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Chapter 6
Environmental Impacts of Switchgrass Management for Bioenergy Production

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Abstract In this chapter, we review major environmental impacts of growing switchgrass as a bioenergy crop, including effects on carbon sequestration, greenhouse gas emissions, soil erosion, nutrient leaching, and runoff. Information from life cycle analyses, including the effects of indirect land use change (iLUC), is examined to quantify the full impact of migration to bioenergy cropping systems on both managed and natural ecosystems. Information on the environmental impacts of switchgrass cultivation is scarce and there exists a critical need for additional research. What limited information there is suggests that switchgrass provides multiple environmental benefits compared to annual crop cultivation. However, benefits generally appear to be similar to other perennial crops.

6.1 Introduction

An evaluation of the environmental impacts of switchgrass (*Panicum virgatum* L.) depends on contrasts to alternative crop species or cropping systems that switchgrass will potentially displace or to which switchgrass might be preferred.

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In the Midwestern United States, an area dominated by maize (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] production, an important contrast will be based on impacts relative to these crops. Alternative perennial crops could be various cool-season (C_3) grasses, forage legumes, or another C_4 grass, *Miscanthus x giganteus* Greef & Deuter ex Hodkinson & Renvoize (hereafter referred to as *miscanthus*). The most relevant contrasts are those that represent realistic alternatives. If switchgrass is grown as a bioenergy crop, it will likely compete for a place on the landscape with maize, soybean, cool-season perennials, and *miscanthus*.

Switchgrass was selected as a model bioenergy crop in the U.S. because it is a native plant, produces substantial above-ground biomass (20 Mg ha^{-1} yr^{-1}), and has an extensive areal range in North America. The development of this selection is described by Wright and Turhollow [1]. Another bioenergy crop, *miscanthus*, offers a particularly interesting alternative to switchgrass. *Miscanthus* has been the focus of bioenergy research in Europe because it produces as much as 40 Mg ha^{-1} yr^{-1} above-ground biomass with similar or fewer N fertilizer inputs as switchgrass [2]. Both *miscanthus* and switchgrass are desirable bioenergy crops because: (1) biomass yield is high; (2) they are perennial rhizomatous plants that cycle nutrients seasonally between the above- and below-ground portions of the plant, thus minimizing fertilizer requirements and corresponding environmental impacts; (3) they provide a clean burning fuel when they are harvested after senescence; (4) planting is required only once, so minimal fuel costs associated with tillage and planting are incurred; and (5) they are both C_4 plants, which are photosynthetically more efficient than C_3 species [2]. A potential advantage of switchgrass over *miscanthus* is that it is reproduced by seeds, whereas *miscanthus* reproduction is vegetative with accompanying higher establishment costs and need for specialized equipment.

In this chapter, we review major environmental impacts of growing switchgrass as a bioenergy crop including effects on carbon sequestration, greenhouse gas emissions, soil erosion, nutrient leaching, and runoff. Where available, information from life cycle analyses, including the effects of indirect land use change (iLUC), will be examined to quantify the full impact of bioenergy crops on both managed and natural ecosystems.

6.2 Climate Change

6.2.1 Carbon Sequestration

One important impact of growing switchgrass or other bioenergy crops will be the potential for C sequestration or loss of C from the soil. The number of studies looking at C sequestration in switchgrass stands is limited, but those that exist have shown that replacing annual crops with perennials such as switchgrass increases C sequestration. In an analysis of published estimates of soil organic carbon (SOC) changes following conversion of natural or agricultural lands to biofuel crops, Anderson et al. [3] found that removing maize grain and residue as a bioenergy feedstock led to rapid loss of SOC at rates up to 4.2 Mg ha^{-1} yr^{-1} compared with management for grain removal only. They calculated that 10 years of maize biomass removal resulted in a loss of about 3 Mg ha^{-1} yr^{-1} at 25% residue removal and about 8 Mg ha^{-1} yr^{-1} at 100% removal. In contrast, SOC accumulation under switchgrass ranged from about 0.4 to 0.7 Mg ha^{-1} yr^{-1} depending on the statistical model used to estimate changes.

Anderson et al. [3] reviewed the existing literature and identified three general principles governing the C balance of biofuel crops. First, conversion of uncultivated soil to biofuel crops initially entails a loss of SOC; second, crops differ in the ability to sequester C with perennial crops outperforming annuals such as maize; and third, a tradeoff exists between biomass removal and C sequestration. In another review of the literature, Blanco-Canqui [4] also concluded that residue removal reduced SOC concentration, whereas, planting warm-season grasses such as switchgrass increased C sequestration.

In a paired comparison of 120 cm deep soil samples from 42 switchgrass/cropland sites, Liebig et al. [5] found that SOC was greater in switchgrass stands at soil depths of 0–5, 30–60, and 60–90 cm and that the differences in SOC were especially pronounced at deeper soil depths. They attributed the difference at depth to the greater switchgrass root biomass below 30 cm compared with cropland sites. In a modeling study, simulations with the DAYCENT model [6] predicted an increase in SOC of 45–300% after 15 years of switchgrass growth compared to cotton production. However, another modeling exercise using DAYCENT suggested that maize had slightly greater SOC than switchgrass [7] and that little change in SOC was predicted to occur over a 10 year period for either cropping system.

In contrast to the studies cited above, Tolbert et al. [8] found that both no-till maize and switchgrass had accumulated SOC after three growing seasons, with no significant difference in accumulation rate between the two. However, SOC was numerically greater under switchgrass even though differences were not significant. In a Canadian study [9] switchgrass increased SOC by 3 Mg ha^{-1} yr^{-1} compared with maize at a higher fertility site, whereas, there was no significant difference after 4 years between switchgrass and maize on a rocky shallow-soil.

Because SOC content changes relatively slowly against a large background pool, long-term studies are often needed before differences in accumulation rates are translated into significant differences in the magnitude of SOC pools. Most of the switchgrass studies cited here, and reviewed by Anderson et al. [3] lasted less than 5 years, highlighting the critical need for more long-term studies.

Comparisons between switchgrass C sequestration and other perennial systems generally revealed little difference between systems, or even lower C sequestration potential for switchgrass sites. Non-significant differences in SOC or C sequestration rates were observed when switchgrass was compared with forests or cool-season grasses [10], smooth bromegrass (*Bromus inermis* Leyss.) [11], or short-rotation poplar plantations [12]. When Chamberlain et al. [6] simulated land use conversion from unmanaged grasses to switchgrass, SOC decreased if the switchgrass was not fertilized, was unchanged when 45 kg N ha^{-1} was applied, and increased when 90 and 135 kg N ha^{-1} were applied. Omonode and Vyn [13] observed little difference in surface SOC content between switchgrass and mixed native warm-season grasses but when SOC mass was calculated to a depth of 1.0 m, SOC was 8% higher under switchgrass than under the native mixture.

Two modeling studies have suggested that C sequestration would be less for switchgrass compared with other perennial species. Growing willow (*Salix alba x glatfelteri* L.) was calculated to increase SOC by 9.0–9.5 Mg ha^{-1} yr^{-1} compared with 3.0–3.5 Mg ha^{-1} yr^{-1} for switchgrass [9]. Davis et al. [7] estimated that SOC under switchgrass would be similar to native prairie but that both would be significantly less than *miscanthus*. While switchgrass appears to have greater C sequestration potential than annual crops, including maize grown as a biofuel, there is no indication that it has any greater C sequestration potential than other perennial systems.

The potential to sequester C depends on a number of factors including initial soil C content, prevailing soils and climate, and management practices. Sequestration is generally greater when existing SOC pools have been depleted, in cool compared with warm climates, in fine-textured compared with course-textured soils, and where soil fertility is high [14]. Perhaps the most widely studied variable is the effect of N fertility. In a switchgrass study in the southern USA, Ma et al. [15] found no difference in SOC among N application rates of 0, 112, and 224 kg N ha^{-1}. They attributed the lack of N effect to the short, 3 year, duration of the study. However, in a study of similar duration in the northern Great Plains, Lee et al. [16, 17] applied 112 or 224 kg N ha^{-1} as either ammonium nitrate or as manure to mature switchgrass stands. Applying N as manure increased SOC accumulation rate to a depth of 90 cm by 33–125% compared with mineral fertilizer, probably because of the additional C input from the manure. All fertilizer rates and sources increased C sequestration compared with no fertilization but there was no difference between the 112 and 224 kg N ha^{-1} application rates. Averaged across N rates, C sequestration was 2.4 Mg ha^{-1} yr^{-1} for plots receiving mineral fertilizer compared with 4.0 Mg ha^{-1} yr^{-1} when manure was applied.

In the simplest terms, C sequestration represents the net balance between C inputs into the system, mainly from photosynthesis but potentially from application of organic sources such as manure or crop residues, and C outputs, mostly from soil and plant respiration but also from removal of harvested material and potentially from runoff or leaching. An assessment of CO_2 fluxes during the first 4 years of switchgrass establishment by Skinner and Adler [18] found that photosynthetic inputs varied little from year to year, ranging from 9.2 to 9.4 Mg-C ha^{-1} yr^{-1}. In contrast, ecosystem respiration ranged from 6.8 to 9.1 Mg-C ha^{-1} yr^{-1}, and harvested biomass removal ranged from 0 to 2.4 Mg-C ha^{-1} yr^{-1}. Mean C sequestration over the 4 years was 0.4 Mg-C ha^{-1} yr^{-1}, and clearly depended more on processes affecting C loss than on C uptake. Similar dependence of sequestration on C loss rather than uptake were observed for cool-season pastures in the northeastern U.S. [19] and for forests along a north–south transect in Europe [20].

Several factors have been found to affect the C dynamics of switchgrass systems. Stepwise regression analysis by Lee et al. [16, 17] found that soil temperature was highly correlated with soil CO_2 flux, whereas, soil moisture was not. Garten and Wullschleger [21] also observed slower decomposition rates in cooler climates. In contrast to the results from Lee et al. [16, 17], Frank et al. [22] measured lower soil CO_2 flux during a drought year compared to CO_2 fluxes during 2 years of above average precipitation.

Lee et al. [16, 17] also found that manure application increased soil respiration but ammonium nitrate application did not. The manure effects were due to increased soil microbial biomass C and potentially mineralizable C. Soil texture may also exert some control over dynamic soil C fractions such as microbial biomass C and thus affect soil respiration. In turn, microbial biomass C will be affected by C input from roots coupled with the influences of soil moisture and temperature [23]. Ma et al. [23] also found that harvest frequency affected soil respiration and attributed the results to the effect of harvest frequency on root lifespan.

Establishment of any new crop, including switchgrass, usually entails an initial loss of soil C, incurring a "carbon debt" that must be repaid before net C sequestration can occur. Corre et al. [10] reported that conversion from cool-season grass to switchgrass initially resulted in a loss of SOC, and that it took 16–18 years after planting for SOC under switchgrass to approach that under the undisturbed cool-season grass. It has been suggested that growing perennial grasses on former cropland soils might result in little or no carbon debt, whereas, replacing uncultivated land could incur a debt that might require decades or even centuries to repay [3, 24]. Whatever the magnitude of the debt, it is important that initial C losses be considered when evaluating the C sequestration potential when replacing existing vegetation with switchgrass or other bioenergy crops.

6.2.2 Nitrous Oxide

Among carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), the three primary greenhouse gases associated with agricultural production systems, the latter has the greatest global warming potential. Although N_2O is found at lower atmospheric concentrations, its global warming potential can be as high as 296 times that of CO_2 over a 100 year period [25]. In nature, soil and oceans are sources of N_2O, where it is produced by microbial processes of nitrification and denitrification. Nitrification is the aerobic oxidation of ammonium to nitrate, and denitrification is the anaerobic reduction of nitrate to nitrogen gas. N_2O is a gaseous intermediate in the reaction sequence of denitrification and a minor by-product of nitrification [26].

Almost all agricultural systems are a significant source of direct (from agricultural lands) and indirect (from volatilization/deposition and leaching/runoff) N_2O emissions through the application of N fertilizers and animal manures.

In general annual crops produce about three times more emissions than unmanaged successional lands and perennial crops such as poplar [27]. According to these authors, the major determinant of N_2O emissions is the amount of nitrogen available in the soil. However, Stehfest and Bouwman [28] indicate that nitrogen fertilization rate, crop type, fertilizer type, soil organic content, soil pH, and texture also play an important role in controlling the activity of nitrifiers and denitrifiers and thus the N_2O emissions from agricultural fields. Therefore, cropping systems and crops such as switchgrass with lower nutrient demands or more efficient utilization of fertilizer inputs are likely to have a greater potential to reduce N_2O emissions and be more profitable to the farmer.

In general, the well developed root systems of grasses like switchgrass have a great capacity for N uptake and large amounts of inorganic N seldom accumulate in soils where they are grown [29]. Moreover, Bransby et al. [30] indicated that switchgrass has the ability to recover about 66% of applied N, which is about 16% higher than an established standard of wheat (*Triticum aestivum* L.) or maize, confirming its high potential to reduce GHG emissions compared to annual crops. Moreover, a comprehensive review of switchgrass and *miscanthus* agronomy indicated that switchgrass has a stronger response to N fertilization than *miscanthus* [2]. This higher response to fertilization could be one of the reasons for the 75% lower N_2O emission from switchgrass than *miscanthus* as reported by Zeri et al. [31] in one of the few side-by-side comparisons of N_2O fluxes from these grasses.

The fertilization rates for switchgrass vary widely; several authors indicated that economically and energetically viable yields can be obtained with 0–100 kg ha^{-1} of N fertilization, depending on site-specific soil conditions, water availability, and crop management [32–34]. Moreover, for optimum biomass yields, Vogel [35] indicated that switchgrass requires between 10 and 12 kg N ha^{-1} for each Mg ha^{-1} of biomass produced.

Since Bransby et al. [30] showed evidence that the N recovery capacity of switchgrass does not change with varieties or harvesting times but only with the yield levels, it is assumed that N_2O emission could be decreased by increasing switchgrass productivity. However, due to the sometimes quadratic [36] dependency between increased fertilization and yields, the effective potential to reduce GHG emission can be counteracted by N_2O emissions that may or may not be proportional with the amount of nitrogen fertilization. For example, Heggenstaller et al. [36] indicated that total plant N content with fertilization rates of 220 kg N ha^{-1} was less than with 140 kg N ha^{-1}, indicating that at higher fertilization rates more N remained in the soil with a greater potential for N loss from the system. The N loss was probably by a combination of volatilization, denitrification, and/or leaching. However, because information is lacking on the relative contribution of each N loss pathway, an exact determination of the greater N_2O emission potential at the higher fertilization rate is difficult.

Actual quantification of N_2O emissions from switchgrass fields is almost nonexistent, not only because installing the measuring chambers is costly, but also because of the high spatial and temporal variability in N_2O fluxes [25]. Some studies have shown that the highest N_2O fluxes occur just after N fertilizer application and/or after large rainfall events [37, 38], making it difficult for spot measurements with small chambers to be representative of total GHG emissions. Therefore, most of the emission values reported in the literature are estimated based on emission factors and calculation guidelines developed by the IPCC [26] and life cycle analysis (LCA) studies such as Qin et al. [39], Adler et al. [40], Crutzen et al. [41], among others. Discrepancies and uncertainties between reported emissions depend on how they were calculated and expressed in the respective LCAs.

Adoption of fertilizer best management practices is one strategy that could reduce N_2O emissions by 30–40% [42]. The appropriate amount, timing, and placement of fertilizers are examples of best management practices [30, 37, 42, 43], but the particular response of switchgrass to such practices depend on climatic, management, and mycorrhizal symbiotic relations [35].

Other agronomic practices such as intercropping with legumes contribute to reduce emissions, although the decomposition of organic residues may contribute to postharvest N_2O emissions. In any case, the limited available results suggest that switchgrass, when compared to other crops, is particularly good at mitigating the soil N_2O emissions associated with N fertilizer applications. However, more studies based on actual measurements of N_2O fluxes are needed to confirm or provide more precise emission factors to be used in LCA and other studies because the general figure that 70% of GHG emission from agricultural activities comes from N_2O emissions seems to be an underestimation [41, 44]. If N_2O emissions are higher than the IPCC estimations, its mitigation will become a priority or at least of equal importance as C sequestration [45].

6.2.3 Methane

CH_4 is a greenhouse gas with global warming potential equivalent to 21 times that of CO_2 [46]. Lately, its atmospheric concentration has increased significantly mainly due to agricultural activities and the use of fossil fuels [44]. Soils can act as sources or sinks for CH_4, depending on land use and climatic conditions [25, 46, 47]. Soil temperature, moisture, pH, and soil N status are factors affecting the capacity of a soil to act as a CH_4 sink [48]. Moreover, forest soils and grasslands are net consumers of CH_4 and have a greater sink potential than cultivated soils, as agronomic and fertilization practices reduce the sink potential of the soil [46–50]. For example, Mosier et al. [51] indicated that annual fertilization increases N_2O fluxes and at the same time decreases CH_4 uptake in the soil by 41%.

In mid and late unmanaged successional forests, N_2O emissions were almost completely offset by CH_4 oxidation [27]. Moreover, in unfertilized and undisturbed grasslands CH_4 uptake was 1.4 and 2 times higher than that in fallow lands and cultivated wheat fields [51]. Since switchgrass is a typical perennial grass with low fertilization and tillage requirements, CH_4 emissions from this crop may be

Table 6.1 N_2O and CH_4 emission factors from switchgrass feedstock production in the power generation chain

Source of emission by activity	Emissions (kg ha^{-1})	
	N_2O	CH_4
Land preparation	2.22E−4	1.23E−2
Crop growth	1.11E−3	5.93E−2
Crop harvest	2.72E−3	1.23E−1
Transport harvested material	2.72E−2	1.41E−0
Production and use of fertilizers and atrazine	5.01E−0	1.6E−0
Use of lime	2.47E−4	1.23E−2
Biomass degradation losses	0	6.10E+1
Combustion in boilers	2.22E−0	3.46E−0
Post combustion activities	4.20E−5	2.07E−3

Data source Qin et al. [39], assumed switchgrass biomass yield 25 Mg ha^{-1}, stand life 10 years, transport distance 40 km

close to zero, or there may be significant CH_4 uptake. For example, Adler et al. [40] indicated that CH_4 uptake by switchgrass was 1.41 g CO_2-eq m^{-2} yr^{-1}. However, another study estimated that during the agronomic practices to establish switchgrass the total CH_4 emission were 23 g CO_2-eq m^{-2} and that during the harvesting operations (mowing, bailing, etc.) emissions were 17.4 g CO_2-eq m^{-2} [52].

Currently, however, limited information is available on CH_4 flux contributions to net GHG emission from switchgrass. Qin et al. [39] in a LCA study estimated that the largest CH_4 emissions are produced during the processing/combustion phase of switchgrass (Table 6.1), but even then they remained of low significance. As far as we know actual CH_4 flux measurements in a switchgrass stand are nonexistent, probably because most studies do not consider it relevant to include these measurements because of the assumed small effect on GHG emissions. Therefore, CH_4 flux based on field measurements are urgently needed to precisely determine the most impacting phases (cultivation, transformation, etc.) of switchgrass when used as a feedstock for diverse purposes.

6.2.4 Life Cycle Assessment

In theory, LCA is an all-inclusive account of the inputs and outputs of a productions cycle [53]. Inputs and outputs can include energy requirement and yield, economic cost and benefit, and environmental impacts whether positive or negative. However, the meaning of 'all-inclusive' can be somewhat nebulous, and a clear definition of comparable system boundaries, both for alternative and traditional fuel sources, is necessary but potentially difficult to achieve when conducting a LCA. The purpose of this review is not to evaluate the appropriateness of various LCAs, but analysis boundaries must be kept in mind when evaluating LCA results.

Because the use of biofuels was prompted by the recognitions of human impacts on global warming and the need to reduce GHG emissions, an appropriate starting point for LCA would be to examine total GHG emissions from various bioenergy systems. In an early LCA comparing switchgrass with other bioenergy crops, Adler et al. [40] found producing ethanol and biodiesel from switchgrass and hybrid poplar reduced GHG emissions by 115% compared with gasoline and diesel. In comparison, maize rotations reduced GHG emissions by 40% and reed canary grass by 85%. They found that displaced fossil fuels were the largest GHG sink, followed by soil C sequestration. They also concluded that GHG reductions resulting from biomass gasification for electricity generation were greater than for biomass conversion to ethanol.

Other studies have found smaller GHG savings from ethanol production from switchgrass. Thus, Cherubini and Jungmeier [43] calculated that the use of switchgrass in a biorefinery reduced GHG emissions by 79% with soil C sequestration responsible for a large part of the GHG benefit. Bai et al. [54] found a 65% reduction in GHG emissions with switchgrass ethanol fuels, and Hsu et al. [55] suggested a 43–57% reduction compared with cars operating on conventional gasoline.

The LCA by Adler et al. [40] suggested that N_2O emissions were the largest GHG source. According to Qin et al. [39] the largest source of N_2O emissions during the crop production phase are: the production and use of fertilizer and other chemicals, the transport, harvest, and growth stages, in that order of importance (Table 6.1). When comparing a biorefinery fed with switchgrass biomass with a traditional fossil fuel refinery, Cherubini and Jungmeier [43] indicated that during the first 20 years of operation of the biorefinery the use of switchgrass had a net reduction in GHG emissions, but that the emissions of N_2O were about 10 times higher than in the fossil fuel refinery. This was due to N_2O emissions from the N fertilizer (112 kg N ha^{-1}) applied to the soil and possibly because of the decomposition of the soil organic matter and dead roots but it seems that this point is not taken into account by the authors. According to their computations, the production phase of switchgrass was responsible for 80% of the GHG emissions and from that 40% were N_2O emissions.

Several other studies also indicated that the crop production phase is the main source of N_2O emissions [40, 56, 57]. Therefore, one of the best options to reduce the large impact of fertilization in GHG emissions would be to minimize the use N fertilizers, or to use and develop more efficient N-use strategies. This would also be the case when manures are the fertilizer source because losses of ammonia to the atmosphere and nitrate to groundwater are larger with manures than from synthetic inorganic fertilizers [45].

It is important to also consider other environmental costs and benefits when evaluating bioenergy production systems. In one such analysis, Harto et al. [58] investigated the life cycle water use of biofuel and other low-carbon transport systems. They found that adoption of electric vehicles and some algae-based and switchgrass systems could contribute to the decarbonization of transportation systems with little additional water consumption. However, use of irrigated biofuel crops could have a significant potential impact on water resources. Whereas, non-irrigated cellulosic ethanol production would require less than 10 gallons of water per gallon of fuel produced, irrigated maize or cellulosic ethanol would consume more than 400 gallons of water per gallon of fuel. They concluded that using irrigated switchgrass to provide 10% of transportation fuel demand in the US would require 7% of total consumptive water demand. Supplying 50% would require 37% of total national annual consumption. Non-irrigated switchgrass, on the other hand, would only consume 1.4% of annual water use to supply 50% of the national fuel demand.

In addition to global warming potential, Bai et al. [54] conducted LCAs for bioenergy production effects on abiotic depletion, eutrophication, photochemical oxidation, ozone layer depletion, human toxicity, eco-toxicity, and acidification. Results were mixed, with bioenergy production reducing global warming, abiotic depletion, and ozone layer depletion, but increasing human toxicity, eco-toxicity, acidification, photochemical oxidation, and eutrophication. Reduced emissions from crude oil production caused the reduction in ozone layer depletion, whereas, abiotic resource depletion decreased due to reduced use of crude oil. They attributed the higher eutrophication score to nitrate leaching from N fertilizer application, whereas, human and eco-toxicity increased due to the use of agro-chemicals. Increased acidification resulted from ammonia emissions from agriculture. Cherubini and Jungmeier [43] also concluded that biofuel production had greater impacts on acidification and eutrophication.

These studies suggest that additional environmental impacts should not be disregarded when evaluating the impact of bioenergy production systems on GHG emissions, but no recommendations were made concerning how to rank the importance of competing environmental impacts, or on how to compute an overall "environmental score" for bioenergy production. Such information will be crucial for evaluating the advisability of using bioenergy sources to replace fossil fuels.

6.2.5 Indirect Land Use Change

The production of switchgrass for energy purposes at an industrial level requires, as with any other crop, large expanses of agricultural croplands. Since projections indicate that the global population will continue to increase, as will their food, feed, and energy demands [59, 60], it is foreseen that the croplands dedicated to produce energy feedstocks such as switchgrass will most probably come from the displacement of existing crops or from the conversion of grasslands and forests. In either case, the new uses of the land will lead to direct and indirect changes in the carbon balance within and outside the boundaries of the newly introduced system with the consequent effects on global climate, food and feed supplies, and ecosystem services [61].

In general, when a natural forest or pasture is replaced by an annual crop the soil and biomass carbon emissions increase significantly [59]. On the other hand, when a cropland cultivated with annual crops or an abandoned cropland is converted to a perennial grass such as switchgrass, large amounts of carbon could be sequestered in the soil and therefore a net reduction in the atmosphere could be expected. However, this process is not always straightforward because many variables are involved [30, 59, 62, 63]. The degree of impact will be a function of the type of crop replaced, root mass, soil depth, soil bulk density, climatic conditions, and crop management and intensity, among others.

In general, it has been estimated that when perennial grasses are introduced into croplands the carbon stocks in the soil increase at a rate between 1.1 and 1.2 Mg C ha^{-1} yr^{-1} [64–66]. In agreement with such general estimations, Garten and Wullscheleger [67] predicted that during a 10–30 year conversion period of a cropland to switchgrass the SOC sequestration rate would be 0.78 and 0.53 Mg C ha^{-1} yr^{-1}, respectively, probably because the large amount of assimilates accumulated in its extended root system. On the other hand converting native grasslands, such as those in U.S., to maize resulted in 5.6 Mg C ha^{-1} yr^{-1} emissions [24].

Land use change (LUC) is not a new topic as the expansion of the agricultural frontier led to, and will continue to lead to significant CO_2 emissions, especially when tropical forest (rich in above-and below-ground carbon) are converted into agricultural lands [68, 69]. Beyond the environmental effects, LUC could have short-term effects on crop prices, stocks, farm incomes, and demand of agricultural products. However, only recently have the direct and iLUC effects been taken into account in the estimations of GHG emission and LCA studies of biofuel and bioenergy production systems.

However, the estimation process has a number of limitations and there is not a conventionally accepted estimation procedure. Since there are not reliable data on LUC effects, the estimations are highly variable and susceptible to economical, political, social, and environmental influences. For example, Fargione et al. [24] and Searchinger et al. [70] indicated that including the effect of LUC and iLUC in the analysis could result in GHG emissions being even higher for bioenergy crops than that of current fossil fuels. On the other hand, other estimates indicate that the positive benefits of biofuels could be seen under certain circumstances and type of crops such as sugarcane or other perennial grasses [60]. Switchgrass, being a perennial grass with low input requirements and high carbon sequestration capacity, may be part of the group of crops with positive environmental effects. But the available information is not sufficient to determine, with an acceptable degree of approximation, its indirect and direct effects.

Changing the preexisting vegetation to bioenergy crops causes removal or sequestration of CO_2, but such changes could be negated or enhanced elsewhere because of the spatial and temporal nature of the replacement effects [71]. Therefore, the iLUC effects are non-local and not specific to a feedstock, so they cannot be quantified directly but only through modeling [71]. Since the dynamics of iLUC are dominated by international trading trends, food and feed prices, agricultural policies, climatic conditions, among others, its global nature makes it very difficult to model. In fact, the validity of the current available methods is hotly debated. But in general two approaches are widely used. In the economic approach, linkages between complex macro- and micro-economic models with biophysical models are used to estimate GHG emissions associated with iLUCs. While in the deterministic approach, the iLUC analysis is based on the export/ import trends of agricultural commodities in the most relevant countries.

Examples of the most common economic and deterministic models used to estimate iLUC effects can be found in Searchinger et al. [70] and Fritsche et al. [61]. In both cases, however, the results and predictions remain vague and variable, mainly because of insufficient analysis of market distortions, complexity of the factors considered, and insufficient analysis of trading levels. A recent study [60], in which seven agro-economic models were compared, indicated a wide range (from 10 to 80 g CO_2 MJ^{-1} of biofuel produced) of overall emissions from iLUC. The large variability mainly depended on the assumptions used in each model. However, in the case of switchgrass none of these models may be applicable as none of them considered the iLUC effects of second generation feedstocks, showing the urgent need to develop estimation procedures that take into account perennial grasses. In the case of the deterministic model, it was shown that adding iLUC plus LUC emissions in LCAs could almost double GHG emissions per unit energy [71].

Some authors consider model simulation approaches to not be sufficiently accurate, therefore they use the risk-adder method, which estimates the average LUC area per additional hectare of bioenergy production [60, 70, 72], to determine the maximum possible effects of iLUC. Based on that approach, Searchinger et al. [70] indicated that even when U.S. maize fields are converted to switchgrass, GHG emissions still increase by 50% over a period of 30 years. Such results raise great concerns about the potential of switchgrass to reduced GHG emissions associated with iLUC. However, this seems to be an overestimation mainly because of the arbitrary assumptions and non-replicable parameters used in the study. But it is clear from this and other studies that the approach of eliminating any iLUC risk provides very rough estimates, which in turn seem insufficient for generalization and rulemaking. Therefore, further studies are needed to define more precise evaluation methods and specific criteria to quantify consistent iLUC values in order to opportunely include them in GHG emission balances.

In any case, it is clear that the production of biofuel and bioenergy leads to GHG emissions associated with iLUC, and controlling them could be an important factor for mitigating the global warming process. Several authors suggest that optimizing the use of byproducts as biofuels feedstocks, maximizing the use of crop residues as biofuels feedstocks, and cultivation of feedstocks on abandoned croplands are measures that to some extent could reduce the iLUC effects on GHG emissions [45, 60]. In addition, technological developments along the supply chain, improved feedstocks, crop management, and improved conversion efficiencies (e.g. bioelectricity instead of biofuels from lignocellulosic crops such as switchgrass) will reduce the impact of the bioenergy feedstock on the GHG balance [60, 73]. Global climate policies with emissions caps would also help to control iLUC effects. In fact, any measure that reduces the land requirements for feedstocks will contribute to mitigate the effects of direct and iLUCs. As for switchgrass, the research window remains completely open as information on the aforementioned aspects is almost nonexistent.

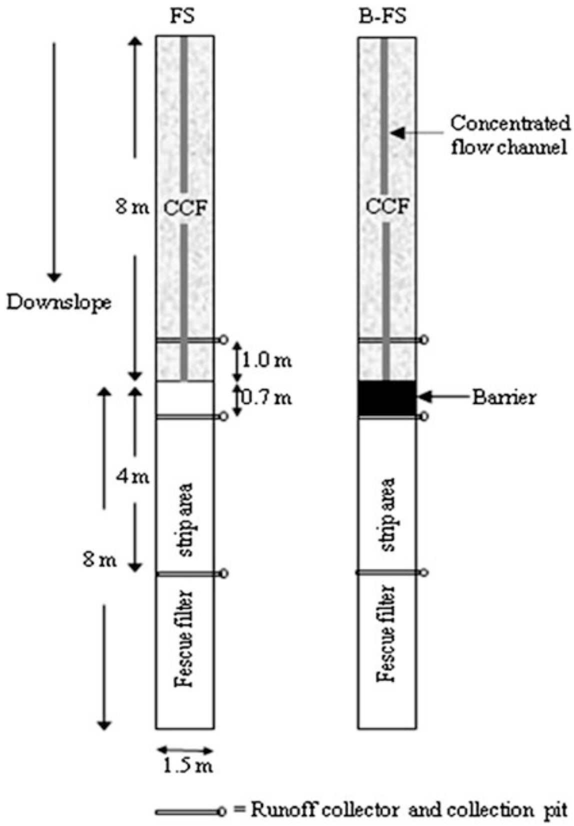
6.3 Impact on Water Quality

6.3.1 Runoff, Nutrient, and Sediment Losses

Before switchgrass became the focus of research as a bioenergy crop, its environmental impact on surface water runoff and quality were considered from the perspective of using this grass species in vegetative buffer strips. Switchgrass has upright and stiff stems and a rhizomatous growth habit, characteristics that make it desirable for intercepting sediment from surface runoff and allowing it to regrow through sediment that has been deposited on the soil surface. The USDA Natural Resource Conservation Service identifies these criteria for selecting desirable species in vegetative buffers (Code 601; [74]). In one study in which several grass species were evaluated as a narrow hedge (0.75–1.0 m) grown along the contour to impede surface water runoff, Dabney et al. [75] determined that the primary explanation for the sediment trapping efficacy of the hedge was the result of flow constriction and water backing up above the hedge. The backed up water slowed runoff and the sediment load was deposited, thus the filtering efficacy of the hedge. In a field study in which the effectiveness of a switchgrass hedge was determined for runoff and sediment loss from maize plots, the hedge reduced runoff (41%) and sediment (63%) losses [76]. While species of grass in the hedge seemed unimportant in these studies as long as there was the physical constraint of backing up the surface runoff, cool-season grasses may not withstand the sediment deposition as well as non-bunch, rhizomatous warm-season grasses like switchgrass.

Rainfall simulations were used in a plot (1.5 × 16 m) study to contrast the effectiveness of filter strips of different grass species, including: tall fescue [*Lolium arundinaceum* (Schreb.) S.J. Darbyshire], tall fescue with a switchgrass barrier (0.7 m wide), and a mixture of grasses and forbs native to the Midwestern United States with a switchgrass barrier [77]. Compared to a tilled plot, tall fescue improved surface water quality on this Mexico silt loam soil by reducing organic N (55%), NO_3–N (27%), NH_4–N (19%), particulate P (36%), and PO_4–P (376%). The addition of the switchgrass barrier (similar to a narrow hedge) provided greater effectiveness, reducing organic N by 67% compared to the tilled plot, and also reducing NO_3–N (68%), NH_4–N (50%), particulate P (53%), and PO_4–P (54%). Expanding this experiment to consider concentrated flow paths for the same fescue strips with switchgrass barriers (Fig. 6.1), Blanco-Canqui et al. [78] determined that dormant or actively growing switchgrass were similarly effective

Fig. 6.1 Diagram of fescue filter strips combined with a narrow switchgrass barrier that was effective in reducing sediment and nutrient losses in runoff (redrawn from Blanco-Canqui et al. [78]; reprinted with permission from the Soil Science Society of America)



in reducing runoff and sediment losses. As with the previous study, the switchgrass barrier was more effective than only tall fescue in reducing sediment and nutrient losses.

In a direct comparison between cool-season grasses [smooth bromegrass, timothy (*Phleum pretense* L.), and tall fescue] and switchgrass, switchgrass was more effective than the cool-season grasses in removing sediment, N, and P from surface runoff [79]. This comparison was based on 3 or 6 m wide filter strips. The 6 m wide strips were generally 50% more effective than the 3 m wide strips in removing sediment and nutrients. While the switchgrass was usually significantly better than the cool-season grasses at filtering runoff sediment and nutrients, differences were generally less than 10% (Table 6.2).

Switchgrass hedges have also been effective in reducing nutrient runoff losses from plots receiving manure or fertilizer [80]. In no-till plots receiving manure or fertilizer, the adjacent (and downslope) hedge reduced runoff concentrations of dissolved P (47%), bio-available P (48%), particulate P (38%), total P (40%), and NH_4–N (60%). In the disked plots, the reduction in runoff concentration was not as great for dissolved P (21%), but was generally comparable for the other nutrients.

Table 6.2 Efficacy of switchgrass and cool-season grasses in reducing sediment and nutrient runoff (redrawn from [79])

Strip							
Width (m)	Area ratio	Grass	Sediment (%)	Total N (%)	NO_3–N (%)	Total P (%)	PO_4–P (%)
6	20:1	Switchgrass	78.2 a*	51.2 a	46.9 a	55.2 a	46.0 a
6	20:1	Cool-season	74.8 b	41.1 b	37.5 b	49.4 b	39.4 b
Overall average			76.5	46.2	42.2	52.3	42.7
3	40:1	Switchgrass	69.0 c	31.7 c	28.1 c	39.5 c	38.1 b
3	40:1	Cool-season	62.0 d	23.5 d	22.3 d	35.2 c	29.8 c
Overall average			65.5	27.6	25.2	37.4	34.0

* Percent within a column for reduction followed by a different letter are significantly different ($P < 0.05$)

Water quality research related to switchgrass hedges has mostly focused on the reduction of runoff losses and improvement in associated water quality characteristics, attributing these reductions to water backing up above the hedge. Improving infiltration and/or hydraulic conductivity of the soil surface within a switchgrass stand would contribute to additional water quality improvements by filtering fine sediment particles and other soluble nutrients. Measuring field-saturated hydraulic conductivity (K_{fs}) was the focus of a study in Iowa on a Monona silt loam soil [81]. In this study K_{fs} was measured at three different locations on the landscape: (1) 7 m upslope from the switchgrass hedge in a maize or soybean field, (2) 0.5 m upslope from the hedge in the sediment depositional area, and (3) within the grass hedge. These hedges had been in place for 10 years. The K_{fs} within the grass hedge (107 and 154 mm h^{-1}) was more than seven times greater than the K_{fs} in the row crop field (13.5 and 22.5 mm h^{-1}) and more than 24 times greater than the depositional area (1.4 and 9.4 mm h^{-1}). Infiltration was measured under conditions of increasing soil water tension. As tension increased to 50 and 100 mm, infiltration within the hedge was still greater than in the row crop field; but as tension was increased to 150 mm, the infiltration within the hedge and row crop field became similar. A reduction in sediment losses due to a switchgrass hedge can probably be attributed mostly to water backing up above the hedge, but the reduction in nutrient loss is likely attributable to the greater infiltration within the hedge.

The efficacy of a 7.1 m buffer of switchgrass was compared to the switchgrass buffer with an additional 9.2 m length of switchgrass and woody species mix [82]. In a 2 h rainfall simulation (22 mm h^{-1}), the switchgrass buffer trapped 70% of the incoming sediment and 64, 61, 72, and 44% of incoming total N, NO_3–N, total P, and PO_4–P, respectively. The additional length of switchgrass-woody species buffer improved these numbers to 92% of the sediment and 80, 92, 93, and 85% of the respective nutrients. The woody species in the additional buffer provided greater infiltration, plus the additional length of buffer contributed to the overall greater effectiveness of the buffer in the latter scenario.

When compared to row crops in small-plot studies, switchgrass is very effective at reducing sediment and nutrient loads in surface runoff. The examples provided here were often side-by-side comparisons for standing crops. Because switchgrass is a perennial crop that will not require additional tillage or re-establishment of the crop, an annual (and long-term) comparison of water quality between a row crops and switchgrass should be even more favorable for switchgrass. When switchgrass was compared to cool-season grasses, such as tall fescue, improvements in the sediment, N, and P from surface runoff were slightly better with switchgrass, but the differences were generally less than 10%. Switchgrass makes an effective ground cover for improving water quality.

6.3.2 Expanding the Spatial and Temporal Scale of Switchgrass Impacts

Numerous land use studies considering changes to switchgrass have evaluated the environmental impact at larger scales, from the small watershed to the Mississippi River basin. Using the Soil and Water Assessment Tool (SWAT), hillslope processes were modeled and the environmental impacts were considered if planting switchgrass to 10, 20, 30, and 50% of the Walnut Creek watershed (51.3 km^2) near Ames, IA [83]. Filter strips of switchgrass representing 10–50% of the sub-basin could lead to a 55–90% reduction in NO_3–N load during an average rainfall year. In the larger Delaware River basin of NE Kansas, SWAT was used to consider sediment yield, surface runoff, NO_3 in surface runoff, and edge-of-field erosion [84]. If the cultivated cropland (119,400 of 300,000 total ha) were converted to switchgrass, sediment loss would be reduced by 99%, surface runoff by 55%, NO_3 loss by 34%, and edge-of-field erosion by 98%. Evaluating a shift to switchgrass in the large, agriculturally dominated Raccoon River watershed of central Iowa (9,364 km^2), results from SWAT indicated that lower water yield will correspond with less NO_3, less P, and less sediment loss [85]. These scientists suggested that even though a shift in land use (i.e. toward more switchgrass) might resemble a pre-1940s land use and land cover, the extensive tile drainage network would prevent the hydrology from ever resembling pre-1940s condition. Tile drainage is meant to move water quickly away from agricultural fields with the inadvertent consequence of carrying its nutrient load with it. Nevertheless, growing maize results in lower annual evapotranspiration and therefore greater runoff and drainage than a perennial cropping system, such as switchgrass; resulting in a shift toward fewer environmental impacts when more switchgrass is grown.

Expanding the spatial scale even further to an area encompassing much of Missouri, Iowa, Nebraska, and Kansas in the Midwestern U.S. (15,100 km^2), Brown et al. [86] used the Erosion Productivity Impact Calculator (EPIC) to consider future environmental impacts of growing more switchgrass in this region. Their model scenarios also extended the temporal scale as well by considering crop yields and sediment loss under increased atmospheric CO_2 (560 μg g^{-1}). Alternative climate conditions were considered using the general circulation model (GCM) from CSIRO. With increased temperature, switchgrass yield increased by 5 Mg ha yr^{-1}, whereas other crop yields decreased (Mg ha^{-1} yr^{-1}): maize, 1.5; sorghum, 1.0; soybean, 0.8; and wheat, 0.5. With additional CO_2 under this otherwise similar future scenario, all crops responded with greater yield compared to the future scenario with only increased temperature. Greater rainfall predicted with climate change corresponded with greater runoff and generally increased sediment loss, except with switchgrass for which sediment loss generally decreased. A stochastic model was used in another study to evaluate the impact of producing cellulosic ethanol (i.e. growing switchgrass) compared to maize-derived ethanol in the Mississippi River basin [87]. They concluded that cellulosic ethanol production would result in a 20% decrease in NO_3 delivered to the Gulf of Mexico.

Growing switchgrass compared to growing a row crop will increase water use and increase infiltration, both of which will have the net effect of reducing runoff. These impacts described and measured at the small plot scale translate to reduced nutrient losses at the larger watershed and basin scales. Including switchgrass on the landscape will have a favorable impact on improving water quality.

6.3.3 Seasonal Nitrogen Dynamics: Implications for Management and Environmental Impacts

Perhaps some of the more interesting questions about growing switchgrass as a bioenergy crop are intertwined in the N dynamics of physiological characteristics, production management, and their corresponding impacts on N in the environment. For example, when is N taken up by switchgrass? When will switchgrass be harvested? What is the biomass N content at harvest? How much N fertilizer will be applied? How do these physiological characteristics impact NO_3 water quality and N_2O emissions? Some of these questions have already been addressed with recent research, though some remain unanswered.

Cropland in the Midwestern U.S. is extensively tile-drained, designed to move water quickly from the field to improve soil conditions for maize and soybean production. This conduit for water also effectively moves NO_3 from agricultural soils to streams and rivers [88, 89], significantly contributing to the increase in NO_3 flux in the lower Mississippi River [89]. Compared to continuous maize or maize–soybean crop rotations, perennial crops will reduce NO_3 leaching in this tile-drained landscape from more than 60 kg N ha^{-1} yr^{-1} to less than 5 kg N ha^{-1} yr^{-1} [90]. This reduction in NO_3 loss from perennial crops can be attributed to both lower NO_3 concentration in the tile water and reduced drainage as a result of greater evapotranspiration.

In a recent study [91], the impact of *miscanthus* and switchgrass managed as bioenergy crops on hydrology and NO_3 leaching was evaluated in the tile-drained landscape of the Midwestern U.S. The soil profile water content was consistently less with *miscanthus* than with either switchgrass or a maize-soybean rotation, especially later in the growing season. Nitrate leaching was much greater in the maize-soybean rotation (34–45 kg N ha^{-1} yr^{-1}) than either switchgrass (0.3–3.9 kg N ha^{-1} yr^{-1}) or *miscanthus* (1.6–6.6 kg N ha^{-1} yr^{-1}). Although N fertilizer was not applied to switchgrass in this study, N applications will likely be a component of switchgrass management as a bioenergy crop; but this study illustrates that a perennial switchgrass crop will contribute much less N to ground water than a crop rotation including maize and soybean.

In a 2 year study in Illinois [92], the biomass and N content of *miscanthus* and switchgrass were evaluated in a side-by-side trial at three locations spanning a 5° latitudinal range (37–42° N). Biomass was harvested on five dates between June and February. As much as 60 Mg ha^{-1} of biomass was harvested for *miscanthus* and more than 20 Mg ha^{-1} for switchgrass. Nitrogen concentrations of the biomass for both species decreased from 1.5 to 2.5% in June to <0.5% in December. Nitrogen concentrations in February were similar to concentrations in December, so there was not an incentive to postpone harvest until February. Harvest in December corresponded with greater biomass than in February and similar N concentrations. When harvested in June, switchgrass removed as much as 187 kg N ha^{-1} and *miscanthus* as much as 379 kg N ha^{-1}, whereas N removal in February corresponded to as little as 5 and <17 kg N ha^{-1}, respectively, because N within the plant was translocated to the rhizomes during winter dormancy. The December harvest, corresponding to low N concentrations and high biomass yield, was the most suitable harvest date for providing a large amount of desirable biofuel feedstock and represents an efficient N recycling cropping system—meeting economic and environmental objectives.

A 5 year study in Knoxville, TN contrasted a two-cut (summer and fall) harvest plan to a one-cut (fall) plan for switchgrass [93]. Biomass yield was similar for the two approaches (17.4–18.7 Mg ha^{-1}), but much less N was removed with the one-cut plan (48 kg N ha^{-1}) than with the two-cut plan (116 kg N ha^{-1}). One other study provided similar results, concluding that less N can be applied and less N removed in the biomass when switchgrass is harvested late in the fall [94]. If less N is removed (but remains in the rhizomes), less N fertilizer will be required; thus reducing loss risks associated with NO_3 leaching and N_2O emissions after N fertilizer applications.

Root distributions and dynamics and total soil respiration were evaluated for an edge-of-field and riparian area that included zones of poplar (*Populus* x *euro-americana*' Eugenei), switchgrass, cool-season grasses (smooth brome, timothy, and tall fescue), and soybean or maize [95]. The fine root biomass for switchgrass increased from 7 to 10 Mg ha^{-1} between May and November, remained relatively constant for poplar and cool-season grasses (6–8 Mg ha^{-1}) during this same period. Fine root biomass was always less than 2 Mg ha^{-1} for maize and soybean. Small root biomass (2–5 mm) was significantly greater in switchgrass (2 Mg ha^{-1}) than for any of the other species (<0.65 Mg ha^{-1}), with no differences across sampling dates. Coarse root biomass (>5 mm) was only observed under poplar (3.8 Mg ha^{-1}), soybean (1.1 Mg ha^{-1}), and maize (0.3 Mg ha^{-1}).

Root density was greater in the poplar, switchgrass, and cool-season grasses than the maize and soybean, for the 0–50, 50–100, and 100–125 cm depths. Soil respiration was greatest in the poplar and cool-season grasses. Soil respiration under switchgrass was less than for poplar or cool-season grasses; but greater than with maize or soybean. The implications from this research is that the additional roots provided by poplar, switchgrass, or cool-season grasses provide a carbon source that is greater and to greater depths than provided by maize or soybean. The additional C contributes to the riparian zone denitrification potential and the presence of growing roots has implication for additional NO_3 removal; consequently, roots deeper in the soil profile represents two effective means of NO_3 removal from ground water.

Switchgrass is a perennial warm-season grass that is native to North America. It grows to a height of about 2 m, has a deep and fibrous root system, and will produce between 5 and 20 Mg ha^{-1} yr^{-1} of above-ground biomass [96]. A stand of switchgrass may maintain this productivity for 15–20 year with much less fertilizer and chemical inputs than usually applied to crops such as maize and soybean. Some of the physiological characteristics that suggest that switchgrass should have a favorable impact on the environment compared to most other agriculture cropping systems include: (1) after crop establishment the soil will remain undisturbed for many years, reducing soil erosion and energy inputs (as fuel); (2) low N fertilizer inputs translates into low energy demand for growing the crop and reduced risk of N_2O emissions and NO_3 leaching that might occur with almost any N fertilizer application; (3) low energy inputs relative to harvested biomass; and (4) a perennial crop with a fibrous root system translates to reduced water and nutrient losses through leaching as well as an effective surface runoff filter. If and when land use currently in traditional cropping systems is converted to switchgrass, environmental impacts should be favorable.

6.4 Conclusions

The number of studies looking at the environmental impacts of switchgrass cultivation is extremely limited, especially when switchgrass was managed for bioenergy production. In particular, additional studies at multiple locations are needed to identify the climatic and edaphic drivers of soil C sequestration. Similar research is needed for N_2O emissions, but in addition, continuous flux measurements throughout the year are needed to indentify the contribution of periodic high-emission events to total annual emission rates. Additional research on how N fertilization rates affect N_2O emissions and NO_3 leaching and on interactions between environmental effects of N fertilization and biomass production is also warranted. What limited information there is suggests that switchgrass provides multiple environmental benefits compared to annual crop cultivation. However, benefits generally appear to be similar to other perennial crops.

References

1. Wright L, Turhollow A (2010) Switchgrass selection as a “model” bioenergy crop: a history of the process. Biomass Bioenergy 34:851–868
2. Heaton E, Voigt T, Long SP (2004) A quantitative review comparing the yields of two candidate C4 perennial biomass crops in relation to nitrogen, temperature and water. Biomass Bioenergy 27:21–30
3. Anderson-Teixeira KJ, Davis SC, Masters MD, Delucia EH (2009) Changes in soil organic carbon under biofuel crops. GCB Bioenergy 1:75–96
4. Blanci-Canqui H (2010) Energy crops and their implications on soil and environment. Agron J 102:403–419
5. Liebig MA, Johnson HA, Hanson JD, Frank AB (2005) Soil carbon under switchgrass stands and cultivated cropland. Biomass Bioenergy 28:347–354
6. Chamberlain JF, Miller SA, Frederick JR (2011) Using DAYCENT to quantify on-farm GHG emissions and N dynamics of land use conversion to N-managed switchgrass in the southern U.S. Agric Ecosyst Environ 141:332–341
7. Davis SC, Parton WJ, Dohleman FG et al (2010) Comparative biogeochemical cycles of bioenergy crops reveal nitrogen-fixation and low greenhouse gas emissions in a *Miscanthus x giganteus* agro-ecosystem. Ecosystems 13:144–156
8. Tolbert VR, Todd DE, Mann LK et al (2002) Changes in soil quality and belowground carbon storage with conversion of traditional agricultural crop lands to bioenergy crop production. Environ Pollut 116:S97–S106
9. Zan CS, Fyles JW, Girouard P, Samson RA (2001) Carbon sequestration in perennial bioenergy, annual corn and uncultivated systems in southern Quebec. Agric Ecosyst Environ 86:135–144
10. Corre MD, Schnabel RR, Shaffer JA (1999) Evaluation of soil organic carbon under forests, cool-season and warm-season grasses in the northeastern US. Soil Biol Biochem 31:1531–1539
11. Al-Kaisi MM, Yin X, Licht MA (2005) Soil carbon and nitrogen changes as influenced by tillage and cropping systems in some Iowa soils. Agric Ecosyst Environ 105:635–647
12. Coleman MD, Isebrands JG, Tolsted DN, Tolbert VR (2004) Comparing soil carbon of short rotation poplar plantations with agricultural crops and woodlots in north central United States. Environ Manage 33:S299–S308
13. Omonode RA, Vyn TJ (2006) Vertical distribution of soil organic carbon and nitrogen under warm-season native grasses relative to croplands in west-central Indiana, USA. Agric Ecosyst Environ 117:159–170
14. Lemus R, Lal R (2005) Bioenergy crops and carbon sequestration. Crit Rev Plant Sci 24:1–21
15. Ma Z, Wood CW, Bransby DI (2000) Soil management impacts on soil carbon sequestration by switchgrass. Biomass Bioenergy 18:469–477
16. Lee DK, Doolittle JJ, Owens VN (2007) Soil carbon dioxide fluxes in established switchgrass land managed for biomass production. Soil Biol Biochem 39:178–186
17. Lee DK, Owens VN, Doolittle JJ (2007) Switchgrass and soil carbon sequestration response to ammonium nitrate, manure, and harvest frequency on conservation reserve program land. Agron J 99:462–468
18. Skinner RH, Adler PR (2010) Carbon dioxide and water fluxes from switchgrass managed for bioenergy production. Agric Ecosyst Environ 138:257–264
19. Skinner RH (2008) High biomass removal limits carbon sequestration potential of mature temperate pastures. J Environ Qual 37:1319–1326
20. Valentini R, Matteucci G, Dolman AJ et al (2000) Respiration as the main determinant of carbon balance in European forests. Nature 404:861–865
21. Garten CT, Wullschleger SD (2000) Soil carbon dynamics beneath switchgrass as indicated by stable isotope analysis. J Environ Qual 29:645–653

22. Frank AB, Derdahl JD, Hanson JD, Liebig MA, Johnson HA (2004) Biomass and carbon partitioning in switchgrass. Crop Sci 44:1319–1396
23. Ma Z, Wood CW, Bansby DI (2000) Carbon dynamics subsequent to establishment of switchgrass. Biomass Bioenergy 18:93–104
24. Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P (2008) Land clearing and the biofuel carbon debt. Science 319:1235–1237
25. Robertson GH, Grace PR (2004) Greenhouse gas fluxes in tropical and temperate agriculture: the need for a full-cost accounting of global warming potentials. Environ Dev Sustainability 6:51–63
26. IPCC-Intergovernmental Panel on Climate Change (2006) N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Guidelines for national greenhouse gas inventories. Agriculture, forestry and other land use, vol 4
27. Robertson GH, Paul EA, Harwood RR (2000) Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289:1922–1925
28. Stehfest E, Bouwman L (2006) N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modelling of global annual emissions. Nutr Cycl Agroecosyst 74:207–228
29. Schimel DS (1986) Carbon and nitrogen turnover in adjacent grassland and cropland ecosystems. Biochemistry 2:345–357
30. Bransby DI, McLaughlin SB, Parrish DJ (1998) A review of carbon and nitrogen balances in switchgrass grown for energy. Biomass Bioenergy 14:379–384
31. Zeri M, Hickman GC, Bernacchi C (2009) Fluxes of nitrous oxide and carbon dioxide over four potential biofuel crops in central Illinois. American geophysical union, fall meeting 2009, abstract #A34A-02
32. Brejda JJ, Moser LE, Vogel KP (1998) Evaluation of switchgrass rhizosphere microflora for enhancing seedling yield and nutrient uptake. Agron J 90:753–758
33. Christian DG, Riche AB, Yates NE (2002) The yield and composition of switchgrass and coastal panic grass grown as a biofuel in southern England. Bioresour Technol 83:115–124
34. Lewandowski I, Scurlock JMO, Lindvall E, Christou M (2003) The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. Biomass Bioenergy 25:335–361
35. Vogel KP (2004) Switchgrass. In: Moser LE, Burson BL, Sollenberger LE (eds) Warm-season (C_4) grasses. Agronomy monogram 45, ASA, CSSA, and SSSA, Madison, Wisconsin
36. Heggenstaller AH, Moore KJ, Liebman M, Anex RP (2009) Nitrogen influences biomass and nutrient partitioning by perennial, warm-season grasses. Agron J 101:1363–1371
37. Burton DL, Zebarth BJ, Gillam KM, MacLeod JM (2008) Effect of split application of fertilizer nitrogen on N_2O emissions from potatoes. Can J Soil Sci 88:229–239
38. Davidson EA (1992) Sources of nitric-oxide and nitrous oxide following wetting of dry soil. Soil Sci Soc Am 56:95–102
39. Qin X, Mohan T, El-Halwagi M, Cornforth G, McCarl BA (2006) Switchgrass as an alternate feedstock for power generation: an integrated environmental, energy and economic life-cycle assessment. Clean Technol Environ Policy 8:233–249
40. Adler PR, Del Grosso SJ, Parton WJ (2007) Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecol Appl 17:675–691
41. Crutzen PJ, Mosier AR, Smith KA, Winiwarter W (2008) N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos Chem Phys 8:389–395
42. CAST-Council for Agricultural Science and Technology (2004) Climate change and greenhouse gas mitigation: challenges and opportunities for agriculture. CAST, Ames
43. Cherubini F, Jungmeier G (2010) LCA of a biorefinery concept producing bioethanol, bioenergy, and chemicals from switchgrass. Int J Life Cycle Assess 15:53–66
44. IPCC-Intergovernmental Panel on Climate Change (2007) Climate change 2007: synthesis report. Summary for policymakers

45. Powlson DS, Whitmore AP, Goulding WT (2011) Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and false. Eur J Soil Sci 62:42–55
46. Chan ASK, Parkin TB (2001) Effect of land use on methane flux from soil. J Environ Qual 30:786–797
47. Dutaur L, Verchot LV (2007) A global inventory of the soil CH_4 sink. Global Biogeochem Cycles 21: GB4013. doi:10.1029/2006GB002734
48. Tlustos P, Willison TW, Baker JC, Murphy DV, Pavlikova D, Goulding KWT, Powlson DS (1998) Short-term effects of nitrogen on methane oxidation in soils. Biol Fert Soils 28:64–70
49. Kara EE, Özdýlek HG (2010) The effect of nitrogenous fertilizers on methane oxidation in soil. Ekoloji 74:1–9
50. Kim DG, Isenhart TM, Parkin TB, Schultz RC, Loynachan TE (2010) Methane flux in cropland and adjacent riparian buffers with different vegetation covers. J Environ Qual 39:786–797
51. Mosier A, Schimel D, Valentine D, Bronson K, Parton W (1991) Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350:330–332
52. Ney RA, Schnoor JL (2002) Greenhouse gas emission impacts of substituting switchgrass for coal in electric generation: The Chariton valley biomass project. Center for global and regional environmental research, University of Iowa
53. Davis SC, Anderson-Teixeira KJ, DeLucia EH (2009) Life-cycle analysis and the ecology of biofuels. Trends Plant Sci 14:140–146
54. Bai Y, Luo L, van der Voet E (2010) Life cycle assessment of switchgrass-derived ethanol as transport fuel. Int J Life Cycle Assess 15:468–477
55. Hsu DD, Inman D, Heath GA, Wolfurm EJ, Mann MK, Aden A (2010) Life cycle environmental impacts of selected U.S. ethanol production and use pathways in 2022. Environ Sci Technol 44:5289–5297
56. Kim S, Dale BR (2004) Cumulative energy and global warming impact from the production of biomass for biobased products. J Ind Ecol 7:147–162
57. Kavdir Y, Hellebrand HJ, Kern J (2008) Seasonal variations of nitrous oxide emission in relation to nitrogen fertilization and energy crop types in sandy soil. Soil Tillage Res 98: 175–186
58. Harto C, Meyers R, Williams E (2010) Life cycle water use of low-carbon transport fuels. Energy Policy 38:4933–4944
59. Bowyer C (2010) Anticipated indirect land use change associated with expanded use of biofuels and bioliquids in the EU—an analysis of the national renewable energy action plan. Institute for European Environmental Policy IEEP, London
60. Croezen HJ, Bergsma GC, Otten MBJ, van Valkengoed MPJ (2010) Biofuels: indirect land use change and climate impact. Delft, CE Delft. http://www.ce.nl. June 2010
61. Fritsche UR, Sims REH, Monti A (2010) Direct and indirect land-use competition issues for energy crops and their sustainable production: an overview. Biofuel Bioprod Biorefin 4:692–704
62. Bullard M, Metcalfe P (2001) Estimating the energy requirements and CO_2 emissions from production of the perennial grasses *Miscanthus*, switchgrass and reed canary grass. ADAS report for the department of trade and industry. ETSU B/U1/00645/REP
63. McLaughlin SB, Ugarte DGDL, Garten CT et al (2002) High-value renewable energy from prairie grasses. Environ Sci Tech 36:2122–2129
64. Gebhart DL, Johnson HB, Mayeux HS, Polley HWJ (1994) The CRP increases soil organic carbon. J Soil Water Conserv 49:488–492
65. IPCC-Intergovernmental Panel on Climate Change (2000) In: Watson RW, Noble IA, Bolin B, Ravindranath NH, Verardo DJ, Dokken DJ (eds) Land use, land use change, and forestry. Cambridge University Press, Cambridge, UK
66. IPCC-Intergovernmental Panel on Climate Change (2003) Good practice guidance for land use, land-use change and forestry. Intergovernmental panel on climate change (IPCC), national greenhouse gas inventories programme

67. Garten CT, Wullschleger SD (2001) Soil carbon and nitrogen dynamics under switchgrass: integration of field data and modeling to ensure sustainability. Bioenergy feedstock development programs: subcontractors and collaborators meeting, Memphis, TN, ORNL, Oak Ridge, TN, 7–9 Nov 2001. http://bioenergy.esd.ornl.gov/02workshop/garten.html
68. Hall DO, Scurlock JMO (1991) Tropical grasslands and their role in the global carbon cycle. In: Esser G, Overdieck D (eds) Modern ecology: basic and applied aspects. Elsevier, Amsterdam
69. Rathmann R, Szklo A, Schaeffer R (2010) Land use competition for production of food and liquid biofuels: an analysis of the arguments in the current debate. Renew Energy 35:14–22
70. Searchinger R, Heimlich R, Houghton R et al (2008) Use of U.S. croplands for biofuels increased greenhouse gases through land-use change. Science 319:1238–1240
71. Fritsche UR, Hennenberg K, Hünecke K (2010a) The "iLUC Factor" as a means to hedge risks of GHG emissions from indirect land use change. Darmstadt: Öko-Institut. http://www.oeko.de/oekodoc/1030/2010-082-en.pdf. Retrieved 20 Aug 2010
72. Burniaux JM, Truong TP (2002) In GTAP technical paper, vol 16. West Lafayette, USA
73. Campbell JE, Lobell DB, Field CB (2009) Greater transportation energy and GHG offsets from bioelectricity than ethanol. Science 324:1055–1057
74. USDA–NRCS (2010) Vegetative barrier. Conservation practice standard, code 601. USDA NRCS Online: http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_026353.pdf
75. Dabney SM, Meyer LD, Harmon WC, Alonso CV, Foster GR (1995) Depositional patterns of sediment trapped by grass hedges. Trans ASAE 38:1719–1729
76. Gilley JE, Eghball B, Kramer LA, Moorman TB (2000) Narrow grass hedge effects on runoff and soil loss. J Soil Water Conserv 55:190–196
77. Blanco-Canqui H, Gantzer CJ, Anderson SH, Alberts EE, Thompson AL (2004) Grass barrier and vegetative strips effectiveness in reducing runoff, sediment, nitrogen, and phosphorus loss. Soil Sci Soc Am J 68:1670–1678
78. Blanco-Canqui H, Gantzer CJ, Anderson SH, Alberts EE, Thompson AL (2004) Grass barrier for reduced concentrated flow induced soil and nutrient loss. Soil Sci Soc Am J 68:1963–1972
79. Lee K, Isenhart TM, Schultz RC, Mickelson SK (1999) Nutrient and sediment removal by switchgrass and cool-season grass filter strips in central Iowa, USA. Agroforest Syst 44:121–132
80. Eghball B, Gilley JE, Kramer LA, Moorman TB (2000) Narrow grass hedge effects on phosphorus and nitrogen in runoff following manure and fertilizer application. J Soil Water Conserv 55:172–176
81. Rachman A, Anderson SH, Gantzer CJ, Thompson AL (2004) Influence of stiff-stemmed grass hedge systems on infiltration. Soil Sci Soc Am J 68:2000–2006
82. Lee K, Isenhart TM, Schultz RC, Mickelson SK (2000) Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. J Environ Qual 29:1200–1205
83. Sahu S, Gu RR (2009) Modeling the effects of riparian buffer zone and contour strips on stream water quality. Ecol Eng 35:1167–1177
84. Nelson RG, Ascough JC II, Langemeier MR (2006) Environmental and economic analysis of switchgrass production for water quality improvement in northeast Kansas. J Environ Manage 79:336–347
85. Schilling KE, Jha MK, Zhang Y, Gassman PW, Wolter CF (2008) Impact of land use and land cover change on the water balance of large agricultural watershed: historical effects and future directions. W00A09. doi:10.1029/2007/WR006644
86. Brown RA, Rosenberg NJ, Hays CJ, Easterling WE, Mearns LO (2000) Potential production and environmental effects of switchgrass and traditional crops under current and greenhouse-altered climate in the central United State: a simulation study. Agric Ecosyst Environ 78:31–47
87. Costello C, Griffin WM, Landis AE, Matthews HS (2009) Impact of biofuel crop production on the formation of hypoxia in the Gulf of Mexico. Environ Sci Technol 43:7985–7991

88. David MB, Gentry LE, Kovacic DA, Smith KM (1997) Nitrogen balance in and export from an agricultural watershed. J Environ Qual 26:1038–1048
89. McIsaac GF, Hu X (2004) Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochemistry 70:251–271
90. Randall GW, Huggins DR, Russelle MP, Fuchs DJ, Nelson WW, Anderson JL (1997) Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa and row crop systems. J Environ Qual 26:1240–1247
91. McIsaac GF, David MB, Mitchell CA (2010) *Miscanthus* and switchgrass production in central Illinois: impacts on hydrology and inorganic nitrogen leaching. J Environ Qual 39:1790–1799
92. Heaton EA, Dohleman FG, Long SP (2009) Seasonal nitrogen dynamics of *Miscanthus x giganteus* and *Panicum virgatum*. GCB Bioenergy 1:297–307
93. Reynolds JH, Walker CL, Kirchner MJ (2000) Nitrogen removal in switchgrass biomass under two harvest systems. Biomass Bioenergy 19:281–286
94. Lemus R, Parrish DJ, Abaye O (2008) Nitrogen-use dynamics in switchgrass grown for biomass. Bioenergy Res 1:153–162
95. Tufekcioglu A, Raich JW, Isenhart TM, Schultz RC (1999) Fine root dynamics, coarse root biomass, root distribution, and soil respiration in a multispecies riparian buffer in central Iowa, USA. Agroforest Syst 44:163–174
96. Sanderson MA, Reed RL, Ocumpaugh WR et al (1999) Switchgrass cultivars and germplasm for biomass feedstock production in Texas. Bioresour Technol 67:209–219