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Research paper

Life cycle air quality impacts on human health from potential switchgrass production in the United States

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ABSTRACT

Switchgrass is a promising bioenergy feedstock, but industrial-scale production may lead to negative environmental effects. This study considers one such potential consequence: the life cycle monetized damages to human health from air pollution. We estimate increases in mortality from long-term exposure to fine particulate matter ($PM_{2.5}$), which is emitted directly ("primary $PM_{2.5}$") and forms in the atmosphere ("secondary $PM_{2.5}$") from precursors of nitrogen oxides (NO_x), sulfur oxides (SO_x), ammonia (NH_3), and volatile organic compounds (VOCs). Changes in atmospheric concentrations of $PM_{2.5}$ (primary + secondary) from on-site production and supporting supply chain activities are considered at 2694 locations (counties in the Central and Eastern US), for two biomass yields (9 and 20 Mg ha^{-1}), three nitrogen fertilizer rates (50, 100, and 150 kg ha^{-1}), and two nitrogen fertilizer types (urea and urea ammonium nitrate). Results indicate that on-site processes dominate life-cycle emissions of NH_3, NO_x, primary $PM_{2.5}$, and VOCs, whereas SO_x is primarily emitted in upstream supply chain processes. Total air quality impacts of switchgrass production, which are dominated by NH_3 emissions from fertilizer application, range widely depending on location, from 2 to 553 \$ Mg^{-1} (mean: 45) of dry switchgrass at a biomass yield of 20 Mg ha^{-1} and fertilizer application of 100 kg ha^{-1} N applied as urea. Switching to urea ammonium nitrate solution lowers damages to 2 to 329 \$ Mg^{-1} (mean: 28). This work points to human health damage from air pollution as a potentially large social cost from switchgrass production and suggests means of mitigating that impact via strategic geographical deployment and management. Furthermore, by distinguishing the origin of atmospheric emissions, this paper advances the current emerging literature on ecosystem services and disservices from agricultural and bioenergy systems.

1. Introduction

Bioenergy is increasingly being considered as a means of enhancing access to clean energy and ensuring energy security, which are fundamental constituents of human wellbeing [1]. Bioenergy feedstock production can also drive ecosystem change, affecting human wellbeing by altering the delivery of ecosystem services from the converted landscapes [2]. At the same time, bioenergy production and use can affect human health [3], another key constituent of human wellbeing [1]. A major concern for human health is mortality arising from long-term exposure to fine particulate matter ("$PM_{2.5}$", particles with a diameter ≤ 2.5 μm) [4]. $PM_{2.5}$ can be emitted directly as "primary" $PM_{2.5}$ or can form in the atmosphere as "secondary" $PM_{2.5}$ through chemical reactions of other pollutants ("precursors"), chiefly ammonia (NH_3), sulfur oxides (SO_x), nitrogen oxides (NO_x), and volatile organic compounds (VOCs). Overall, the air quality effects of agriculture in the US negatively impact human health. For example, in the US, the agricultural sector contributes around half of the surface-level mass of anthropogenic $PM_{2.5}$ in the atmosphere [5], with agricultural sources of outdoor air pollution in the US estimated to have been responsible for around 16,000 premature deaths in 2010 [6].

In the US, the dominant prospective lignocellulosic feedstock for bioenergy is the perennial herbaceous crop, switchgrass (*Panicum virgatum*). It has many attractive attributes concerning feasibility; *e.g.*, high yield [7] [8], long stand life (~10 years [9]), and harvestable using conventional techniques [10]. Furthermore, owing to low agricultural inputs and perenniality, switchgrass has the potential to provide other valuable ecosystems services such as soil erosion control

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[11], carbon sequestration [12] [13], habitat provision for biodiversity [14] [15] [16] [17], and reduction of water pollution [11]. Still, switchgrass production, and potentially the production of any bioenergy feedstock, can affect air quality, from both biogenic [18] [19] [20] and anthropogenic emissions [21] [22] [23].

The aim of this study is to evaluate the potential air quality effects of switchgrass production in the US through an ecosystem services framework. Some previous research has investigated the effects of switchgrass production on air quality, but much of that work either has not been carried to the end point of estimating impacts on human health and wellbeing [21] [24], has not been at a sufficiently high resolution to demonstrate potential impacts of geographic variability in production [22], or has not included biogenic emissions alongside anthropogenic emissions [21]. Within an ecosystem services framework, the emissions considered in this study entail a trade-off. While these emissions are the outcome of acquiring a provisioning ecosystem service (*i.e.*, switchgrass feedstock) that enhances human wellbeing through energy security and access [1], they can negatively affect another constituent of human wellbeing, namely health. Some authors refer to such processes as "ecosystem disservices" [25] [26] [27].

We first describe potential yield, fertilization, and location scenarios for growing switchgrass (Section 2.1), ultimately describing impacts for 2694 locations (counties in the Central and Eastern US), 2 yields, 3 fertilization rates, and 2 fertilizer types. We examine such a range of production locations and practices because there is considerable uncertainty as to where and how switchgrass might be grown in the US [28]. Only ∼11 Gg of switchgrass were produced in 2012 (for comparison, corn grain: ∼260 Tg y^{-1}) [29]. Next, we construct a life cycle inventory of emissions of $PM_{2.5}$ and its precursors for switchgrass production, normalized to 1 Mg (dry basis) of switchgrass (Section 2.2). We then model the resulting annual average changes in concentrations of total $PM_{2.5}$, both primary and secondary, and estimate the subsequent monetized mortality impacts for each of these scenarios (Section 2.3). Section 3 outlines the main results of the life cycle assessment (LCA) study for the US. Section 4 puts these findings into perspective, comparing them with other studies and discussing them within the context of ecosystem services.

2. Methodology

2.1. Growing scenarios

The air quality impacts of switchgrass on human health depend on many parameters, including the following:

- Yields can vary widely in a given location owing to genotype or environmental conditions. Genetic engineering, cross-breeding, and improved management practices might increase future yields [30], which, all else being equal, might reduce the negative air quality impacts of biomass production.
- Higher N fertilization results in higher emissions, both from fertilizer production and use. Emissions also depend upon fertilizer type and timing of application, among other factors. Precision agriculture or high N use efficiency cultivars could reduce emissions [31].
- All else being equal, health impacts are generally greater if emissions occur near a densely populated area [32].

Because of the importance and uncertainty of these variables, we consider multiple scenarios for each. A "baseline scenario" was chosen, at a yield of 20 Mg ha^{-1} switchgrass, and 100 kg ha^{-1} N applied as urea (see 2.1.3). "Low" and "high" scenarios for fertilization rate were chosen at 50 kg ha^{-1} and 150 kg ha^{-1} N respectively. Application of N in the form of urea ammonium nitrate (UAN) was also considered. A "low yield" scenario was chosen at 9 Mg ha^{-1} switchgrass.

This paper considers scenarios for all combinations of the chosen biomass yield, N fertilizer rate, and N fertilizer type. These variables are strongly coupled [33] [34] [35], and work has been done to model how they relate to each other [33] [36] [37]. We examine them across a wide range of locations in the Central and Eastern US.

2.1.1. Biomass yield

"Low" and "high" yield scenarios are taken as 9 Mg ha^{-1} and 20 Mg ha^{-1} dry switchgrass, respectively. This choice of yields was informed by the following studies. Wullschleger et al. [33] compiled 1190 yield observations from 39 US field trials, concluding that the mean (± standard deviation) yield was 8.7 ± 4.2 Mg ha^{-1} for the upland ecotype and 12.9 ± 5.9 Mg ha^{-1} for the lowland ecotype. EPA's yield range for perennial grasses [28] is higher, from 7.6 to 22.2 Mg ha^{-1} (national average: 20.4 Mg ha^{-1}), whereas DOE reports a range of 9.2–18.0 Mg ha^{-1} (national average: 13.5 Mg ha^{-1}).

2.1.2. Nitrogen fertilizer rate

Although switchgrass can be grown without N fertilization, its profitable production in monocultures likely requires N application [38]. The N fertilization rate in the baseline scenario is 100 kg ha^{-1} y^{-1} of N. "Low" and "high" fertilization scenarios are chosen at 50 kg ha^{-1} y^{-1} and 150 kg ha^{-1} y^{-1}, respectively.

N fertilization guidelines are commonly derived from expected yields and reported per unit mass of switchgrass produced. Argonne's GREET (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) Model has a default fertilization rate of 8 kg Mg^{-1} of N per unit mass of dry switchgrass. Iowa State Extension describes 5 kg Mg^{-1} N for the Liberty cultivar [39], which is also recommended by Penn State Extension for switchgrass cultivars in general [40]. For the Great Plains and Midwest, Mitchell et al. recommend 10 kg Mg^{-1} of N when harvested in the growing season, or 6–7 kg Mg^{-1} if harvesting after a killing frost [41].

2.1.3. Nitrogen fertilizer type

Because switchgrass is a perennial crop, there is aboveground biomass throughout most of its stand life, which contraindicates fertilization methods involving injection, subsurface banding, or incorporation [42]. The choice of N fertilizer type is therefore limited. Urea, ammonium nitrate, and a solution mix of both are possible options. Enterprise budgets from Oklahoma State University [43] and several field plot studies [44] [45] [46], suggest the use of urea, whereas GREET [47] [48] and Mississippi State University [49] suggest ammonium nitrate. However, the use of pure ammonium nitrate has strongly declined in recent years [50].

There are two reasons to justify the consideration of UAN solution as a fertilizer in this study. First, a recent study compiling the emissions inventory of switchgrass [21] states that this is likely to be the primary fertilizer used. Second, urea can have 15% N loss by volatilization, compared to 8% for an UAN solution [51]. Large-scale switchgrass farmers might act to be more efficient in N-use, so as to be more cost-efficient [52]. We consider both urea and UAN for scenarios in our analysis. This is because of the uncertainty in whether switchgrass farmers will prefer urea or UAN as discussed above, and the fact that the large difference in volatilization rate for these fertilizer types leads to a large difference in NH_3 emissions.

2.1.4. Location

We constrain growing locations to US states on and east of the 100th meridian west, which is the historic range of switchgrass [53], comprising 2694 counties in the Central and Eastern US. Research efforts have been made to determine the land available for switchgrass production at subcounty resolution. However, many of these depend on some specification of "marginal" land, whether this be idle, fallow, or abandoned land [54], polluted land [55], unproductive cropland [31], or conservation land [56]. As many of these studies differ in crucial assumptions regarding where switchgrass will be profitable or high-yielding [35] [57], for the sake of generality, the centroid of each county is taken as the growing location. Counties with little agricultural land are unlikely to grow switchgrass, including urban areas where air quality impacts are likely to be high as there is a large population. We therefore calculate a cropland area-adjusted average impact across all counties in the domain using USDA Farm Service Agency estimates of total area planted for all crops [58].

Finding spatially-resolved changes in concentrations of $PM_{2.5}$ from growing switchgrass also requires spatially allocating off-site emissions from processes upstream in the supply chain [59]. In this study, locations and capacities of facilities were found for the following off-site processes: nitric acid production, sulfuric acid production, ammonia production, ammonium nitrate production, and urea production. Monetized damages were found for producing 1 Mg of the respective chemical at each of these plants (Section 2.3), and the production-weighted average was found by weighting the damages at each plant by the plant's capacity.

2.2. Life cycle emissions inventory

Emissions associated with growing switchgrass come from many supply chain processes. Prior research has found that important sources of emissions include primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors from the production and use of fertilizer, using farm equipment fuel, VOC emissions from herbicide application, and fugitive dust emissions of $PM_{2.5}$ from agricultural operations [21] [22]. The scope of the emissions inventory for the current study includes emissions from "on-site processes", which occur on the farm, as well as "off-site processes", which occur elsewhere in supply chain activities. The system boundaries of this life cycle inventory include all processes in the GREET life cycle for switchgrass farming (Section 2.2.1) [47] [48] as well as the processes outlined in Section 2.2.2–2.2.5. The functional unit (*i.e.*, the output of the product system that provides the service whose impacts are being assessed) is 1 Mg of dry switchgrass (15% mass fraction of water) at the farm gate, before taking into account dry matter loss (Section 2.2.1).

2.2.1. GREET model

This study makes substantial use of the switchgrass farming pathway in the GREET 2015 Model [47] [48]. GREET default parameters are used, with the exceptions discussed below.

First, the emissions rates of SO_x for the production of sulfuric acid (H_2SO_4) were modified. H_2SO_4 is used in phosphate fertilizer production. The default emissions factor in GREET, for SO_x emitted per unit mass of H_2SO_4 produced, is 20 kg Mg^{-1}. However, regulatory documents [60] and other publicly available data [61] suggest that an emissions factor of 2 kg Mg^{-1} better reflects the current industry.

Second, dry matter loss is set to zero. GREET estimates biomass losses from handling, harvesting and storing biomass, such that between 1.09 and 1.24 Mg of dry switchgrass must be grown for every dry megagram of switchgrass output at the farm gate, depending on the storage method. These losses were not included in the current analysis for simplicity, but results per dry megagram can be readily scaled to account for dry matter loss, as health impacts are modeled as linear (Section 2.3).

Third, GREET parameters were changed to reflect the scenarios chosen in this study (Section 2.1). The N application rate was varied accordingly, and the N pathway mix was changed to 100% urea, or 100% UAN solution, depending on the scenario.

GREET default stand life and yield assumptions are used, with an assumed stand life of 10 years, including 2 years for the establishment phase. No fertilizer is applied during the establishment period, but electricity and diesel fuel are used in cultivation and planting. All parameters are averaged over the entire stand life to arrive at yearly inputs and outputs [47] [48].

Sections 2.2.2–2.2.5 describe additions made to this GREET pathway to construct a comprehensive life cycle emissions inventory. Section 2.2.6 provides, as a point of comparison, an emissions inventory for unmanaged land that might be displaced due to switchgrass production.

2.2.2. Ammonia emissions

Ammonia (NH_3) emissions occur during the life cycle of switchgrass production, in both on-site and off-site processes [21] [22]. The GREET Model does not model NH_3 emissions, so these are added in our analysis.

On-site NH_3 emissions occur when N fertilizer is applied to soils, and some fraction of this N volatilizes into the atmosphere. The magnitude of these emissions depends on N application rate, fertilizer type, and soil, water and climate conditions [50]. The fraction of volatilized N can vary substantially depending on these conditions [62] [63], which are difficult to model precisely [51]. We use the CMU (Carnegie Mellon University) Ammonia model [64] emissions factors of volatilized N as a percentage of applied N. For UAN solutions, 8% of the N applied to the soil is released as ammonia. For urea, this value is 15% throughout most of the domain used in this study, except in South Dakota and Texas where it is 20%.

Off-site NH_3 emission factors were estimated by using the National Emissions Inventory (NEI), which classifies emissions by sector, industry, and process [65]. The processes within the life cycle of switchgrass production, and for which the NEI reported NH_3 emissions, are the following: ammonia production, urea production, and ammonium nitrate production. The emissions were divided by total US production for ammonia, urea, and ammonium nitrate respectively, giving emissions factors in mass of pollutant per unit mass of product.

2.2.3. Fugitive dust emissions

Fugitive dust from agricultural sources contributes to particulate matter emissions [66], but it is not included in the GREET Model. Activities that disrupt the soil emit coarse particulate matter ("PM_{10}") [67], 20% of which is taken to be $PM_{2.5}$ [68]. Following Zhang et al. [21], we use a 45 g Mg^{-1} emissions factor of $PM_{2.5}$ from fugitive dust emissions for growing switchgrass.

Emission inventories commonly overestimate dust emissions, because they often do not account for atmospheric removal by land cover within ~100 m of the source [69], caused by dry deposition and reduction in wind speed [70]. Dust emitted in dense vegetation is almost completely captured, thus having almost no health impact, but dust emitted in a barren landscape is not well captured [70]. Because of the local scale of this capturing effect, it can be corrected by modifying the emission inventory. Pace [70] derived a "transport fraction" (the fraction of dust which is not locally captured) for each county; the average transport fraction across all counties in the Central and Eastern US is 46%. Values from Pace [70] are applied to this analysis.

2.2.4. On-site VOC emissions

Plants emit VOCs from their leaves and other organs, which can affect air quality [19] [20] [71]. These emissions increase greatly as a result of injury such as during harvesting, especially when harvested in the growing season. Eller et al. [18] use laboratory chamber measurements to estimate the yearly emissions from switchgrass plantations. These biogenic VOC emissions are included in the emissions inventory.

VOCs are also emitted during herbicide application [21]. In general, these emissions depend on the herbicide formulation, including active ingredients and inert ingredients (*e.g.*, buffers and surfactants) [72]. There is considerable variation in the amount, type of active ingredient and formulation of herbicide used for establishing and maintaining switchgrass [73]. In GREET, 465 g ha^{-1} of glyphosate are applied per dry short ton of switchgrass [47] [48] in field preparation, although the formulation and application method are not given. For some glyphosate formulations (*e.g.*, Roundup Weathermax) the VOC emissions factor is negligible [74], while for other formulations it can be higher. Following Zhang et al. [21] and EPA [75], we assume the VOC emissions factor to be the mass of active ingredient applied (465 g ha^{-1} from GREET) multiplied by the evaporation rate (0.9) and the VOC content (0.835), to give 349 g ha^{-1} (17.5 g Mg^{-1} high yield; 38.8 g Mg^{-1} low yield).

2.2.5. Off-site production of farm equipment

The GREET 2015 Fuel-Cycle Model reports inputs, outputs, emissions, and lifetimes of the production of farm equipment. Here we use a list of farm equipment required for switchgrass production as suggested by Zhang et al. [21]. All types of farm equipment are assumed to be necessary for growing switchgrass and are only used for switchgrass production. To estimate the amount of machinery used per hectare of switchgrass grown, we require the size of a switchgrass farm, which we assume to be 220.9 ha (*i.e.*, the same as the GREET default for corn). The mass of machinery used per dry ton of switchgrass produced then depends on the yield scenario.

2.2.6. Emissions inventory for unmanaged grassland

We also estimate emissions from unmanaged land displaced by the production of 1 Mg of switchgrass. Biogenic VOC emissions (Section 2.2.4) from unmanaged grassland are obtained from Lamb et al. [76], and NO_x emissions from Williams et al. [77]. Emissions factors are derived as the mass of pollutant emitted per hectare. From these, together with our yield scenario (Section 2.1.1), we determine the emissions that would be displaced from producing 1 Mg of switchgrass in place of unmanaged grassland.

This addition is relevant for three reasons. First, it is likely that switchgrass will be grown on unmanaged grassland, because it may not be profitable enough to outcompete other crops [14]. Second, growing switchgrass on unmanaged grassland ensures that competition with food crops is minimized [13]. Third, there is considerable policy scope for bringing grassland, such as CRP (Conservation Reserve Program) land, into production for bioenergy [57].

2.2.7. Spatial description of life cycle inventories

The emissions inventories shown in Fig. 1 include all on-site and off-site processes in the GREET model switchgrass production pathways with the additions noted above (*e.g.*, fugitive dust). Due to spatial data limitations for some off-site processes (*e.g.*, farm equipment and high sulfur diesel production) the health impacts were calculated from emissions from all on-site processes and those off-site processes for which spatial data were readily available. Emissions from these processes account for 100% of NH_3 emissions, 99% of VOC emissions, 84% of NO_x emissions, 77% of primary $PM_{2.5}$ emissions, and 3% of SO_x emissions. This suggests the life cycle health impact estimates presented in this paper are underestimates. In particular, the small fraction the life cycle SO_x emissions included in the impact analysis is a result of the exclusion in the air quality modeling of several of the upstream processes related to the production of high sulfur diesel.

2.3. Health impacts

After compiling the spatially-allocated emissions inventory, the air quality model InMAP (Intervention Model for Air Pollution) [78] was used to model the steady-state annual average changes in $PM_{2.5}$ concentrations resulting from these emissions, as well as the resulting changes in mortality. InMAP is a reduced-form air quality model, which uses the annual average outputs of a single run of a full Eulerian chemical transport model (WRF-Chem) to calculate changes in the concentrations of pollutants arising from changes in emissions. The grid size used in InMAP is set to vary so that there is higher resolution in areas with higher population and lower resolution in the upper atmosphere and areas with low population. EASIUR [79], another reduced form air quality model, was also used here for comparison (Fig. S2), although air quality impacts from VOC emissions are not calculated in EASIUR. We note that NO_x and VOCs also lead to tropospheric ozone (O_3) formation, which can also lead to premature mortality. However, we do not account for the impacts of tropospheric ozone in this study, as its mortality effects are generally minor compared to those of $PM_{2.5}$ [6] [4]. The Value of Statistical Life [87] was used to calculate the monetary value of health impacts. EPA's value of 7.4 M$ (2006) was used, and adjusted for inflation to arrive at 8.78 M$ (2016) [88].

The link between long-term exposure to $PM_{2.5}$ and mortality has been established in multiple studies, including two major long-term cohort studies in the US: the American Cancer Society (ACS) study and the Harvard 6-City (H6C) study. The results of these studies have been confirmed over several reanalyses, most recently in Krewski et al. [80] for the ACS study and Lepeule et al. [81] for the H6C study. Results from these studies are used to evaluate the benefits of EPA regulations for reducing particulate matter concentrations [82]. Here, we employ the dose-response relationship between human mortality and long-term exposure from $PM_{2.5}$ concentrations given by Krewski [80]. It is worth mentioning that these cohort studies find that $PM_{2.5}$ affects mortality rates even at levels below those required by regulations; furthermore, the studies do not find evidence of a threshold. Pope [83] discusses relevant research on threshold effects with particulate matter, showing that the effect is nearly linear. Since then, there has been evidence suggesting that the effect is "supralinear", meaning that there are larger marginal effects at lower concentrations levels compared with higher concentrations [80] [84] [85]. However, the convention in air pollution analysis assumes a nearly linear concentration-response relationship with no threshold [86].

3. Results

3.1. Emissions inventory

Fig. 1a shows the emissions inventory for growing one dry megagram of switchgrass in scenarios with a yield of 20 Mg ha^{-1} and a N fertilization rate of 100 kg ha^{-1}, applied as urea or UAN. Changing the fertilizer type from urea to UAN dramatically reduces on-site NH_3 emissions. All other on-site emissions remain the same, while all off-site emissions increase slightly. Reducing the N input by 50%, from 100 kg ha^{-1} to 50 kg ha^{-1}, halves the ammonia emissions (Fig. 1b). In this sense, it is similar to the effect of changing fertilizer type to UAN, although NO_x emissions, which also depend on the amount of N applied, are also reduced. Off-site emissions are also reduced because less N fertilizer is manufactured. Biogenic VOC emissions are constant per megagram of switchgrass.

3.2. Monetized health impacts

County-level monetized damages from health impacts for the baseline scenario are shown in Fig. 2. The cropland area-weighted air quality impact for growing switchgrass across all counties is 45 $ Mg^{-1} of switchgrass. Maine is the state with the lowest average cropland-area-weighted air quality impact across its counties at 3.71 $ Mg^{-1}, followed by North Dakota at 9.34 $ Mg^{-1}. Pennsylvania has the highest average cropland-area-weighted air quality impact across its counties at 82.60 $ Mg^{-1}. Fig. 2 does not show where impacts are occurring; rather, it shows how the total monetized health impacts, which occur across North America, vary depending on where switchgrass is grown. $PM_{2.5}$ can travel a great distance from its source and is both a near-source and a long-distance concern for health. Overall, Fig. 2 shows a general trend that growing switchgrass in, or upwind from, areas with higher population density gives rise to higher health impacts.

3.3. Monetized health impacts by pollutant

Fig. 3 shows for the baseline scenario the monetized health impacts per megagram of switchgrass for different species of primary pollutants (VOCs, Primary $PM_{2.5}$, and NO_x). For all pollutants, the mortality impact is calculated from the resulting change in atmospheric

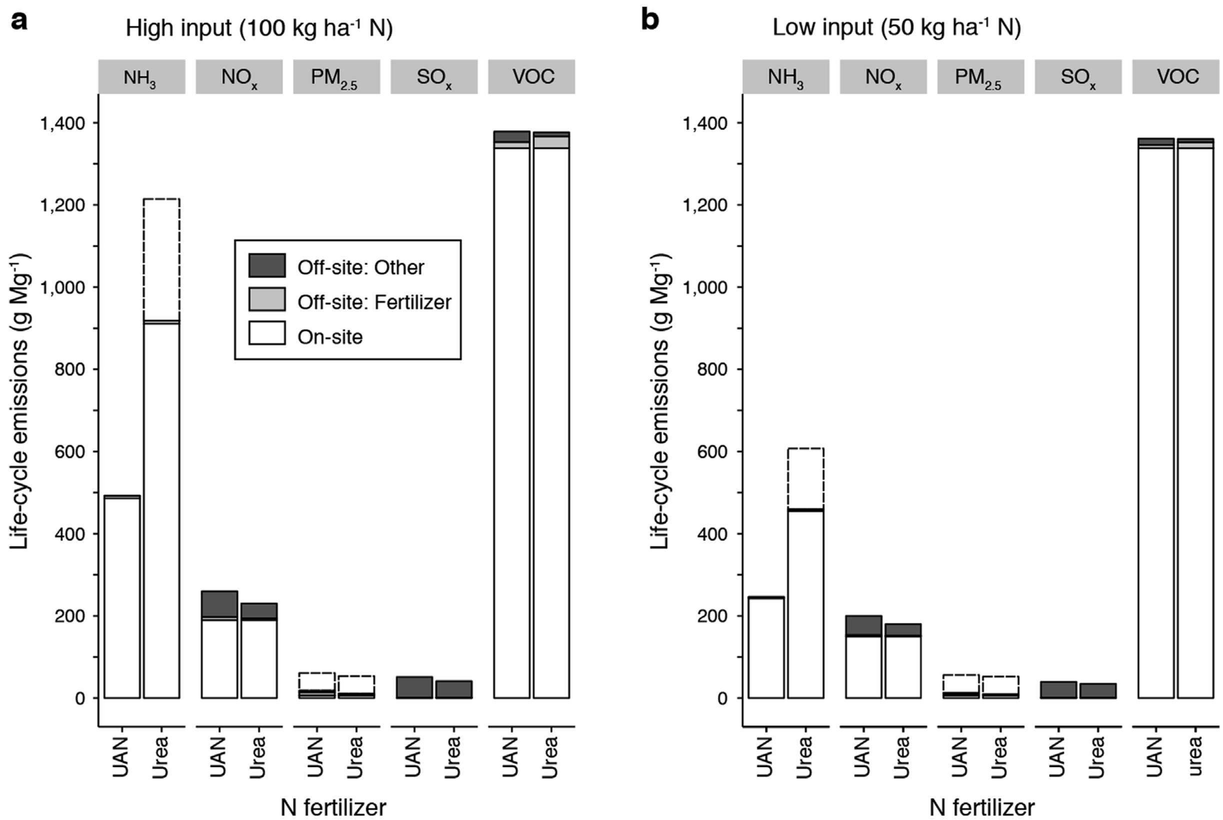

Fig. 1. Life cycle emissions inventory for switchgrass production for two scenarios **(a)** high input (yield 20 Mg ha^{-1}, N fertilization rate 100 kg ha^{-1} N) and **(b)** low input (yield 20 Mg ha^{-1}, fertilization rate 50 kg ha^{-1} N).

Note: "Off-site: Fertilizer" refers to the production of sulfuric acid, potassium oxide, phosphoric acid, ammonia, urea, ammonium nitrate, and nitric acid. Emissions from GREET for all other upstream processes are aggregated as "Off-site: Other". The dashed bar for NH_3 urea is for South Dakota and Texas, which are modeled with a higher emissions factor for ammonia volatilization from urea application. The dashed bar for $PM_{2.5}$ is for dust emissions, which vary by county (mean: 20.7 g Mg^{-1} biomass).

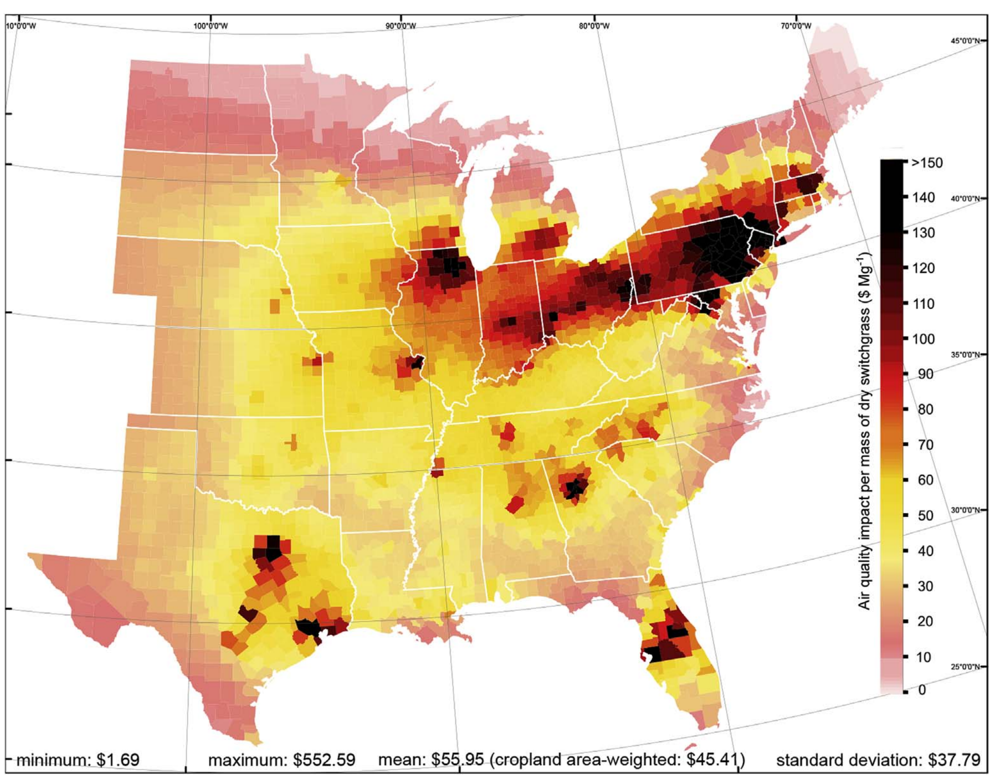

Fig. 2. Total monetized air quality impacts of switchgrass production (yield 20 Mg ha^{-1}, fertilization rate 100 kg ha^{-1} N as urea).

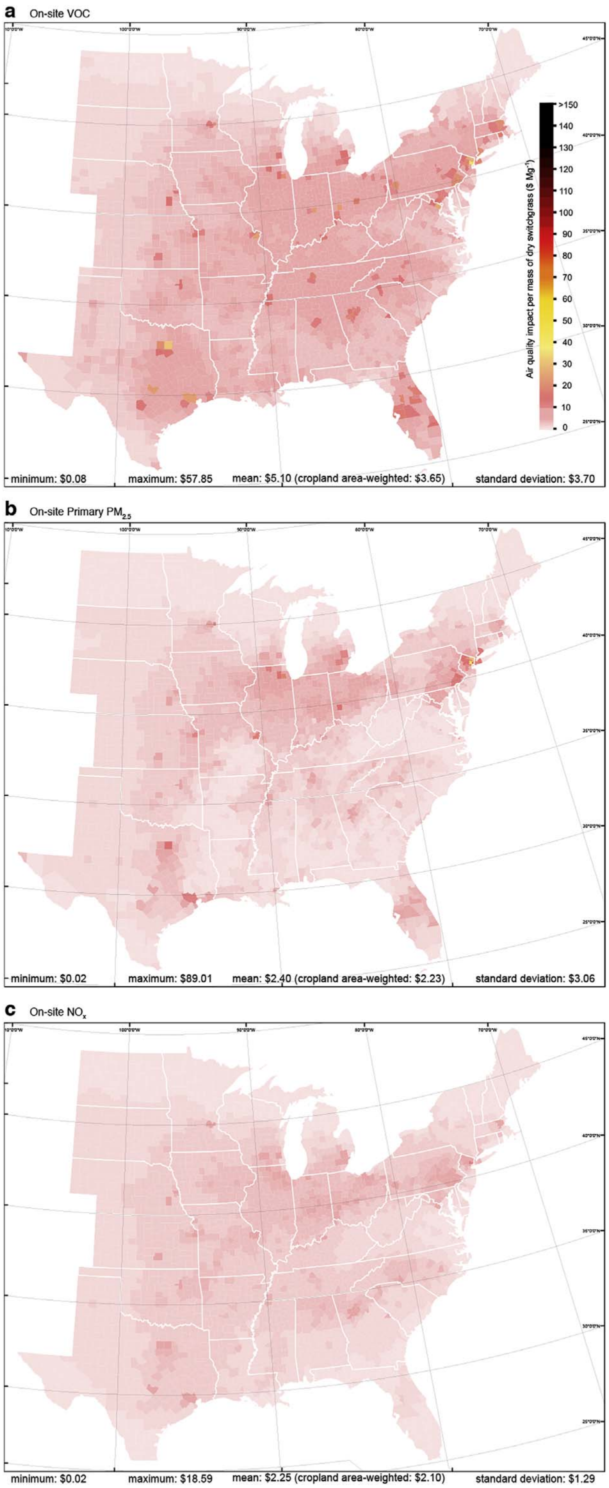

Fig. 3. Monetized air quality impacts of on-site **(a)** VOC, **(b)** $PM_{2.5}$, and **(c)** NO_x emissions for switchgrass production (yield 20 Mg ha^{-1}, fertilization rate 100 kg ha^{-1} N as urea).

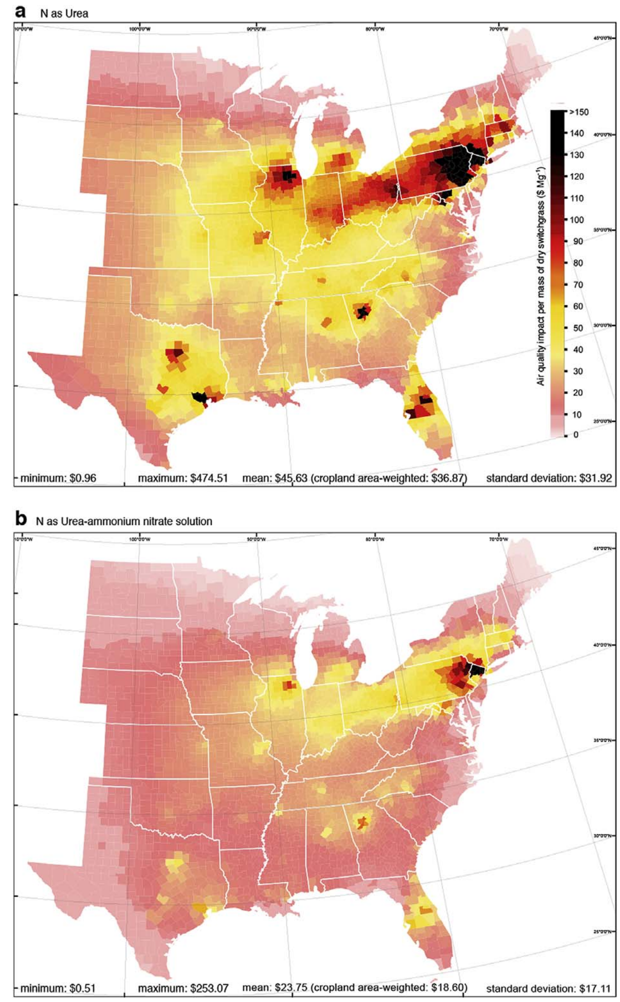

Fig. 4. Monetized air quality impacts of on-site NH_3 emissions for switchgrass production (yield 20 Mg ha^{-1}, fertilization rate 100 kg ha^{-1} N as **(a)** urea and **(b)** urea-ammonium nitrate solution).

concentrations of $PM_{2.5}$ that is caused by emissions of primary $PM_{2.5}$ and of secondary $PM_{2.5}$ precursors of VOCs, NO_x, NH_3, and SO_x (Section 2.3).

Fig. 3a shows the health impacts from VOC emissions from growing switchgrass. Although this includes VOC emissions from herbicide application, as well as other on- and off-site emissions, 98% of the impacts shown here are biogenic (see 2.2.4). Although biogenic VOC emissions are low for switchgrass throughout most of the year [18], these emissions increase to 26.3 kg ha^{-1} y^{-1} in the high yield scenario due to injury from harvest. In the low yield scenario, the biogenic VOC emissions from switchgrass are 11.8 kg ha^{-1} y^{-1}, close to the 13.7 kg ha^{-1} y^{-1} of biogenic VOC emissions for unmanaged grassland [76]. Fig. S1 shows the estimated impacts from biogenic VOC emissions for unmanaged grassland that might be displaced by growing switchgrass at yields of 9 Mg ha^{-1} y^{-1} and 20 Mg ha^{-1} y^{-1}. Almost all of these impacts result from biogenic VOC emissions.

The spatial distribution of all pollutants is such that areas upwind of highly populated areas tend to have higher impacts for growing switchgrass. For primary $PM_{2.5}$ emissions (Fig. 3b), the impact pattern is more varied across space, owing to the difference in dust emissions in different locations (Section 2.2.3). Impacts of NO_x emissions (Fig. 3c) and SO_x are small relative to the other $PM_{2.5}$-related species.

Fig. 4(a)-(b) show the impacts of NH_3 emissions from urea and UAN fertilization, respectively, at the baseline yield of 20 Mg ha^{-1} and N fertilization rate of 100 kg ha^{-1}. The air quality impact of NH_3 emissions from fertilizer application dominates that of any other pollutant from any on-site process, accounting for 81% of the average total impacts across all counties for urea fertilization and 68% for UAN fertilization (Fig. 5).

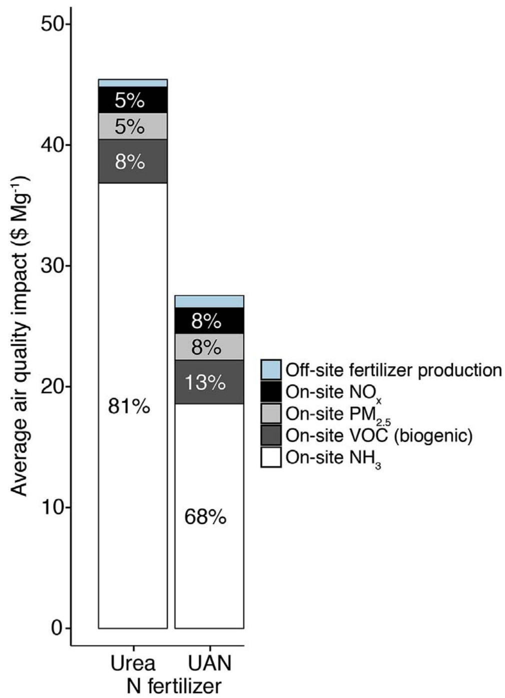

Fig. 5. Monetized air quality impacts of switchgrass production for each pollutant, averaged across all counties, also shown a percentage of total air quality impacts. Note: "Off-site fertilizer production" refers to the production of sulfuric acid, potassium oxide, phosphoric acid, ammonia, urea, ammonium nitrate, and nitric acid. Impacts of non-biogenic on-site VOC emissions and SO_x are negligible.

4. Discussion

4.1. Comparison with other studies

The life cycle emissions inventory in this study augments the GREET life cycle inventory, as detailed in Section 2.2.2–2.2.6. Many of the same processes were included in the recent inventory compiled by Zhang et al. [21]. However there are some differences with the study of Zhang et al., notably their assumption of a UAN solution as N fertilizer [21], which the present study considers in addition to urea. The method employed in the present study for estimating the fugitive dust emissions is the same as the method by Zhang et al., apart from the transport fraction being taken into account in the present study, which reduces the primary $PM_{2.5}$ emissions from dust by 46% on average across all counties in the study domain. The approach for estimating VOC emissions from herbicide application in this study also reflected Zhang et al. [21]; however, neither biogenic emissions, which dominate the life cycle VOC emissions inventory, nor the production of farm equipment were included in Zhang et al. [21].

The study of Hill et al. [22] estimates the air quality impacts of the life cycle of switchgrass-derived ethanol production and use. Unlike the present study, which provides impact estimates for different switchgrass growing scenarios, Hill et al. [22] considers only one scenario (yield: 7.1 Mg ha^{-1}, fertilization rate: 74 kg ha^{-1}, fertilizer: ammonium nitrate, location: CRP land). The life cycle emissions inventory of feedstock production in Hill et al., as a percentage of the life cycle emissions from one of the scenarios studied in this paper (yield: 9 Mg ha^{-1}, fertilization rate: 50 kg ha^{-1}, fertilizer: UAN), are: 6% of VOCs, 84% NO_x, 20% $PM_{2.5}$, 64% SO_x, and 53% NH_3. These differences can be explained by (a) the difference in fertilization rate, (b) the inclusion of more processes in the present study (*i.e.*, biogenic VOCs and dust emissions) and (c) the use of ammonium nitrate fertilizer in Hill et al. [22], for which only 2% of the applied nitrogen is volatilized as NH_3 [when compared with the higher volatilization rates UAN (8%) and urea (15–20%)].

Hill et al. [22] estimate the mean human health cost for the production and combustion of an additional 3.78 hm^3 of switchgrass-derived cellulosic ethanol to be 188 M$ (2016) [88]. When normalized to the yield of switchgrass, this amounts to 16.90 $ Mg^{-1}. This health cost can be compared with the cropland-area-weighted mean air quality impact of 33.01 $ Mg^{-1} found in the scenario from this study. Correcting for the difference in NH_3 volatilization rate, this impact becomes 17.51 $ Mg^{-1}. However, the human health cost reported by Hill et al. [22] takes into account the reduced impacts from displaced fuels, end-use of the fuel, and health effects additional to mortality, although mortality accounts for 93% of the monetized health effects.

4.2. Ecosystem disservices

This paper adopts the distinction of ecosystem disservices proposed by Shackleton et al. [27] (Section 1), noting that what constitutes exactly an ecosystem disservice is not settled in the literature. Some researchers say ecosystem disservices must have a direct causal link with ecosystem functions; otherwise, they are simply social costs of managing or acquiring ecosystem services [27]. Following this distinction, of all the different emissions associated with switchgrass production, only biogenic VOC emissions can be considered as an ecosystem disservice. Other emissions are merely "anthropogenic" social costs associated with acquiring an ecosystem service (*i.e.*, biofuel feedstock, a provisioning service).

However, some anthropogenic emissions have ecosystem components. For example, applying urea fertilizer to soil results in ammonia volatilization, which is responsible for the majority of the monetized air quality-induced mortality impacts from growing switchgrass in our baseline scenario. This volatilization process involves urea hydrolysis, catalyzed by urease from soil microorganisms [89]. Thus, ammonia volatilization from urea fertilization can be construed as an ecosystem function, albeit one arising as a response to human activity. Some authors explicitly consider cases such as this to be "ecosystem disservices" [90].

Ecosystem disservices exist on a continuum between natural and social hazards [27]. In agricultural systems, they can arise through interactions between anthropogenic activity and ecosystem functions [91]. Although switchgrass emits VOCs without human intervention (*i.e.*, biogenic VOC emissions), the rate of emissions increases dramatically when the plant material is harvested [18]. So, similar to urea volatilization, this ecosystem disservice can, in part, be construed as a side effect of anthropogenic crop management.

In general, there is a lack of literature on the economic valuation of ecosystem disservices. Here, we attempt to monetize these disservices as a function of switchgrass management, where farmers can reduce biogenic VOC emissions by harvesting after the growing season. The average monetized air quality impact from biogenic VOC emissions of switchgrass amounts to 8% of the total health damages in the baseline scenario, at 3.60 $ Mg^{-1}, but can be as high as 57 $ Mg^{-1} in extreme cases (*e.g.*, in Hudson, NJ). The emissions factors used here are based on Eller et al. [18], in which switchgrass is harvested during the growing season. Harvesting post-senescence is likely to reduce plant wound response, lowering biogenic VOC emissions dramatically. In this case, the health damages may be closer to those of VOC emissions from unmanaged and unharvested grassland, giving rise to an average air quality impact of 1.90 $ Mg^{-1} (*i.e.*, an average reduction of 53% in health damages).

4.3. Impact intervention

We have shown that NH_3 emissions from fertilizer application are the largest on-site contributor to the air quality impact of growing switchgrass. Using UAN instead of urea as an N fertilizer reduces this impact.

These results highlight the importance of ammonia abatement strategies. Such strategies include:

- reducing the amount of N applied, perhaps by growing switchgrass in locations that require less N [92];
- changing fertilizer type, as in the case of urea to UAN;
- breeding switchgrass to increase N use-efficiency;
- changing timing of application relative to soil, water, and climate conditions [93];
- using fertilizer coatings [94];
- incorporating N into soil using selective tilling or irrigation [95]; and
- growing mixes of diverse prairie species that include legumes to reduce N input by increasing nitrogen fixation [96].

Ammonia emissions from agricultural sources can have impacts on human wellbeing other than via the health effects related to air quality. Ammonia deposition may provide additional biomass from croplands and forests, while deposition in freshwater bodies may decrease salmonid fish populations, which may reduce food production [97] [98]. These trade-offs from switchgrass can have interesting effects locally and nationally, and should be the focus of future research.

4.4. Future work

This study estimates the air quality impacts of growing switchgrass, compared across many locations, fertilization rates, yield scenarios and fertilizer types. Further considerations of the expected human health effects of switchgrass production may include:

4.4.1. Downstream impacts of using the feedstock for fuel

Switchgrass can be used to produce cellulosic ethanol, gasification-based fuels [99], and electricity to power electric vehicles. Emissions from biomass burning, refineries, or tailpipe emissions are known to have large human health impacts [100] [101] [102].

4.4.2. Emission reductions from displaced fuels

Using switchgrass-derived fuel may reduce the use of gasoline in the transport sector. Thus, future studies should consider the reduced air quality effects of no longer producing the displaced gasoline.

4.4.3. Deployment scenario

The efficiency of industrial and agricultural operations tends to increase as the scale increases, meaning that a larger-scale operation may have lower emissions factors. While the emissions factors and yields used in this study are suitable for large-scale deployment, they may vary depending on the exact deployment scenario considered.

4.4.4. Comparisons with other feedstocks

Corn is likely to have higher N application rates. This would probably increase off-site emissions of SO_x, $PM_{2.5}$, NO_x and VOCs due to fertilizer production, and increase NO_x emissions from fertilizer application. However, NH_3 emissions from fertilizer application will depend on the form of the fertilizer used (Section 2.2.3). For annual crops, more fertilizer may be applied using incorporation or subsurface banding [103], which may drastically reduce N volatilization [42]. However, annual crops are also likely to entail more extensive soil disruption from tilling, increasing dust $PM_{2.5}$ emissions (Section 2.2.3). Biogenic VOC emissions for corn are also likely to be greater [104].

4.4.5. Other health impacts

Nitrogen fertilization can also affect human health through water quality [105]. The form of N applied, and the amount lost to volatilization, may have affect the amount of N available for leaching or runoff. Further, human health impacts of pesticide and herbicide toxicity may be appreciable, and should be the focus of future research studies.

5. Conclusions

Our results show that air quality-induced human health impacts from growing switchgrass vary greatly by location in the US and are notably higher when using urea fertilizer compared to UAN. In the life cycle emissions inventory for growing 1 Mg of switchgrass, VOC emissions are larger by mass than for any other pollutant considered; however, NH_3 emissions give rise to the greatest air quality-induced health impacts. Further research should consider how switchgrass compares to other bioenergy crops in terms of human health impact related to air quality, as well as considering the impacts of different uses of the feedstock, whether for producing cellulosic ethanol, electricity, renewable fuels, or other types of bioenergy and bioproducts. Our work also demonstrates the usefulness of evaluating the environmental impacts of agriculture and bioenergy within an ecosystem service context. In particular, an accounting of the ecosystem services and disservices from switchgrass production identified biogenic emissions of VOCs from switchgrass harvest as a potentially large contributor to reduced air quality. Management practices such as harvesting post-senescence can reduce this ecosystem disservice. Further research should explore both the contribution of biogenic VOCs from switchgrass harvest to tropospheric ozone formation and other means of biogenic VOC mitigation.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.biombioe.2017.10.031.

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