Also, while the USEtox represents one of the most advanced LCIA methods, uncertainties exist throughout its modeling, especially regarding human and ecotoxic effects (Rosenbaum et al., 2008).

4. Conclusion

This study evaluates the life cycle human health cancer, noncancer, and freshwater ecotoxicity effects of U.S. corn ethanol (E85) and gasoline based on the USEtox model. Overall, corn ethanol generates potentially larger freshwater ecotoxicity and noncancer effects than gasoline, whereas their human health cancer effects are comparable. Major contributors to corn ethanol’s life cycle freshwater ecotoxicity effect are pesticide emissions at the corn cultivation stage, particularly atrazine, acetochlor, chlorpyrifos and cyfluthrin. Regarding cancer effect, nitrogen and phosphate fertilizers, diesel combusted in farm equipment, and process energy used in ethanol conversion are identified as key contributors. Regarding noncancer effect, heavy metals, mainly cadmium, contained in phosphate fertilizers are identified as the primary contributor in the corn ethanol system. The results indicate that replacing gasoline with corn ethanol could further degrade water quality and exacerbate noncancer related human health issues. Improving fertilizer and pesticide management at corn growth stage and energy efficiency at both corn growth and ethanol conversion stages have the potential to reduce corn ethanol’s overall toxic impacts.

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Appendix A. Supplementary data

Supplementary date related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2013.04.009.

References


Bare, J., Norris, G., Pennington, D., Mckone, T., 2003. The tool for the reduction and assessment of chemical and other environmental impacts. J. Ind. Ecol. 6, 49–78.


