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Initial soil conditions outweigh management in a cool-season dairy farm's carbon sequestration potential

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Initial soil conditions outweigh management in a cool-season dairy farm's carbon sequestration potential

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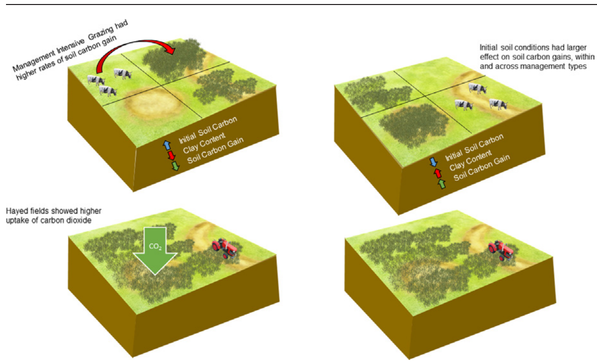
HIGHLIGHTS

- Initial soil conditions controlled soil carbon sequestration rates.
- Management intensive grazing sequestered more carbon than other grazing strategies
- Hayed fields outperformed grazed fields in greenhouse gas uptake.

GRAPHICAL ABSTRACT



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ABSTRACT

Pastures and rangelands are a dominant portion of global agricultural land and have the potential to sequester carbon (C) in soils, mitigating climate change. Management intensive grazing (MIG), or high density grazing with rotations through paddocks with long rest periods, has been highlighted as a method of enhancing soil C in pastures by increasing forage production. However, few studies have examined the soil C storage potential of pastures under MIG in the northeastern United States, where the dairy industry comprises a large portion of agricultural use and the regional agricultural economy. Here we present a 12-year study conducted in this region using a combination of field data and the denitrification and decomposition (DNDCv9.5) model to analyze changes in soil C and nitrogen (N) over time, and the climate impacts as they relate to soil carbon dioxide (CO_2) and nitrous oxide (N_2O) fluxes. Field measurements showed: (1) increases in soil C in grazed fields under MIG ($P = 0.03$) with no significant increase in hayed fields ($P = 0.55$); and (2) that the change in soil C was negatively correlated to initial soil C content ($P = 0.006$). Modeled simulations also showed fields that started with relatively less soil C had significant gains in C over the course of the study, with no significant change in fields with higher initial levels of soil C. Sensitivity analyses showed the physiochemical status of soils (i.e., soil C and clay content) had greater influence over C storage than the intensity of grazing.

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More extensive grazing methods showed very little change in soil C storage or CO_2 and N_2O fluxes with modeled continuous grazing trending towards declines in soil C. Our study highlights the importance of considering both initial system conditions as well as management when analyzing the potential for long-term soil C storage.

1. Introduction

Management of pasture systems, which comprise ~70% of global agricultural land cover, has been studied to both optimize farm productivity while maximizing ecosystem services such as soil carbon (C) sequestration (Conant, 2012; Teague et al., 2008). Much focus has been placed on the impacts of grazing intensity as it applies to stocking rates under continuous grazing (Abdalla et al., 2018; Han et al., 2008; Xu et al., 2014), showing that higher stocking rates are detrimental to rangeland ecosystems by reducing vegetation growth, soil nutrients (Teague et al., 2011), and C storage, and leading to undesirable forage composition due to preferential feeding habits of livestock (Watkinson and Ormerod, 2001).

Management intensive rotational grazing (MIG, also referred to as multi-paddock grazing and intensive rotational grazing) concentrates large numbers of animals for short periods of time into small paddocks rotated across fields. There are many different forms of rotational grazing depending on the number of paddocks and the amount of time a group of animal spends in each paddock (Undersander et al., 2002). Rotated paddocks may be given permanent fences or may use movable fencing to allow dynamic adjustment of paddock area based on available forage. Some rotated paddocks may allow back grazing, i.e. where a new paddock area is opened up for grazing but the previous paddock continues to be accessible. Rotational grazing can be thought of as a spectrum, with extensive grazing and no rotation at one end, and MIG at the other, with varying moderate intensity grazing approaches in between.

MIG is perceived as a management strategy with the potential for increasing soil C sequestration (Conant et al., 2017) via positive impacts on primary production and forage quality that bring ecological and economic advantages to farmers (Teague et al., 2015; Wang et al., 2018). By allowing longer periods of regeneration and reducing grazing preferences, MIG can support higher stocking densities and increase forage species diversity as compared to continuous grazing (Guretzky et al., 2005; Teague et al., 2008). However, there is debate to the ubiquitous benefits of MIG (Briske et al., 2008) with some results showing greater soil C sequestration occurring in drier environments (Wang et al., 2018). A recent study comparing soil C accumulation between fields under long-term MIG or hay production (Contosta et al., 2021) showed that the agroecological benefits of MIG can be mixed, with soil C concentrations and stocks comparable between fields under long-term MIG or hay production even as soil nitrogen (N) was higher and the C:N ratio was lower in MIG fields.

Realizing the possibility of MIG to increase soil C may also have the unintended consequence of enhancing the N cycle causing increased N loss. This loss may be in the form of increased emissions of nitrous oxide (N_2O) (Freney, 1997; Luo et al., 2010) and ammonia, or leaching (Bowles et al., 2018). Contosta et al. (2021) show higher soil N_2O emissions in fields under MIG, which may be driven by higher soil N stocks, lower soil C:N, higher rates of soil N cycling and net N-degrading enzyme activity, or shifts in forage species composition. Changes in N cycling in response to increased soil C storage has been variable across studies and regions (Salinas-Garcia et al., 1997; Wright et al., 2005; Wright et al., 2004; Zhang et al., 2015; Zhou et al., 2020) suggesting the need for a more detailed study of factors such as the forms, quantities, and timing of N inputs, transformations, and losses, and how they change over space and time in relation to soil C sequestration.

Regional specificity and complex interactions between livestock management, productivity, and soils may limit the transferability of results of MIG across farms (Herrero et al., 2013; Rojas-Downing et al., 2017). The effects of MIG on forage productivity and soil C storage depend upon soil characteristics such as texture and bulk density, landscape features such as slope, and climate variables, including temperature and precipitation. The ways livestock interact with these factors can vary over space and time, affecting how MIG might enhance forage yields and soil C accumulation. Spatial variations based on slope and soil type may impact soil moisture (Reid, 1973) and nutrient leaching (Brouwer and Powell, 1998) which can also have feedbacks on forage productivity and soil C storage. Interannual climate variability also influences growth and C allocation, with productivity significantly reduced when conditions deviate outside temperature and precipitation optima. Within a MIG system, animals move from one paddock to the next often using movable fencing that can adjust paddock size to account for changes in forage availability. As livestock graze, they may alter plant growth and biomass C allocation patterns, compact the soil, increase bulk density, and deposit manure, creating N hotspots, all of which may affect spatiotemporal forage yields and patterns in soil C storage and N_2O emissions that are difficult to measure and predict.

Connecting the drivers of soil C storage to the drivers of increased forage production in MIG systems is a critical need to employ MIG as a sustainable management practice with optimal climate outcomes, especially in regions where the effects of MIG are poorly understood, such as the northeastern US. While studies have shown positive agroecosystem impacts of MIG in warm-season pastures or rangelands (Conant et al., 2003; Follett et al., 2000; Oates and Jackson, 2014; Teague et al., 2011), the effects of management on agroecological outcomes such as forage productivity and soil C storage in cool-season pasture systems under MIG are much less studied. Process-based models of soil C dynamics can help investigate the impact of these management practices as they provide an opportunity for scenario testing, sensitivity analysis, and can be run over large spatial and temporal scales that are difficult to achieve in field studies (Brilli et al., 2017). However, model analyses of grazed systems are often constrained by limited data availability, as the intersection of forage growth and animal movement create dynamics that are complex and variable spatially, temporally, as well as in terms of environmental impacts. Accurately representing grazed systems at the scale of an individual field requires many different types of information in order to overlay animal treatment (i.e. location, number of animals, supplemental feed if applicable, paddock fence location if movable) with forage (i.e. species, amendments if applicable, non-animal removal if applicable) and identify sensitive factors affecting soil C and other ecosystem properties (Fetzel et al., 2017).Even at a field scale, temporally and spatially resolved data are rare, with data for paddocks within fields under MIG even more so. Paddock-scale modeling is also extremely demanding of time and computational resources, making MIG paddocks exceptionally difficult to represent in a model, despite the importance of paddocks to the field's temporal and spatial dynamics.

Here we used the process-based biogeochemical model DNDC at a field scale to study the potential for soil C storage promotion under MIG in cool-season pastures typical of the northeastern US, comparing results against field measurements of soil C and N, forage productivity, and greenhouse gas (GHG; both CO_2 and N_2O unless otherwise noted) exchange. Our objectives were to determine how (1) initial system conditions, including soil C and clay content, will influence the magnitude and rate of increase in soil C across fields; and (2) determine soil C gains in grazed compared to hayed fields while acknowledging accompanying N_2O emissions that may offset the climate mitigation benefit. In addition to investigating these objectives, we also explored how simulated changes in initial soil physiochemical conditions, grazing intensity, and vegetation composition alter soil C storage and GHG emissions to identify factors that might maximize soil C sequestration while minimizing soil GHG emissions.

2. Methods

In this study, we analyzed soil samples taken at the same locations 10 years apart to assess how soil conditions have changed a decade after converting to a certified organic, MIG dairy operation. We used these data to parameterize the DeNitrification-DeComposition (DNDC) model (Li et al., 1997; Li et al., 1992) to simulate the effects of MIG on soil C storage and GHG dynamics. We then conducted sensitivity analyses to explore how antecedent conditions, grazing intensity, and soil physical properties influence changes in soil C and N stocks and GHG fluxes. In addition to cumulative CO_2 and N_2O emissions, a CO_2 equivalent (CO_2-eq) was calculated as the sum of net CO_2 exchange and N_2O flux $\times$ 298 to account for the 298$\times$ warming potential of N_2O compared to CO_2 (IPCC, 2019).

2.1. Site description

This study took place at the University of New Hampshire (UNH) Organic Dairy Research Farm (ODRF; Fig. 1). Established in 2005, the ODRF (https://colsa.unh.edu/facility/organic-dairy-research-farm) is in Lee, New Hampshire, USA (43.09°N, 70.99°W), approximately 12 km from the UNH campus. The site is characterized by a humid continental climate with a mean annual temperature of 8.7 °C, and an annual precipitation of 1174 ± 184 mm (mean ± sd) from 1985 to 2017 (NOAA NCEI, 2021). Unusually wet years include 2006 and 2008 with 1629 and 1609 mm of annual precipitation and 2015 and 2016 were unusually dry with 969 and 919 mm of annual precipitation, respectively. The ODRF contains 40 ha of certified organic pastures divided into 14 fields. Two fields (13 ha) are under MIG, here denoted as G1 and G2, while the remaining fields are managed for hay, here designated as H1 and H2 (Fig. 1, Table 1). Both the grazed and ungrazed fields consist of species typical of Northeast naturalized swards: Kentucky bluegrass (*Poa pratensis*), orchard grass (*Dactylis glomerata L.*), timothy (*Phleum pratense*), red clover (*Trifolium pratense*), and white clover (*Trifolium repens*). Soils are marine terraces with parent material of glacial till or outwash and are primarily of Buxton, Hollis-Charlton, Hinckley, Scantic, and Swanton series, providing heterogeneity in soil characteristics. Textures range from loamy sands to silt loams (NRCS Soil Survey Staff, 2020). Harvest records since 1980 indicate the farm had been primarily managed for hay until the creation of the ODRF (Cousineau et al., 2008). The first cows arrived in late fall 2006, and milk deliveries began in January 2007. The farm currently supports 79 Jersey cows, 60 milkers and 19 heifers and calves. It meets organic requirements for grazing and dry matter intake by keeping the lactating herd on pasture ~120 days per year. When grazing, cows rotate through a series of paddocks, each averaging 0.40 ha and occupied for ~24 h. From 2014

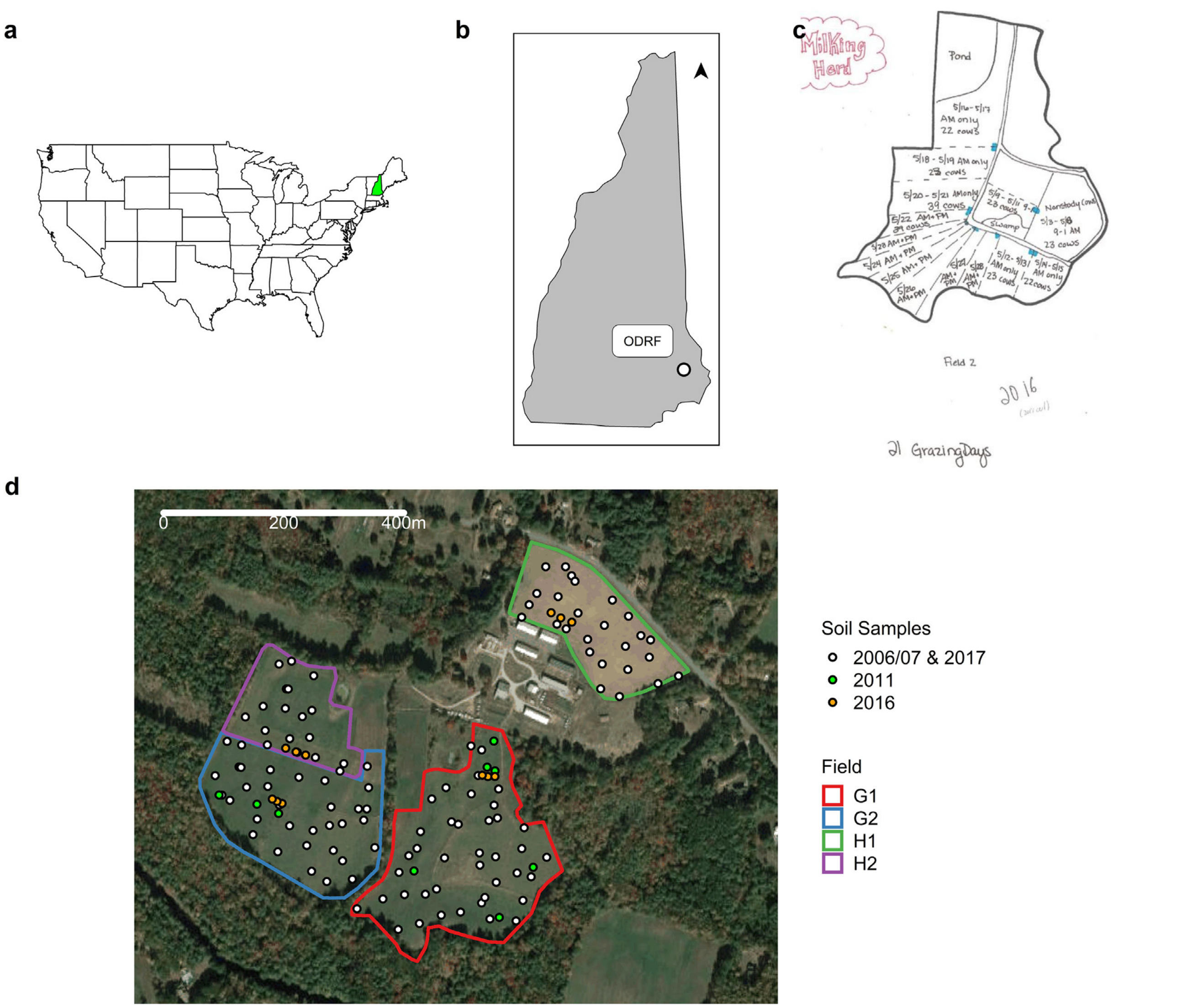

Fig. 1. (a) Location of the state of New Hampshire (NH) highlighted in green in the contiguous United States. (b) Location of the Organic Dairy Research Farm (ODRF) in Lee, NH. (c) Example of a hand-drawn map provided by the farm showing the paddocks and dates grazed. (d) Satellite image showing the four fields analyzed in this study (Maxar Technology imagery extracted using "ggmap" (Kahle and Wickham, 2013)). Points are locations of soil sample points collected over the course of the study.

Table 1
Initial soil conditions at the four fields showing the total carbon (C) and nitrogen (N) concentration as well as texture breakdowns.

Field	Total C	Total N	Sand	Silt	Clay	Texture Class
	%	%	%	%	%	–
G1	4.11 ± 0.14	0.32 ± 0.01	53.63 ± 2.05	33.59 ± 1.27	12.78 ± 1.11	Sandy Loam
G2	2.84 ± 0.11	0.22 ± 0.01	61.41 ± 1.28	27.9 ± 0.9	10.7 ± 0.46	Sandy Loam
H1	3.34 ± 0.32	0.27 ± 0.04	78.36 ± 0.78	14.91 ± 0.63	6.74 ± 0.24	Loamy Sand
H2	3.73 ± 0.23	0.29 ± 0.01	56.05 ± 1.92	32.2 ± 1.24	11.76 ± 0.95	Sandy Loam

to 2017, paddock maps were hand drawn to show movable fencing locations, the number of animals, and the duration of their stay within each paddock (Fig. 1c).

2.2. Soil sample collection and analysis

Baseline soil sampling at the ODRF occurred in 2006 and 2007, which was during the transition to an organic production system that utilized MIG as a pasture management strategy (Fig. 1d). For this baseline sampling, the farm was divided into 40 m × 40 m grid cells, with sampling points randomly determined within each grid cell using ArcGIS ($n = 172$ points). Locations were geo-referenced with a Trimble Pathfinder 4000 GPS with <1 m error (Trimble Navigation, Sunnyvale, CA, USA), and each point was considered the centroid of a 1 m × 1 m square. Samples were collected to 15 cm depth at the centroid and at the four corners of the square and were hand-homogenized in the field. About half of these samples were collected in fields now under MIG while the other half were in fields currently managed for hay. Baseline soil samples were transported to the USDA Agricultural Research Service New England Plant, Soil, and Water Research Lab in Orono, Maine, USA and were kept at 4 °C until air dried and sieved to 2 mm. Soil texture was analyzed using Kettler et al. (2001), and ~ 20 g of subsamples were archived for future analysis. Archived soil samples were transported to UNH in 2016 and were analyzed for C and N as described below.

Ten years following the transition to an organic MIG system, soils were resampled at the ODRF June 26 – June 29, 2017. Fields or areas of fields included in the baseline soil sampling that did not feature consistent grazing or haying management were omitted from the resampling effort, leaving $n = 119$ points. These original points were located with a Trimble GeoExplorer 6000 Series Geo XT GPS with submeter accuracy. As with the baseline sampling, soils were cored to 15 cm using hand augers, with five subsamples per location. Soils were transported to UNH and stored at 4 °C pending processing and analysis. This consisted of weighing soil field wet (for bulk density determination), sub-sampling to measure soil moisture (65 °C for 48 h), sieving (8 mm), and air-drying. Bulk density was calculated using the moisture-corrected weight of the fine earth fraction divided by the total soil volume (Throop et al., 2012). Soils were then bulked within each sampling location to provide a single, aggregated sample for C and N analysis. Total C and N contents for both newly collected and archived soils were determined using a COSTECH ECS 4010 CHNS-O elemental analyzer (Valencia, CA, USA), with samples run in triplicate for three analytical replicates per sample.

Additional soil sampling occurred at the ODRF in 2011 and 2016 as part of ongoing research efforts at the site (Contosta et al., 2021; Contosta et al., in prep). In 2011, nine plots were established at the farm as part of a long-term study examining feedbacks between land use and climate across a mixed land use landscape (Contosta et al., in prep). All nine plots were in fields under MIG management, six in the field designated as G1 in this study, three in the field designated G2. Six soil cores were taken from each plot using a 9 cm, hollow-core concrete drill attached to a gas-powered auger (Briggs & Stratton). The cores were taken to 50 cm depth, divided into ten cm depth increments, and returned to UNH. Analysis followed the same protocols for sample processing, bulk density, texture, C and N as described above. In 2016, twelve additional plots were sampled at the ODRF as part of a study examining soil C storage and greenhouse gas emissions in fields managed for rotational grazing or hay production (Contosta et al., 2021). In this case, three plots were established in each of two grazed fields and two hayfields within each farm. Each plot was configured in same way as the plots established during the baseline soil sampling and consisted of a 1 m × 1 m square. Soil cores were collected to 50 cm depth at the centroid and at the four corners of each square using a 9 cm, hollow-core concrete drill attached to a gas-powered auger. Cores were separated into three depth increments, 0–15 cm, 15–30 cm, and 30–50 cm, and were processed as above for soil texture, bulk density, and total C and N. For both the 2011 and 2016 sample collection, separate plots were considered the experimental units, and soil physiochemical values measured across multiple cores per plot ($n = 6$ in 2011; $n = 5$ in 2016) were averaged to produce a single value.

2.3. Model calibration & validation

DNDCv9.5 was used for all modeling. The DNDC model is a process-based model that simulates the cycling of C and N through plant, soil, and climate interactions (Li et al., 1992). Data from the original 2006, 2007 field sampling event was used to parameterize soil variables (i.e., clay fraction, bulk density, surface soil C, and pH). The 0–15 cm soil depth was used to parameterize the surface soils (0–10 cm) in the model was used in validation in addition to the whole soil profile (0–50 cm). A model spin-up of 10 years was applied to all models to ensure calibration and equilibration of parameters at the start of the experiment period (i.e. 2006). Management records from the ODRF were available from 2011 to 2017 and were used to inform grazing, cutting, and manure applications. Grazing was modeled at the field scale, by converting paddock-scale management records (number of animals, time they were in paddocks, number of paddocks used during a completed field-level rotation) into field-scale approximations of MIG during the time when paddock-scale data were available.

Modeling rotational grazing comes with challenges given spatiotemporal variation in fields and grazing intensity (Wang et al., 2020). There are two potential field-level approximations of MIG – one is to keep the duration of time during which the whole field is grazed matching reality and reduce the intensity of grazing across this time (Hsieh et al., 2005). The other approach is to keep the intensity of grazing matching reality and reduce the time in which grazing is taking place (Saggar et al., 2007). For the first approach, forage will not be impacted accurately, as smaller portions will be removed over a longer time than occurs in MIG fields, and this approach would be more representative of continuous grazing. In the second approach, the field is modeled as if it were a single paddock. This means forage will be removed at an accurate rate, but there will be some degree of temporal mismatch relative to environmental conditions. DNDC inputs for grazing are on a head per ha basis for specified periods of time. To create DNDC inputs, using an example from one of the fields in the study: for the 1st approach, 45 head rotating through a 5.3 ha field with 14 paddocks, one day per paddock, would be converted at a field scale to DNDC inputs of 45/5.3 ha = ~8 head per ha for 14 days for the entire field. For the 2nd approach, the same information would be converted to DNDC inputs of 14 days/5.3 ha = ~3 days for all 45 heads for the entire field. In either case grazing intensity is approximately the same, although rounding with either approach can cause some discrepancy.

In our analysis, we chose the 2nd approach, to more accurately represent the impact of MIG on forage. Using this method allowed us to extrapolate results across the whole field assuming conditions each paddock experienced was similar. Possible error may have been introduced in this method because it does not represent the spatial heterogeneity of the paddocks and does not model grazing pressure under all climate conditions (e.g., anomalously hot dry days could be missed although cows were always grazing). Cuts were done in hayed fields three times a summer with 80% of the aboveground biomass cut each time. Manure spreading was done according to management records (see SI).

Models were validated following the approaches of similar studies (Gopalakrishnan et al., 2012; Tonitto et al., 2007) utilizing the R^2 when appropriate and normalized root mean square error (NRMSE, eq. 1 where RMSE is the root mean square error and *σ(obs)* is the standard deviation of measured values). Models were considered to have performed acceptable if the NRMSE was < two representing model data falling with two standard deviations of measured data. Variables included in model validation were aboveground biomass production (forage in grazed fields and baleage in hayed fields), surface soil C and N, ecosystem respiration, and N_2O fluxes described in Contosta et al. (2021). Briefly, biomass data were collected weekly from May to October 2017 by harvesting biomass from 1 m^2 plots and using a cumulative sum to understand growth over the season and compare to model results. Baleage data were taken from farm records of hay collected during harvests and compared to the cutting of hayed fields in DNDC model runs. Ecosystem respiration and N_2O fluxes were obtained weekly during the growing seasons of 2016 and 2017 using static chambers and collecting headspace with a syringe and using the linear change over time to determine the flux rate. Soil C and N were evaluated primarily at the surface where 3–4 sampling events took place over the course of the study. The full column was compared when available in two intermediate years (2011, 2016; see SI).

$$NRMSE = \frac{RMSE}{\sigma(obs)} \tag{1}$$

2.4. Sensitivity analyses

Model sensitivity analyses (Hamby, 1994) were conducted to determine the relative importance of management variables including stocking density and rotation length, forage composition and quality (particularly of N-fixing legumes), and soil attributes including soil texture and baseline soil C content. We simulated a range of grazing practices to analyze the impact of MIG on organic dairy pastures in the northeastern US ranging from continuous grazing to more intensive MIG. Other parameters were chosen due to their noted impact on soil C dynamics and cycling (Minasny et al., 2017) and the heterogeneity in soils in the region (Aubertin and Leaf, 1976). These analyses were performed by varying each parameter one at a time, holding the rest constant, re-running the model, and examining predictions of soil C and N stocks, GHG fluxes, and forage production (Li et al., 1992; Saggar et al., 2007). Various grazing approaches were created, similar to Saggar et al. (2007) and Wang et al. (2018), through space and time to assess different stocking densities in the two grazed fields (SI Table 2 and SI Fig. 8). This included one more intensive scenario (In1), two increasingly extensive rotational scenarios (Ex1, Ex2) and a continuous grazing scenario (Cont.). Differences in forage composition and quality were assessed by altering the composition of leguminous vegetation via the nitrogen fixation index in the model; originally set at 1.5 (N1.5; a ratio of total N in the plant to plant N uptake from soil), values used ranged from 1 (N1; no legumes) to two (N2; pure leguminous vegetation like alfalfa is typically ~4). Variability in initial soil C and clay content were assessed by relatively increasing and decreasing the mean initial soil C and clay content by 25 and 50% (SI Table 4). The grazing sensitivity analysis was only done in fields G1 and G2 as it only applied to grazing management whereas the soil conditions (initial soil C and clay content) and legume composition sensitivity analyses were run for all fields.

2.5. Statistical analysis

All statistical analyses were done in R v4.0.4 (R Core Team, 2020). Change over time in soil C, soil N, and the C:N ratio in field data were evaluated with two-tailed *t*-tests using the “t.test” function in R with the null hypothesis that the mean change in the test parameters (e.g., final C – initial C) is not different from zero. All parameters were tested to ensure a normal distribution and similar variance. Differences in baseline and newly collected soils were evaluated separately for fields under MIG or hay production to determine the effect of management on soil C and N. A mixed effects modeling framework (Pinheiro and Bates, 2000) was used to analyze the role of initial system conditions in influencing change over time in measured soil C, N and C:N using the “nlme” package (Pinheiro et al., 2018). Each model was fit with the change in the soil variable (C concentration, N concentration, or C:N) as a function of the initial value, management, and their interaction (e.g., $\Delta C \sim$ initial C + management + initial C $\times$ management). Possible random intercept effects of plot and unequal variance structures between management grouping factors were assessed in models but were not found to improve model fit and were therefore dropped. Changes over time in estimated soil C and N in the DNDC modeling efforts and sensitivity analyses were evaluated according to Wang and Swail (2001) using the “zyp” package (Bronaugh and Werner, 2019) to ensure issues of temporal autocorrelation in time-series data were addressed. Briefly, autocorrelation is assessed in the timeseries and if found, is removed before conducting a Mann-Kendall test. The linear trends are then assessed using a Theil Sen approach (Sen, 1968). In the sensitivity analyses, further comparisons were made in the trends of soil C and N using analysis of covariance (ANCOVA). Final cumulative GHG emissions were compared numerically. The net ecosystem exchange (NEE) of CO_2 was assessed as well as N_2O fluxes. NEE is calculated as the ecosystem respiration – gross primary productivity (GPP). The net CO_2 equivalent (CO_2-eq) was assessed as NEE + N_2O times 298 in accordance with IPCC (2019). Both NEE and N_2O fluxes were transformed to total mass of CO_2 and mass of N_2O before applying the multiplier, and then was converted back to mass of CO_2-C for comparisons with total NEE. A negative value for any GHG measurement denotes and sink from the atmosphere and negative CO_2-eq values are referred to as net GHG sinks.

3. Results

3.1. Change over time in biomass and soil carbon and nitrogen

3.1.1. Field data

Soils sampled from the same locations in 2006 and 2007 and then again in 2017 showed a 3.2 ± 2.9 g kg^{-1} increase in soil C concentration in the grazed fields but no significant change in soil C in fields managed for hay (Fig. 2; grazed: $p = 0.030$, $t = 2.22$, df = 64; hayed: $p = 0.548$, $t = 0.61$, df = 32). The same was the case for soil N, which exhibited a 0.79 ± 0.21 g kg^{-1} increase in N in the grazed fields but showed no change in the hayed fields (grazed: $p < 0.001$, $t = 7.60$, df = 61; hayed: $p = 0.506$, $t = 0.67$, df = 29). The C:N significantly decreased in both grazed ($p < 0.001$, $t = -12.14$, df = 64, -1.38 ± 0.23) and hayed ($p = 0.001$, $t = -3.51$, df = 32, -0.69 ± 0.40) fields. Mixed effects models showed that initial C and N concentrations were negatively correlated to the respective changes in soil C ($p = 0.006$) and N ($p = 0.005$) resulting in greater gains in soil C when initial C was lower, and losses when soil C was higher during baseline sampling (Fig. 2a). This relationship extended across management systems. The C:N also changed over time as a function of initial system conditions in both grazed fields ($p < 0.001$), with fields under MIG showing decreases in soil C:N and hayfields exhibiting both increases and decreases in C:N values.

3.1.2. Modeled soil dynamics

The modeled soil C increased in all fields (Fig. 3a, Table 2). Soil C increased the fastest (steepest slope) in G2, the grazed field with the lowest soil C ($p < 0.001$). Both hayed fields increased at similar rates ($p = 0.460$). When analyzing between grazed and hayed fields, G1 and H1 increased at statistically similar rates ($p = 0.106$), however, G1 increased significantly faster than H2 despite slightly higher initial soil C. Model results showed significant increases in soil N across all fields except H2 (Fig. 3b,

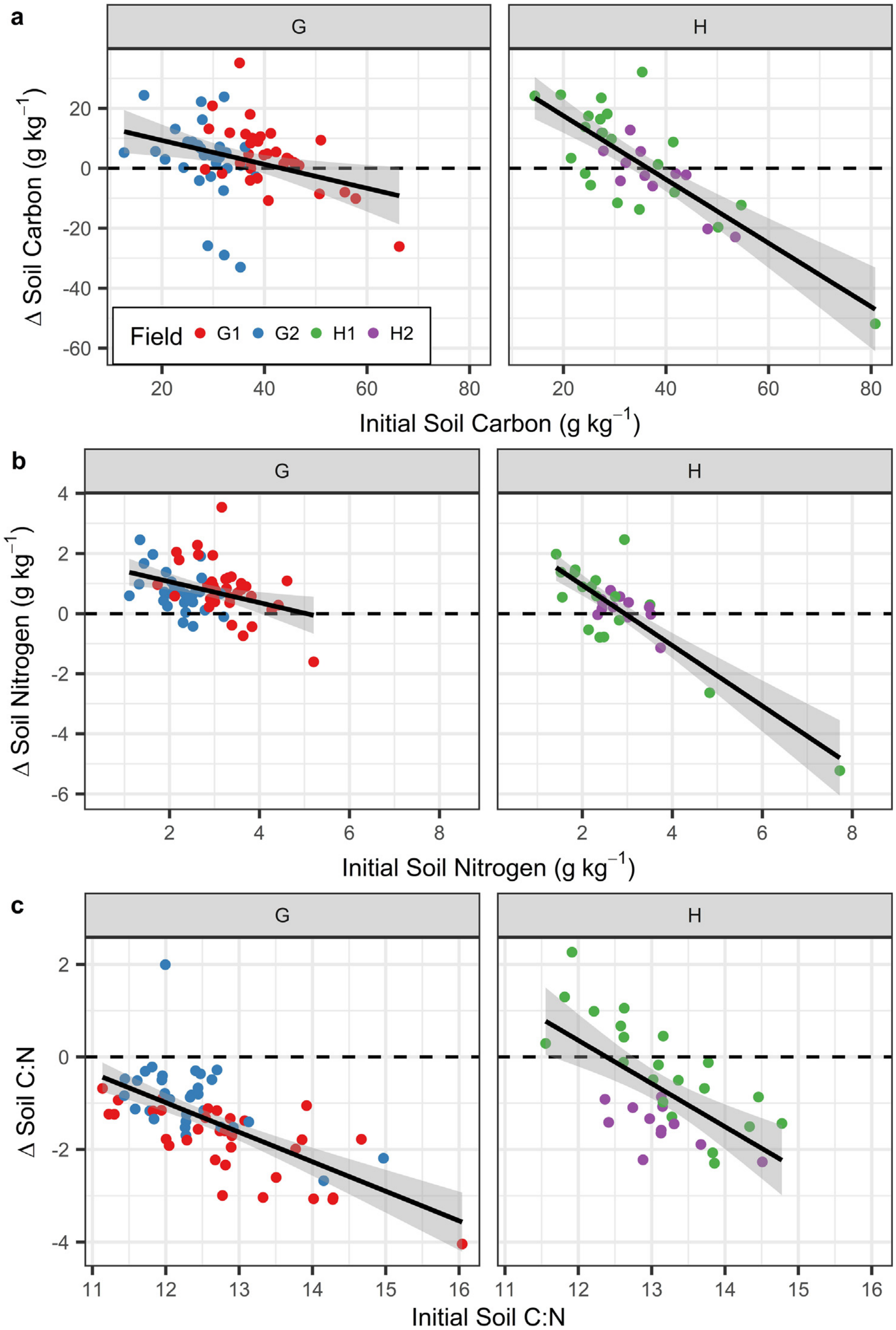

Fig. 2. Differences in (a) soil carbon (C), (b) soil nitrogen (N), and the C:N (c) measured in 2017 compared to the initial concentrations measured 2006 or 2007. Differences were assessed within management systems, grazed (G) and hayed (H). The solid line shows the relationship between initial system conditions and change over time. The dashed line indicates zero change between the initial and final soil sampling. Points that appear above the dashed line increased between the initial and final soil sampling. Points that occur below the dashed line decreased over the same period.

Table 2). Patterns of increase differed by management. G2 had the fastest rate of increase greater than G1 ($p < 0.001$) and G1 was greater than H1 ($p < 0.001$). When comparing visually, N increases co-occurred with fertilization in hayed fields while it was steadier in grazed fields where manure was directly deposited on the field during grazing (SI Fig. 13).

3.2. Greenhouse gas fluxes

Grazed fields (G1 & G2) were a smaller CO_2 sink than hayed fields (H1 & H2), with final CO_2 uptake (negative values representing uptake from the atmosphere and positive emission to the atmosphere) of −503 and − 7943 kg CO_2-C ha^{-1} in G1 and G2 and − 22,540 and − 16,599 kg CO_2-C ha^{-1} in H1 and H2, respectively (Fig. 4, Table 2). Total cumulative N_2O emissions had mixed trends, with G1 having the highest emission at 34.5 kg N_2O-N ha^{-1}, followed by H2 with 23.8 kg N_2O-N ha^{-1}. Fields with lower N stocks had lower N_2O emissions of 19.2 kg N_2O-N ha^{-1} and 16.2 kg N_2O-N ha^{-1}2, respectively in fields G2 and H1. The total CO_2-eq was calculated to evaluate the net warming impact of the four fields. Fields G1 was a source of warming with a final CO_2-eq of 3900, and G2 was a sink at −5485 kg CO_2-eq ha^{-1}. The hayed fields were larger net GHG sinks with

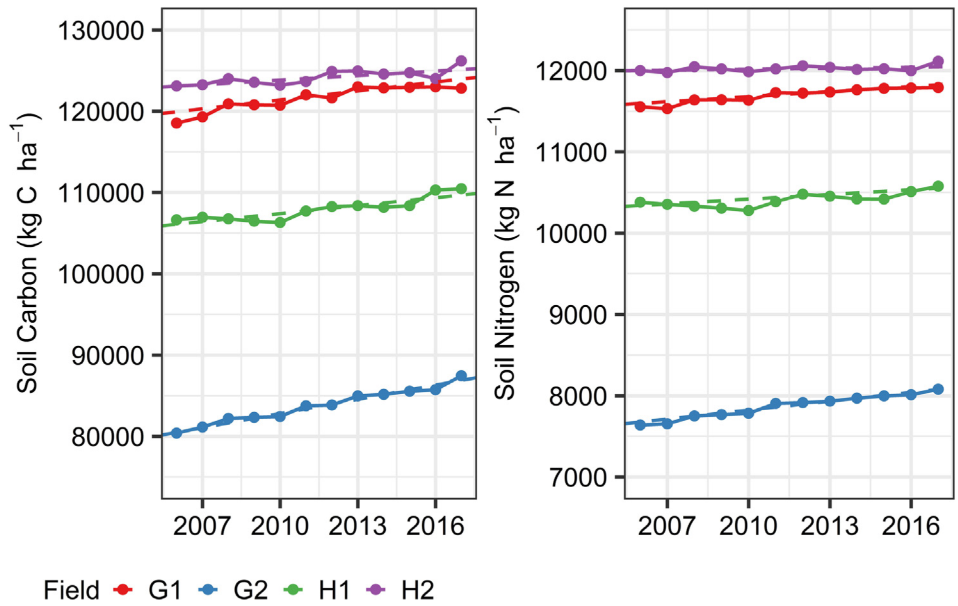

Fig. 3. Modeled trends in soil carbon (C) and nitrogen (N). (a) Soil C increased in all fields with faster rates in fields with lower starting levels of C (i.e., G2 and H1) and (b) N significantly increased in all fields except H2 with consistent increases in grazed fields and increases co-occurring with fertilization in H1. Dotted lines show the Theil-Sen slope for each time series.

−20,469 and − 13,559 kg CO_2-eq ha^{-1}, in H1 and H2 respectively, meaning that the CO_2 absorbed outweighed the N_2O released in terms of climate warming potential. Most of the total N_2O released in all fields occurred in October 2006 during relatively large (>600 kg N/ha) waste slurry spreading events during the start of the farm (Fig. 4b, SI Fig. 13).

3.3. Validations of DNDC models

All error parameters in this section are reported in the following order unless otherwise noted: G1, G2, H1, and H2. Validations of vegetation biomass over the growing season showed good performance across fields (SI Fig. 2). The R^2 values were 0.91, 0.97, 0.96, and 0.90 and the NRMSE was 1.13, 0.38, 1.11, and 0.88 with the model over-predicting field measured biomass in hayed fields and underpredicting in G1 (N = 18 per field). Baleage performed acceptably when assessed by the NRMSE (1.10, and 1.42 for H1 and H2, respectively; N = 13 per field), although it did not follow a linear trend using the daily time steps (SI Fig. 3). Ecosystem respiration followed similar trends to baleage with a NRMSE of 1.65, 1.93, 1.29, and 1.03 (SI Fig. 4; N = 31 per field, 29 in H2), with the model over predicting mid-summer values. Conversely, the model under predicted daily N_2O fluxes (SI Fig. 5; N = 46, 35, 21, and 21) with NRMSEs of 1.22, 1.44, 1.12, and 1.11. Surface soil C resulted in an NRMSE of 0.95 and soil N had an NRMSE of 0.96 (N = 14; SI Figs. 6 & 7). Validation for whole soil column (0–50 cm) soil C and N content included just one to two time points (2011 and 2016) and represented only three points for those sampling years. Whole soil column model estimates of soil C and N were generally within the error range of field samples (SI Figs. 6 & 7).

3.4. Sensitivity analyses

In all sensitivity analyses, p-values and trends represent the statistics for each individual scenario and % change represents the relative shift from year 1 (2006) to year 12 (2017) within each scenario.

3.4.1. Grazing intensity

Adjusting grazing intensity had mixed effects on soil C and N stocks and GHG fluxes between fields. In G1, the field with the highest initial soil C (4.11 ± 0.13%, mean ± standard error), the standard MIG (best reflecting actual farm management practices) stored more C when compared to more intensive (In1), more extensive (Ex1 & Ex2), and continuous (Cont.) grazing scenarios (Table 2). Continuous grazing showed no change in soil carbon as did the Ex1 scenario. These scenarios also had the lowest average production levels and net source NEE. Results differed in G2, where similar rates of soil C sequestration occurred with all rotational grazing scenarios. However, less soil C was sequestered under the continuous grazing scenario when compared to the regular MIG (ANCOVA results, p = 0.014), which also had lower forage production. Nitrogen stocks did not change significantly with more extensive and continuous grazing scenarios in field G1 and increased at similar rates to regular MIG in Ex1 and Ex2 in G2. Both In1 and continuous grazing resulted in lower rates of soil N increase (p = 0.002 and 0.037, respectively) than regular MIG. Differences in the GHG balance showed mixed results with grazing, both between fields and among grazing scenarios. In G1, NEE flipped based on production patterns and timing (SI Fig. 9) and G2 had similar NEE between scenarios. N_2O flux rates were similar to the regular MIG scenario across both fields leading shifts in NEE to drive the CO_2-eq in both fields.

3.4.2. Nitrogen fixation index

We represented the relative composition of legumes in fields with the nitrogen fixation index. The effects of this index on soil C and N stocks and GHG emissions varied between fields and among N-fixation index scenarios. In the N1 scenario where no leguminous vegetation is in fields, no significant increase in soil C was observed in any field and in hayed fields, N1.25 also showed no significant increase in soil C. All fields except G2 didn't show a significant change in the rate of soil C gain compared to the standard N1.5 scenario used in model calibration to represent normal field conditions. In G2, soil C increased at a slower rate in the N1.25 scenario (p = 0.004) and similar rates at higher levels. Like soil C, the no legume scenario (N1) had no significant increase in soil N. Field H2 had no significant increase in soil N in all scenarios, following the baseline for that field, except in N1 where there was a small significant decrease in soil N (Table 2). Rates of soil N increase were similar across other different treatments except in field G2 where the rate of soil N increase was lower under N1.25 (p = 0.044). For GHG fluxes, higher relative proportions of legumes typically resulted in both increased CO_2 uptake and N_2O emissions. However, benefits in increased CO_2 uptake often outweighed increased N_2O emissions with lower CO_2-eq with a higher N fixing index (Table 2, SI Fig. 10). Production also responded to the nitrogen fixation index with lower production under lower levels with gains leveling off above N1.5.

Table 2
Statistics for the sensitivity analyses showing variation of change over time in soil carbon (C) and nitrogen (N) stocks and cumulative greenhouse gas (GHG) fluxes. Gas fluxes are in kg CO_2-C ha^{-1} for CO_2 and CO_2 equivalents (CO_2-eq) obtained by the sum of CO_2 fluxes plus $N_2O \times 298$ in accordance with IPCC (2007). N_2O fluxes are in kg N_2O-N ha^{-1}. *P*-values and trend statistics represent the change over time in each individual scenario analyzed according to Wang and Swail (2001). The percent (%) change is the relative change from the end of 2006 to the end of 2017. Baseline conditions are bolded in the treatment column and are the standard MIG ("Sta-In") in grazing, "N1.5" in the vegetation composition, and "mean" for soil texture and C.

Test Variable	Field	Treatment	SOC		SON		Cumulative GHG fluxes			Production
			P-value	Trend (kgC/ha/yr)	P-value	Trend (kgN/ha/yr)	CO_2	N_2O	CO_2-eq	kg C/Ha
Default	G1	NA	0.013	390 ± 51	0.001	23.8 ± 2.4	−503	34.5	3900	4255 ± 393
	G2		<0.001	576 ± 32	<0.001	39.5 ± 2.4	−7943	19.2	−5485	5010 ± 231
	H1		0.020	346 ± 55	0.043	18.9 ± 4.8	−22,540	16.2	−20,469	5077 ± 235
	H2		0.029	202 ± 50	0.244	5.0 ± 2.9	−16,599	23.8	−13,559	4984 ± 261
Grazing	G1	Cont.	0.640	235 ± 76	0.087	16.2 ± 2.8	1130	34.4	5527	3719 ± 629
		Ex2	0.020	299 ± 59	0.276	19.0 ± 2.9	−2736	34.3	1642	4192 ± 519
		Ex1	0.350	278 ± 76	0.087	20.5 ± 3.0	2037	34.9	6488	3891 ± 490
		In1	0.005	228 ± 69	0.020	12.1 ± 2.5	−3802	34.9	651	3965 ± 777
Grazing	G2	Cont.	0.008	373 ± 68	0.020	26.0 ± 3.0	−8183	19.6	−5685	4142 ± 578
		Ex2	<0.001	599 ± 30	<0.001	40.5 ± 2.3	−8240	19.6	−5815	5073 ± 237
		Ex1	<0.001	586 ± 31	<0.001	39.9 ± 2.3	−8046	19.2	−5592	5036 ± 232
		In1	<0.001	505 ± 47	<0.001	32.0 ± 2.4	−8897	20.2	−6316	4482 ± 524
Legume Composition	G1	N1	0.755	149 ± 63	0.640	11.4 ± 3.3	−311	27.9	3252	3623 ± 331
		N1.25	0.043	349 ± 52	0.001	21.0 ± 2.6	−219	31.9	3860	4151 ± 368
		N1.75	0.013	400 ± 51	<0.001	25.0 ± 2.4	−680	36.3	3960	4281 ± 401
		N2	0.005	415 ± 49	<0.001	25.9 ± 2.4	−1374	37.7	3443	4354 ± 405
Legume Composition	G2	N1	0.640	9 ± 73	0.755	13.5 ± 4.1	−3127	14.8	−1238	3111 ± 261
		N1.25	0.013	343 ± 64	0.013	30.7 ± 3.3	−5911	17.6	−3667	4461 ± 328
		N1.75	<0.001	609 ± 30	<0.001	42.0 ± 2.3	−8564	21.0	−5886	5099 ± 251
		N2	<0.001	623 ± 30	<0.001	43.5 ± 2.2	−9001	22.3	−6151	5136 ± 259
Legume Composition	H1	N1	0.213	114 ± 73	0.161	5.0 ± 4.8	−13,088	13.1	−11,419	3367 ± 163
		N1.25	0.062	274 ± 60	0.043	13.9 ± 4.8	−19,367	14.5	−17,512	4589 ± 185
		N1.75	0.013	368 ± 56	0.043	20.4 ± 4.9	−23,865	17.9	−21,579	5185 ± 241
		N2	0.013	383 ± 58	0.043	21.6 ± 4.9	−24,709	19.3	−22,238	5237 ± 245
Legume Composition	H2	N1	0.213	−87 ± 53	0.043	−7.4 ± 3.0	−8649	19.7	−6128	3517 ± 103
		N1.25	0.161	116 ± 51	0.732	1.6 ± 2.9	−14,105	21.9	−11,305	4625 ± 174
		N1.75	0.020	221 ± 50	0.150	6.5 ± 2.8	−17,346	25.7	−14,067	5053 ± 280
		N2	0.008	232 ± 50	0.064	7.7 ± 2.8	−17,832	27.2	−14,362	5087 ± 288
Texture	G1	−50%	0.020	323 ± 45	<0.001	15.1 ± 2.1	−8011	31.3	−4007	4217 ± 381
		−25%	0.013	352 ± 48	0.001	19.0 ± 2.3	−5410	33.1	−1178	4237 ± 389
		+25%	0.008	434 ± 56	<0.001	29.1 ± 2.6	4533	36.7	9216	4255 ± 398
		+50%	0.008	478 ± 58	<0.001	34.8 ± 2.8	8300	38.2	13,180	4262 ± 404
Texture	G2	−50%	<0.001	484 ± 29	<0.001	31.4 ± 2.1	−11,745	17.5	−9509	4920 ± 225
		−25%	<0.001	527 ± 32	<0.001	35.1 ± 2.2	−10,894	18.5	−8525	4959 ± 229
		+25%	<0.001	624 ± 31	<0.001	44.1 ± 2.5	−3722	19.8	−1192	5049 ± 234
		+50%	<0.001	663 ± 31	<0.001	48.4 ± 2.6	248	20.5	2865	5065 ± 235
Texture	H1	−50%	0.029	307 ± 50	0.043	15.7 ± 4.6	−22,959	14.5	−21,104	5022 ± 231
		−25%	0.020	324 ± 52	0.043	17.1 ± 4.7	−22,926	15.8	−20,909	5055 ± 234
		+25%	0.008	371 ± 57	0.043	20.9 ± 5.0	−21,639	16.6	−19,519	5105 ± 236
		+50%	0.008	395 ± 59	0.043	23.1 ± 5.1	−20,054	17.0	−17,878	5135 ± 236
Texture	H2	−50%	0.087	133 ± 46	0.451	−1.4 ± 2.5	−21,170	21.9	−18,378	4894 ± 251
		−25%	0.087	162 ± 48	0.945	1.4 ± 2.7	−19,859	22.8	−16,942	4923 ± 256
		+25%	0.013	243 ± 51	0.062	9.1 ± 3.0	−12,411	25.2	−9192	5010 ± 268
		+50%	0.008	286 ± 52	0.013	13.6 ± 3.1	−8891	26.4	−5523	5037 ± 275
Soil C	G1	−50%	0.003	609 ± 59	<0.001	44.0 ± 2.7	−1187	26.4	2190	4248 ± 401
		−25%	0.005	499 ± 55	<0.001	33.8 ± 2.6	−789	30.6	3115	4252 ± 399
		+25%	0.062	279 ± 49	0.003	14.1 ± 2.4	182	38.9	5149	4220 ± 389
		+50%	0.213	173 ± 47	0.213	4.6 ± 2.3	667	43.1	6176	4198 ± 383
Soil C	G2	−50%	<0.001	717 ± 31	<0.001	52.7 ± 2.6	−8245	14.9	−6347	5002 ± 222
		−25%	<0.001	646 ± 31	<0.001	46.0 ± 2.5	−8119	17.0	−5945	5011 ± 227
		+25%	<0.001	504 ± 32	<0.001	33.0 ± 2.3	−7739	21.5	−4991	5003 ± 234
		+50%	<0.001	434 ± 33	<0.001	26.8 ± 2.2	−7508	23.8	−4468	4995 ± 237
Soil C	H1	−50%	0.003	543 ± 56	0.013	36.7 ± 4.8	−21,981	11.7	−20,481	4906 ± 224
		−25%	0.008	448 ± 55	0.043	28 ± 4.8	−22,420	13.9	−20,648	5040 ± 228
		+25%	0.087	244 ± 56	0.087	9.5 ± 4.9	−22,473	18.8	−20,077	5071 ± 237
		+50%	0.213	140 ± 58	0.350	0.3 ± 4.9	−22,307	21.4	−19,573	5051 ± 239
Soil C	H2	−50%	0.001	420 ± 49	<0.001	26.6 ± 3	−16,975	15.3	−15,026	4867 ± 228
		−25%	0.005	317 ± 49	0.013	15.9 ± 2.9	−16,966	19.5	−14,473	4959 ± 248
		+25%	0.244	82 ± 50	0.062	−5.6 ± 2.8	−16,056	28.2	−12,451	4981 ± 265
		+50%	0.537	−38 ± 51	0.001	−16.1 ± 2.8	−15,426	32.7	−11,246	4968 ± 263

3.4.3. Soil texture

Scenarios of increased clay content generally supported increased soil C storage in all fields, albeit there were not many cases of rate significantly changing in the scenarios tested. Decreased clay proportions in H2, the field with the highest clay and initial soil C content, even resulted in nonsignificant soil C gains (Table 2). Significant changes in soil C sequestration rates were in field G2 where the minus 50% scenario had significantly lower rates than the mean value ($p = 0.046$). Soil N had similar trends where clay content was positively correlated to rates of soil N sequestration. Specifically, grazed fields had significantly different rates of soil N increase

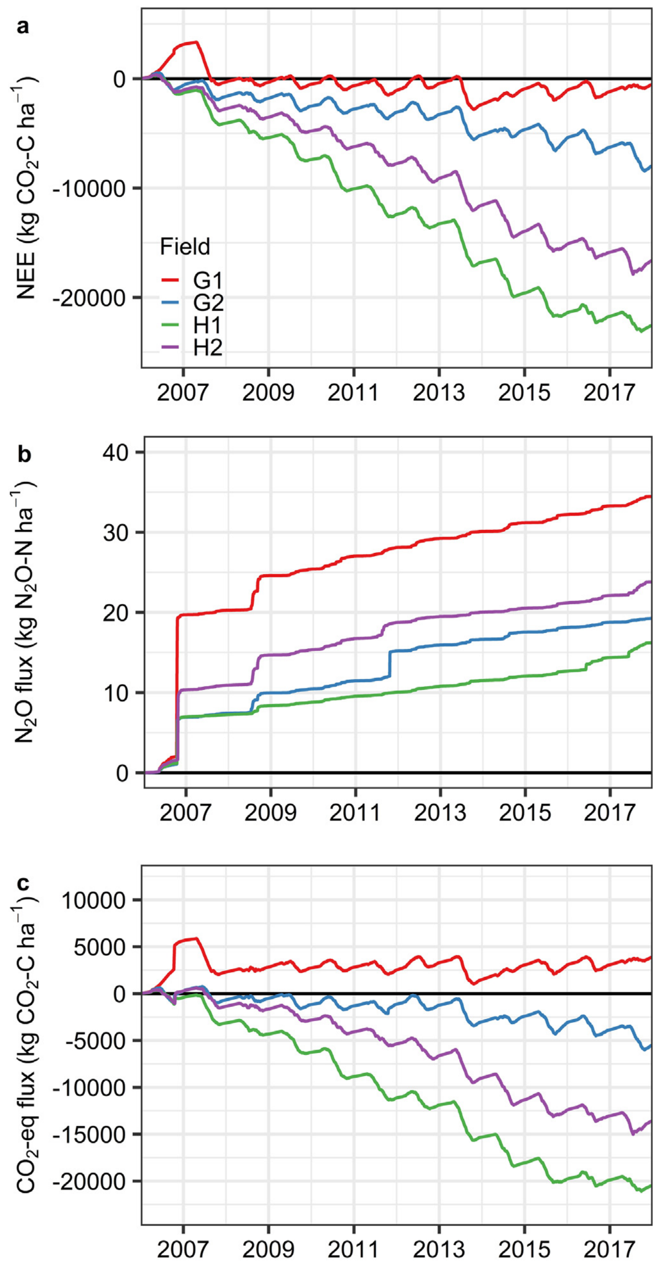

Fig. 4. Cumulative modeled greenhouse gas (GHG) fluxes from each field. (a) Net ecosystem exchange (NEE; CO_2 flux) over the 12-year study showing sinks from each of the fields with stronger sinks from hayed fields. (b) Nitrous oxide (N_2O) emissions showed pulses corresponding to fertilization. (c) Gas fluxes as CO_2 equivalents (N2O multiplied by 298 to account for stronger warming; IPCC, 2019) showing an overall source of warming in G1, a weak sink in G2, and stronger sinks in hayed fields.

at the plus 50% ($p = 0.007$ and $p = 0.019$ for G1 and G2, respectively) and minus 50% scenarios ($p = 0.014$ and $p = 0.019$). Higher clay content also resulted in lower uptake of CO_2 and larger N_2O emissions. G2 became a CO_2 source in the +50% treatment at 248 kg CO_2-C ha^{-1}. The hayed fields remained CO_2 sinks in all treatments but with the same patterns of an increased sink with reduced clay content and vice versa when clay content was increased. Likewise, net CO_2-eq followed the same trends as cumulative CO_2 and N_2O loss, with higher total emissions occurring with higher clay content. A net GHG sink began in G1 in the −25% clay scenario from the increased CO_2 sink and decreased N_2O emissions.

3.4.4. Initial soil carbon

As with measured and modeled changes over time in soil C, sensitivity analyses also indicate that lower initial soil C concentrations were linked to higher increases in soil C stocks across all fields. In all fields except G2, the field with the lowest initial soil C, adding any additional initial soil C caused carbon sequestration to cease (Table 2). At the −50% treatment, all fields had significantly greater increases in soil C than at their respective baseline conditions ($p = 0.011, 0.005, 0.021$, and 0.006 for G1, G2, H1 and H2, respectively). In grazed fields, more instances of soil C sequestration were observed when initial soil C concentrations below ~120,000 kg C ha^{-1}. Lower levels of initial soil C were also correlated to higher rates of soil N increase across fields and management. In H2, with the highest measured levels of initial soil C, N accumulation stopped at the +25% and began to decrease at the +50% treatment. Cumulative GHG exchange trended in a similar direction with lower initial soil C resulting in greater sinks of CO_2 and higher initial levels resulting a source of CO_2 in G1. Emissions of N_2O responded more strongly to sensitivity analysis of initial soil C content as compared to clay content, with lower N_2O emissions in scenarios with lower initial soil C and higher N_2O emissions in scenarios with increased soil C. The different soil conditions, including soil C and texture, did not result in large changes in production levels relative to baseline.

4. Discussion

4.1. Initial system conditions

Our study used a combination of spatiotemporally intensive measurements and process-based modeling to explore how MIG might increase soil C storage in cool-season pastures that are characteristic of the northeastern US and other temperate areas globally. We show that MIG can enhance soil C concentrations by about 4% over time (Oates and Jackson, 2014; Teague et al., 2008; Teutscherová et al., 2021), but that these increases are strongly influenced by baseline soil physiochemical conditions. Our field-based measurements revealed a negative correlation between initial soil C concentrations and change over time in soil C content. Modeling results indicated significant increases in soil C as well with larger increases in grazed fields than hayed fields, agreeing with field data trends. Further, as initial C content was altered in sensitivity analyses, a threshold like response at about 120,000 kg C ha^{-1} was observed where soil C would not continue to increase if soils started near this level, regardless of management. The farm studied here has comparable stocks of soil C relative to other farms in the region which range from ~170,000 kg C ha^{-1} to ~70,000 kg C ha^{-1} (Contosta et al., 2021). The sensitivity analysis of soil textures shows similar but less impactful trends than initial soil C with higher values supporting increased uptake in all fields. This is important for regions such as the Northeastern US where soil textures may be highly spatially variable due to the glacial history. Sensitivity analysis also demonstrated greater soil C gains under scenarios where initial soil C content was reduced below baseline values, and even pauses in C gains when soil C was increased. Previous studies have illustrated the role that initial system conditions play in driving soil C gains with management (McSherry and Ritchie, 2013; Minasny et al., 2017), with some of the largest C increases occurring in highly degraded soils. Yet even when soils start with relatively low C content, they can be expected to eventually reach equilibrium, after which soil C gains may be difficult to achieve (Chan et al., 2011; Nemo et al., 2016; Poeplau et al., 2011) so even fields with C gains observed here may start to have diminishing returns over time.

4.2. Tradeoffs between soil C storage and GHG emissions

Our modeling results also demonstrate the importance of considering how soil N dynamics can change with management approaches aimed at enhancing soil C storage, particularly within the context of site-specific conditions. Field measurements from 2006/2007 and 2017 show that total soil

N increased significantly across all fields over the course of the study (Figs. 2b, 3b), but patterns differed between MIG and hayed fields. Model simulations exhibited a similar dynamic while also illustrating how management events (i.e., fertilization) may drive rapid changes in soil N content and N losses. Generally, soil N co-increased with soil C across a range of modeling scenarios, and higher levels of soil N corresponded to higher total N_2O emissions. Prior research has also documented that elevated N_2O emissions can accompany increases in soil C storage (Contosta et al., 2021; Gu et al., 2017; Lugato et al., 2018; Xia et al., 2018), which can offset climate mitigation benefits achieved with C sequestration. However, over 50% of N_2O emissions followed fertilization events, specifically in the first year of the farm (2006), so reductions in large fertilizer use could greatly reduce N_2O from these farms (Bell et al., 2016; Schwenke and Haigh, 2016). Reducing fertilizer is not universally beneficial, fertilization can conserve natural forest land by increasing production on smaller areas that can increased the net C sink and can outweigh increased N_2O emissions (Snyder et al., 2009). Further, sometimes more important consideration is the timing of application in relation to expected precipitation events, which are a vital factor to the amount of N_2O produced (Zimmermann et al., 2018).

Timing is particularly important regarding soil N losses through leaching and emissions of N_2O given the relation between precipitation or irrigation to run-off (Di and Cameron, 2002) and denitrification occurring during “hot moments” when nutrient and redox conditions in “hot spots” on the landscape (Wagner-Riddle et al., 2020). In grazed fields, gradual increases in soil N and cumulative N_2O emissions were punctuated by rapid gains that coincided with manure fertilization. By contrast, hayed fields saw sudden increases in soil N and cumulative N_2O fluxes with spreading events that plateaued between manure additions. These differences were due to regular deposition of manure from grazing animals in grazed fields. Although most attention on fertilizer management is on the amount, type, and timing of fertilizers used (Snyder et al., 2009), the relative proportion of leguminous species may also have influenced soil N inputs, storage and losses (Fuchs et al., 2020). When we varied the relative abundance of legumes within the fields, we observed significant increases in soil C, soil N, and cumulative N_2O emissions, particularly in field G2, where we observed the greatest gains in soil C according to both our field measurements and baseline model output. As with other findings, this result emphasizes the importance of management recommendations that account for existing environmental conditions.

4.3. Model performance and sensitivity analysis

The DNDC simulations performed well for aboveground vegetation, which is one of the key impacts of grazing management on forage production (Mwendera et al., 1997), with cascading effects on soils biogeochemical cycling. The robust relationship between measured and modeled values of aboveground biomass production gives us confidence in simulated soil C and N values, especially given good model predictions of soil C and N in surface soils total column estimates within the uncertainty range of observations of soil C and N, and the slow change in total soil C over time (Smith, 2007). Predictions of GHG exchange were less accurate on a daily timescale, but this is not unusual based on the spatiotemporal variability in these parameters (Shah et al., 2020). In fact, respiration was generally over-estimated meaning that our modeled estimates were likely conservative with regards to the C sink. While we cannot validate the results of our sensitivity analyses, they provide key insights into the relative roles of grazing intensity, vegetation composition, initial soil C content, and soil texture in driving soil C storage and GHG emission that can be further evaluated with new field experiments and data collection. We examined how a variety of grazing intensity regimes might impact soil C sequestration and GHG emissions given the role of stocking density in driving historic increases and decreases in soil C (Potter et al., 2001). Results from these analyses suggest that MIG in cool-season pastures, like the northeastern US, can store soil C while maintaining a net GHG sink under conditions where legume composition is high enough, initial soil C content is below 4%, and clay content is at least 10%.

Taken together, our field-based measurements and model simulations indicate that initial soil conditions should be evaluated when assessing the soil C storage potential of a particular system. They also illuminate the ways management might enable soil C storage, particularly given aggressive grazing and tillage practices that are historically responsible for large losses of C from soils (Salinas-Garcia et al., 1997; Wright et al., 2005). Strategies such as MIG (Heiberg and Syse, 2020; Stanley et al., 2018), crop-pasture integration (de Faccio Carvalho et al., 2010), and reduced tillage have been successful in many instances in sequestering soil C, particularly when initial soil C levels are low (Chivenge et al., 2007; Krauss et al., 2017). In fact, many of the scenarios included in this study, which varied initial C content, grazing intensity, or clay content, sequestered more than 4 per mil C annually, which is a goal of global climate priorities such as the United Nations 2016 Conference of the Parties (Minasny et al., 2017). Despite the overall performance of the model, our study showed the importance of considering the initial ecosystem conditions when considering the benefits of different management on organic dairy operations. Our study, by modeling at a field scale, also treats each field in a homogenous way that is not reflective of soil heterogeneity typical in glacial impacted regions such as New England. Thus, there may variability between paddocks that was not captured in this analysis and would be worth exploring if sufficient data and computational capacity can be brought to bear to parameterize with MIG and model with greater spatial and temporal accuracy. This type of analysis would most benefit by extensive paddock-scale measurements, ideally paired with strong inter-paddock signals of impacted ecosystem properties. Nonetheless, our results provide useful insights for efforts in upscaling results from field to regional scales that may inform where MIG may best help climate goals.

5. Conclusions

This study highlights the importance of site conditions when analyzing climate benefits and impacts of pasture ecosystems. In four fields on the same farm, differences were observed under similar management techniques despite being in the same region and under the same climate regime. Our results build on research that has shown soils with less soil C have greater capacity for further soil C storage. We observed C sequestration rates higher than the 4 per mil benchmark, showing the potential for regional agricultural systems to contribute to meeting global climate mitigation goals, however, when considering net warming potential the hay fields had higher net GHG uptake. Higher initial soil C levels may reduce the potential for further storage, but management decisions related to grazing intensity and vegetation composition may help to retain soil C within an existing system. We highlight the need for more spatially explicit research in how farm management with high physiochemical heterogeneity can maximize soil C sequestration while minimizing soil GHG emissions.

CRediT authorship contribution statement

Kyle A. Arndt: Conceptualization, Methodology, Validation, Formal analysis, Writing – original draft, Visualization. **Eleanor E. Campbell:** Conceptualization, Methodology, Validation, Writing – review & editing. **Chris D. Dorich:** Investigation, Writing – review & editing. **A. Stuart Grandy:** Investigation, Writing – review & editing, Funding acquisition. **Timothy S. Griffin:** Investigation, Writing – review & editing. **Peter Ingraham:** Methodology. **Apryl Perry:** Investigation, Writing – review & editing. **Ruth K. Varner:** Supervision, Funding acquisition, Writing – review & editing. **Alexandra R. Contosta:** Formal analysis, Investigation, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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