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Assessing Short-Term Impacts of Management Practices on N2O Emissions From Diverse Mediterranean Agricultural Ecosystems Using a Biogeochemical Model

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Key Points:
- We tested the capability of a process-based model (DNDC) for predicting N_2O emissions from various cropping systems
- N_2O mitigation efficiencies of management practices have been assessed for the investigated cropping systems
- DNDC performed better than the emission factor approach in quantifying N_2O emission from regions with diverse cropping systems

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Assessing Short-Term Impacts of Management Practices on N_2O Emissions From Diverse Mediterranean Agricultural Ecosystems Using a Biogeochemical Model

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Abstract Croplands are important sources of nitrous oxide (N_2O) emissions. The lack of both long-term field measurements and reliable methods for extrapolating these measurements has resulted in a large uncertainty in quantifying and mitigating N_2O emissions from croplands. This is especially relevant in regions where cropping systems and farming management practices (FMPs) are diverse. In this study, a process-based biogeochemical model, DeNitrification-DeComposition (DNDC), was tested against N_2O measurements from five cropping systems (alfalfa, wheat, lettuce, vineyards, and almond orchards) representing diverse environmental conditions and FMPs. The model tests indicated that DNDC was capable of predicting seasonal and annual total N_2O emissions from these cropping systems, and the model's performance was better than the Intergovernmental Panel on Climate Change emission factor approach. DNDC also captured the impacts on N_2O emissions of nitrogen fertilization for wheat and lettuce, of stand age for alfalfa, as well as the spatial variability of N_2O fluxes in vineyards and orchards. DNDC overestimated N_2O fluxes following some heavy rainfall events. To reduce the biases of simulating N_2O fluxes following heavy rainfall, studies should focus on clarifying mechanisms controlling impacts of environmental factors on denitrification. DNDC was then applied to assess the impacts on N_2O emissions of FMPs, including tillage, fertilization, irrigation, and management of cover crops. The practices that can mitigate N_2O emissions include reduced or no tillage, reduced N application rates, low-volume irrigation, and cultivation of nonleguminous cover crops. This study demonstrates the necessity and potential of utilizing process-based models to quantify N_2O emissions from regions with highly diverse cropping systems.

1. Introduction

Intensive applications of nitrogen (N) fertilizers, here defined as single application or split applications of high N concentration, are commonly used in cropping systems to increase crop yields. However, high rates of N application often have low N use efficiency (defined as efficiency of crop N uptake) and result in a significant portion of reactive N being released into the environment, leading to air and water pollution (Cassman et al., 2002; Galloway et al., 2003). The greenhouse gas (GHG) nitrous oxide (N_2O) is an important component of the N cycle, contributing significantly to global warming due to its high global warming potential (Intergovernmental Panel on Climate Change (IPCC), 2013). In addition, N_2O is a dominant anthropogenic, ozone-depleting substance, responsible for the destruction of stratospheric ozone (Ravishankara et al., 2009). The atmospheric concentration of N_2O was 323 ppb in 2009, increasing at a rate of 0.73 ppb/year during the past three decades (IPCC, 2013). Globally, N_2O released from agriculture is approximately 4.1 Tg (10^{12} g) N year^{-1} (IPCC, 2013; Syakila & Kroeze, 2011), which is primarily attributed to the use of synthetic fertilizers and organic manure (Davidson, 2009).

There is an urgent need for quantifying N_2O emissions and assessing mitigation potential from croplands (e.g., Bouwman et al., 2002; Decock et al., 2015; Smith et al., 2008; Venterea et al., 2012). However, complex mechanisms underlying the high variability observed in N_2O emissions make this quantification difficult.

Nitrous oxide emissions from soils are primarily produced through microbial-mediated nitrification and denitrification and are subject to controls involving interactions of environmental factors, such as concentrations of mineral N, availability of dissolvable organic carbon (DOC), soil water content, redox potential, and temperature (e.g., Butterbach-Bahl et al., 2013; Robertson & Groffman, 2007). The temporal and spatial variability of these controlling factors results in enormous heterogeneity in N_2O fluxes (e.g., Bouwman et al., 2002; Groffman et al., 2009). Therefore, it is impractical to quantify N_2O emissions from regional or global croplands based on field measurements alone. To extrapolate the measurements taken at specific sites and during specific periods to large regions or over extended time spans, a variety of approaches have been developed, ranging from simple regressions to complex process-based models. Simple regression approaches, such as emission factor (EF) methods (Bouwman, 1996), are useful tools to estimate N_2O emissions at regional or global scales (e.g., Stehfest & Bouwman, 2006; Syakila & Kroeze, 2011). However, such approaches generally become less accurate at finer temporal and spatial scale because they usually ignore some natural or management factors that are critical controls on N_2O emission (Butterbach-Bahl et al., 2013; Chen et al., 2008). By neglecting specifics of farming management practices (FMPs), empirical methods may not be suitable for identifying mitigation opportunities for N_2O emission (Butterbach-Bahl et al., 2013; Giltrap et al., 2010). Process-based models, such as the DeNitrification-DeComposition (DNDC) model, have taken into account important regulating factors to support the quantification of N_2O emissions and thus have been recognized as useful tools to evaluate effects of management practices on N_2O emissions from agricultural ecosystems (e.g., Butterbach-Bahl et al., 2013; Chen et al., 2008; De Gryze et al., 2011; Giltrap et al., 2010). However, large uncertainty still exists in applying process-based models to estimate N_2O emissions from regions with diverse agriculture. Models have been usually tested and applied for a single or limited type cropping system (mostly field crops; e.g., Chen et al., 2008; Giltrap et al., 2010). For example, we are not aware of testing or application of process-based models for quantifying N_2O emissions from orchard or vineyard systems. Considering that characteristics of N_2O emission and its controlling factors are often variable across different cropping systems and that model testing or application has been limited for multitype cropping systems, it has been especially challenging to reliably quantify N_2O emission and its mitigation potential for areas with diverse agriculture.

To quantify N_2O emissions from regions with diverse agriculture and evaluate mitigate strategies, we applied DNDC to simulate N_2O emissions from various cropping systems in California where agriculture is extremely diverse consisting of over 400 commodities across a wide range of landscape and geographical conditions (National Agricultural Statistics Service, 2012). California is a major agricultural producer with the highest crop cash receipts in the United States (National Agricultural Statistics Service, 2012). It is also the second largest emitter of GHGs in the United States (U.S. Energy Information Administration, 2018). With the passage of the Global Warming Solution Act in 2006, which aims to reduce GHGs emissions from California to 1990 levels by 2020, California is at the frontier of quantifying and mitigating GHGs. There are approximately 3.4×10^6 ha of harvested croplands, 34% of which are orchards and vineyards, 23% are alfalfa and hay, and 14% are devoted to vegetable crops (University of California Agricultural Issues Center, 2009). Agricultural management practices, such as tillage, fertilization, and irrigation are highly variable. However, they are generally characterized as intensive because standard tillage operations, high rates of N fertilizer application, and intensive furrow irrigation are common in the majority of croplands (Suddick et al., 2010). For example, over 90% of California vineyards and orchards are irrigated and fertilized using microirrigation systems (Smart et al., 2011), although many orchard growers often flood following harvest, while field crops are usually flood irrigated (Tindula et al., 2013), and fertilized through surface spraying, injection, or broadcast. The wide variation in the type of cropping systems and FMPs could further exacerbate the temporal and spatial variability of N_2O emissions from croplands, thus making the quantification of N_2O emissions a challenging task (De Gryze et al., 2011).

In this study, we applied the DNDC model to assess impacts of FMPs on N_2O emissions from five major cropping systems in California: alfalfa, wheat, lettuce, vineyards, and almond orchards. Our objectives were (1) to test DNDC against field observations of N_2O emissions from these cropping systems, and verify if process-based models, such as DNDC, can be utilized to quantify N_2O emissions from diverse cropping systems; and (2) to apply the model to assess the impacts of FMPs on N_2O emissions and quantify the mitigation potential of different management strategies.

Table 1
General Characteristics and Soil Properties of Study Fields Where Measurements of N_2O Fluxes Were Used for Model Tests

County	Coordinate	Period[a]	Crop type	T or P[b]	Soil texture	Clay (%)	BD[c]	pH	SOC[d]	Reference[e]
Yolo	38°35′N, 121°50′W	November 2010 to December 2011	Alfalfa	2-year stand	Myers clay	34	1.43	7.7	12.6	(1)
Yolo	38°35′N, 121°50′W	November 2010 to December 2011	Alfalfa	5-year stand	Myers clay	34	1.43	7.7	12.6	(1)
Solano	38°26′N, 121°52′W	November 2009 to May 2010	Wheat	DNF-1[f]	Silty clay loam	25	1.35	7.2	12.8	(1)
Solano	38°26′N, 121°52′W	November 2010 to May 2011	Wheat	DNF-2[g]	Silty clay loam	35	1.29	7.4	14.9	(1)
Monterey	36°40′N, 121°36′W	June 2009 to March 2011	Lettuce	DNF-3[h]	Loam	17	1.58	7.6	12.7	(1)
Napa	38°26′N, 121°52′W	January 2009 to December 2010	Vineyards	Grape row	Loam	25	1.24	5.6	23.0	(2)
Napa	38°26′N, 121°52′W	January 2009 to December 2010	Vineyards	Alley	Loam	25	1.24	5.6	23.0	(2)
Colusa	39°03′N, 121°59′W	March 2009 to March 2011	Vineyards	Grape row	Silty clay	19	1.3	7.2	12[i]	(3)
Colusa	39°03′N, 121°59′W	March 2009 to March 2011	Vineyards	Alley	Silty clay	19	1.3	7.2	12[i]	(3)
Solano	38°58′N, 122°05′W	March 2010 to March 2011	Almonds	Tree row	Sandy loam	13	1.6	7.6	4.4[i]	
Solano	38°58′N, 122°05′W	March 2010 to March 2011	Almonds	Alley	Sandy loam	13	1.6	7.6	4.4[i]	

[a]Period during which measurements of N_2O fluxes were used for model tests. [b]T or P, treatments included in the field studies or positions of the fields. [c]BD, bulk density (g cm^{-3}). [d]SOC, content of soil organic carbon (g C kg^{-1} dry soil). [e](1) Burger and Horwath (2012); (2) Steenwerth et al. (2010), Smart et al. (2011); (3) Garland et al. (2011, 2014). [f]DNF-1, different nitrogen fertilization. Measurements of N_2O fluxes under five treatments were used for model tests. The treatments included different practices of nitrogen fertilization, with 254 (NH_4^+-N + urea), 203 (anhydrous ammonia + urea), 151 (NH_4^+-N + urea), 91 (urea), and 0 kg N ha^{-1} applied during the wheat growing season. Details of the fertilization treatments are described by Burger and Horwath (2012). [g]DNF-2, different nitrogen fertilization. Measurements of N_2O fluxes under five treatments were used for model tests. The treatments included different practices of nitrogen fertilization, with 266 (NH_4^+-N + urea), 210 (NH_4^+-N + urea), 210 (anhydrous ammonia + urea), 154 (NH_4^+-N + urea), and 0 kg N ha^{-1} applied during the wheat growing season. Details of the fertilization treatments are described by Burger and Horwath (2012). [h]DNF-3, different nitrogen fertilization. Measurements of N_2O fluxes under five treatments were used for model tests. The treatments included different practices of nitrogen fertilization, with 336, 252, 168, 84, and 11 kg UAN32-N ha^{-1} applied during the lettuce growing season. UAN32: $(NH_2)_2CO \cdot NH_4NO_3$. Details of the treatment setting are described by Burger and Horwath (2012). [i]Estimated based on typical soil properties at the study fields, as on-site observations were not available.

2. Materials and Methods

2.1. The Study Sites and Field Data

Field data used to support the model applications were collected during 2009 to 2011 at seven sites located in five counties in California. The study fields were cultivated with different crops, including alfalfa, winter wheat, lettuce, grape vine, and almond tree, representative of a range of typical cropping systems in California, including hay, cereal crops, vegetables, vineyards, and perennial orchards. The study sites generally experience a Mediterranean climate that consists of hot dry summers and wet cool winters. Table 1 summarizes general characteristics and soil properties of the tested fields.

All field experiments included a treatment representing typical FMPs in the local area. In addition, treatments of different N fertilization were set in winter wheat and lettuce to investigate the impact on N_2O emissions (Table 1). To quantify the impacts of alfalfa age on N_2O emissions, the experiment measured N_2O emissions from 2- and 5-year-old alfalfa fields (Table 1). For the vineyards and almond orchards, N_2O fluxes were measured at two different locations, that is, tree rows and alley, to capture spatial variability within a field (Garland et al., 2011, 2014; Steenwerth et al., 2010). Additional details regarding the FMPs performed at the sites are summarized in Tables 1 and 2 and were described by Burger et al. (2016), Burger and Horwath (2012), Garland et al. (2011, 2014), Smart et al. (2011), Steenwerth et al. (2010), and Zhu-Barker et al. (2015).

The field measurements of N_2O fluxes were performed using the vented static chamber method (Hutchinson & Mosier, 1981). In general, static chambers were sealed onto bottom collars and gas samples were taken using air-tight polypropylene syringes at regular intervals at sampling time. Nitrous oxide concentrations were analyzed using similar gas chromatographs across the studies, each equipped with an electron capture detector. Gas fluxes were calculated from the rate of change of N_2O concentration, chamber volume, and soil surface area (Hutchinson & Mosier, 1981). Nitrous oxide fluxes were usually measured daily or once every 2 days following events such as tillage, fertilization, precipitation, irrigation, or harvest, until the high N_2O fluxes induced by the event receded to background levels. For other periods, the measurements of N_2O fluxes were performed less frequently but more often than once every 2 weeks. Seasonal or annual total N_2O emissions were generally available in these field studies and were calculated by linear interpolation of the measured daily fluxes (Burger & Horwath, 2012; Garland et al., 2011, 2014). When measuring the N_2O fluxes, the local climate, soil properties, crop yield or aboveground biomass, and FMPs were often recorded as well. The rich data sets from these field studies therefore provided an opportunity to evaluate the DNDC

Table 2
Farming Management Practices and Simulated Annual Total N_2O Emissions Under the Baseline and Alternative Scenarios

Cropping system	Scenario	Planting[a]	Harvest	Tillage[b]	N ($kg\ N \cdot ha^{-1} \cdot year^{-1}$)[c]	Irrigation[d]	Cover crop	N_2O emission ($kg\ N \cdot ha^{-1} \cdot year^{-1}$)
Alfalfa	Baseline		Six times per year	NT	8.5	1,290-mm flood	None	3.93
Alfalfa	A1		Six times per year	NT	8.5	711-mm sprinkler	None	3.12
Alfalfa	A2		Six times per year	NT	8.5	508-mm sprinkler	None	2.76
Wheat	Baseline	November	June	CT	210	150 mm	None	1.82
Wheat	W1	November	June	CT	168	150 mm	None	1.38
Wheat	W2	November	June	RT	210	150 mm	None	1.60
Lettuce	Baseline	June	August	CT	252	227-mm drip	None	1.59
Lettuce	L1	June	August	CT	202	227-mm drip	None	1.08
Lettuce	L2	June	August	CT	252	381-mm sprinkler	None	1.62
Lettuce	L3	June	August	RT	252	227-mm drip	None	1.47
Vineyards	Baseline		October	CT	5.4	160-mm drip	Legume	2.25
Vineyards	V1		October	CT	5.4	160-mm drip	None	1.71
Vineyards	V2		October	CT	5.4	160-mm drip	Nonlegume	0.83
Vineyards	V3		October	RT	5.4	160-mm drip	Legume	1.97
Almonds	Baseline		September	NT	230	965-mm sprinkler	None	0.78
Almonds	A1		September	NT	230	1,300-mm flood	None	1.04
Almonds	A2		September	NT	184	965-mm sprinkler	None	0.71
Almonds	A3		September	CT	230	965-mm sprinkler	None	0.79

[a]No planting during production years for perennial crops. [b]CT, conventional tillage. The soil was tilled (20-cm depth) twice on days of planting and harvest for wheat and lettuce. The soil in alley was tilled twice (30-cm depth) in spring and late fall for vineyards and almonds orchard. RT, reduced tillage. The soil was tilled (10-cm depth) twice on days of planting and harvest for wheat and lettuce. The soil in alley was tilled twice (5-cm depth) in spring and late fall for vineyards. NT = no tillage. [c]For the alfalfa fields, nitrogen fertilizers were applied one time per 2 years at a rate of 17 kg N ha^{-1}. [d]Irrigation practices in each scenario, including water amount (mm) and irrigation method, were determined from either field records or cost and return studies published by the University of California, Davis (2016).

model to predict N_2O emissions and investigate mitigation options for multitype cropping systems in a region with diverse agriculture. The technical details regarding the N_2O measurements, and the relevant auxiliary variables are described by Burger et al. (2016), Burger and Horwath (2012), Garland et al. (2011, 2014), Smart et al. (2011), Steenwerth et al. (2010), and Zhu-Barker et al. (2015).

2.2. The DNDC Model

The DNDC model (Li et al., 1992a, 1992b) was developed for quantifying C sequestration and emissions of GHGs and has been utilized worldwide during the last two decades (e.g., Beheydt et al., 2007; Giltrap et al., 2010). The model has incorporated a relatively complete suite of biogeochemical processes governing C and N cycling, including decomposition, fermentation, ammonia volatilization, nitrification, and denitrification. DNDC is composed of two components. The first component consists of the soil climate, crop growth, and decomposition submodels and converts primary drivers, such as climate, soil properties, vegetation, and anthropogenic activity, into soil environmental factors such as soil temperature and moisture, pH, redox potential, and substrate concentration. The second component consists of the nitrification, denitrification, and fermentation submodels and simulates C and N transformations that are mediated by soil microbes. In DNDC, soil N primarily exists in several pools—organic N, ammonium, ammonia, and nitrate. Dynamics of soil N in each pool are simulated at an hourly or daily time step through a series of biogeochemical reaction: decomposition, microbial assimilation, plant uptake, ammonia volatilization, ammonium adsorption, nitrification, denitrification, and nitrate leaching. Fluxes of N gases (i.e., NO, N_2O, and N_2) are predicted as either products or intermediate products by simulating the relevant N transformation processes, primarily nitrification and denitrification.

DNDC parameterizes FMPs, such as tillage, fertilization, flooding, irrigation, and cultivation of cover crops, as primary drivers regulating soil environmental conditions and/or substrate concentrations, and thereby simulates the influence of management practices on the rates of C and N biogeochemical reactions. For example, the application of N fertilizers affects soil N pools based on N application rate, method of application, and type of fertilizer and influences crop growth, nitrate leaching, and emissions of C and N gases. Flooding and irrigation directly control soil moisture and redox potential, which influence crop growth and all

Table 3
DNDC Crop Physiological Parameters

Crops	MP[a]	SRF[b]	C/N[c]	TDD[d]	WR[e]	NFI[f]
Alfalfa	10000	0.7/0.3	14	5000	200	5.0
Wheat	9000	0.83/0.17	35	2000	200	1.0
Lettuce	1875	0.8/0.2	11	1000	500	1.0
Grapes	3333	0.9/0.1	22	5000	200	1.0
Almonds	3333	0.8/0.2	19	5000	150	1.0

[a]MP, the maximum productivity under optimum growing conditions (unit: kg dry matter ha^{-1}). The values were estimated by calibrating against the observed crop yields. [b]SRF, the shoot and root fractions [c]C/N, carbon to nitrogen ratio of the plant biomass [d]TDD, the required cumulative air temperature heat sum (in °C days) above a 0 °C threshold during the growing period for full crop growth. [e]WR, amount of water required by the crop to produce dry matter (in g water g^{-1} dry matter). [f]NFI, index of biological nitrogen fixation. The value of 1.0 indicates no N-fixation, and 5.0 indicates 80% of crop nitrogen demand is came from N-fixation.

biogeochemical reactions in the soil. Further details regarding DNDC structure, inputs, and outputs, as well as the physical, chemical, and biogeochemical processes incorporated into the model's framework, are available in Gilhespy et al. (2014), Li (2000), and Li et al. (2012).

2.3. Model Application

2.3.1. Model Test

Field data to test DNDC, including measurements of N_2O fluxes and soil properties and FMPs, were acquired from the respective researchers (Burger & Horwath, 2012; Garland et al., 2011, 2014; Smart et al., 2011; Stehfest & Bouwman, 2006; Zhu-Barker et al., 2015). Daily meteorological data, including maximum and minimum air temperatures, and precipitation were obtained from either on-site measurements or local meteorological stations. Primary soil input parameters, including soil texture, clay fraction, bulk density, pH, and soil organic carbon (SOC) content, were determined using on-site measurements at the alfalfa, wheat, and lettuce fields, and the vineyard in Napa County. For the vineyard in Colusa county and the almond orchard, we used on-site observations to determine these soil parameters except SOC, which was estimated based on the soil properties of the local dominant soil types from the SSURGO database (Natural Resources Conservation Service, 2015), because the on-site observations were not available (Table 1). FMPs (Table 2) applied in each treatment included planting and harvest dates, tillage, fertilization, irrigation, and cultivation and incorporation of cover crop and were derived from field records. We set up the input parameters of FMPs by strictly following field records in order to represent all variations in FMPs in the simulations. The phenological and physiological parameters related to crop growth (Table 3) were estimated by referring to on-site observations, calibrating the parameters against the crop yields or using model defaults that were derived from a large collection of literature values. DNDC was run separately for the 2- and 5-year-old alfalfa fields (Table 1) with the primary difference being the length of the model simulations (i.e., 2 and 5 years; Burger et al., 2016). For the vineyards and almond orchard, the model was run for crop rows and alleys separately to account for spatial variation of N_2O fluxes associated with typical FMPs applied to rows and alleys in vineyards or orchards. During the simulations, we did not calibrate any soil biogeochemistry parameters or functions, including those calculating the processes of N_2O production, consumption, and emission. No site-specific modification was performed if not mentioned above. We used a 1-year model spin-up to initialize the soil climate and mineral N conditions and then proceeded with the simulations for the DNDC tests. The modeled N_2O fluxes were compared against the measured records. The data set of N_2O fluxes used for model tests included a total of 32 unique site-treatment-year combinations.

We also calculated the seasonal or annual N_2O emissions using the EF approach (Tier 1), in which the EF is defined as the loss rate via N_2O emission of nitrogen applied to soils, and is a fixed value of 0.01 (Bouwman, 1996; IPCC, 2006). The EF approach is often used to develop N_2O inventories from agricultural soil management in many regions, including those regions with diverse agriculture (e.g., California Air Resources Board, 2011, 2014). The EF estimates of N_2O emissions were compared to the field measurements and DNDC simulations to assess if the use of the process-based model, such as DNDC, can improve the N_2O emissions inventory. The EF-based seasonal or annual N_2O emissions were calculated as follows:

$$N_2O_D = (N_{SF} + N_{CR}) * EF_1, \quad (1)$$

where N_2O_D is direct N_2O emissions (kg N_2O-N) from agricultural managed soils, N_{SF} is amount of synthetic fertilizer N (kg N) applied to soils, N_{CR} is amount of N in crop residues (kg N) returned to soil, and EF_1 is EF for N_2O emissions from N inputs (kg N_2O-N kg^{-1} N), which was set as 0.01 by following California Air Resources Board (2011, 2014). Note that amounts of N from organic fertilizers and managed manure were 0 based on the field records.

We used zero-intercept linear regression between simulations and observations to evaluate DNDC performance. The slope and determination coefficient (R^2) of the regression indicate the consistency and

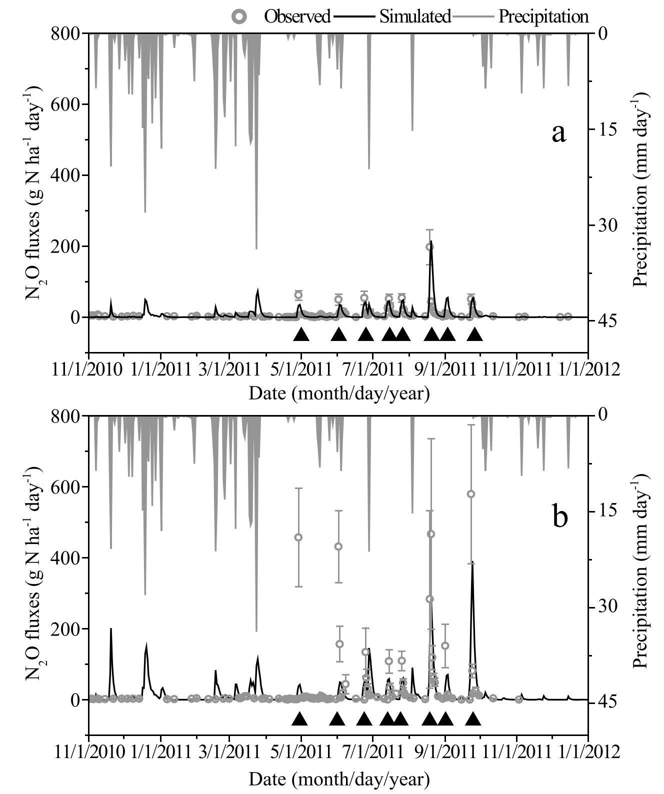

Figure 1. Precipitation and simulated and measured daily N_2O fluxes from (a) second-year and (b) fifth-year alfalfa fields. The triangles indicate the dates of flood irrigation events. The flux measurements are mean values from Burger and Horwath (2012); the vertical bars indicate standard errors of replicates (n = 6).

correlation between simulations and observations, respectively (Moriasi et al., 2007). The normalized root-mean-square error (RMSE) was also used for quantitative comparisons between the simulations and observations.

2.3.2. Scenarios of FMPs

To investigate impacts of FMPs on N_2O emissions, we conducted a series of simulations varying FMPs for each studied cropping system (Table 2). The simulations under alternative management practice scenarios were compared to the baseline simulations for alfalfa in Yolo County, wheat in Solano County, lettuce in Monterey County, vineyard in Napa County, and an almond orchard in Solano County. The N fertilizer application rates were 8.5, 210, 252, 5.4, and 230 kg $N \cdot ha^{-1} \cdot year^{-1}$, respectively, for the alfalfa, wheat, lettuce, vineyard, and almond orchard under the baseline.

Alternative scenarios were set by exclusively changing a single FMPs from the baseline scenarios. The alternative management practices tested (Table 2) were generally extracted from Cost and Return Studies (University of California, Davis, 2016) and reflect specific practices for each crop type. For alfalfa, because no-till and low rates of N fertilizers are common, while irrigation practices vary (Tindula et al., 2013), we ran two alternative irrigation practices using sprinkler irrigation. For the cropping systems with relatively intensive tillage and nitrogen applications, including wheat, lettuce, and almond, three or four alternative scenarios were set by exclusively changing tillage, rate of N applications, or irrigation (Table 2). For the vineyards, which have diverse cover crop management and where small amounts of water and N are commonly applied using drip irrigation systems, we conducted three alternative scenarios changing either tillage or planting of cover crops (Table 2).

The DNDC model was run for baseline and alternative scenarios. For each cropping system, a single FMP change was evaluated (i.e., tillage, fertilization, irrigation, or cover cropping), but other conditions (i.e., climate, soil, crop type, and other practices) were kept the same under the different scenarios. To consider potential impacts of stand age on N_2O emissions from the alfalfa fields (Burger & Horwath, 2012), we ran DNDC for 5 years from 2007 to 2011 (a full alfalfa growth cycle) using the meteorological data of these years, and the modeled average annual N_2O emissions of the 5 years were used for analysis. For other cropping systems, the DNDC model was run for 2 years, and the modeled annual N_2O emissions for the second year were used for analysis.

3. Results

3.1. Model Tests

3.1.1. N_2O Fluxes

Figure 1 illustrates seasonal patterns of the measured and simulated daily N_2O fluxes from the alfalfa fields. The measurements showed similar seasonal patterns between the fields with second- and fifth-year alfalfa, with high N_2O peaks occurring on days following each flood irrigation event (Burger et al., 2016; Burger & Horwath, 2012). In comparison with the measurements, DNDC generally captured the seasonal patterns of daily N_2O fluxes, although the magnitudes of some modeled N_2O peaks were not consistent with the field observations of the fifth-year alfalfa (e.g., late April and early June 2011, Figure 1b). In addition, the model successfully predicted the impact of alfalfa stand age on N_2O emissions. Both the DNDC simulations and field observations indicated higher N_2O peaks for the site with fifth-year alfalfa (Figure 1) and therefore produced a higher rate of annual cumulative N_2O emission for this site (Table 4). DNDC predicted higher N_2O emission for the fifth-year alfalfa primarily due to the simulated higher soil N that accumulated over time as a result of alfalfa N fixation. The modeled annual total N_2O emissions were 2.8 and 6.4 kg N_2O-N ha^{-1} for the second- and fifth-year alfalfa fields. In comparison with the measured data of annual total N_2O emissions (2.3 and

Table 4
Comparison Between the Simulated (S) and Observed (O) Seasonal or Annual N_2O Emissions

Period[a]	Cropping system	T or P[b]	O[c]	SE[c]	S	RMSE
December 2010 to November 2011	Alfalfa	2-year stand	2.3	0.26	2.79	21%
December 2010 to November 2011	Alfalfa	5-year stand	5.2	0.79	6.42	23%
November 2009 to June 2010	Wheat	254 N (NH_4^+ + urea)	0.5	0.13	0.8	60%
		203 N (anhydrous ammonia + urea)	1.31	0.35	1.0	24%
		151 N (NH_4^+ + urea)	0.57	0.12	0.6	5%
		91 N (NH_4^+ + urea)	0.31	0.08	0.16	48%
		0 N	0.24	0.07	0.14	42%
November 2010 to June 2011	Wheat	266 N (NH_4^+ + urea)	2.15	0.23	2.26	5%
		210 N (anhydrous ammonia)	2.05	0.17	1.79	13%
		210 N (NH_4^+ + urea)	1.42	0.1	1.75	23%
		154 N (NH_4^+ + urea)	0.88	0.18	1.35	53%
		0 N	0.72	0.22	0.65	10%
June 2009–may 2011	Lettuce	366 N (UAN32)	1.51	0.27	1.7	13%
		252 N (UAN32)	1.09	0.08	1.31	20%
		168 N (UAN32)	0.69	0.07	0.84	22%
		84 N (UAN32)	0.71	0.07	0.63	11%
		11 N (UAN32)	0.58	0.05	0.51	12%
June 2010–may 2011	Lettuce	366 N (UAN32)	1.42	0.22	1.54	8%
		252 N (UAN32)	1.14	0.14	1.21	6%
		168 N (UAN32)	1.13	0.2	0.82	27%
		84 N (UAN32)	0.56	0.03	0.31	45%
		11 N (UAN32)	0.59	0.13	0.19	68%
January 2009 to December 2009	Vineyards	Grape row	0.22	NA	0.14	36%
January 2010 to December 2010		Grape row	0.17	NA	0.2	18%
January 2009 to December 2009	Vineyards	Alley	0.27	NA	0.21	22%
January 2010 to December 2010		Alley	0.37	NA	0.3	19%
Jan uary 2009 to December 2009	Vineyards[d]	Vineyard	0.26	NA	0.19	25%
January 2010 to December 2010		Vineyard	0.32	NA	0.28	14%
March 2009 to March 2010	Vineyards	Grape row	0.48	NA	0.6	25%
March 2010 to March 2011		Grape row	0.6	NA	0.14	77%
March 2009 to March 2010	Vineyards	Alley	5.39	NA	3	44%
March 2010 to March 2011		Alley	0.54	NA	0.67	24%
March 2009 to March 2010	Vineyards[d]	Vineyard	3.92	NA	2.28	42%
March 2010 to March 2011		Vineyard	0.56	NA	0.51	8%
March 2010 to March 2011	Almonds	Tree row	0.92	NA	0.84	9%
March 2010 to March 2011	Almonds	Alley	0.64	NA	0.52	19%
March 2010 to March 2011	Almonds[d]	Orchard	0.81	NA	0.71	12%

Note. RMSE = root-mean-square error; NA = not available.
[a]Period during which measurements of N_2O fluxes were used for model tests. [b]T or P, treatments included in the field studies or positions of the fields. Details of the treatment setting were described by Burger and Horwath (2012), Steenwerth et al. (2010), Smart et al. (2011), Garland et al. (2011, 2014), and Zhu-Barker et al. (2015). [c]Observed data with standard error (SE) of the field measurements are reported in Burger and Horwath (2012), Steenwerth et al. (2010), Smart et al. (2011), Garland et al. (2011, 2014), and Zhu-Barker et al. (2015). [d]Annual N_2O emissions from the whole vineyard or almond orchard were calculated by weighting the N_2O emissions with the relative row and alley widths of each location across the vineyard or orchard (Garland et al., 2011, 2014; Smart et al., 2011; Steenwerth et al., 2010).

5.2 kg N_2O-N ha^{-1}), the DNDC simulations had an RMSE of 21% and 23%, respectively, for the fields with second- and fifth-year alfalfa.

In the fertilized wheat fields, peaks of N_2O fluxes were often observed on days following heavy rainfall, while N_2O fluxes from the unfertilized wheat fields remained consistently at relatively low levels (Figure 2). As compared with the field measurements, the DNDC model generally captured the peaks of daily N_2O flux induced by heavy precipitation, although discrepancies remained between the magnitude of the modeled N_2O peaks and the corresponding observations. As compared with the field records, the DNDC model also predicted more frequent N_2O peaks after rainfall events during the winter rainy season, which were not always observed in the field studies (e.g., on late January 2010 and end March 2011, Figure 2). The model indicated, as was observed in the measurements, that N fertilization treatments exerted substantial impacts on the N_2O emissions from the wheat fields (Table 4). The simulated N_2O emissions generally increased along with

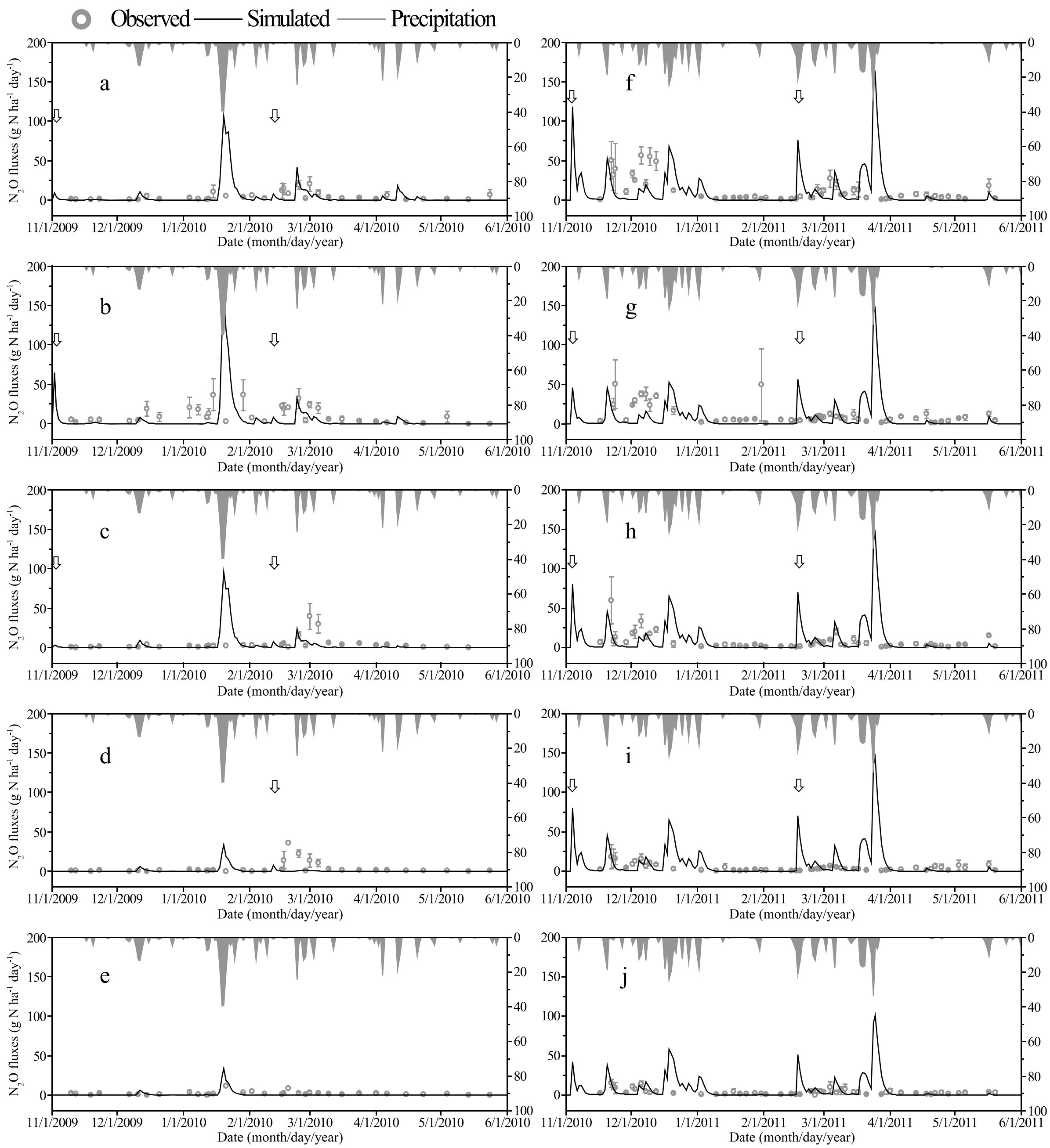

Figure 2. Precipitation and simulated and measured daily N_2O fluxes from winter wheat fields with different nitrogen fertilization rates. N fertilizers were applied at rates of (a) 254, (b) 203, (c) 151, (d) 91, and (e) 0 kg N ha^{-1} during the November 2009 to May 2010 growing season and at (f) 266, (g) 210, (h) 210, (i) 154, and (j) 0 kg N ha^{-1} during the November 2010 to May 2011 growing season. The arrows indicate the dates of fertilization events. The measured flux data are means from Burger and Horwath (2012) and Zhu-Barker et al. (2015); the vertical bars indicate standard errors of replicates (n = 3).

increasing N application rate if fertilizer type was kept the same; this was consistent with the measured data (Burger & Horwath, 2012; Zhu-Barker et al., 2015). The simulated seasonal N_2O emissions for wheat varied from 0.14 to 2.26 kg N-N_2O ha^{-1}, comparable with the observed seasonal N_2O emissions that ranged from 0.24 to 2.15 kg N-N_2O ha^{-1} (Table 4), with RMSE values between the modeled and observed seasonal N_2O emissions ranging from 5% to 60%.

DNDC reproduced similar N_2O peaks for lettuce (Figure 3), although the magnitudes of the simulated peaks were not fully consistent with the observations. DNDC also predicted more frequent N_2O pulses following

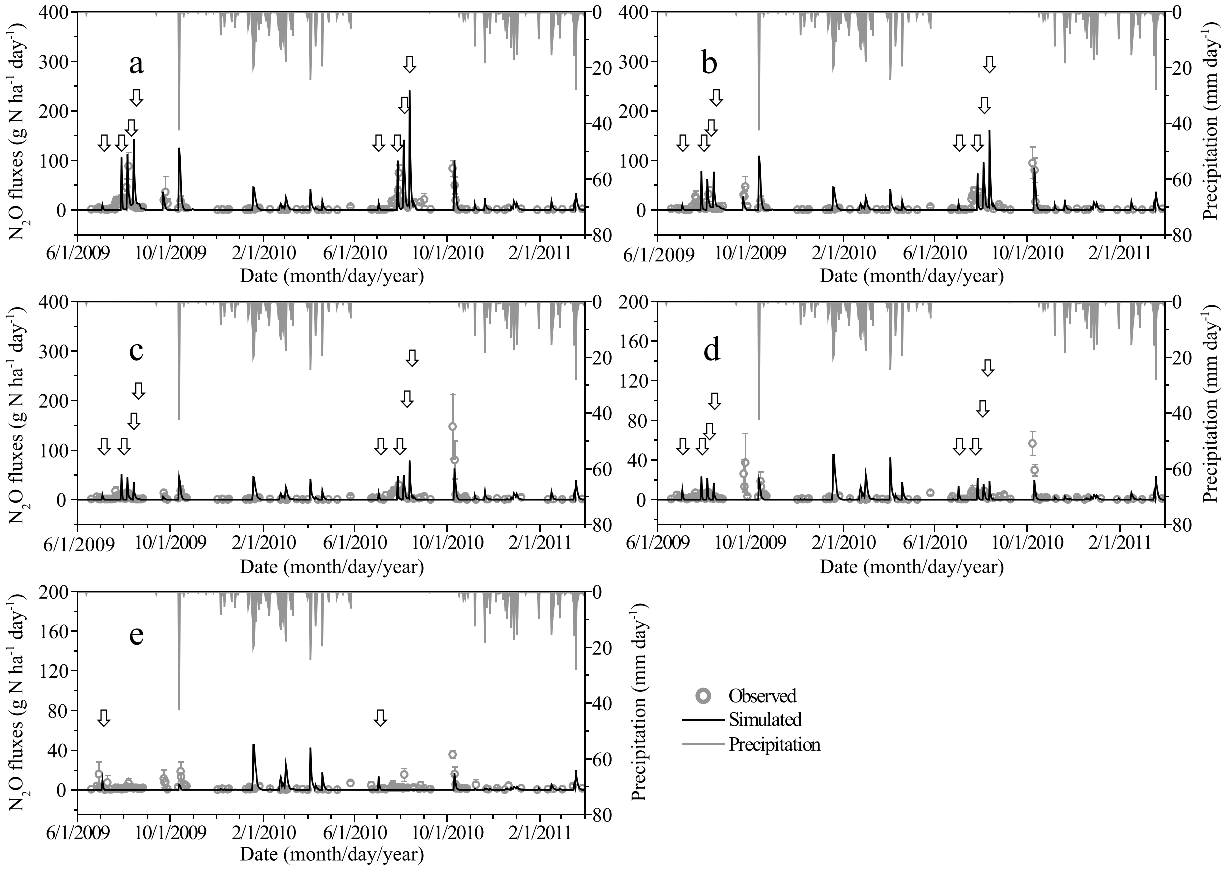

Figure 3. Precipitation and simulated and measured daily N_2O fluxes from lettuce fields under different nitrogen fertilization rates. N fertilizers were applied at rates of (a) 336, (b) 252, (c) 168, (d) 84, and (e) 11 kg N ha^{-1} during each growing season. The arrows indicate the dates of fertigation events. The fields were irrigated many additional times with small volumes of water by drip irrigation systems, and the dates of these irrigation events are not shown. The flux measurements are means from Burger and Horwath (2012); the vertical bars indicate standard errors of replicates (n = 4). Note that the vertical axis scales for N_2O fluxes in panels (a) to (c) are different from the scales in panels (d) and (e).

high precipitation during the winter rainy season (Figure 3), which did not always appear in field records (e.g., on late January 2010). The simulations showed an increasing trend in N_2O emission with increasing N application rate for the lettuce fields; this was consistent with the measurements (Burger & Horwath, 2012; Zhu-Barker et al., 2015). Of the 10 studied treatment-year combinations, the simulated annual N_2O emissions varied from 0.19 to 1.70 kg N_2O-N·ha^{-1}·year^{-1}. The values of RMSE between the modeled and observed annual cumulative N_2O emissions ranged from 6% to 68% across the 10 lettuce field cases (Table 4).

Figure 4 illustrates seasonal patterns of the measured and simulated daily N_2O fluxes from the tested vineyards. In the grape rows, high N_2O fluxes were detected on days following fertigation or high precipitation (Figures 4a and 4c), while peaks of N_2O fluxes only appeared following heavy precipitation in alleys (Figures 4b and 4d; Garland et al., 2011, 2014; Steenwerth et al., 2010). DNDC successfully captured the observed peaks of daily N_2O fluxes from both the grape rows and alleys, and the magnitudes of the simulated N_2O pulses were close to the field observations for most occasions (Figure 4). As Figure 4d shows, a significant N_2O peak was observed on mid-October 2009 in the vine alleys in Colusa County following heavy precipitation. This peak may be related to the planting and incorporation of leguminous cover crops in the alleys (Garland et al., 2014). The DNDC model successfully captured this peak as well. However, the model overestimated another N_2O peak induced by heavy precipitation during mid-January 2010 at this vineyard (Figure 4d). The observed annual total N_2O emissions varied from 0.17 to 0.60 kg N_2O-N ha^{-1} and 0.27 to 5.39 kg N_2O-N ha^{-1}, respectively, for the grape rows and alleys (Table 4; Garland et al., 2011, 2014;

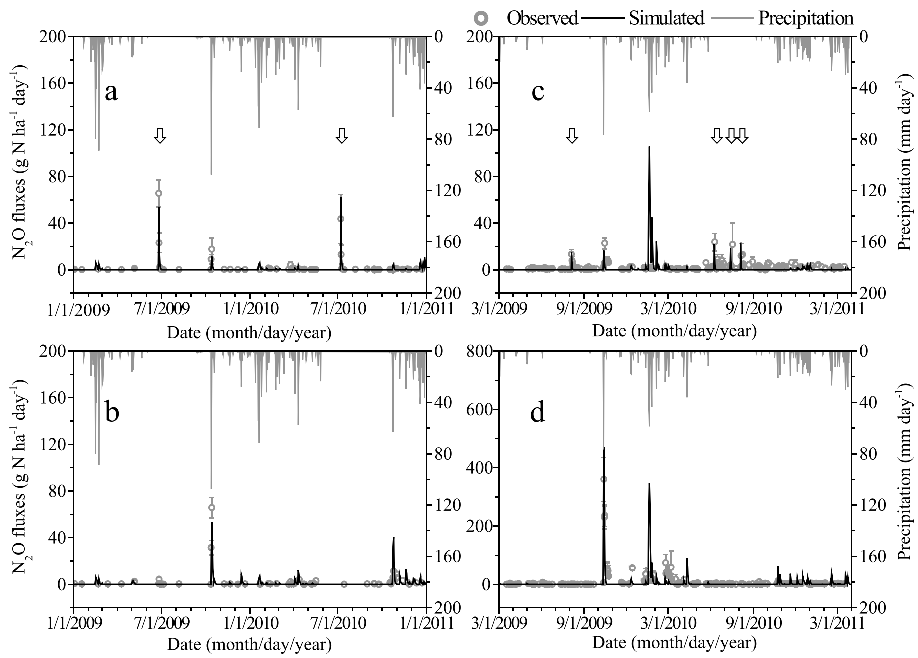

Figure 4. Precipitation and simulated and measured daily N_2O fluxes from vineyards in the (a and b) Napa and (c and d) Colusa counties. The data in panels (a) and (c) and (b) and (d) are for vine rows and alleys, respectively. The arrows indicate the dates of fertigation events. The grape rows were irrigated many additional times with small volumes of water by drip irrigation, and the dates of these irrigation events are not shown. The measured data are means from Steenwerth et al. (2010) and Garland et al. (2011, 2014); the vertical bars indicate standard errors of replicates (*n* = 3 for a and b and *n* = 4 for c and d). Note that the vertical axis scale for N_2O fluxes in panel (d) is different from the scale in other panels.

Steenwerth et al., 2010). The corresponding simulations varied from 0.14 to 0.60 kg N_2O-N ha^{-1} and 0.21 to 2.7 kg N_2O-N ha^{-1}, respectively. The RMSE values between the modeled and observed annual total N_2O emissions ranged from 18% to 77% and 19% to 50%, respectively, for the grape rows and alleys.

DNDC simulated different patterns of N_2O emissions between almond tree rows and alleys as well. Peak N_2O fluxes were predicted following fertigation, irrigation, or heavy precipitation at the tree rows (Figure 5a). On the contrary, the simulated N_2O fluxes from the alleys were only peaked following heavy precipitation (Figure 5b). DNDC generally captured the peaks of daily N_2O fluxes in comparison with the field measurements. However, DNDC predicted more frequent N_2O peaks after heavy rainfall events in both almond rows and alleys, some of which (e.g., 11 April and late October 2010) were not observed in the field studies. The predicted emissions (0.85 kg N_2O-N for rows and 0.52 kg N_2O-N for alleys) were close to the observed cumulative N_2O emissions from the tree rows and alleys, which were 0.92 and 0.64 kg N_2O-N, respectively. The RMSE between the observed and predicted emissions was 8% and 19%, respectively, for the tree rows and alleys (Table 4).

3.1.2. Total N_2O Emissions

The measured seasonal or annual total N_2O emissions varied between 0.24 (for wheat growing season from November 2009 to June 2010) and 5.20 kg N_2O-N ha^{-1}(for the fifth-year stand alfalfa) across the cropping system-treatment-year combinations tested (Table 4). The corresponding simulations had a similar range (minimum: 0.14 kg N_2O-N ha^{-1}, for the wheat growing season from November 2009 to June 2010; maximum: 6.42 kg N_2O-N ha^{-1}, for the fifth-year stand alfalfa). The RMSE values varied from 5% to 68% (Table 4). These results indicate a general agreement between the simulated and measured seasonal or annual N_2O emissions, although the goodness of fit varied across the test cases.

A zero-intercept linear regression with an R^2 of 0.92 ($P < 0.001$) and a slope of 1.0 could be obtained, relating DNDC predicted emissions to those measured in the field (Figure 6), indicating that overall the DNDC model reliably predicted the seasonal and annual N_2O emissions without statistical biases. The

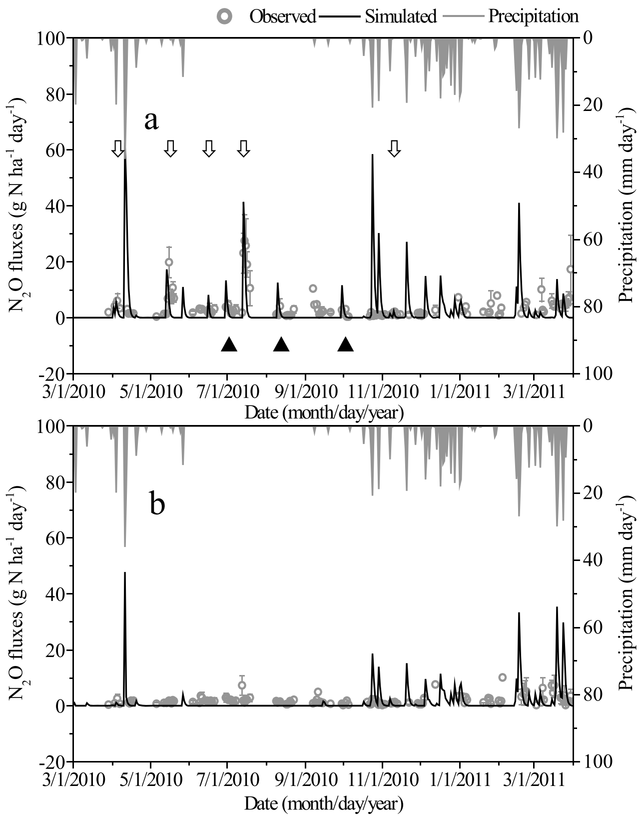

Figure 5. Precipitation and simulated and measured daily N_2O fluxes from almond (a) rows and (b) alleys. The arrows and triangles indicate the dates of fertigation and irrigation events, respectively. The measurements are the means and the vertical bars indicate standard errors of replicates (n = 3).

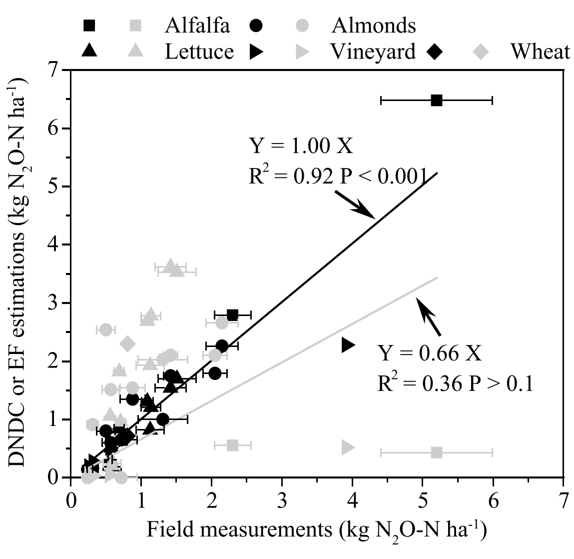

Figure 6. Comparison of DeNitrification-DeComposition (DNDC) simulated (black) and emission factor calculated (gray) seasonal and annual cumulative N_2O emissions against field measurements for all the tested cropping systems (different symbols). The functions shown describe the zero-intercept fitted regression lines. The horizontal bars indicate standard errors of replicate field measurements (n = 3 to 6, depending on the crop system).

EF-derived regression showed an R^2 of 0.36 ($P > 0.1$) and a slope of 0.66 (Figure 6). It is clear that DNDC performed better than the EF approach in estimating seasonal or annual N_2O emissions from the tested cropping systems.

3.2. Impacts of FMPs on N_2O Emissions

Table 2 lists the simulated annual total N_2O emissions from the five cropping systems under the baseline and alternative scenarios. The baseline scenario modeled annual N_2O emissions were 3.93, 1.82, 1.60, 2.30, and 0.78 kg N ha^{-1} for the alfalfa, wheat, lettuce, vineyard, and almond orchard, respectively.

All changes in FMPs under the alternative scenarios affected N_2O emissions. Compared to conventional tillage, reduced or no tillage slightly reduced the N_2O emission and the efficiency of reduction varied across the tested cropping systems (Figure 7). The annual total N_2O emissions were decreased by 12%, 8%, 13%, and 2%, respectively, for the wheat, lettuce, vineyard, and almond orchard.

Reducing N application would reduce concentrations of soil mineral N and thus the N_2O emissions, especially in cropping systems with intensive N inputs. Compared to the baseline, a decrease of N application by 20% would lower the annual N_2O emissions by 24%, 32%, and 8% for the wheat, lettuce, and almond orchard, respectively (Figure 7).

Irrigation practices affected N_2O emissions as well, and the simulated N_2O emissions under low-water irrigation were generally lower than those receiving high-water irrigation (Figure 7). Compared to the baseline, reducing the water inputs in furrow-irrigated alfalfa fields (1,290 mm in the baseline) to sprinkler systems with lower water inputs (711 mm in A1 and 508 mm in A2, Table 2) decreased the N_2O emission by 21% and 30%, respectively. By contrast, increasing the water input from 227 to 381 mm in lettuce and from 965 to 1300 mm in almonds increased the N_2O emissions by 2% and 33%, respectively.

Finally, changing cover crop management is another practice that can influence N_2O emissions substantially. N_2O emissions can be reduced by up to 51% in vineyards planted with nonleguminous cover crops relative to no cover crops and 63% relative to leguminous cover crops, if the N fertilizer application remains unchanged (Figure 7).

4. Discussion

4.1. DNDC Tests

In this study, we tested a process-based biogeochemical model, DNDC, against field measurements of N_2O emissions from five cropping systems representing a wide range of soil types and FMPs. The comparisons between the simulations and field records demonstrated that DNDC reliably predicted the seasonal and annual N_2O emissions from the cropping systems studied, despite variability in characteristics of N_2O emissions and regulating factors for N_2O emissions across these systems (Figure 6 and Table 4). The observed impacts of N fertilization practices for wheat and lettuce, and of different age-stand for alfalfa on N_2O emissions, and the spatial variability of N_2O fluxes in both vineyard and almond orchard were also generally captured by the model. DNDC clearly performed better than the EF approach in simulating seasonal

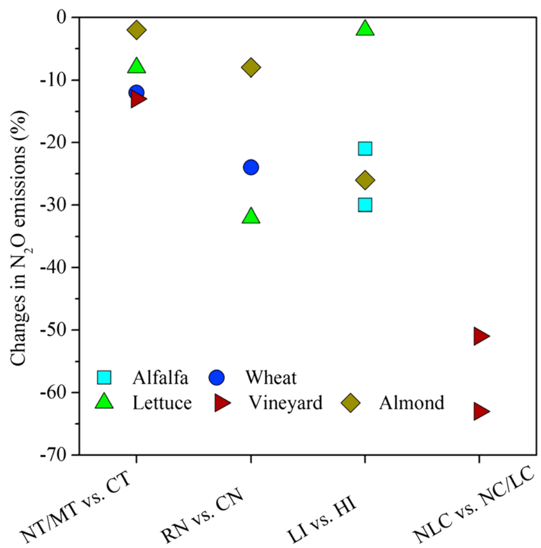

Figure 7. Changes in N_2O emissions as a percentage of the reference treatment, for comparisons between treatments with versus without potential mitigation options. NT = no tillage; RT = reduced tillage; CT = conventional tillage; RN = reduced nitrogen application; CN = conventional nitrogen application; LI = low-volume irrigation; HI = high-volume irrigation; NLC = cultivation of nonleguminous cover crops; NC = no cover crop; LC = cultivation of nonleguminous cover crops.

and annual N_2O emissions in these systems (Figure 6). The EF-based N_2O emissions were not significantly correlated with the field observations ($P > 0.1$), indicating that using a fixed EF value cannot reliably estimate the N_2O emissions from the diverse cropping systems studied. Furthermore, the EF approach underestimated the N_2O emissions from alfalfa fields and vineyards in Colusa County, where the high N_2O emissions may be partially attributable to planting leguminous crops capable of fixing atmospheric nitrogen gas (N_2), or to intensive irrigation, rather than solely to N fertilizer application (Burger et al., 2016; Burger & Horwath, 2012; Garland et al., 2014). The capability of DNDC to capture impacts of factors other than the N fertilizer application on N_2O emissions is important for improving inventory estimates, but it is also essential for identifying and evaluating management options that can reduce N_2O emissions. In addition, the model generally captured the episodic patterns of daily N_2O fluxes (Figures 1–5). These results demonstrate that process-based models, like DNDC, are better than the EF approach in quantifying N_2O emissions, as well as their mitigation potentials, for cropping systems with diverse management practices. This conclusion is consistent with previous studies that also showed DNDC was more accurate than the IPCC EF methods in predicting N_2O emissions from cropping systems (e.g., Uzoma et al., 2015). We note that the agreement between the simulations and observations in seasonal and annual N_2O emissions could sometimes have resulted from compensating discrepancies between the simulated and observed daily N_2O fluxes.

There are discrepancies between the simulated seasonal or annual emissions and field measurements. For example, DNDC underestimated the N_2O emissions from vineyard alleys in Colusa during March 2009 to March 2010 (3.00 versus 5.39 kg N_2O-N ha^{-1}, Table 4) and overpredicted the emissions from the fifth-year alfalfa fields (6.42 versus 5.20 kg N_2O-N ha^{-1}, Table 4). These discrepancies could be partially due to uncertainties in data processing of field records for estimating seasonal or annual N_2O emissions. In this study, all N_2O measurements used for model tests were carried out at irregular intervals and were event based. Errors may occur in calculating seasonal or annual N_2O emissions from the discrete flux measurements, due to extrapolation or interpolation uncertainties (Parkin, 2008). In addition, there are uncertainties in model inputs. For example, if on-site observations were unavailable, weather data from nearby meteorological stations and typical soil property values were used as model inputs. Both climate and soil properties have substantial influence on N_2O emissions in DNDC (Li et al., 1992a), and therefore, potential biases in these inputs could affect the simulated N_2O fluxes.

We also identified some discrepancies that may result from inaccurate mechanisms or algorithms in the model. After most but not all heavy rainfall events across all tested cropping systems, the DNDC model predicted large pulses of N_2O fluxes. These simulated pulses were due to enhanced denitrification resulting from an increase in the soil anaerobic volume under heavy precipitation. However, flux pulses were not always observed in the field (e.g., late January 2010 and late March 2011 for wheat, Figure 2; late January 2010 for lettuce, Figure 3; mid April and late October 2010 for almond, Figure 5). Possible explanations for these more frequent N_2O pulses predicted by DNDC include (1) model overestimation of the duration of soil anaerobic conditions or of the sensitivity of denitrification rate to changes in soil anaerobic conditions, (2) discrepancies in simulating denitrifier substrates (soil nitrate and DOC) during these occasions, (3) underestimation of the impact of factors on reducing denitrification (e.g., less favorable soil nitrate content, DOC, and soil temperature), and/or (4) discrepancies in simulating the loss rate of N_2O during denitrification. It should be noted that the reasons for the model-observation discrepancies may be different across the different cropping systems. The overpredicted N_2O pulses following heavy precipitation probably did not result from overestimations of soil nitrate for those cropping systems with zero or low N application, since DNDC predicted relatively low soil mineral N status for them. However, it is hard to identify the reasons for the overpredicted N_2O pulses for other cropping systems, because DNDC was only tested against the N_2O flux data in this study. The overall denitrification rate and N_2O production and consumption rates in denitrification are all highly uncertain, and there is not a thorough understanding of underlying mechanisms controlling these variations (e.g., Bouwman et al., 2013; Butterbach-Bahl et al., 2013). To reduce the biases of N_2O fluxes after heavy rainfall that may result from inaccurate mechanisms, further studies should focus on clarifying processes or mechanisms controlling the impacts of soil anaerobic condition and other factors on denitrification rates and N_2O emission.

4.2. Practices Influencing N_2O Emission and Potential Mitigation Options

The DNDC simulations demonstrate that tillage, the amount of applied N fertilizers, irrigation, and management of cover crops influence N_2O emissions. The model predicted lower emission rates under the practices of reduced or no tillage, reducing the rate of N application, low-volume irrigation, or cultivation of nonleguminous cover crops. However, the efficiencies for N_2O mitigation were variable across the studied cropping systems and practices (Table 2 and Figure 7), suggesting that crop-specific FMPs should be designed for N_2O mitigation in regions with diverse cropping systems.

The simulations also provide information about the mechanisms responsible for the impacts of these FMPs on N_2O emission. Both reduced and no tillage (versus conventional tillage) and reducing the rate of N application decreased the content of soil mineral N, either indirectly by reducing mineralization of soil organic matter (tillage) or directly by reducing synthetic N addition. This resulted in 2% to 32% lower N_2O emission rates (Figure 7). Although reduced or no tillage can enhance N_2O emission through increasing soil water content and denitrification, this did not occur in these Mediterranean agricultural ecosystems, because the relative dry soil water status was usually lower than the threshold favoring denitrification (Rochette, 2008).

Irrigation practices affected N_2O emissions primarily through influencing soil water and oxygen status. The model simulated higher N_2O emissions under flood irrigation, primarily because of transient conditions of near saturation after each irrigation event. This, in combination with warm summer temperatures and relatively high soil nitrate generated during periods without irrigation, could lead to optimal conditions for denitrification and N_2O production (Davidson & Verchot, 2000). In contrast, lower N_2O emissions simulated under practices of low-volume irrigation (e.g., drip irrigation) were due to restricted N_2O production through denitrification, because dry soil conditions were not strongly affected by the low amount of water applied and the limited water distribution in soil profiles (Kallenbach et al., 2010) following the frequent low-volume irrigation events.

The DNDC simulations also demonstrate that reduced N_2O emission in the vineyard with nonleguminous cover cropping was primarily due to immobilization of soil residual N through N uptake by the cover crop, which led to a low availability of soil mineral N during winter rainy season. In contrast, planting of a leguminous cover crop in winter added more N into the system and therefore stimulated N_2O emissions.

However, it should be noted that we only simulated short-term impacts of the FMPs on N_2O emission, and long-term impacts arising from persistently applying these practices may differ from the predicted short-term impacts. For example, strategies that can help increase soil C sequestration and decrease short-term N_2O emission (e.g., reduced or no tillage, Figure 7) could also stimulate N_2O emission over the long term due to the slow accumulation of SOC (Li et al., 2005). It should also be noted that the efficiency of the FMPs on N_2O mitigation is highly variable, depending on specific conditions of climate, soil, and crop species. For example, the model predicted different emission mitigation efficiencies across different cropping systems for a single practice (Figure 7). Therefore, evaluation of these FMPs by considering long-term impacts on N_2O emission, climate variability, and specific environmental conditions are needed before they can be recommended for adoption.

5. Conclusions

A process-based biogeochemical model, DNDC, was tested against field measurements of N_2O emissions from five cropping systems representing a range of environmental conditions and FMPs. The results indicate that DNDC reliably predicted the seasonal or annual N_2O emissions from the studied cropping systems without statistical biases. The model also captured the impacts of the setting treatments, including different N fertilization and alfalfa age, on N_2O emission, and the spatial variability of N_2O fluxes in both vineyards and almond orchards. DNDC clearly performed better than the EF approach in simulating seasonal or annual

N_2O emissions from the cropping systems studied. The model tests also suggest that DNDC overestimated N_2O fluxes following some heavy rainfall events. To reduce the biases of N_2O fluxes following heavy rainfall, further studies should focus on clarifying processes or mechanisms controlling the impacts of soil anaerobic conditions and other factors on the dynamics of denitrification rates and N_2O emission. After testing against the field measurements, the model was applied to assess impacts on N_2O emissions of alternative management practices, including tillage, fertilization, irrigation, and planting of cover crops. The simulations demonstrated that N_2O emissions could be mitigated by reducing tillage, reducing the N application rate, using low-volume irrigation, and reducing the period of fallow through cultivation of nonleguminous cover crops, although the efficiencies for N_2O mitigation were variable across the cropping systems and practices.

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