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Reduced nitrogen losses after conversion of row crop agriculture to alley cropping with mixed fruit and nut trees

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ABSTRACT

Agriculture across the temperate zone is dominated by a maize-soybean rotation (MSR) characterized by a "leaky" nitrogen (N) cycle. MSR N losses have considerable negative impacts on water quality via N leaching and climate change via soil emissions of nitrous oxide (N_2O), a potent greenhouse gas. Alley cropping (AC) focused on food- or fodder-producing tree crops has the potential to substantially reduce environmental N losses while maintaining agricultural productivity. To compare the N cycling of MSR and AC, this study (1) summarized literature values of N pools and fluxes in both systems, (2) directly measured N leaching and N_2O emissions in a side-by-side trial of MSR and an establishing AC over four years, and (3) used AC yield projections to estimate the trajectory of yield-scaled N losses as AC grows to productive maturity. Ample literature data on MSR permitted the construction of a robust working N budget, while a paucity of existing data on N cycling in AC revealed gaps and high uncertainty in our existing knowledge. In the field trial, AC quickly reduced both N leaching and N_2O emissions compared to MSR. Nitrate leaching at 50 cm depth in MSR ranged from 21.6 to 88.5 kg N ha^{-1} yr^{-1}, whereas leaching was reduced by 82–91% in AC. Cumulative annual net N_2O fluxes in MSR ranged from 0.4 to 2.0 kg N ha^{-1}, but AC reduced annual fluxes by 25–83%. Overall, conversion of MSR to AC reduced unintended N losses over four years by 83% from 240 to 41 kg N ha^{-1}. Even when accounting for the low yield in AC during the establishment years studied here, yield-scaled N leaching in AC and MSR were not significantly different. In contrast, yield-scaled N_2O fluxes were an average of 4.8 times higher in AC across years and were only estimated to reach a comparable range to MSR after reaching productive maturity. Our results demonstrate rapid tightening of the N cycle and a competitive trajectory of yield-scaled N losses as row crop agriculture is converted to AC.

1. Introduction

Row crop agriculture is a dominant land-use around the world, with maize and soybean alone covering over 346 million hectares worldwide (FAO, 2017). The maize-soybean rotation (MSR) typically relies on large nitrogen (N) inputs and intensive disturbance, which can increase environmental N losses. The two most concerning avenues of unintended N loss from agricultural systems are N leaching and soil nitrous oxide (N_2O) emissions (David et al., 2009; Hernandez-Ramirez et al., 2009). In North America, agricultural N leaching contributes around 80% of the 1.2 million tons of N entering the Gulf of Mexico and results in hypoxia (David et al., 2010; USEPA, 2007). Although the absolute amount of N lost via N_2O emissions is small, N_2O is a potent greenhouse gas and driver of climate change (IPCC, 2014). The leaky agricultural N cycle produces 55% of global N_2O emissions (USEPA, 2012). Ammonia (NH_3) volatilization can also contribute substantial N loss, with an average of 18% of applied N lost as NH_3 globally (Pan et al., 2016) and as much as 46% of applied N lost in temperate pasture systems (Vaio et al., 2008). Many agronomic techniques have been proposed to reduce N losses from row crop agriculture, such as adjusting N fertilization to crop needs, application of nitrification inhibitors, cover crops, and water management in irrigated crops (Abalos et al., 2016; Quemada et al., 2013). A meta-analysis of the many techniques intended to reduce N losses in maize found that they can reduce N leaching by 14–37% and N_2O emissions by 5–40% (Xia et al., 2017). However, much greater reductions are needed to meet hypoxia reduction goals (Scavia et al., 2004) and climate change mitigation goals (IPCC, 2014).

Alley cropping (AC), the integration of trees with crops, is a

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transformative departure from the incremental improvements to MSR that focus on minor agronomic improvements or field margins (Gold and Hanover, 1987; Wilson and Lovell, 2016). In particular, AC with food- or fodder-producing "tree crops" (e.g. nut or fruit trees), could maintain high agricultural yields while promoting substantial ecological benefits in a "land sharing" land-use approach (Anderson-Teixeira et al., 2012; Lovell et al., 2017; Wolz et al., 2018). By integrating trees and crops throughout a field, temperate AC can promote carbon sequestration, improved soil structure, increased biodiversity, and soil stabilization (Jose, 2009; Thevathasan and Gordon, 2004; Tsonkova et al., 2012). In addition, AC has potential to reduce N losses.

Like cover crops or buffer strips, tree roots can provide a "safety-net" by catching N that leaches beyond the crop rooting depth or growing season (Allen et al., 2004). For example, AC reduced nitrate (NO_3^-) leaching compared to monoculture crops by 46% at 0.3 m depth and 71% at 0.9 m depth (Allen et al., 2004). The greater leaching reduction with depth illustrates the cumulative effect of tree roots. Lower in the soil profile, Dougherty et al. (2009) found 46% lower NO_3^- levels in tile effluent under AC than monoculture maize, which directly translates into impacts on surface water quality. Even compared to perennial pasture, which has deeper roots and a longer growing season than annual crops, integrating trees reduced peak NO_3^- concentrations at 1.2 m depth by 56% (Bambo et al., 2009). The efficacy of leaching reductions in AC varies with alley crop species (Dai et al., 2006) and soil texture (Bergeron et al., 2011).

Agroforestry also has potential as a mitigation tool for climate change through reduced N_2O emissions (Schoeneberger et al., 2012). For example, studies of hedgerows and shelterbelts found up to 74% lower N_2O emissions compared to adjacent cropland (Amadi et al., 2016; Baah-Acheamfour et al., 2016). However, in a synthesis of N_2O emissions in agroforestry, Kim et al. (2016) reported an increase of annual N_2O emissions of 0.64 ± 0.26 kg N ha^{-1} in AC compared to adjacent agricultural fields. This value was based on only a single study (Guo et al., 2009) – the only study of N_2O emissions in AC with sufficient sampling to generate annual flux estimates – and clearly demonstrates the paucity of data on N_2O emissions in AC. Although not providing an annual total, Beaudette et al. (2010) found that AC reduced soil N_2O emissions by 72% on four dates without impacting alley crop yields.

Studies of N cycling in AC have focused on mature systems, leaving uncertain the trajectory of N losses during establishment. Other perennial crops can reduce N losses soon after conversion from MSR. For example, perennial grasses grown as bioenergy crops reduced NO_3^- leaching and N_2O emissions by over 90% in just four years (Smith et al., 2013). Young woody bioenergy crops can reduce NO_3^- leaching by more than 99% over the first 11 years (Syswerda et al., 2012) and N_2O emissions by 81% over the first nine years (Robertson et al., 2000). It is important to note, however, that perennial bioenergy crop are typically not fertilized due to the wide C:N ratios of the harvested biomass. Instead, fertilization can increase N losses unnecessarily (Balasus et al., 2012; Behnke et al., 2012). In contrast, AC with tree crops will likely require greater N replenishment due to the narrower C:N ratios of fruit/nut yields. These higher N inputs could negate the potential of AC to reduce N losses.

As a land sharing approach, a complete comparison of AC with MSR requires the use of yield-scaled N losses (Linquist et al., 2012), in which N losses are scaled by caloric food yields to determine N loss per unit yield. The yield-scaled concept has only recently been applied in perennial crops (Schellenberg et al., 2012) but is especially important when comparing AC and MSR due to the low yields of immature tree crops and the contrasting fertilization regimes typically used. Many years of high yield-scaled N losses during the establishment phase could outweigh lower values at maturity.

Understanding the N cycle of AC is critical for its evaluation as a viable agricultural practice in the temperate zone. The objective of this study was to quantify changes in the N cycle when transitioning from MSR to AC. Three approaches were used: (1) To provide context on the possible range of N pools and fluxes in temperate MSR and AC, we constructed working N budgets from literature values and agricultural statistics. (2) We conducted a side-by-side trial of AC and MSR to evaluate changes in the N cycle over the first 5 years after AC establishment. (3) Using projections of AC yield, we estimated the trajectory of yield-scaled N losses as AC grows to reproductive maturity. We hypothesized that transitioning from MSR to AC would (1) substantially reduce N losses, although (2) yield-scaled N losses would only become competitive with MSR once the tree crops reach reproductive maturity.

2. Materials and methods

2.1. Working N budgets

Working N budgets for MSR and temperate AC were constructed using a combination of agricultural statistics, climate statistics, and literature values. Literature values were primarily gathered from existing reviews of various components of the N cycle. All budget values were summarized as ranges, with values greater than 20 kg N ha^{-1} yr^{-1} rounded to the nearest 10 units. Complete details on the derivation of N budgets are provided in the Supplemental Materials.

2.2. Site description and experimental design

Our study site was located at the University of Illinois Pomology Research Farm (40°4′45.05″N, 88°12′57.45″W, ~220 m above sea level). The site previously grew soybeans (2009–2011), silage maize (2006–2008), and alfalfa (2002–2005), although was historically in a traditional MSR. Average annual temperature during the study ranged from 9.8–12.9° C, and annual precipitation ranged from 861 to 1331 mm (Illinois Climate Network, 2017). Monthly precipitation and temperature data are shown in Tables S1 and S2. Soils are a Flanagan silt loam (Fine, smectitic, mesic aquic agriudolls), typical of the deep, poorly drained mollisols of central Illinois. At the time of establishment, mean organic C content in the top 30 cm of soil ranged from 17.3–25.2 g kg^{-1}, and soil pH was 7.3. The study site has a ~2% slope and contains four-inch drain tile oriented E to W at 30 m spacing.

The two treatments studied were: MSR and an establishing AC. Plots were established in spring 2012 in a randomized complete block design with four 0.2-ha replicates and mowed grass buffers (Fig. 1). Neither treatment was irrigated during the study period. MSR was managed using typical practices of central Illinois. Soybean was planted in 80-cm rows on 17 May 2013 and 22 May 2015. Maize was planted in 75-cm rows in 23 Apr. 2014 and 6 May 2016. Glyphosate was applied in all years approximately one month after planting. Grain harvest was completed on 28 Oct. 2013, 18 Nov. 2014, 22 Oct. 2015, and 5 Oct. 2016. All MSR plots were conventionally tilled annually.

The AC design was based on Shepard (2013), containing six different food-producing tree and shrub species with grass-clover hay alleys (Fig. 1). Multiple tree and shrub species were included in the AC design because the site is part of a collection of experiments exploring the impact of tree crop diversity in AC (see Lovell et al., 2017; Wolz et al., 2018). Grass-clover alleys, rather than row-crop alleys, were included in this study because this is the approach most commonly used by farmers adopting AC in the region. All woody plants were planted between 12 May and 4 Jun. 2012, and the hay was seeded on 1 Oct. 2012. Except for raspberries, which had a 40% survival rate and were replanted on 19 May 2013, all species exhibited ~90% survival. Herbicide (29.4% S-metolachlor, 11% atrazine, 2.94% mesotrione) was applied prior to planting on 24 Apr. 2012. In 2013, a 1.4 m band of pre-emergent herbicide was applied in the tree rows (oryzalin) on 7 May and the alley crop (prodiamine) on 17 May. On 31 Jan. 2014, Dutch white clover was broadcast under the tree rows to serve as a living mulch. From 2014 on, the 0.5 m on either side of tree rows was mowed monthly. Weeds within rows were managed using a string trimmer. To

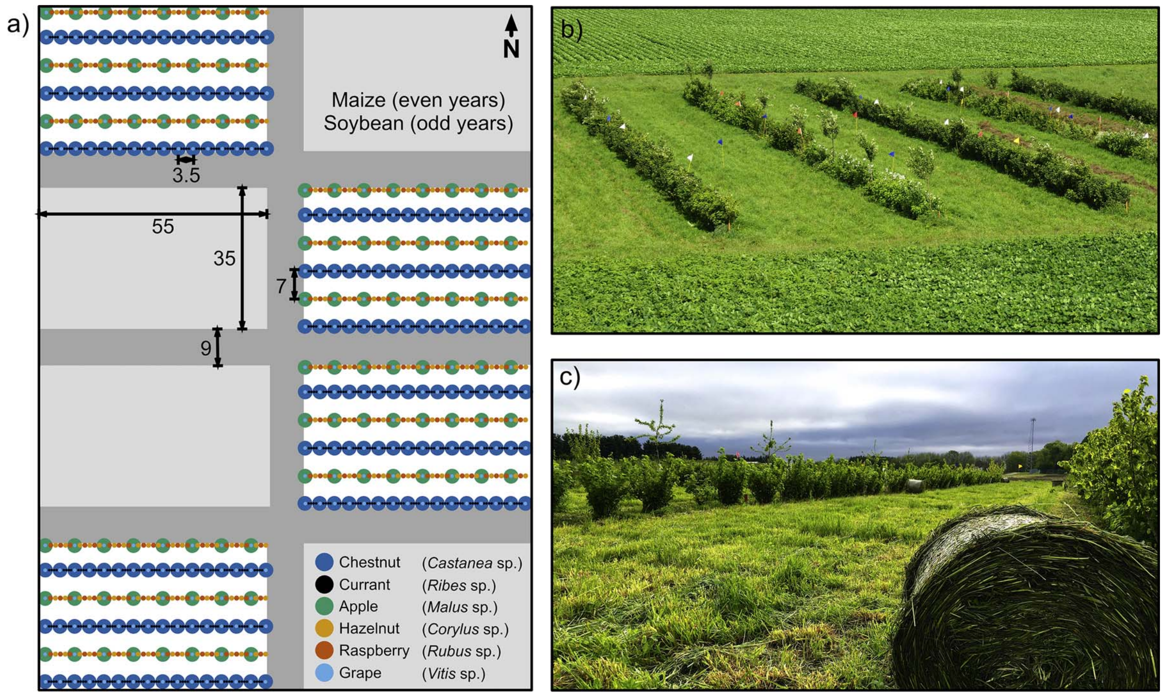

Fig. 1. (a) Map of the study site. All dimensions are shown in meters. (b) Aerial view of AC at the study site in 2015, with soybean plots in the foreground and background. (c) Early spring in AC at the study site, just after the first hay cutting and with leaves on tree species still emerging.

lower soil pH into the range for optimal chestnut growth, 300 g of granular elemental sulfur was spread around the base of each chestnut tree on 16 Apr. 2015 and 5 May 2016. The hay alleys were mowed monthly in 2013, but no hay was harvested. In 2014–2016, hay was cut, baled, and removed four times each season. Small fruit/nut harvests were removed each year. Beginning in 2014, all species were pruned each winter. Raspberries were cut to the ground after the 2014 and 2016 seasons, and grapes were cut to the ground each year.

2.3. Soil N pools and fluxes

To characterize baseline (2012) conditions and quantify the total soil N pool, 3.8-cm-diameter soil samples were collected using a tractor-mounted hydraulic probe (Giddings Machine Co., Windsor CO USA) from 10 random locations to 100-cm depth in each plot. Each sample was air-dried and weighed to calculate bulk density, and a subsample was oven-dried at 105 °C to correct for moisture content. Subsamples were then crushed, sieved (2 mm), finely ground with a modified coffee grinder, oven-dried at 65 °C, and analyzed for total C and N concentrations with an elemental analyzer (Elemental Combustion System, Costech Analytical Technologies, Inc.).

Maize was fertilized each year at the time of planting with 200 kg N ha^{-1} of liquid urea ammonium nitrate (UAN), a mixture of urea (51% of applied N) and ammonium nitrate (49% of applied N) (Millar et al., 2010; Sawyer et al., 2006). The UAN was applied to the soil surface and then mechanically incorporated to a depth of 6 cm. Soybean was not fertilized. AC was not fertilized during the first three years after establishment (a standard practice allowing trees to establish) and then uniformly fertilized with 100 kg N ha^{-1} of granular urea on 22 May 2015 (year 4) and 5 May 2016 (year 5). The fertilization rate used in AC is comparable to recommended rates for the various woody crops and was selected so that the mean annual N fertilizer input was the same in MSR and AC over 2015–2016. All fertilizer applications were timed to coincide with precipitation events to minimize NH_3 volatilization that can occur in the absence of precipitation.

Total N inputs from N_2 fixation were estimated using empirical relationships with aboveground [N] for soybean (Gelfand and Robertson, 2015) and clover (Høgh-Jensen et al., 2004). Clover biomass was visually estimated as 20% of hay biomass. Wet N deposition (NH_4–N + NO_3–N) was obtained from the National Atmospheric Deposition Program at Bondville, IL (~15 km from the study site) for 2013–2016, and dry deposition was estimated as 70% of wet (McIsaac et al., 2002; USEPA, 2007). N deposition in Jan-Apr 2017 was estimated as the 1980–2016 mean.

Annual N leaching fluxes were measured using ion exchange resin-based lysimeters, which bind NH_4^+ and NO_3^- ions as soil water passes through them (Smith et al., 2013; Susfalk and Johnson, 2002). Each lysimeter encloses a layer of 10 g dry resin (Rexyn I-300 H-OH Beads, Fisher Chemical) between two layers of nutrient-free silica sand within a 5.1 cm-diameter, 5.5 cm-long solid PVC pipe. The silica sand was exposed to the soil at the top of the lysimeters and held in place with landscape fabric at the bottom. Lysimeters were buried at 50-cm depth beneath intact soil profiles. Three lysimeters were randomly placed in each MSR plot, and one lysimeter was randomly placed within each of the three zones (two tree row types and the alley crop) in each AC plot. Lysimeters were initially installed on 7 May 2013 and replaced annually. Lysimeters were only retrieved annually to minimize damage to the perennial crops. Ammonium (NH_4^+) and NO_3^- loads in the resin were obtained via KCl extraction followed by colorimetric flow injection analysis (Lachat QuickChem 8000). Annual fluxes on an area basis were calculated by dividing the extracted loads by the lysimeter cross-sectional area.

N_2O flux was measured using vented static PVC chambers with 20-cm diameter and 3.2-L volume (Smith et al., 2013). Chambers were placed on PVC ring bases (20 cm diameter, 10 cm height) that were inserted ~5 cm into the soil and maintained free of vegetation. The bases remained in the soil throughout the experiment, though temporarily removed in MSR for tilling. In each MSR plot, one ring was randomly placed in each of three zones: within row, between rows, and splitting these two. In each AC plot ring placement was stratified as for lysimeters. Sampling occurred in late morning to minimize soil temperature variability, and measured fluxes were assumed to represent the mean daily flux. Sampling began in early spring and continued throughout the growing season approximately every two weeks.

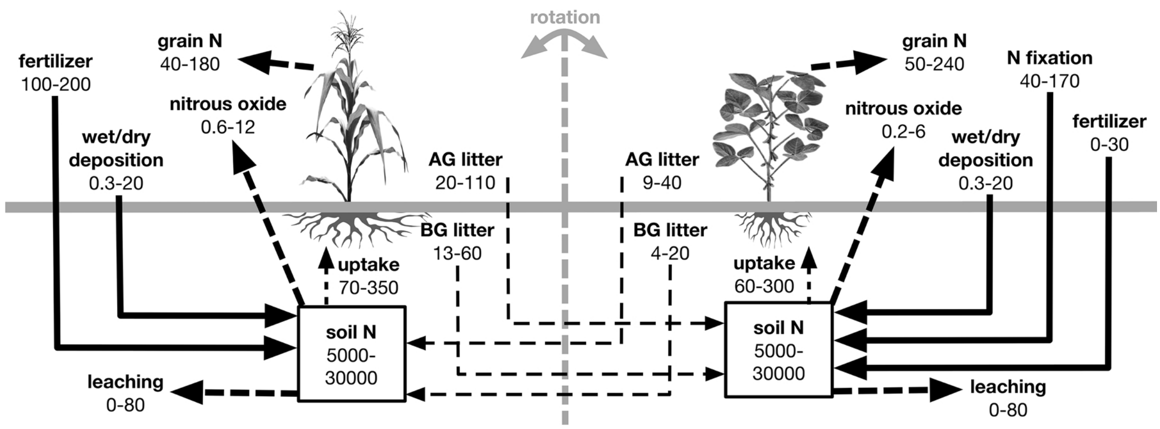

Fig. 2. Summary of literature values and working N budget for maize-soybean rotation. Values are ranges of literature-derived values. All arrows represent N fluxes in kg N ha^{-1} yr^{-1}. Solid arrow are N inputs, thick dashed arrows are N losses, and thin dashed arrows indicate internal cycling. All boxes represent N pools in kg N ha^{-1}.

Sampling frequency increased surrounding fertilization events and slowed in late summer to once every three to four weeks. For flux measurements, chambers were secured on the rings for 30-min incubations. Every 10 min, 15-mL gas samples were collected into previously evacuated 10-mL vials sealed with molded gray butyl rubber, Teflon-coated septa (Sun SRI). Gas samples were analyzed by gas chromatography (Shimadzu GC-2014 Greenhouse Gas Analyzer, Shimadzu Scientific Instruments) alongside known gas standards (Scott Specialty Gases) using an autosampler. Cumulative N_2O fluxes were linearly interpolated. Fluxes of NH_3 were not measured.

2.4. Harvested and standing biomass

To quantify N removed in yields, aboveground biomass samples were collected from three randomly placed 0.5 m^2 quadrats in each plot during harvest of MSR and AC alleys. Yield subsamples were analyzed for total C and N concentration as described above for soil samples. All harvested fruits and nuts from AC were weighed during harvest, and N content was calculated using literature values of N concentration. All woody biomass removed via pruning was weighed.

To quantify net annual N uptake by woody plants, standing aboveground woody biomass in AC was estimated each year using stem caliper measurements and species-specific allometric relationships. Allometric relationships were constructed using pruning events and several destructive harvests (Figs. S1 and S2). Belowground woody biomass was estimated from literature values, when available, or using root:shoot ratios measured during destructive harvests. To estimate N content of woody biomass, above- and belowground wood samples of each species were harvested in Mar. 2013 while all species were still dormant. Collected samples were analyzed for total C and N concentration as described above.

2.5. Yield-scaled N losses

To calculate yield-scaled N losses during the four study years, measured N losses were divided by measured yield. Yield was defined as the caloric content of the economic products of each system. This included fruits, nuts, and hay for AC and grain and beans for MSR. Yields from top individuals in AC were used rather than site means because of high yield variability due to rodent damage. For each tree crop, yield biomass was converted to caloric yield using standard conversions (USDA, 2016). Based on hay biomass production, alleys were assumed to support production of zero, one, two, and two beef steers per hectare (225 kg beef per steer) in years two through five, respectively (Pratt and Rasmussen, 2001). To estimate trajectories of yield-scaled externalities of AC beyond year five, yield projections were taken from Wolz et al. (2018). Although N losses are likely to decrease as trees mature, we made the conservative assumption that N losses beyond year five will remain equivalent to the mean of years four to five.

2.6. Statistical analyses

All statistical analyses were performed on plot-level means. AC plot means were calculated as area-weighted means of the stratified samples. Statistical analyses were performed using the R statistical computing software version 3.3.2 (R Core Team, 2017), with differences in means significant at a probability level of $p < 0.05$. Prior to comparing treatment means, all data were assessed for normality (Shapiro-Wilk) and variance homogeneity (Levene). All data exhibited a normal distribution and homogeneous variance, except for unequal variance in NO_3^- flux data between treatments. One alley subsample value for NO_3^- leaching within one AC block in 2015 was identified as an extreme outlier (higher than the mean) using Turkey's method and replaced by a mean of the alley subsamples in the remaining three blocks. Nitrate fluxes were analyzed for significant differences between treatments for each year using a Welch modified two-sided *t*-test with unequal variance. The same test was performed for differences in NH_4^+ fluxes, cumulative N_2O fluxes, and yield-scaled N losses except that a pooled variance was used. Significant differences across years for each treatment were assessed using a one-way ANOVA. When overall F-tests were significant, Tukey's HSD was used to separate means. A blocking factor was not used since preliminary analyses revealed that blocking was non-significant for all tests.

3. Results

3.1. Working N budgets

Ample available data from crop statistics and literature reviews made it possible to construct a robust working N budget for MSR (Fig. 2). Fertilizer and N fixation dominated N inputs for maize and soybean, respectively, with N deposition much smaller. The largest output in both crops was harvested N in grain, though N leaching was the largest unintended loss. In contrast, net N_2O emissions constituted negligible N loss, underscoring its disproportionate impact as a greenhouse gas. Ranges for all pools and fluxes were wide due to the variable climate, soil, and management in MSR across the temperate zone.

A paucity of data on N cycling in AC revealed gaps in our knowledge and generally low precision in the estimated ranges of N fluxes (Fig. 3). Net N_2O emissions were comparable to those of soybean, but lower than in maize. Leaching losses exhibited an extremely wide range, although data were derived from only three studies. Overall, the high variance in all estimates also represents the broad array of trees, crops,

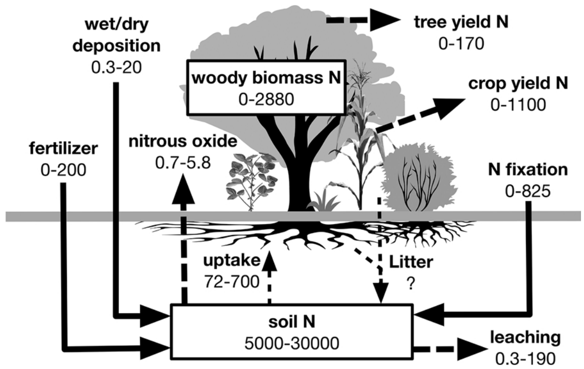

Fig. 3. Summary of literature values and working N budget for AC. Values are ranges of literature-derived values. All arrows represent N fluxes in kg N ha^{-1} yr^{-1}. Solid arrow are N inputs, thick dashed arrows are N losses, and thin dashed arrows indicate internal cycling. All boxes represent N pools in kg N ha^{-1}.

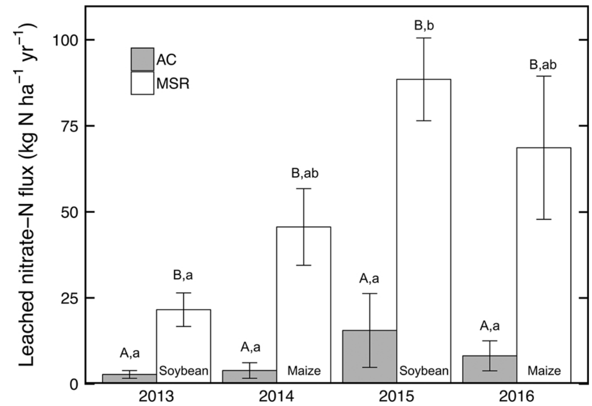

Fig. 4. Annual NO_3^- leaching (May–May) at 50 cm depth using resin lysimeters for AC (grey bars) and MSR (white bars) during 2013–2016 (mean ± SE). Treatment means within each year with the same capital letter, and means within each treatment across years with the same lowercase letter, are not significantly different. Maize was fertilized with 200 kg N ha^{-1}. AC was fertilized in 2015 and 2016 with 100 kg N ha^{-1}.

designs, and management regimes that can constitute AC.

3.2. Initial soils and harvested biomass

Initial soil C and N content was consistent between treatments (Table S3). Organic C content in the top 30 cm of soil ranged from 17.3–25.2 g kg^{-1} and declined to as low as 3.0 g kg^{-1} at 50–100 cm depth. Total soil N followed the same pattern, ranging from 1.71–2.65 g kg^{-1} in surface soils and declining to a low of 0.46 g kg^{-1} at 50–100 cm depth. Organic C and total N pools in the top meter of soil averaged 140 Mg C ha^{-1} and 14.0 Mg N ha^{-1}.

As stover was incorporated into the soil each fall using a disc, grain yields constituted the only biomass harvested from MSR and were typical of the region (Table S4). Fruit/nut yields in AC generally increased throughout the study, although peak yields were not reached for any crop. Hay dominated AC harvested biomass, with yields increasing each year and more than doubling after the initiation of fertilization in 2015. Woody biomass prunings also generally increased throughout the study.

3.3. Standing biomass

Standing woody biomass in AC generally increased throughout the study, with some irregularities due to pruning regimes (Table S5). The shrub species dominated the standing biomass as they were the most numerous and fastest-growing species, occupying the space in the bottom canopy strata in each tree row. Root:shoot ratios for woody components in AC species ranged from 0.78–2.00, with the large shrub species exhibiting the largest ratios. N content in aboveground woody biomass ranged from 0.7–1.1% across species, with the range in belowground N content slightly higher from 0.9–1.8%.

3.4. Environmental N losses

Large quantities of NO_3^- leached in MSR at 50 cm depth in all years, ranging from 21.6-88.5 kg N ha^{-1} yr^{-1} and representing a substantial portion of the annual N inputs to the system (Fig. 4). Leaching under MSR did not vary significantly across years except for leaching between the two soybean years (2013 and 2015), which were the highest and lowest leaching rates recorded in MSR. In the second and third years after establishment of AC, NO_3^- leaching was significantly lower than in MSR at just 2.7 and 3.9 kg N ha^{-1} yr^{-1}, respectively. Once fertilization commenced in AC in 2015 and 2016, NO_3^- leaching rose to 15.5 and 8.2 kg N ha^{-1} yr^{-1}, respectively, but was still significantly lower than in MSR. Nitrate leaching at 50-cm depth was significantly reduced in AC compared to the MSR by 87, 91, 82 and 88% in 2013–2016, respectively. Ammonium leaching rates remained low (1.5–5.0 kg N ha^{-1} yr^{-1}) throughout the study (Fig. S3).

Net N_2O fluxes in AC and soybean were low throughout each growing season, generally remaining below 10 g N ha^{-1} day^{-1} (Fig. 5). In contrast, net N_2O fluxes in maize were substantially higher, reaching as high as 157 g N ha^{-1} day^{-1} in 2016, and only remaining consistently low in the fall season. Prominent spikes in net N_2O flux lasting approximately 20 days were observed after all fertilization events in maize and AC. In maize, these initial spikes accounted for 28% and 50% of 2014 and 2016 N_2O emissions, respectively. Additional but smaller spikes were observed in maize fluxes throughout both seasons, typically after heavy rains. In AC, the initial post-fertilization spikes accounted for 37% and 31% of 2015 and 2016 N_2O emissions, respectively. No additional large spikes were observed in AC fluxes, even after large rain events.

Cumulative growing season net N_2O fluxes in MSR ranged from 0.4–0.9 kg N ha^{-1} in soybean and 1.4–2.0 kg N ha^{-1} in maize, representing around 0.4% and 0.8% of annual N inputs, respectively (Fig. 6). Cumulative fluxes were reduced in AC compared to MSR by 55, 83, 25 and 65% in 2013–2016, respectively ($p < 0.05$ for all years except 2015), although these values do not account for any potential emissions from Nov.-Apr. While cumulative net N_2O flux in AC increased more than three-fold with the onset of fertilization in 2015 and 2016, the difference was not statistically significant. Net N_2O fluxes varied significantly between years for MSR, with fluxes higher in maize compared to soybean.

3.5. Overall nitrogen fluxes

Overall N budgets were developed for both experimental treatments (Table 1). Fertilization in maize was comparable to the annual N fixation in soybean. Together, these dominated N inputs for MSR. In AC, fertilizer was also the major N input, followed by clover N fixation. Atmospheric deposition added little N to both treatments. The largest N outputs were grain yield for MSR and harvested hay in AC. Fruit/nut yields and woody biomass prunings removed relatively little N from AC. Nitrate leaching was the largest unintended export of N from both systems, constituting 29.6% and 5.5% of annual N inputs to MSR and AC, respectively, over the four study years. Ammonium leaching and N_2O emission were much smaller N losses, constituting 1.5% and 1.6%, respectively, of N inputs in both systems. MSR was consistently a net N exporter across years, with a net export of 306 kg N ha^{-1} over the four study years. While AC was also a net exporter of N over the four study years, the net export was much lower at only 132 kg N ha^{-1}. Overall, conversion of MSR to AC reduced unintended N losses by 83% over four years from 240 to 41 kg N ha^{-1}.

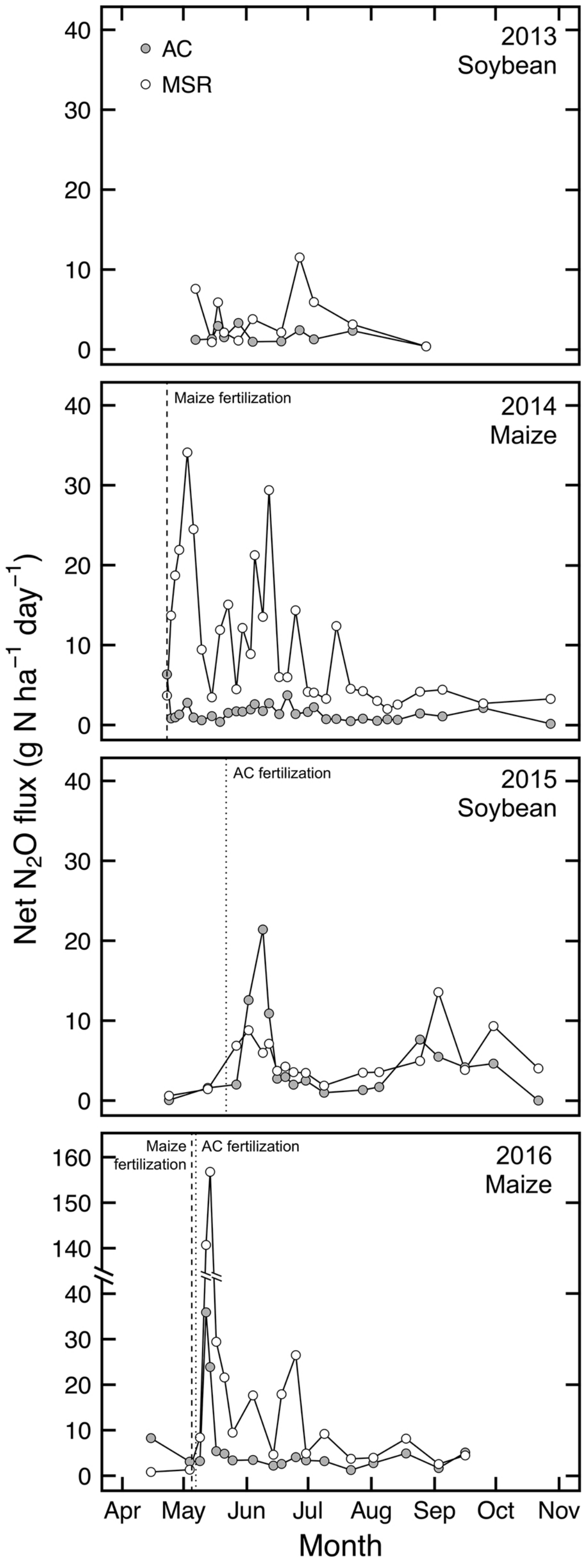

Fig. 5. Net N_2O fluxes from chamber measurements during 2013–2016 for AC (filled circles) and MSR (open circles). Fertilization dates for maize and AC are indicated by vertical dashed and dotted lines, respectively. The number of sampling days in 2013–2016 were 11, 32, 18, and 19, respectively.

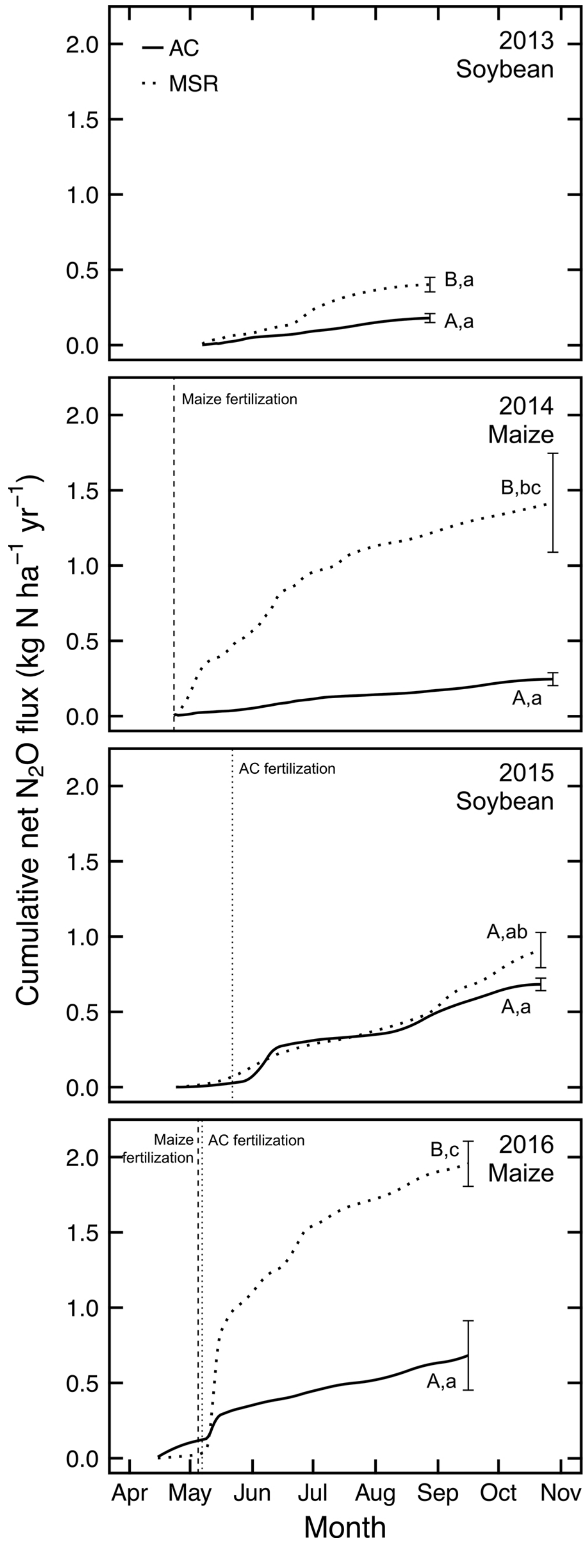

Fig. 6. Cumulative net N_2O fluxes during 2013–2016 for AC (solid line) and MSR (dotted line). Fertilization dates for maize and AC are indicated by vertical dashed and dotted lines, respectively. Final cumulative fluxes within each year with the same capital letter, and within each treatment across years with the same lowercase letter, are not significantly different. Error bars represent standard errors of final cumulative fluxes.

3.6. Yield-scaled N losses

Mean yield-scaled NO_3^- leaching during the four study years was 2.8 kg N MCal^{-1} and 2.3 kg N MCal^{-1} for AC and MSR, respectively, with no significant differences between systems (Fig. 7a). Mean yield-scaled N_2O flux during the four study years was 0.17 kg N MCal^{-1} and 0.04 kg N MCal^{-1} for AC and MSR, respectively, with AC significantly higher than MSR in all years except year five (Fig. 7b). The trajectory of yield-scaled N losses in AC was projected to decrease until around year

Table 1
Nitrogen fluxes (May–May) for the second through the fifth years of AC establishment. All values were measured in this study unless otherwise noted.

	Maize-soybean rotation				Alley cropping			
	Soy 2013	Maize 2014	Soy 2015	Maize 2016	2013	2014	2015	2016
	kg N ha^{-1} yr^{-1}							
Inputs								
Fertilizer	0.0	200	0.0	200	0.0	0.0	100	100
Atm. deposition[a]	6.4	10.3	7.8	8.9	6.4	10.3	7.8	8.9
N_2 fixation[b]	148	–[c]	173	–	44.3	44.3	81.6	81.1
Total in	154	210	181	209	50.7	54.6	189	190
Outputs								
NH_4^+ leaching	1.8	2.0	2.5	5.0	2.1	1.5	1.7	3.5
NO_3^- leaching	21.6	45.6	88.5	68.6	2.7	3.9	15.5	8.2
N_2O efflux	0.4	1.4	0.9	2.0	0.2	0.2	0.7	0.7
Woody biomass	–	–	–	–	0.0	8.0	5.2	10.2
Hay biomass	–	–	–	–	0.0	94	201	238
Grain/Fruit/Nut yield	211	194	272	142	5.4	2.5	4.3	7.4
Total out	235	243	364	218	10.4	110	228	268
Net N Input	−81	−33	−183	−9	40	−55	−39	−78
Woody biomass growth								
Aboveground	–	–	–	–	4.3	9.9	10.1	12.6
Belowground	–	–	–	–	9.7	18.9	−4.6	12.1
Total woody biomass growth	–	–	–	–	14.0	28.8	5.5	24.7

[a] National Atmospheric Deposition Program at Bondville, IL (~15 km from the study site).
[b] Estimated using empirical relationships with aboveground [N] for soybean (Gelfand and Robertson, 2015) and clover (Høgh-Jensen et al., 2004).
[c] Not applicable.

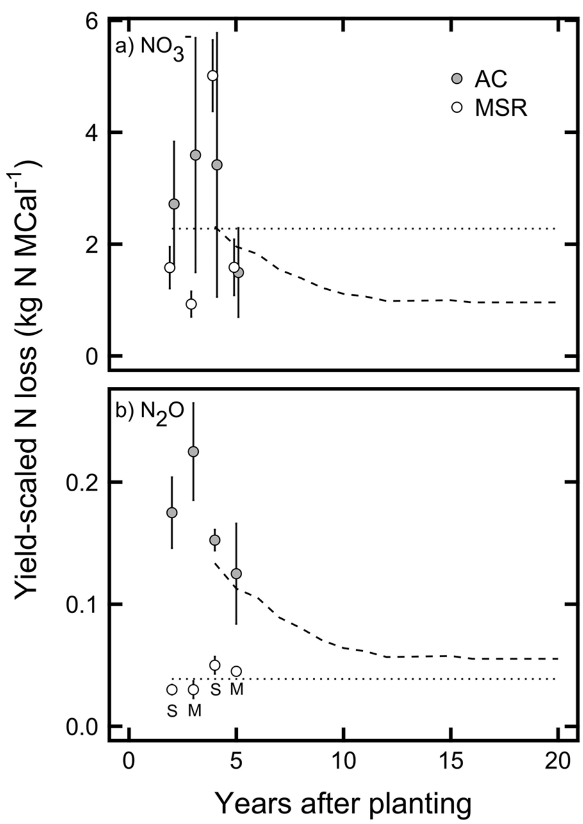

Fig. 7. Yield-scaled annual NO_3^- (a) and N_2O (b) losses for AC (filled circles) and MSR (open circles). Points in years two to five are data from this study. The horizontal dotted line in each panel shows the mean MSR value of the four study years. The dashed line shows the theoretical trajectory for AC using yield projections from Wolz et al. (2018) and the conservative assumption that N losses beyond year five remain equivalent to the mean of years four and five. Letters in (b) indicate soybean and maize years.

12 and then plateau once all component tree crops reach mature yields. Yield-scaled NO_3^- leaching and N_2O flux were projected to plateau at approximately 42% and 143% of the measured MSR mean, respectively.

4. Discussion

Our results support our first hypothesis by demonstrating that a system-level transition from MSR to AC can rapidly decrease environmental N losses via NO_3^- leaching and soil N_2O emissions. Reduction of N losses in AC occurred even when mean annual N fertilizer inputs were equivalent to MSR (as in 2015 and 2016) and despite the soil disturbance and low plant biomass of establishment years. Support of our second hypothesis regarding yield-scaled N losses when leveraging tree crops in AC was different for NO_3^- and N_2O losses. The trajectory of yield-scaled NO_3^- loss in AC was competitive with MSR soon after establishment, whereas yield-scaled N_2O loss was projected to approach that of MSR only after tree crop maturity. These results improve our knowledge of the N cycle during the transition from MSR to AC and support AC as a multifunctional, land sharing land-use approach. Widespread adoption of AC across the Midwest could rapidly reduce absolute and yield-scaled N losses, reducing hypoxia in surface waters and mitigating global climate change.

Leaching rates in both systems were within the ranges of the working N budgets except for leaching under soybean in 2015, which was unexpectedly high. This may be explained by 2015–2016 being the wettest year of the study (1331 mm). Most leaching loss in the Midwest US occurs in winter and spring when vegetation is absent, and much of the loss can occur during a short period of high precipitation (Royer et al., 2006). Precipitation during Nov. 2015–Jan. 2016 (310 mm) was triple that of the same period for soybean in 2013–2014 (104 mm), likely driving the higher leaching in 2015 (Table S1). Furthermore, some of the measured NO_3^-, which moves slowly through the fine-textured soils of the region, may have been applied during the previous maize season. At a nearby site, Smith et al. (2013) similarly documented higher leaching losses under soybean than in some maize years.

Even if the soybean leaching rate in 2015 were capped at the working budget maximum for soybean (80 kg N ha^{-1} yr^{-1}; Fig. 2), this would only reduce leaching by 11% and not alter associated interpretations. The low NH_4^+ leaching fluxes observed in both systems were as expected (Allen et al., 2004).

One important limitation of assessing N leaching losses using resin lysimeters at 50 cm is that deep roots below 50 cm can scavenge N before it is lost to drain tile. When measuring N leaching rates at a nearby site over 3 years using both resin lysimeters at 50 cm and individual plot drain tile outflow (2 m), Smith et al. (2013) found rates measured via tile outlets to be an average of 41% lower than rates measured via resin lysimeters at 50 cm. However, individually tiled plots are costly, and burying lysimeters deeper than 50 cm can cause substantial disturbance to the measured soil profile or the root systems of perennial crops. Furthermore, since over two-thirds of roots in both annual and perennial crops are above 50 cm (Black et al., 2017), any N leached below 50 cm can be considered lost from an agricultural perspective.

Cumulative N_2O emissions measured here were on the low end of the ranges developed in the working N budgets for MSR and AC. One possible reason for this is that N_2O sampling did not occur during winter months. In some crops, winter N_2O emissions can account for around half of annual emissions (Kaiser and Ruser, 2000). Winter N_2O emissions, however, would likely disproportionately increase MSR emissions. Consequently, we expect that the overall trends found here could underestimate reductions due to conversion to AC.

Ammonia volatilization was also not measured in this study. Since the UAN fertilizer in maize was incorporated into the soil after application, NH_3 losses were likely low in MSR (Pan et al., 2016). Incorporation of fertilizer, however, is not an option in perennial systems, so the granular urea broadcast in AC may have resulted in substantial NH_3 losses. Options for reducing NH_3 losses in AC include the application of urease inhibitors and controlled release fertilizers, which were estimated in a global meta-analysis by Pan et al. (2016) to decrease NH_3 volatilization by 54 and 68%, respectively.

In fertilization years, N_2O emissions were driven by prominent post-fertilization spikes in both systems. These spikes support previous calls for split fertilizer applications as an approach to reduce N_2O fluxes (Dinnes et al., 2002). The extremely high post-fertilization spikes in 2016 for both maize and AC are especially noticeable. These high peaks, however, are not easily explained by precipitation and soil temperature, which were both higher in 2014–15 than in 2016. This supports previous observations that the relationship between N_2O fluxes and soil moisture/temperature is inconsistent (Amadi et al., 2016; Baah-Acheamfour et al., 2016). Though already low in AC by the fifth year of this study, N_2O emissions are expected to continue to decrease as the trees mature, as was found for afforested pasturelands across climate zones (Allen et al., 2009).

Soil disturbance associated with transitioning between agricultural systems (e.g. planting trees, tillage, drilling seed, low root biomass etc.) has the potential to stimulate an increase in N losses by aerating soil and reducing soil structure. This effect was demonstrated by Smith et al. (2013), where leaching was higher in the establishment year compared to the subsequent three years after conversion from MSR to three different perennial biofuel crops. While no data was collected here during the initial year of AC (2012–2013), we do not expect that leaching or N_2O emissions were substantially higher than during 2013–2014 because the soil disturbance during seedling establishment is minimal compared to the high disturbance required to establish perennial biofuel crops. Furthermore, 2012–2013 hosted an intense growing season drought and the lowest Dec.–Jan. precipitation of the study years (96 mm). Furthermore, any disturbance-driven increase in N losses during the first year will likely become negligible over the long lifespan of AC.

This study utilized a system-level comparison between MSR and AC under standard management practices. Therefore, direct comparisons of N loss mechanisms were not practical due to several differences between systems, including (1) plant functional type, (2) soil disturbance regime, (3) annual N fertilization rate, (4) source of fertilizer N (liquid UAN vs. granular urea), and (5) fertilizer N placement (surface vs. incorporated). Identifying the specific treatment differences and mechanisms driving reductions in N losses during the transition to AC will require further measurements of the evolution of soil mineral N concentrations, soil moisture, and microbiological activity (Abalos et al., 2016; García-Marco et al., 2014). Additional treatments using non-standard management to maintain consistent N inputs between systems may also be useful in future studies.

Possible mechanisms for N loss reductions in AC include (1) more efficient uptake of residual N or excess soil water that would otherwise be available for leaching and gaseous losses (Allen et al., 2004; Beaudette et al., 2010), and (2) altered microbiological controls on nitrification and denitrification resulting from decreased soil disturbance and changing soil organic matter inputs (Lee and Jose, 2003). The improved N uptake efficiency in AC may also result in beneficial tradeoffs in other gaseous losses, such as NH_3 volatilization or nitric oxide (NO), which have not been measured in our study but are of major environmental and economic concern (Ussiri and Lal, 2013).

Although temperate AC is typically applied using a single tree species (Wolz and DeLucia, 2018), emphasizing woody polycultures, such as in the system studied here, could expand its potential (Lovell et al., 2017; Wolz et al., 2018). Evidence from forests has shown that N retention can increase with diversity (Lang et al., 2014; Schwarz et al., 2014). Leveraging multiple tree species within AC could similarly increase its potential to reduce N losses. Further research is needed to determine the optimal fertilization rates and management practices in mixed species systems that lead to the highest system productivity and lowest yield-scaled N losses (Malézieux et al., 2009).

Beyond ecological sustainability, which includes the mitigation of area-scaled and yield-scaled N losses measured here, widespread adoption of AC will also require economic viability and cultural acceptance (FAO, 2016; Jordan and Warner, 2010; Robertson and Swinton, 2005). Even simple AC systems with timber trees can be economically competitive with row crops in many contexts, especially when including potential incentives for carbon sequestration (Frey et al., 2010; Yemshanov et al., 2007). Emphasizing well-developed, highly productive tree crops with robust markets can further promote AC profitability and scalability (Wolz et al., 2018). To further catalyze economic and cultural acceptability, AC could initially be established on marginal farmland where row crops are unprofitable and contribute disproportionally to N losses (Brandes et al., 2016, 2018).

5. Conclusions

Row crop agriculture continues to contribute substantially to water quality issues and global climate change through large environmental N losses. Our results demonstrate that AC quickly reduced N losses via NO_3^- leaching and soil N_2O emissions during establishment years, even when mean annual fertilizer N inputs remain the same. These results build on prior work that has demonstrated reduced N losses in mature agroforestry systems by evaluating the underexplored establishment phase. Furthermore, our approach provides a more thorough comparison between AC and MSR via yield-scaled N losses. Future work evaluating the long-term yields and biogeochemical consequences of AC are critical to widespread adoption of this transformative agricultural alternative.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.agee.2018.02.024.

References

Abalos, D., Jeffery, S., Drury, C.F., Wagner-Riddle, C., 2016. Improving fertilizer management in the U.S. and Canada for N_2O mitigation: understanding potential positive and negative side-effects on corn yields. Agric. Ecosyst. Environ. 221, 214–221. http://dx.doi.org/10.1016/j.agee.2016.01.044.

Allen, S.C., Jose, S., Nair, P., Brecke, B.J., Nkedi-Kizza, P., Ramsey, C.L., 2004. Safety-net role of tree roots: evidence from a pecan (*Carya illinoensis* K. Koch)–cotton (*Gossypium hirsutum* L.) alley cropping system in the southern United States. For. Ecol. Manag. 192, 395–407. http://dx.doi.org/10.1016/j.foreco.2004.02.009.

Allen, D.E., Mendham, D.S., Singh, B., Cowie, A., Wang, W., Dalal, R.C., Raison, R.J., 2009. Nitrous oxide and methane emissions from soil are reduced following afforestation of pasture lands in three contrasting climatic zones. Aust. J. Soil Res. 47, 443–458. http://dx.doi.org/10.1071/SR08151.

Amadi, C.C., Van Rees, K.C.J., Farrell, R.E., 2016. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in shelterbelts compared with adjacent cropped fields. Agric. Ecosyst. Environ. 223, 123–134. http://dx.doi.org/10.1016/j.agee.2016.02.026.

Anderson-Teixeira, K.J., Duval, B.D., Long, S.P., DeLucia, E.H., 2012. Biofuels on the landscape: is land sharing preferable to 'land sparing'? Ecol. Appl. 22, 2035–2048.

Baah-Acheamfour, M., Carlyle, C.N., Lim, S.-S., Bork, E.W., Chang, S.X., 2016. Forest and grassland cover types reduce net greenhouse gas emissions from agricultural soils. Sci. Total Environ. 571, 1115–1127. http://dx.doi.org/10.1016/j.scitotenv.2016.07.106.

Balasus, A., Bischoff, W.A., Schwarz, A., Scholz, V., Kern, J., 2012. Nitrogen fluxes during the initial stage of willows and poplars in short-rotation coppices. J. Plant Nutr. Soil Sci. 175, 729–738. http://dx.doi.org/10.1002/jpln.201100346.

Bambo, S.K., Nowak, J., Blount, A.R., Long, A.J., Osiecka, A., 2009. Soil nitrate leaching in silvopastures compared with open pasture and pine plantation. J. Environ. Qual. 38, 1870–1877. http://dx.doi.org/10.2134/jeq2007.0634.

Beaudette, C., Bradley, R.L., Whalen, J.K., McVetty, P.B.E., Vessey, K., Smith, D.L., 2010. Tree-based intercropping does not compromise canola (*Brassica napus* L.) seed oil yield and reduces soil nitrous oxide emissions. Agric. Ecosyst. Environ. 139, 33–39. http://dx.doi.org/10.1016/j.agee.2010.06.014.

Behnke, G.D., David, M.B., Voigt, T.B., 2012. Greenhouse gas emissions, nitrate leaching, and biomass yields from production of *Miscanthus* × *giganteus* in Illinois, USA. Bioenergy Res. 5, 801–813. http://dx.doi.org/10.1007/s12155-012-9191-5.

Bergeron, M., Lacombe, S., Bradley, R.L., Whalen, J., Cogliastro, A., Jutras, M.-F., Arp, P., 2011. Reduced soil nutrient leaching following the establishment of tree-based intercropping systems in eastern Canada. Agrofor. Syst. 83, 321–330. http://dx.doi.org/10.1007/s10457-011-9402-7.

Black, C.K., Masters, M.D., LeBauer, D.S., Anderson-Teixeira, K.J., DeLucia, E.H., 2017. Root volume distribution of maturing perennial grasses revealed by correcting for minirhizotron surface effects. Plant Soil 419, 391–404. http://dx.doi.org/10.1007/s11104-017-3333-7.

Brandes, E., McNunn, G.S., Schulte, L.A., Bonner, I.J., Muth, D.J., Babcock, B.A., Sharma, B., Heaton, E.A., 2016. Subfield profitability analysis reveals an economic case for cropland diversification. Environ. Res. Lett. 11, 014009. http://dx.doi.org/10.1088/1748-9326/11/1/014009.

Brandes, E., McNunn, G.S., Schulte, L.A., Muth, D.J., VanLoocke, A., Heaton, E.A., 2018. Targeted subfield switchgrass integration could improve the farm economy, water quality, and bioenergy feedstock production. Glob. Change Biol. Bioenergy 10, 199–212. http://dx.doi.org/10.1111/gcbb.12481.

Dai, X.Q., Sui, P., Xie, G.H., Steinberger, Y., 2006. Water use and nitrate nitrogen changes in intensive farmlands following introduction of poplar (*Populus* × *euramericana*) in a semi-arid region. Arid Land Res. Manag. 20, 281–294. http://dx.doi.org/10.1080/15324980600904734.

David, M.B., McIsaac, G.F., Darmody, R.G., Omonode, R.A., 2009. Long-term changes in mollisol organic carbon and nitrogen. J. Environ. Qual. 38, 200–211. http://dx.doi.org/10.2134/jeq2008.0132.

David, M.B., Drinkwater, L.E., McIsaac, G.F., 2010. Sources of nitrate yields in the Mississippi River basin. J. Environ. Qual. 39, 1657–1667. http://dx.doi.org/10.2134/jeq2010.0115.

Dinnes, D.L., Karlen, D.L., Jaynes, D.B., 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. Agron. J. 94, 153–171. http://dx.doi.org/10.2134/agronj2002.1530.

Dougherty, M.C., Thevathasan, N.V., Gordon, A.M., Lee, H., Kort, J., 2009. Nitrate and *Escherichia coli* NAR analysis in tile drain effluent from a mixed tree intercrop and monocrop system. Agric. Ecosyst. Environ. 131, 77–84. http://dx.doi.org/10.1016/j.agee.2008.09.011.

FAO, 2016. Food and Agriculture Key to Achieving the 2030 Agenda for Sustainable Development.

FAO, 2017. Food and Agriculture Organization of the United Nations, Statistics Division. FAOSTAT Last Updated: Feb 2017.

Frey, G.E., Mercer, D.E., Cubbage, F.W., Abt, R.C., 2010. Economic potential of agroforestry and forestry in the lower Mississippi alluvial valley with incentive programs and carbon payments. South. J. Appl. For. 34, 176–185.

García-Marco, S., Ravella, S.R., Chadwick, D., Vallejo, A., Gregory, A.S., Cárdenas, L.M., 2014. Ranking factors affecting emissions of GHG from incubated agricultural soils. Eur. J. Soil Sci. 65, 573–583. http://dx.doi.org/10.1111/ejss.12143.

Gelfand, I., Robertson, G.P., 2015. A reassessment of the contribution of soybean biological nitrogen fixation to reactive N in the environment. Biogeochemistry 123, 175–184. http://dx.doi.org/10.1007/s10533-014-0061-4.

Gold, M.A., Hanover, J.W., 1987. Agroforestry systems for the temperate zone. Agrofor. Syst. 5, 109–121. http://dx.doi.org/10.1007/BF00047516.

Guo, Z.L., Cai, C.F., Li, Z.X., Wang, T.W., Zheng, M.J., 2009. Crop residue effect on crop performance, soil N_2O and CO_2 emissions in alley cropping systems in subtropical China. Agrofor. Syst. 76, 67–80. http://dx.doi.org/10.1007/s10457-008-9170-1.

Høgh-Jensen, H., Loges, R., Jørgensen, F.V., Vinther, F.P., Jensen, E.S., 2004. An empirical model for quantification of symbiotic nitrogen fixation in grass-clover mixtures. Agric. Syst. 82, 181–194. http://dx.doi.org/10.1016/j.agsy.2003.12.003.

Hernandez-Ramirez, G., Brouder, S.M., Smith, D.R., Van Scoyoc, G.E., 2009. Greenhouse gas fluxes in an eastern Corn Belt soil: weather, nitrogen source, and rotation. J. Environ. Qual. 38, 841–854. http://dx.doi.org/10.2134/jeq2007.0565.

IPCC, 2014. Climate change 2014: mitigation of climate change. Contribution of Working Group III to the Fifth Assessment Report to the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.

Illinois Climate Network, 2017. Water and Atmospheric Resources Monitoring Program. Illinois State Water Survey, Champaign, IL. http://dx.doi.org/10.13012/J8MW2F2Q.

Jordan, N., Warner, K.D., 2010. Enhancing the multifunctionality of US agriculture. Bioscience 60, 60–66. http://dx.doi.org/10.1525/bio.2009.60.1.10.

Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor. Syst. 76, 1–10. http://dx.doi.org/10.1007/s10457-009-9229-7.

Kaiser, E.-A., Ruser, R., 2000. Nitrous oxide emissions from arable soils in Germany — an evaluation of six long-term field experiments. J. Plant Nutr. Soil Sci. 163, 249–259. http://dx.doi.org/10.1002/1522-2624(200006)163:3<249:aid-jpln249>3.0.co;2-z.

Kim, D.-G., Kirschbaum, M.U.F., Beedy, T.L., 2016. Carbon sequestration and net emissions of CH_4 and N_2O under agroforestry: synthesizing available data and suggestions for future studies. Agric. Ecosyst. Environ. 226, 65–78. http://dx.doi.org/10.1016/j.agee.2016.04.011.

Lang, A.C., Oheimb, G., Scherer-Lorenzen, M., Yang, B., Trogisch, S., Bruelheide, H., Ma, K., Härdtle, W., 2014. Mixed afforestation of young subtropical trees promotes nitrogen acquisition and retention. J. Appl. Ecol. 51, 224–233. http://dx.doi.org/10.1111/1365-2664.12157.

Lee, K.H., Jose, S., 2003. Soil respiration and microbial biomass in a pecan — cotton alley cropping system in Southern USA. Agrofor. Syst. 58, 45–54. http://dx.doi.org/10.1023/A:1025404019211.

Linquist, B., Groenigen, K.J., Adviento Borbe, M.A., Pittelkow, C., Kessel, C., 2012. An agronomic assessment of greenhouse gas emissions from major cereal crops. Glob. Change Biol. 18, 194–209. http://dx.doi.org/10.1111/j.1365-2486.2011.02502.x.

Lovell, S.T., Dupraz, C., Gold, M., Jose, S., Revord, R., Stanek, E., Wolz, K.J., 2017. Temperate agroforestry research: considering multifunctional woody polycultures and the design of long-term field trials. Agrofor. Syst. 263, 1–19. http://dx.doi.org/10.1007/s10457-017-0087-4.

Malézieux, E., Crozat, Y., Dupraz, C., Laurans, M., 2009. Mixing plant species in cropping systems: concepts, tools and models: a review. Agron. Sustain. Dev. 29, 43–62. http://dx.doi.org/10.1051/agro:2007057.

McIsaac, G.F., David, M.B., Gertner, G.Z., Goolsby, D.A., 2002. Relating net nitrogen input in the Mississippi River basin to nitrate flux in the lower Mississippi River: a comparison of approaches. J. Environ. Qual. 31, 1610–1622.

Millar, N., Robertson, G.P., Grace, P.R., Gehl, R.J., Hoben, J.P., 2010. Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (maize) production: an emissions reduction protocol for US Midwest agriculture. Mitig. Adapt. Strat. Glob. Change 15, 185–204. http://dx.doi.org/10.1007/s11027-010-9212-7.

Pan, B., Lam, S.K., Mosier, A., Luo, Y., Chen, D., 2016. Ammonia volatilization from synthetic fertilizers and its mitigation strategies: a global synthesis. Agric. Ecosyst. Environ. 232, 283–289. http://dx.doi.org/10.1016/j.agee.2016.08.019.

Pratt, M., Rasmussen, G.A., 2001. Determining Your Stocking Rate, 993rd ed. Utah State University, Logan, UT.

Quemada, M., Baranski, M., Lange, M.N.J.N.-D., Vallejo, A., Cooper, J.M., 2013. Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. Agric. Ecosyst. Environ. 174, 1–10. http://dx.doi.org/10.1016/j.agee.2013.04.018.

R Core Team, 2017. R: A Language and Environment for Statistical Computing. R

Foundation for Statistical Computing, Vienna, Austria.
Robertson, G.P., Swinton, S.M., 2005. Reconciling agricultural productivity and environmental integrity: a grand challenge for agriculture. Front. Ecol. Environ. 3, 38–46. http://dx.doi.org/10.1890/1540-9295(2005)003[0038:RAPAEI]2.0.CO;2.
Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922–1925. http://dx.doi.org/10.1126/science.289.5486.1922.
Royer, T.V., David, M.B., Gentry, L.E., 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: implications for reducing nutrient loading to the Mississippi River. Environ. Sci. Technol. 40, 4126–4131. http://dx.doi.org/10.1021/es052573n.
Sawyer, J., Nafziger, E.D., Randall, G., Bundy, L., Rehm, G., Joern, B., 2006. Concepts and Rationale for Regional Nitrogen Rate Guidelines for Corn, 105th ed. Iowa State University, Ames, IA.
Scavia, D., Justic, D., Bierman, V.J., 2004. Reducing hypoxia in the Gulf of Mexico: advice from three models. Estuaries 27, 419–425.
Schellenberg, D.L., Alsina, M.M., Muhammad, S., Stockert, C.M., Wolff, M.W., Sanden, B.L., Brown, P.H., Smart, D.R., 2012. Yield-scaled global warming potential from N_2O emissions and CH4 oxidation for almond (*Prunus dulcis*) irrigated with nitrogen fertilizers on arid land. Agric. Ecosyst. Environ. 155, 7–15. http://dx.doi.org/10.1016/j.agee.2012.03.008.
Schoeneberger, M., Bentrup, G., de Gooijer, H., Soolanayakanahally, R., Sauer, T., Brandle, J., Zhou, X., Current, D., 2012. Branching out: agroforestry as a climate change mitigation and adaptation tool for agriculture. J. Soil Water Conserv. 67, 128A–136A. http://dx.doi.org/10.2489/jswc.67.5.128A.
Schwarz, M.T., Bischoff, S., Blaser, S., Boch, S., Schmitt, B., Thieme, L., Fischer, M., Michalzik, B., Schulze, E., Siemens, J., Wilcke, W., 2014. More efficient aboveground nitrogen use in more diverse Central European forest canopies. For. Ecol. Manag. 313, 274–282. http://dx.doi.org/10.1016/j.foreco.2013.11.021.
Shepard, M., 2013. Restoration Agriculture. Acres U.S.A., Austin, TX.
Smith, C.M., David, M.B., Mitchell, C.A., Masters, M.D., Anderson-Teixeira, K.J., Bernacchi, C.J., DeLucia, E.H., 2013. Reduced nitrogen losses after conversion of row crop agriculture to perennial biofuel crops. J. Environ. Qual. 42, 219–228. http://dx.doi.org/10.2136/sssaj2003.0889.
Susfalk, R.B., Johnson, D.W., 2002. Ion exchange resin based soil solution lysimeters and snowmelt solution collectors. Commun. Soil Sci. Plant Anal. 33, 1261–1275. http://dx.doi.org/10.1081/CSS-120003886.
Syswerda, S.P., Basso, B., Hamilton, S.K., Tausig, J.B., Robertson, G.P., 2012. Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA. Agric. Ecosyst. Environ. 149, 10–19. http://dx.doi.org/10.1016/j.agee.2011.12.007.
Thevathasan, N.V., Gordon, A.M., 2004. Ecology of tree intercropping systems in the north temperate region: experiences from southern Ontario, Canada. Agrofor. Syst. 61, 257–268. http://dx.doi.org/10.1023/B:AGFO.0000029003.00933.6d.
Tsonkova, P., Böhm, C., Quinkenstein, A., Freese, D., 2012. Ecological benefits provided by alley cropping systems for production of woody biomass in the temperate region: a review. Agrofor. Syst. 85, 133–152. http://dx.doi.org/10.1007/s10457-012-9494-8.
USDA, 2016. USDA National Nutrient Database for Standard Reference, Release 28 (Slightly Revised). US Department of Agriculture, Agricultural Research Service, Nutrient Data Laboratory, Washington, DC Version Current: May 2016. https://ndb.nal.usda.gov/ndb/.
USEPA, 2007. Hypoxia in the Northern Gulf of Mexico, an Update by the EPA Science Advisory Board. EPA-SAB-08-003. U.S. Environmental Protection Agency, Washington, DC.
USEPA, 2012. Global Anthropogenic Non-CO_2 Greenhouse Gas Emissions: 1990–2030. U.S. Environmental Protection Agency, Washington, DC.
Ussiri, D., Lal, R., 2013. Soil Emission of Nitrous Oxide and Its Mitigation. Springer, Netherlands.
Vaio, N., Cabrera, M.L., Kissel, D.E., Rema, J.A., Newsome, J.F., Calvert, V.H., 2008. Ammonia volatilization from urea-based fertilizers applied to tall fescue pastures in Georgia, USA. Soil Sci. Soc. Am. J. 72, 1665–1671. http://dx.doi.org/10.2136/sssaj2007.0300.
Wilson, M.H., Lovell, S.T., 2016. Agroforestry—the next step in sustainable and resilient agriculture. Sustainability 8, 574–589. http://dx.doi.org/10.3390/su8060574.
Wolz, K.J., DeLucia, E.H., 2018. Alley cropping: global patterns of species composition and function. Agric. Ecosyst. Environ. 252, 61–68. http://dx.doi.org/10.1016/j.agee.2017.10.005.
Wolz, K.J., Lovell, S.T., Branham, B.E., Eddy, W.C., Keeley, K., Revord, R.S., Wander, M.M., Yang, W.H., DeLucia, E.H., 2018. Frontiers in alley cropping: transformative solutions for temperate agriculture. Glob. Change Biol. 24, 883–894. http://dx.doi.org/10.1111/gcb.13986.
Xia, L., Lam, S.K., Chen, D., Wang, J., Tang, Q., Yan, X., 2017. Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis. Glob. Change Biol. 23, 1917–1925. http://dx.doi.org/10.1111/gcb.13455.
Yemshanov, D., McKenney, D., Fraleigh, S., D'Eon, S., 2007. An integrated spatial assessment of the investment potential of three species in southern Ontario, Canada inclusive of carbon benefits. For. Policy Econ. 10, 48–59. http://dx.doi.org/10.1016/j.forpol.2007.03.001.