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Synergistic water quality and soil organic carbon sequestration benefits of winter cover crops

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Abstract

Winter cover crops (WCCs) are promising best management practices for reducing nitrogen and sediment pollution and increasing soil organic carbon (SOC) sequestration in agricultural fields. Although previous watershed studies assessed the water quality benefits of growing WCCs in the Chesapeake Bay watershed, the SOC sequestration impacts remain largely unknown. Here, we designed six WCC scenarios in the Tuckahoe Watershed (TW) to understand potential synergies or tradeoffs between multiple impacts of WCCs. Besides corroborating the nitrate reduction benefits of WCCs that have been reported in previous studies, our results also demonstrated comparable reduction in sediment. We also found that the six WCC scenarios can sequester 0.45-0.92 MgC ha^{-1} yr^{-1}, with early-planted WCCs having more than 70% SOC sequestration benefits compared with their late-planted counterparts. With a linear extrapolation to all the cropland in Maryland, WCCs hold potential to contribute to 2.1-4.4% of Maryland's 2030 Greenhouse Gases reduction goal. Additionally, we showed that WCCs could noticeably increase evapotranspiration and decrease water yield and streamflow, which could negatively impact aquatic ecosystem health and water supply. Overall, this study highlights the potential synergistic water quality and SOC sequestration benefits of WCCs in the Chesapeake Bay watershed. Meanwhile sustainable adoption of WCCs into existing crop rotations will also require careful assessment of consequential water availability.

Keywords:

Cover crops, nitrate, sediment, sustainability, SWAT-Carbon, tradeoffs, water availability

1. Introduction

The U.S. agriculture sector not only secures food, feed, fiber, and fuel supply, but also contributes significantly to job growth and economic development (Hatfield, 2014). Accompanying those great benefits are substantial environmental impacts, such as soil erosion, water quality degradation, greenhouse gas (GHG) emissions and biodiversity loss (Tilman, 1999). Over the last few decades there has been an ever-growing interest from multiple sectors to minimize these negative effects. For example, the "US Agriculture Innovation Agenda" aims to reduce detrimental environmental impacts of U.S. agriculture, such as achieving a net reduction of the agricultural sector's carbon footprint and reducing nutrient loss by 30 percent (USDA, 2020). To sustainably meet the growing demand of agricultural products in the U.S. and the world, the multi-dimensional environmental impacts of diverse cropping systems need to be assessed to inform sustainable agricultural development.

Agriculture is a major source of carbon (C) emissions to the atmosphere, however, certain agricultural practices can also provide opportunities to remove carbon from the atmosphere (IPCC, 2022). CO_2 emissions from cropland mainly come from SOC change and agronomic inputs and operations, such as seeds, fertilizer/herbicides, tillage, harvesting (Gelfand et al., 2013). SOC change is often a major contributor to the life cycle global warming impacts of cropland (Lee et al., 2020). Therefore, agricultural practices, such as cover crops (Kaye and Quemada, 2017), agroforestry (Don et al., 2011), conservation tillage, and biochar application (Gross et al., 2021), that can help sequester carbon into soils are being promoted as important measures to offset CO_2 emissions from the agriculture sector. For example, uptake and storage by agricultural soils hold promise to reduce approximately 3 $PgCO_2$ emissions from the global agricultural sector (NASEM, 2019). It is worth noting that other CO_2 utilization techniques such

as converting CO_2 into valuable products in industrial sectors, or biological conversion of CO_2 in both industrial and agricultural sectors have been developed (Ahmad et al., 2023a; Ahmad et al., 2022; Yousaf et al., 2022). Coordination between those techniques in different social-economic contexts may help maximize the potential of total GHG emissions reduction.

In addition to the significant role of agriculture-based measures for C removal from the atmosphere to achieve our climate goals, IPCC (2022) also highlighted the importance of impacts, risks and co-benefits of the land-based C mitigation measures. A major co-benefit of growing cover crops is water quality improvement, such as nitrate and sediment reduction % (Blanco‐Canqui, 2018). Currently, winter cover crops (WCCs) are recommended as an agricultural best management practice (BMP) to reduce nitrogen and phosphorus loading to the Chesapeake Bay, thereby enhancing the economic viability and environmental and biological integrity of this region. Note that previous field experiments well-documented the noticeable impacts of growing cover crops on terrestrial hydrology. Cover crops can increase infiltration (McVay et al., 1989) and evapotranspiration (Logsdon et al., 2002), and decrease soil moisture (De Bruin et al., 2005) and runoff (Kaspar et al., 2001). Although diminished water yield can mitigate water quality issues by decreasing yield of sediment and nitrate from agricultural fields, growing cover crops could exacerbate drought conditions as live vegetation during winter can cause greater evapotranspiration (Dabney et al., 2001). A modeling study showed that growing cover crops under semi-arid conditions could increase ET by 15.8% as compared to the fallow soil, but has no significant impacts on the summer crop growing period (Bodner et al., 2007). The enhanced ET led to lower soil moisture and less water yield from the scenarios with cover crops. Watershed scale modeling studies also showed that the water use of cover crops can have a noticeable impact on streamflow discharge. In the TW, a previous study simulated an 8%

reduction in streamflow (Yeo et al., 2014). The modeling study conducted by Getahun and Keefer (2023) also showed that growing cover crops could modestly increase evapotranspiration (2%) and decrease water yield (5%) from the Big Ditch watershed in Illinois, USA.

Thee impacts of agricultural conservation measures are highly variable (depending on site-specific context, scale, and other factors), and can cause trade-offs with other sustainability goals (Pandeya et al., 2016; Zhang et al., 2010). Despite being a widely used agricultural measures, the synergies and trade-offs of growing winter cover crops have not been extensively studies at the watershed scale. Given the high variability field data and cost of conducting exhaustive field experiments across large scales, there is an urgent need of watershed modeling tools for quantifying the environmental benefits and costs of adoption of agricultural conservation measures over large scales. In this study, we applied the Soil and Water Assessment -Carbon (SWAT-Carbon) model (Zhang et al., 2013) in the TW to assess different cover crops for SOC sequestration and water quality impacts at the watershed scale. Specifically, we (1) calibrated and validated the SWAT-Carbon model for simulating streamflow, sediment yield, and nitrate pollution, as well as assessed its credibility for simulating eddy covariance tower observed carbon fluxes in the TW, (2) designed and assessed a total of seven modeling scenarios, including the baseline and six cover crop scenarios, (3) analyzed the impacts of growing cover crops on streamflow, sediment yield, nitrate loading, and SOC sequestration with different choices of crop species and planting dates. The results presented here demonstrate the environmental impacts of cover crops and provide useful information for decision- and policy-makers to promote sustainable adoption of cover crops into future watershed management plans.

2. Materials and methods

2.1. Study area description

The Chesapeake Bay (CB) is the largest estuary in North America and its nearly 167,000 km^2 watershed is home to over 18 million people and is an extremely productive and complex ecosystem. The health of streams and estuaries in the CB Watershed (CBW) is experiencing multiple coupled human and natural stresses, and has been declining for several decades. Excessive amounts of nutrient and sediment pollution are the three major contributors to the poor health of the streams, rivers, and the Chesapeake Bay (Chesapeake Bay Foundation, 2016; Clune et al., 2021; Kleinman et al., 2019). The water quality challenge in the CBW raised wide concerns and led to a Presidential Executive Order that tasked federal agencies "to contribute significantly to improving the health of the Chesapeake Bay" (EPA, 2009). Off-site movement of nitrogen and phosphorus into surface or groundwater negatively impacts water quality (e.g., eutrophication, hypoxia) (Love and Nejadhashemi, 2011; Robertson et al., 2011; Welch et al., 2010). Approximately one-third of the CBW is dedicated to farming. Not surprisingly, agriculture is a major cause of the degradation of CBW's health and its living resources. It is the largest single source of pollution, and contributes ca. 40% of the nitrogen and 60% of the sediment entering the Bay (CBP, 2022). Deployment of agricultural conservation practices in the CBW was estimated to have reduced a significant amount of N and P (USDA-Natural Resources and Conservation Service, 2013). Despite numerous regulation and conservation efforts, the water quality of the CBW is still in poor condition (CBF, 2021), threatening the economic viability and environmental and biological integrity of this region.

The land and waters of the CBW provide economic benefits totaling $107.2 billion (in 2013 $) annually. The Chesapeake Clean Water Blueprint (CBF, 2022), if fully implemented, would

increase the ecosystem services by $22.5 billion for a total of $129.7 billion annually (Surrick, 2014). Note that, the valuation of the natural benefits included air and water filtering, agricultural and seafood production, enhancement of property values, and protection from floods and hurricanes, but did not account for carbon sequestration. The Scientific and Technical Advisory Committee advised CBP to identify climate/carbon sequestration as an important co-benefit of nutrient and sediment and pollutant best management practices (BMPs) (e.g., winter cover crops or WCC) that should be quantified and considered in planning future ecosystem restoration (McGee et al., 2017). In 2017, the establishment of a new Maryland Healthy Soils Program (MHSP) was legislated (Maryland State Government, 2017), which charged Maryland Department of Agriculture (MDA) to improve soil health on Maryland farms. The MHSP is expected to increase biological activity and carbon sequestration by promoting practices based on emerging soil science, as well as help curb the state's greenhouse gases (GHG) emissions, lessen drought damage, reduce flooding and erosion and limit nutrient runoff.

In 2009, Maryland legislated the Greenhouse Gas Emissions Reduction Act (GGRA). The 2030 GGRA plan aims at a total of 52.7 TgCO2-eq (or 14.4 TgC) reduction in GHG emissions in 2023, as compared to 2006 (Maryland Department of the Environment, 2022). This is a comprehensive, multi-sector, multi-agency plan, and sets forth a comprehensive set of measures to reduce and sequester GHGs. Notably, cropland management practices are also identified as potential measures for reducing GHG emissions. More recently, the Maryland Healthy Soils Program (MHSP) was legislated (Maryland State Government, 2017) to improve soil health on Maryland farms by increasing soil organic matter (SOM), improving water and nutrient holding capacity, sequestering C and reducing GHG emissions. The state of Maryland has 1.6 M acres (or 0.68 M ha) of cropland that grows corn, soybean, and wheat

(https://www.nass.usda.gov/Quick_Stats/Ag_Overview/stateOverview.php?state=MARYLAND)

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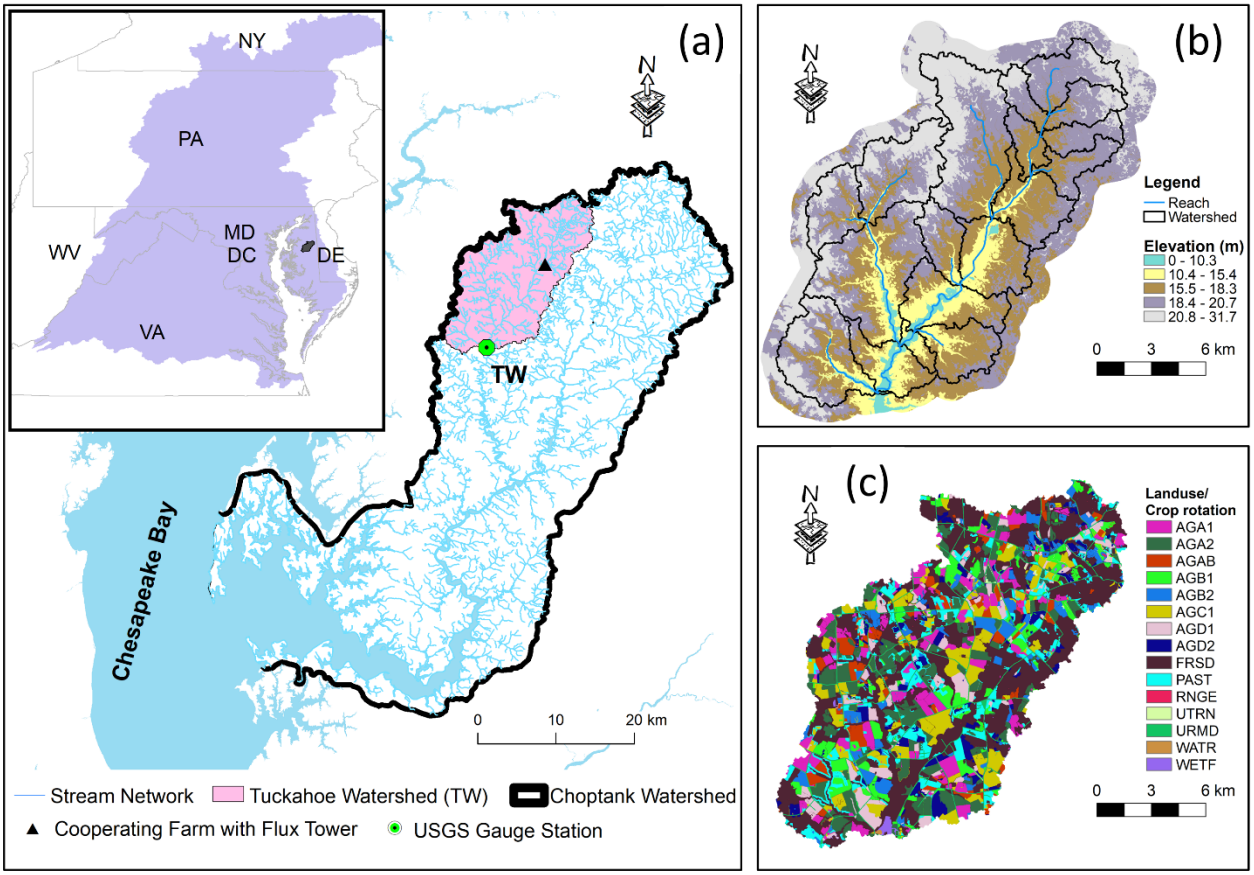


Fig. 1. Location of the study area. (a) location of the TW within the CBW in the Northeastern U.S. (b) the delineated subbasins and river networks based on the digital elevation model. (c) the major land use and crop rotations. The crop rotation is defined in Table 1.

The Tuckahoe Watershed (TW, Fig. 1), located on the Delmarva Peninsula, is part of the Choptank River Watershed (CRW) and a major tributary of Chesapeake Bay. The TW comprises of 54% of cropland and has been identified as an “impaired” water body by the Environmental Protection Agency (EPA) under Section 303(d) of the Clean Water Act due to excessive nutrient and sediment from agricultural land (McCarty et al., 2008). Federal and state conservation

programs are supporting farmers in the TW to adopt conservation practices. Including cover crops in annual cropping systems has the potential to provide year-round vegetation cover and reduce soil erosion and nutrient leaching (Ha and Wu, 2017; Krueger et al., 2012; Pantoja et al., 2016). The use of WCCs in Maryland rose from 27% of corn and 9% of soybean fields in 2008 to 89% of corn fields and 46% of soybean fields in 2016. The benefits of WCCs for reducing nitrate leaching in the Chesapeake Bay watershed have been demonstrated in multiple previous modeling studies (Hively et al., 2020; Lee et al., 2016; Yeo et al., 2014). However, the potential impacts of cover crops on sediment reduction and SOC sequestration remain understudied at the regional scale.

The TW is part of the USDA-Agricultural Research Service's Long-Term Agricultural Research network (USDA-ARS; Baffaut et al., 2020) and has been intensively monitored and studied (McCarty et al., 2008). Major land use types in TW include cropland (54%), forest (33%), pasture (8%) and Urban (4%). Approximately 1% of the watershed is covered by water. The National Agricultural Imagery Program Digital Orthophoto Quad Imagery (1:12,000) was used to delineate agricultural field boundaries and major crop rotations (Fig. 1 and Table 1) were derived from the crop data layer maps and mainly comprises corn, soybean and winter wheat (Johnson and Mueller, 2010; Lee et al., 2016). Soil map and properties are derived from the Soil Survey Geographic Database (Soil Survey Staff, 2024). The majority of the soils are moderately well or poorly drained. The watershed configuration (e.g., subbasin and river network delineation) was derived from a processed Light Detection and Ranging (LiDAR)-derived Digital Elevation Model (Lee et al., 2016). The TW has a warm maritime climate with long-term average annual precipitation and temperature of 1100 mm and 13 °C, respectively. Daily precipitation and temperature were obtained from two National Oceanic Atmospheric

Administration (NOAA) National Climate Data Center (NCDC) stations (i.e., USC00181750 and USC00187806) as the climate inputs to the SWAT-Carbon model.

Table 1. Major crop rotations in the Tuckahoe Watershed.

Rotation	Number of years in rotation	Crop sequence	Proportion (%)
AGA1	2	Corn - Dbl WW/Soyb	14.5
AGA2	2	Dbl WW/Soyb - Corn	21.9
AGAB	4	Corn - Soyb - Corn - Dbl WW/Soyb	7.7
AGB1	2	Corn - Soyb	11.3
AGB2	2	Soyb - Corn	9.8
AGC1	1	continuous corn	17.1
AGD1	3	Soyb - Corn - Soyb	10.2
AGD2	3	Soyb - Soyb - Corn	7.5

2.2. The SWAT-Carbon model

The Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998) is a widely used watershed model for assessing the effects of land use on water quality (Gassman et al., 2007; Wellen et al., 2015). The SWAT model delineates a watershed into subbasins that are linked by river networks. For each subbasin, the model combines soil and land use layers to derive a homogeneous modeling unit – hydrologic response unit (HRU). At the HRU scale, the model simulates coupled agronomic, hydrologic, and biogeochemical processes; the water, sediment and nutrients leaving a HRU are routed through the river networks to calculate the export of different water quality parameters at the outlet of a watershed. Since 2013, numerous efforts have been conducted to enable the SWAT model to represent the soil organic carbon (SOC) cycling to form the SWAT-Carbon model. Zhang et al. (2013) incorporated the SOC algorithms from Daily CENTURY (DayCENT) and Environmental Policy Integrated Climate (EPIC) into SWAT-Carbon, and demonstrated that the model can capture well daily Net Ecosystem

Exchange of multiple cropping systems, including rainfed and irrigated continuous corn and corn soybean rotations. Later on, the SOC cycling processes were further improved by accounting for factors such as soil erosion, dissolved organic carbon leaching, and tillage perturbation (Du et al., 2019; Liang et al., 2022; Zhang, 2018). Liang et al. (2022) showed that the SWAT-Carbon model can satisfactorily reproduce long-term SOC dynamics across 47 cropping system treatments over the U.S. Midwest. In addition, the SWAT-Carbon model has been successfully tested for simulating hydrologic, C and nitrogen processes in multiple watersheds (Dangol et al., 2023; Liang et al., 2023a; Liang et al., 2023b; Mukundan et al., 2023; Qi et al., 2020a; Qi et al., 2020b). It is also worth noting that the energy balance algorithms within SWAT-Carbon are based on physical processes, which further enhance the model performance for simulating hydrologic and water quality processes as compared with the empirical approaches used in the SWAT model (Qi et al., 2021; Qi et al., 2019; Wang et al., 2020; Wang et al., 2021). Based on the previous results, the SWAT-Carbon model represents a state-of-the-art watershed model for assessing water quality and SOC sequestration impacts of diverse cropping systems.

2.3. Cover crop scenarios

Although the benefits of cover crops for SOC sequestration and water quality have been widely studied, it is difficult to generalize the results from fields or modeling studies from other regions to the current study. The performance of cover crops depends on numerous factors, such as soil properties, crop species, planting and termination dates, fertilization and tillage practices (Blanco‐Canqui, 2018; Mazzoncini et al., 2011). Here, we designed six cover crop scenarios that include three winter cover crops (i.e., wheat (*Triticum aestivum* L.), winter barley (*Hordeum vulgare* L.), and rye (*Secale cereale* L.)) and two planting dates (i.e., early (Oct. 1st) and late (Nov. 2nd)) based on the guideline of the Maryland Agriculture Cost Share (MACS) cover crop

program and agricultural statistics (Lee et al., 2016): Barley Early (BE), Barley Late (BL), Rye Early (RE), Rye Late (RL), Wheat Early (WE), and Wheat Late (WL) The practices for summer crops were set to be the same in baseline and WCC scenarios. General crop management practices for the baseline and WCC scenarios are provided in Table S1.

2.4. Model evaluation

Before applying the SWAT-Carbon model, we calibrated and validated the model performance for simulating streamflow and water quality indicators of concern. We compiled observed daily streamflow and sediment concentration data from the USGS Ruthsburg station (USGS#01491500), which is located at the outlet of the TW. The daily streamflow data of the calibration (2014-2015) and validation (2016-2017) periods were aggregated to the monthly time scale. There were a total of 148 sediment concentrations over 2014-2017. We used the Load Estimator (LOADEST) computer program to estimate monthly sediment loads by pairing concurrent streamflow and sediment concentration data and deriving regression relationships (Runkel et al., 2004). We evaluated nine regression models within the LOADEST program and eventually chose the following one:

$$a_0 + a_1 \ln Q + a_2 \sin(2\pi dtime) + a_3 \cos(2\pi dtime) + a_4 dtime$$

where $\ln Q$ is the difference between $\ln streamflow$ at a data point and $center$ of $\ln(streamflow)$; $dtime$ is the difference between $decimal$ time and center of decimal time; a_0, a_2, a_3, and $a_4$are model regression coefficients. This model explains approximately 87% of the observed sediment concentrations.

For nitrate, we used the photo-spectrometer observed half-hourly riverine nitrate concentrations, which were multiplied by streamflow to estimate monthly nitrate loads. We

assessed the performance of the SWAT-Carbon model performance using widely recognized evaluation metrics such as Nash-Sutcliffe Efficiency (NSE) (Nash and Sutcliffe, 1970) and the percent bias (PBIAS) (Gupta et al., 1999). During model calibration, we used NSE as the objective function and further evaluated the model performance with PBIAS. The model calibration and validation was performed using the R-SWAT (Nguyen et al., 2022). R-SWAT is an open-source, user-friendly software package designed for calibrating different versions of the SWAT model. In this study, we calibrated our model using the SUFI-2 method (Abbaspour et al., 2004). In this approach, we generated 2000 random parameter sets with given parameter ranges listed in Table 2 and the best parameter set was selected based on the NSE.

Table 2. Parameters calibrated to match observed flow, sediment, and nitrate loads.

Parameters	Brief definition	Adjustment method	Lower bound	Upper bound	Calibrated value
ALPHA_BF.gw	Baseflow recession factor	replace	0.01	1	0.905
ALPHA_BNK.rte	Baseflow alpha factor for bank storage	replace	0.01	1	0.472
CH_K2.rte	Effective hydraulic conductivity in main channel alluvium	replace	0.1	150	91.879
CH_N2.rte	Manning's "n" value for the main channel.	replace	0.01	0.3	0.13
CN2.mgt	SCS runoff curve number	relative	-0.25	0.25	-0.153
EPCO.hru	Plant uptake compensation factor	replace	0	1	0.068
ESCO.hru	Soil evaporation compensation factor	replace	0.01	1	0.979
GW_DELAY.gw	Groundwater delay (days)	replace	10	120	83.013
GW_REVAP.gw	Groundwater "revap" coefficient	replace	0.02	0.2	0.064
LAT_TTIME.hru	Lateral flow travel time (days)	replace	0.01	24	19.757
SFTMP.bsn	Snowfall temperature.	replace	0	3	2.457
SMTMP.bsn	Snowmelt base temperature.	replace	-2	2	-0.119
SOL_AWC.sol	Available water capacity of the soil layer.	relative	-0.25	0.25	-0.163
SOL_K.sol	Soil erodibility	relative	-0.25	0.25	-0.101
SURLAG.hru	Surface runoff lag time	replace	0.5	24	12.467
PRF_BSN.bsn	Peak rate adjustment factor for sediment routing in the main channel	replace	0.5	1.5	1.438
SOL_BD.sol	Soil bulk density	relative	-0.2	0.2	-0.004
SLSUBBSN.hru	Average slope length	relative	-0.5	0.5	-0.039
USLE_K.sol	USLE equation soil erodibility (K) factor	relative	-0.5	1	-0.293
USLE_P.mgt	USLE equation support practice factor	replace	0	1	0.799
N_UPDIS.bsn	Nitrogen uptake distribution parameter	replace	0	30	7.706
NPERCO.bsn	Nitrogen percolation coefficient	replace	0.01	1	0.427
SHALLST_N.gw	Concentration of nitrate in groundwater contribution to streamflow from subbasin (mg N/l)	relative	-1	1	-0.362

CDN.bsn	Denitrification exponential rate coefficient.	replace	0	3	1.394
SDNCO.bsn	Denitrification threshold water content	replace	0	1.1	0.958
HLIFE_NGW.gw	Half-life of nitrate in the shallow aquifer (days)	replace	0	200	109.207
ERORGN.hru	Organic N enrichment ratio	replace	0	5	4.663
HRU_SLP.hru	Average slope steepness	relative	-0.5	0.5	0.328
CH_S1.sub	Average slope of tributary channels	relative	-0.5	0.5	-0.403
CH_N1.sub	Manning's "n" value for the tributary channels	relative	-0.5	0.5	0.208
CH_S2.rte	Average slope of main channel	relative	-0.5	0.5	0.346
OV_N.hru	Manning's "n" value for overland flow	relative	-0.5	0.5	0.22

Note: * “replace” the calibrated parameter values were used to replace the default inputs. # “relative” the default values are adjusted by a fraction (i.e., the calibrated parameter values).

3. Results

3.1. Model calibration and validation

Table 3. Performance of the SWAT-Carbon model for simulating flow, sediment and nitrate loads for calibration and validation periods, respectively.

		NSE	PBIAS
Calibration	Flow	0.60	12.1%
	Sediment	0.60	17.3%
	Nitrate	0.57	8.8%
Validation	Flow	0.63	18.9%
	Sediment	0.57	26.2%
	Nitrate	0.70	7.6%

We calibrated parameters (Table 2) relevant to the hydrologic cycle, sediment generation and nitrogen cycle to match the model simulated water quantity and quality variables with observations (Fig. 2). For streamflow, sediment, and nitrate, NSE was greater than 0.5 for both calibration and validation periods (Fig. 2 and Table 3). In addition, PBIAS was less than 20% for all three variables during calibration and validation, except for sediment during validation (26.2%). In general, these statistical metrics illustrate the satisfactory performance of the calibrated model (Moriasi et al., 2007).

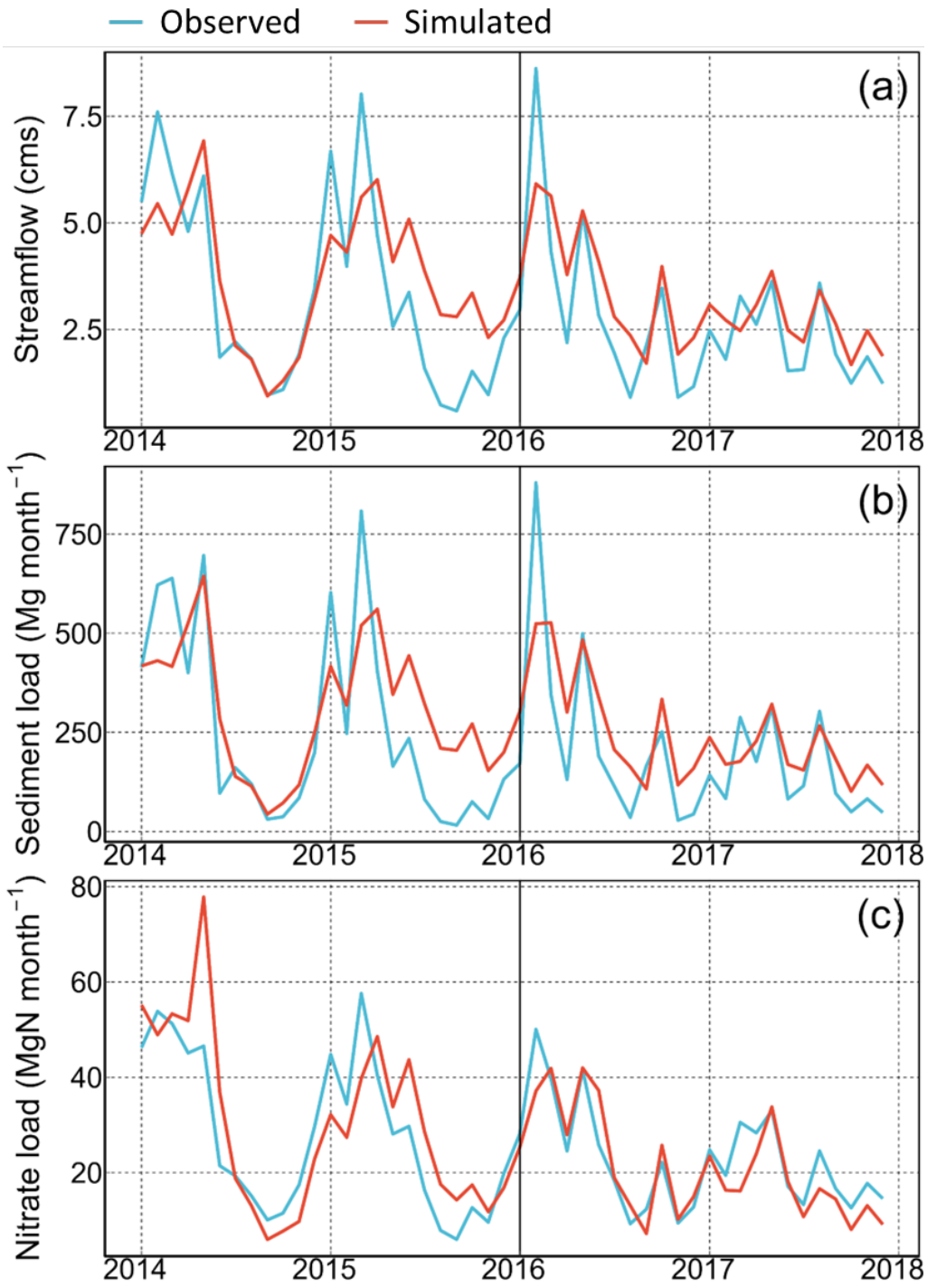


Fig. 2. Simulated and observed monthly streamflow, sediment, and nitrate loads for calibration (2014-2015) and validation (2016-2017) periods, respectively, in the Tuckahoe Watershed.

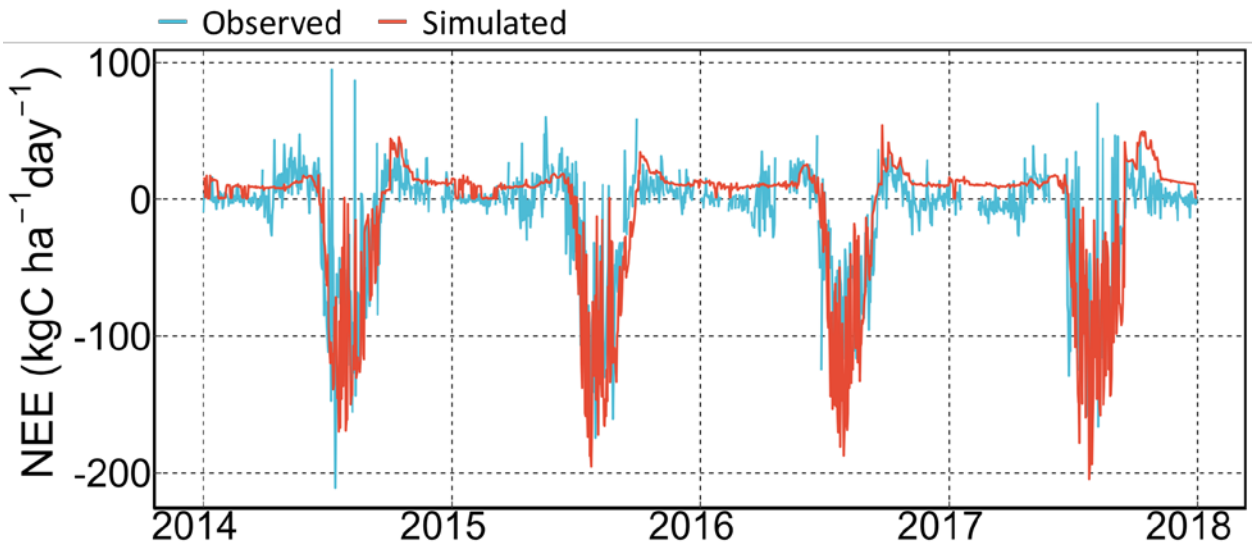


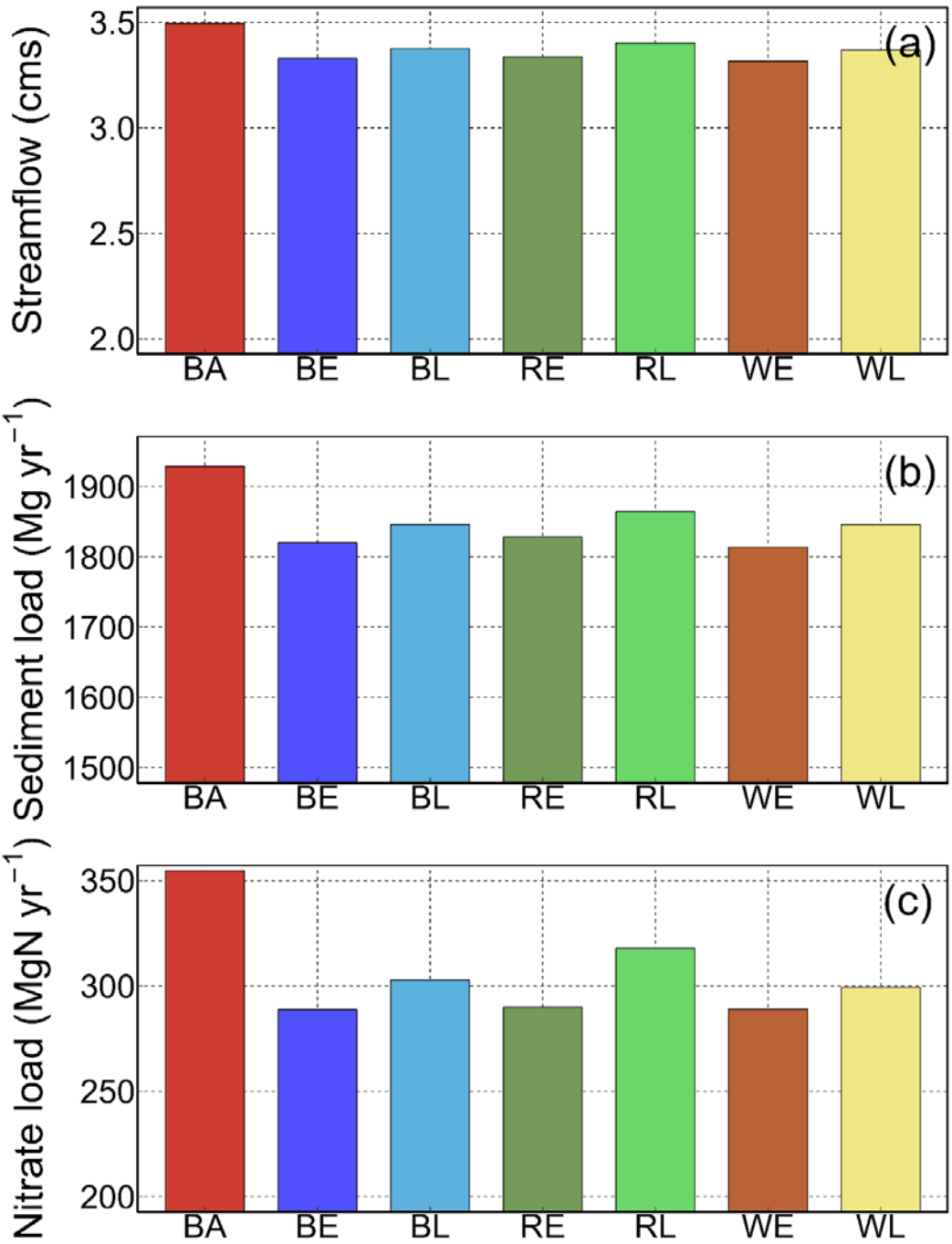
Figure 3. Simulated and flux tower observed NEE from a corn field in the TW. The location of the cooperating farm is shown in Fig. 1.

After calibrating the SWAT-Carbon model for simulating flow, sediment and nitrate, we also evaluated the model for simulating Net Ecosystem Exchange (NEE) observed by a flux tower at a corn field within the TW (Fig. 3). With the carbon cycling parameters determined from previous studies (Liang et al., 2022), the SWAT-Carbon model simulated average daily NEE was -13.4 kgC ha^{-1}, slightly lower than the observed daily average of -12.4 kgC ha^{-1}. The correlation between the time series of observed and simulated daily NEE is 0.73. In general, the evaluation results indicate that the SWAT-Carbon model reliably captured the magnitude and temporal variability of carbon fluxes from the corn field.

3.2. Water quality benefits of growing cover crops

Based on the SWAT-Carbon modeled scenarios, WCCs exhibited a notable capacity to reduce streamflow (Fig. 4a). For the baseline scenario, the annual streamflow rate was around 3.5 cubic meters per second (cms), while the simulated streamflow under different cover crop scenarios was between 3.32-3.40 cms. This represents a reduction in streamflow by 2-5%. Note

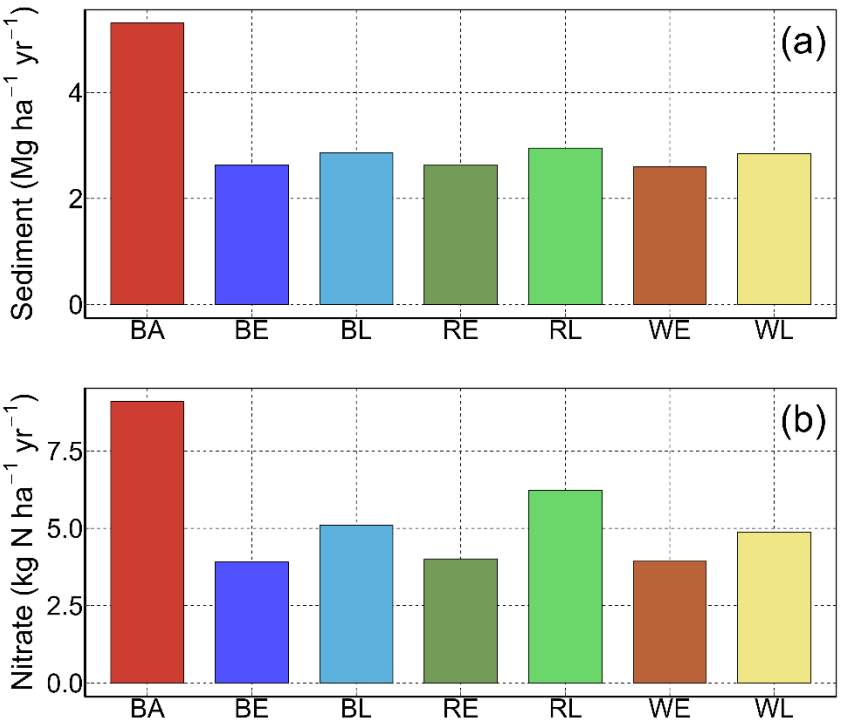
that early-planted crops demonstrated a more pronounced capacity to cause reductions in streamflow in comparison to their late-planted counterparts, and this effect remains consistent across various WCC types.



306 Figure 4. Annual average simulated (a) streamflow, (b) sediment, and (c) nitrate loads at the outlet
307 subbasin in the TW. Barley Early (BE), Barley Late (BL), Rye Early (RE), Rye Late (RL), Wheat
308 Early (WE), and Wheat Late (WL)
309

310 In addition, growing WCCs could cut back on sediment yield and nitrate loading as
311 compared with the baseline scenario. In general, adoption of WCCs could reduce approximately
312 3-6% sediment and 10-19% nitrate loads (Fig. 4b and 4c). Like streamflow, we also observed
313 that early planting of cover crops has a greater efficacy in reducing both sediment and nitrate.

314 The adoption of WCCs can reduce nitrate and sediment transported from agricultural fields
315 to river networks (Fig. 5). The net effects on reducing nutrient and sediment load at the
316 watershed outlet is also influenced by the share of cropland and riverine routing through the
317 watershed. Therefore the watershed nutrient and sediment load reduction is often not equal to the
318 reduction of nutrient and sediment yield from agricultural fields. Here, we examined the effects
319 of different WCCs at the field scale to better understand their role in reducing terrestrial load to
320 the aquatic environment. Also, as most field experiments regarding the environmental impacts of
321 cover crops have been conducted at the plot or field scale, such analysis facilitates further
322 comparison between our modeling results and those reported in previous field studies. The six
323 WCCs scenarios reduced nitrate leaching by 38-71% (Fig. 5b). For example, early-planted Rye
324 cut back on nitrate leaching by 64%, which is much higher than the 38% reduction by late-
325 planted Rye. Planting WCCs early-provides opportunities for the cover to develop and uptake
326 soil nitrate, thereby resulting in much more substantial environmental benefits than their late-
327 planned counterparts.

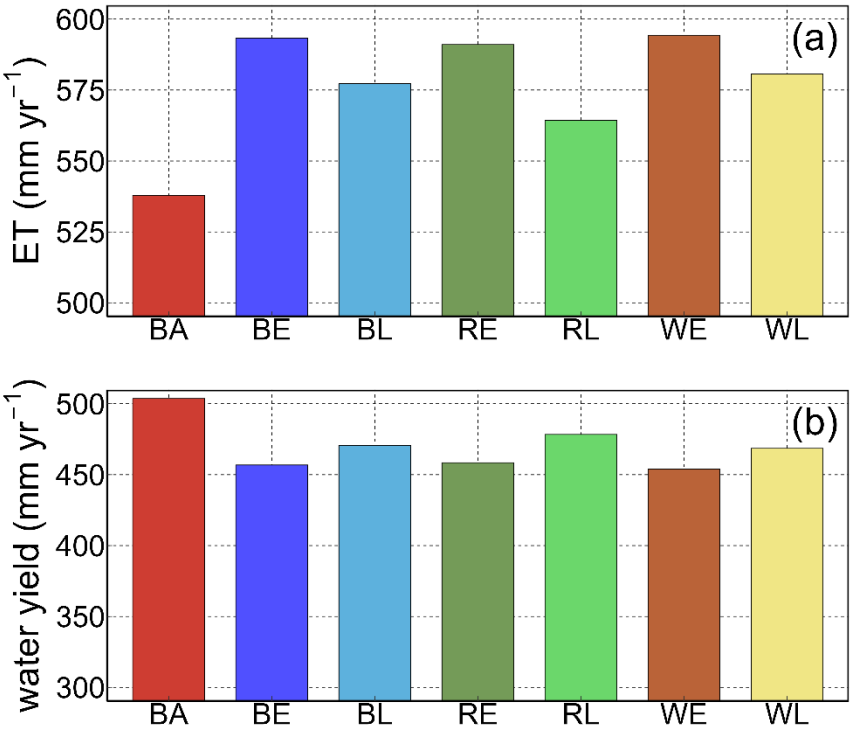


328

329 Figure 5. Average simulated annual (a) sediment, and (b) nitrate yield from cropland in the TW.
330 Barley Early (BE), Barley Late (BL), Rye Early (RE), Rye Late (RL), Wheat Early (WE), and
331 Wheat Late (WL)

332

3333.3. Enhanced ET and reduced water yield by cover crops



334

335 Figure 6. Average simulated annual (a) ET, (b) water yield on croplands in the Tuckahoe
336 Watershed.
337

338 In the TW, ET increased 5-10% across different WCC scenarios (Fig. 6a). For early-planted
339 cover crops, ET increases were around 10%, while for late-planted cover crops ET increases
340 ranged from 5-8%. As a result of the increased ET, the model simulation results showed that
341 water yield from the agricultural fields decreased 5-10% (Fig. 6b), with late-planted cover crops
342 exhibiting greater reduction than early-planted cover crops. Such a modest reduction in water
343 availability may not have profound impacts on the water use by summer crops in the humid
344 study region, but could threaten crop yields in dry regions where water use is a limiting factor for
345 crop growth.

3.4. Soil carbon sequestration benefits of WCCs

The modeling results show that WCCs had a discernible impact on the accumulation of SOC stocks on croplands (Fig. 7). As compared with the baseline, the presence of WCCs led to a modest elevation in SOC stocks, typically ranging from 1.8 to 3.7 additional Mg C ha^{-1} over the 4-year simulation period. Notably, among the spectrum of scenarios considered, those involving early-planted cover crops exhibit a greater propensity for sequestering organic carbon in the soil when compared with their late-planted counterparts. That is, early-planted WCCs accrued 3.1-3.7 Mg C ha^{-1}, which compares to 1.8-2.1 MgC ha^{-1} by late-planted WCCs.

Substantial disparities in SOC storage are evident among various crop rotation types, with values spanning from approximately 90 to in excess of 160 Mg ha^{-1} yr^{-1}. Across all crop rotation types considered, the baseline scenario consistently exhibits the lowest levels of SOC stocks, while early-planted winter wheat consistently attains the highest levels of SOC content.

In our study, early-planted WCCs in general produced more biomass and sequestered more SOC than late-planted WCCs by a factor of ca. 1.72-1.76 or over 70%. Although variations in crop species can also have an impact on cover crop biomass and SOC sequestration, the impact was much smaller. For example, the range of SOC sequestration of the three WCCs species (i.e., Barley, Rye, and Winter Wheat) was 0.78-0.92 MgC ha^{-1} yr^{-1} for early planting and 0.45-0.53 MgC ha^{-1} yr^{-1} for late planting.

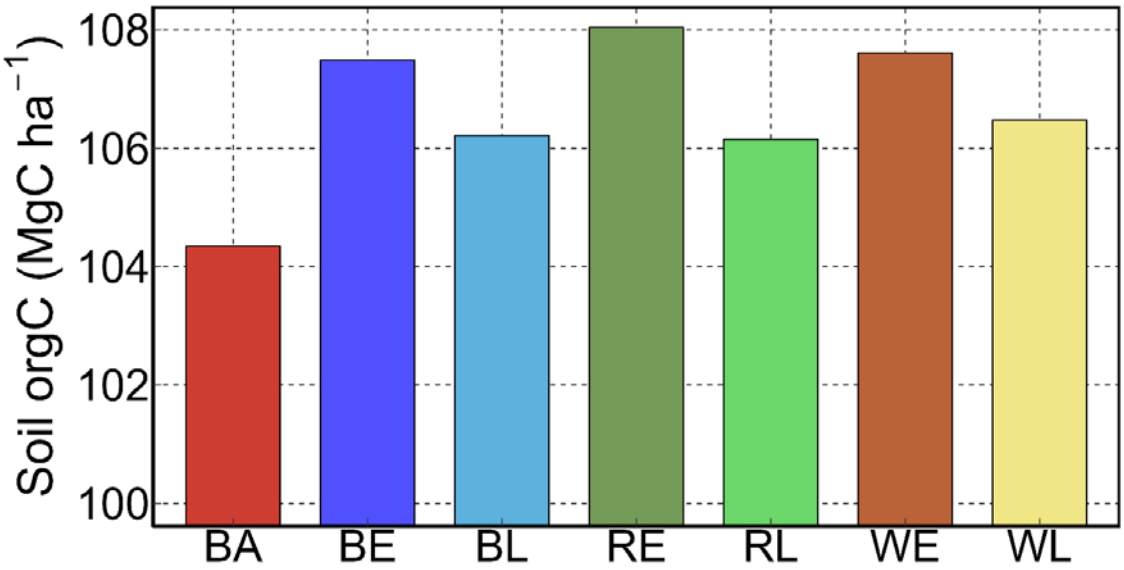


Figure 7. Soil organic carbon on croplands at the end of the 4-year simulation period. Barley Early (BE), Barley Late (BL), Rye Early (RE), Rye Late (RL), Wheat Early (WE), and Wheat Late (WL)

4. Discussion

4.1. Potential contribution of cover crops to multi-dimensional environmental benefits

Our society relies on agricultural conservation to achieve multiple environmental benefits, such as the USDA's Agriculture Innovation Agenda which aims at net reduction of the agricultural sector's carbon footprint and reducing nutrient loss by 30%. In the state of Maryland, multiple legislations have been passed to cut back on GHG emissions and improve environmental quality, emphasizing the importance of simultaneously addressing different sustainability goals to avoid inadvertent consequences.

Since 2010, the Chesapeake Bay Total Maximum Daily Load (TMDL) uses Watershed Implementation Plans (WIPs) to drive nutrient and sediment reductions because clean water is

the foundation for healthy fisheries, habitats and communities across the region (Chesapeake Bay Program, 2020). Relative to the 2017 nitrogen load (54.2 million pounds per year), Maryland's Phase III WIP (2019-2025) aims to further reduce nitrogen load to 44.8 million pounds, a reduction of ca. 17.4%. The nitrate load reduction by WCCs range between 10-19% in the TW, indicating a potentially significant role of WCCs to contribute to Maryland's watershed nutrient reduction goals. Such contributions of WCCs depend on many factors, such as the WCC species, planting and termination dates, and other management practices, therefore we cannot directly extrapolate the results to other watersheds in Maryland or other regions. It is worth noting that our simulation results are in general consistent with previous watershed modeling and experimental results. For example, the watershed modeling results by Getahun and Keefer (2023) showed that nitrate loading could be reduced up to 20.9–24.3% in the Big Ditch watershed in Illinois, USA. Speir et al. (2022) conducted riverine nitrogen measurements in two agricultural watersheds in Indiana, USA, and found that the watershed loading of nitrate was reduced by 2%-67% at five out of a total of eight sites in Spring via the use of cover crops. As such, achieving a reduction of watershed nitrate loading of 20% or less is plausible with growing WCCs.

Our modeling results show a reduction of approximately 3-6% in watershed sediment load as a result of adoption of different WCCs, which is less effective than nitrate. Sediment is mainly transported by surface runoff from agricultural fields, while nitrate loads can be substantially reduced through cover crop uptake. The differences in transportation pathways of nitrate and sediment partially explains the varied effectiveness of cover crops. Maryland's Phase III WIP sets a goal of reducing 4% of sediment loads by 2025, which is comparable to the model simulated reduction from growing WCCs in the TW. This indicates that WCCs could be an

effective tool for farmers and water managers to achieve the goal of reducing sediment load to the Chesapeake Bay.

By scaling the estimated SOC sequestration benefits of 0.45-0.92 MgC ha^{-1} yr^{-1} estimated for the TW cropland (see section 3.4) to the Maryland framing land, the potential SOC sequestration across the state would be 0.3-0.63 TgC yr^{-1}, approximately 2.1-4.4% of the 2030 GGRA plan's GHG reduction goal. The above estimate assumes all corn, soybean, and wheat land in Maryland will adopt cover crops, which is costly to achieve, therefore should be taken as the upper limit of the GHG reduction from WCCs. This result indicates that adoption of cover crops can provide beneficial contributions, but could not serve as a major technique. Techniques in the electricity, transportation, and building energy use sectors are likely the major contributors to Maryland's GHG reduction goals.

The water quality and SOC sequestration benefits of WCCs have potential to simultaneously contribute to the sustainability goals of USDA and the state of Maryland. Notably, WCCs are also identified as a potential agricultural practice by other states and regions, such as the Chesapeake Bay Program that is coordinating the ecosystem conservation and restoration in Delaware, New York, West Virginia, District of Columbia, Maryland, Pennsylvania, and Virginia. The results of this study demonstrate the potential of using cover crops to meet environmental sustainability goals in the jurisdictions within the Chesapeake Bay Watershed, particularly for nutrient load mitigation.

4.2. Field scale nitrate and sediment reduction by cover crops

Cover crops can uptake nutrients that might otherwise be lost through leaching or runoff during the non-growing season (Adetunji et al., 2020; McDaniel et al., 2014). The cultivation of cover crops contributes to soil N and enhances N cycling, and can reduce potential N loss. This is achieved through several processes, such as atmospheric N fixation by leguminous cover crops and the assimilation and recycling of residual soil N at the end of the growing season (Tribouillois et al., 2015). These actions significantly reduce nitrate leaching into groundwater and prevent the downward movement of N beyond the root zone. For instance, a 15-year study in Kansas demonstrated that hairy vetch increased soil N by 258 kg ha^{-1} under soybean and 279 kg ha^{-1} under sun hemp (Blanco-Canqui et al., 2011). Furthermore, in no-till corn-soybean rotations, rye and hairy vetch were found to enhance soil aggregate stability, total porosity, and plant-available water (Villamil et al., 2006). Meta-analyses have shown that cover crops can reduce nitrate leaching by up to 94% (Kaspar et al., 2012a). Non-leguminous cover crops, in particular, are effective at scavenging excess nutrients from the soil, thereby mitigating the risk of water pollution from inorganic fertilizers or animal waste (Blanco-Canqui et al., 2015). Note that the effectiveness of cover crops varies by species. In a 5-year experiment in Iowa, rye significantly reduced nitrate concentration in subsurface drainage by 48%, while oat only achieved a 26% reduction (Kaspar et al. (2012b).

Our simulation results are in general consistent with previous modeling studies. For example, Yeo et al. (2014) reported that WCCs could reduce annual nitrate yield from agricultural land from 14 kg ha^{-1} yr^{-1} to 4.6-10.1 kg ha^{-1} yr^{-1}, or a reduction of 28-67%. Both Lee et al. (2016) and Hively et al. (2020) reported the nitrate reduction benefits of different WCCs varied widely from 45-79%, and showed that early-planted WCCs provide more nitrate leaching reduction benefits. Reported field results also showed a wide range of nitrate reduction by cover crops between 18

449 and 95% (Blanco‐Canqui, 2018). For example, Griffith et al. (2020) reported that their field
450 study in northern New York, USA which showed negligible reduction by a Rye cover crop in
451 nitrate leaching from tile drainage. In contrast, Speir et al. (2022) reported that cover crops
452 reduced nitrate from tile drainage from 27 to 72% in two agricultural watersheds in northern
453 Indiana, USA.

454 Growing WCCs provide biomass to cover fallow soils, thereby reducing kinetic energy of
455 precipitation and protecting soils from erosion (Dabney et al. 2001). Previous studies reported a
456 wide range of field scale sediment loss reduction benefits of cover crops, from 22 to100%
457 (Blanco‐Canqui, 2018). Our results indicate that the six WCCs scenarios can reduce sediment
458 loss by 45-52% (Fig. 5a), with early-planted WCCs providing more reduction. In general, the
459 large variability of the field experimental results indicate that they are likely applicable to the
460 specific climate, terrain, soils, and management practices of a specific site or region. As such,
461 although our simulation results agree with previous modeling studies and fall within the range
462 reported in field studies, the large variability and uncertainties call for more field experiments
463 and coordinate modeling efforts to further ensure the fidelity of large-scale assessment of the
464 WCCs' nitrate reduction benefits.

465 **4.3. SOC sequestration by WCCs**

466 Field experimental data showed that the SOC sequestration benefits of adoption of cover
467 crops could vary widely (Eagle et al., 2012; Poeplau and Don, 2014). For example, a field
468 experimental study in Santa Fe, Argentina showed higher benefits of cover crops for increasing
469 SOC by 1.15-4.92 MgC ha^{-1} (or 0.2-0.82 MgC ha^{-1} yr^{-1}) over 2007-2012 (Duval et al., 2016).
470 Higashi et al. (2014) showed that cover crops could sequester 0.36-0.89 MgC ha^{-1} yr^{-1} from

2003-2011 in the Japanese province of Kanto. Blanco‐Canqui et al. (2015) reviewed studies in temperate soils and found 0.1–1 Mg ha^{-1} yr^{-1} SOC sequestration by cover crops. In general, our results indicate that the SOC sequestration benefits (0.45-0.92 MgC ha^{-1} yr^{-1} in the TW) fall within the range of previously reported data.

Note that some studies showed minimal improvements in soil physical properties and SOC content after adoption of cover crops (Steele et al., 2012). Chaplot and Smith (2023) examined six experiments and found that most (four) of them showed negligible impact of growing cover crops on SOC, one displayed decreased SOC, and one showed modest positive impact (0.03 MgC ha^{-1} yr^{-1}). Chaplot and Smith (2023) pointed out that there are fundamental issues with the existing field study of cover crops impacts, such as not comparing based on equivalent soil mass. Despite potential uncertainties associated with the effectiveness of cover crops for SOC sequestration, McClelland et al. (2021) highlighted the strong positive impacts of cover crops on SOC accruement based on the meta-analysis of 181 observations across the world, with an average 1.11 MgC ha^{-1} more SOC compared with no cover crop control. Given that most studies showed positive impacts of cover crops on SOC, we cautiously contend that adoption of WCCs in the TW could serve as a useful tool for GHG emissions reduction. Meanwhile, we acknowledge that more robustly designed field experiments are needed to further demonstrate the SOC sequestration benefits of cover crops.

The meta-analysis by McClelland et al. (2021) and Joshi et al. (2023) showed that the SOC sequestration of cover crops is influenced by planting and termination date (which determines the growing season length of cover crops), cover crop biomass production, soil clay fraction, and initial SOC. We also found that soil clay fraction and SOC sequestration is positively correlated, which is in line with findings by McClelland et al. (2021). However, the R-square between soil

clay fraction and SOC sequestration is not significant at the significance level of 0.05, which is largely explained by the lack of strong correlation between soil clay fraction and initial SOC in the TW. In addition, the soil clay fraction is negatively correlated with percent of SOC accruement by cover crops, consistent with findings by Joshi et al. (2023). Our results also agree with Joshi et al. (2023) in terms of a significant negative correlation between initial SOC and percent of SOC accruement by cover crops. Collectively, our modeling results agree with previous meta-analysis in terms of how management and soil texture factors influence SOC sequestration benefits of cover crops. The planting-termination window and initial SOC are the most important predictors of the SOC sequestration by WCCs in the TW, which can be used to inform optimal management and optimal locations for SOC sequestration benefits.

Notably, the carbon photosynthesized and released into the soil through cover crops can also nourish soil microorganisms and improve overall soil health (Adetunji et al., 2020; McDaniel et al., 2014). Beyond nutrient management, cover crops can increase the diversity of plants and soil biota compared to continuous corn or corn-soybean rotations, making cropping systems more resilient to climate variations in the long term. A meta-analysis of 122 studies by McDaniel et al. (2014) revealed that increasing crop biodiversity through cover crops enhances soil microbial biomass, as well as soil C and N levels. These improvements in microbial biomass contribute significantly to soil nutrient cycling and physical processes like aggregate formation. The inclusion of cover crops increased soil carbon and N by 8.5% and 12.8%, respectively, compared to the 3.6% and 5.3% increases observed when adding one or more cash crops to a monoculture. Notably, the effectiveness of cover crops in building SOC is highly site-specific, depending on factors such as biomass quantity, duration before termination, initial soil carbon levels, soil type,

cover crop species, tillage management, and climate conditions (Blanco‐Canqui et al., 2015). Those factors should also be considered in the adoption of WCCs.

4.4. Water availability impacts of WCCs

Similar to the results reported in the previous studies, our modeling results indicate that water yield from cropland in the TW could reduce as a result of growing WCCs. This reduction in water yield is primarily attributable to the heightened annual evapotranspiration (ET) induced by the WCCs. Compared with the baseline scenario, growing cover crops can cause greater ET as those plants can extract more soil water through the transpiration (Bodner et al., 2007). The impact of streamflow reduction in lower flow periods deserve further attention, as low streamflow often has a significant role for aquatic ecosystem integrity (Qiu et al., 2021). In addition, in watersheds that contribute to downstream water supply reservoirs, the impacts of reduction in streamflow should be carefully considered (Getahun and Keefer, 2023). Collectively, although diminished water yield can benefit water quality issues by decreasing yield of sediment and nitrate from agricultural fields, the simulated modest reduction in water availability resulting from the adoption of WCCs should be concerned in regions where water demand by crop growth, aquatic ecosystem health, and water supply could be impacted under current or future climate.

4.5. Future directions

The simulation results are in general aligned with previous modeling studies that growing WCCs into existing crop rotations help significantly reduce watershed nitrate loading. The results also agree with field experiment results that cover crops can sequester more carbon into soils, reducing sediment and nutrient loss, and enhancing ET. The novelty of this study mainly

lies in the simultaneous quantification of the multi-dimensional impacts of growing diverse WCCs at the watershed scale. Field studies are often costly and require a long time, therefore cannot be efficiently applied across large scales for environmental impacts quantification. As such, watershed models are often used to assess the environmental benefits and costs of different conservation practices. Previous watershed modeling studies mainly focused on the role of WCCs in reducing nitrate loading, but did not examine SOC, sediment reduction and water availability implications. Our study provided a systematic assessment and analyzed the potential synergies and trade-offs of the multi-dimensional impacts of growing WCCs. Such information provides scientific support for decision makers to balance those impacts in designing future policies and regulations. In addition, this study demonstrated the usefulness of the SWAT-Carbon model for assessing not only the water quality implications of WCCs, but the contributions to GHG reduction goals, both of which are being considered by policy makers in the Chesapeake Bay and other watersheds.

While WCCs offer significant benefits for climate change adaptation, including reducing CO_2 emissions, mitigating soil erosion, and minimizing nutrient loss, their effectiveness is also influenced by the impacts of climate change. For instance, warming temperatures may create narrow transition windows between WCC termination and cash crop planting, which may hinder the successful establishment of cash crops (Kaye and Quemada, 2017). Kaye and Quemada (2017) suggested that warming temperatures may also lead to increased soil N mineralization rates that coincide with the WCC growth period. In such cases, CCs can serve as an adaptive management tool to reduce N leaching. Additionally, adjusting WCC termination dates and employing mulching techniques can help adapt to drought conditions by managing WCC water

560 use and reducing soil evaporation. WCCs that enhance soil infiltration and increase rooting depth
561 are also promising strategies for improving drought resilience.

562 Our study highlights the significant environmental benefits of winter WCCs, offering crucial
563 insights for policymakers in promoting the adoption of WCCs. By selecting appropriate WCC
564 species, farmers can enhance soil fertility while capturing atmospheric CO_2 and converting it into
565 organic matter. Similar to industrial approaches with simultaneous adsorption and catalytic
566 biodegradation of N and organic carbon in industrial wastewater treatment (Ahmad et al., 2017;
567 Ahmad et al., 2023b; Ahmad et al., 2019; Han et al., 2023), cover cropping systems offers a
568 promising biological and ecological approach to achieve multiple environmental benefits
569 simultaneously in agricultural soils. The feasibility of these approaches depends on careful
570 planning, species selection, and process optimization to maximize their effectiveness and
571 sustainability. For example, early planting of WCCs nearly doubles the environmental benefits
572 compared to late planting. In addition, policymakers must also consider the potential trade-offs,
573 such as the risk of reduced water availability for subsequent cash crops. To maximize the
574 effectiveness of WCCs, incentives should encourage early planting dates while carefully
575 balancing the environmental and economic interests of farmers. The introduction of WCCs is
576 also associated with direct and indirect costs—such as seeds, herbicides, and field operations
577 (Gabriel et al., 2013), improper management can lead to adverse effects on cash crops, including
578 competition for water and nutrients, increased risks of disease (Larkin and Honeycutt, 2006), or
579 even nutrient loss (Liang et al., 2019). Despite these challenges, WCCs can generate economic
580 benefits, such as revenue from selling WCC biomass as animal feed and savings on fertilizer
581 costs (Gabriel et al., 2013). Therefore, to promote the widespread adoption, policymakers and
582 agricultural stakeholders should prioritize both economic viability and environmental

sustainability by offering tailored incentives, such as cost-share subsidies through USDA's programs like the Environmental Quality Incentives Program (EQIP) and Conservation Stewardship Program (CSP). These programs should be flexible, allowing farmers to adapt cover-cropping strategies to their specific needs and conditions. Moreover, integrating WCCs into crop insurance policies without penalizing farmers, coupled with education and awareness initiatives like on-farm trials, can further enhance adoption rates. By aligning economic incentives with environmental goals, policymakers can successfully integrate WCCs into sustainable agricultural practices, benefiting both farmers and the environment (Bergtold et al., 2017).

5. Conclusions

WCCs are being promoted as a best management practice to reduce nitrogen and sediment inputs from land areas to the Chesapeake Bay in the jurisdictions located within the Chesapeake Bay Watershed. Meanwhile, the state of Maryland encourages adoption of WCCs for not only water quality benefits, but also GHG emissions reduction. Although previous modeling studies showed the potential benefits of growing cover crops to reduce nitrogen pollution, the potential impacts on sediment load and SOC sequestration remain largely unknown. In this study, we designed six WCC scenarios defined by three crop species and two planting-termination windows to understand the potential water quality and SOC sequestration impacts. With the open-source SWAT-Carbon model (https://sites.google.com/view/swat-carbon) and the R-SWAT (https://github.com/tamnva/R-SWAT), we conducted watershed simulations and found that adoption of WCCs can pronouncedly reduce nitrate and sediment loads to the Chesapeake Bay from the TW by 3-6% and 10-19%, respectively, depending on WCC scenarios. In general, early-planted WCCs nearly doubled the water quality benefits from late-planted WCCs. In

606 addition, the six WCC scenarios can sequester 0.45-0.92 MgC ha^{-1} yr^{-1}, with early-planted
607 WCCs having more than 70% SOC equation benefits compared with the late-planted WCCs. If
608 linearly scaled to the farming land across the state of Maryland, WCCs hold potential to
609 contribute to 2.1-4.4% of Maryland's 2030 GHG reduction goal of 14.4 TgC. Further analysis
610 showed that the SOC sequestration benefits of WCCs is mainly determined by planting-
611 termination window, but also influenced by crop species and initial SOC content. Overall, this
612 modeling study demonstrated that WCCs hold potential to provide pronounced synergistic water
613 quality and SOC sequestration benefits in the Chesapeake Bay Watershed. However, one
614 tradeoff that deserves further attention is the increase in evapotranspiration and decrease in water
615 yield and streamflow, which could have negative implications for aquatic ecosystem health and
616 water supply.

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