APPLIED ECOLOGY

Inconsistencies undermine the conclusion that agriculture is a dominant source of NO_X in California

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Almaraz *et al*. reported that agricultural soils are a dominant source of NO_x pollution in California (20 to 32% of total statewide NO_x emissions). However, this conclusion may be undermined by the lack of agreement between their modeled estimates and previously reported empirical measurements, the extrapolation of NO_x fluxes during hot moments to derive annual estimates, and the overestimation of nitrogen fertilizer consumption in California.

Almaraz *et al*. (*1*) estimated that the mean NO_x emission flux from California's agricultural soils was 19.8 kg of N ha^{-1} $year^{-1}$, in comparison to 1 kg of N ha^{-1} $year^{-1}$ from natural ecosystems. The study estimated that 161,100 metric tons of NO_x-N $year^{-1}$ was emitted from California's soils, and that 127,000 metric tons of this was derived from agricultural land. A bottom-up model was scaled up on the basis of the N surplus and theoretical NO, N_2O, and N_2 partitioning based on soil conditions, and verified using top-down airborne NO_x measurements. Using these two approaches, the study identified agricultural regions in California that are potential hot spots for NO_x emissions due to the spatial distribution of soil environmental conditions and N fertilizer inputs. We agree that accurate accounting of all sources of NO_x is important, particularly for the management and mitigation of direct emissions from agricultural soils.

The magnitude of NO_x emitted from agricultural soils presented in this work, however, is considerably greater than the magnitudes reported in previously published work. Given that background NO_x fluxes were estimated at 1 kg of N ha^{-1} $year^{-1}$ and fertilizer inputs averaged 132 kg ha^{-1}, attributing 19.8 kg of NO_x-N ha^{-1} $year^{-1}$ to agricultural land implies a fertilizer-induced emission factor in excess of 14%. A recent global meta-analysis reported a fertilizer-induced NO emission factor of 1.2% (*2*). Another earlier global assessment concluded that the emission factor for NO is similar to that for N_2O (*3*). While measurements in California were not included in previous global assessments, measurements from similar agroecosystems were included. The assertion that the state of California has twice the NO flux and an emission factor eightfold greater than the previous global assessment for vegetable croplands (*2*) warrants a high degree of scrutiny. Emission factors can vary regionally owing to soil texture, climate, and other factors, but Almaraz *et al*. did not discuss their findings within the context of these global assessments nor provide an analysis of uncertainty needed to interpret their estimate.

The authors concluded that their estimates of total NO_x flux were "slightly higher than, although comparable with, the few number of empirical [chamber-based] measurements of NO_x emissions from the San Joaquin Valley's cropland soils (made between July and September 1995)." This discrepancy was attributed to greater fertilizer use and population growth in the last 20 years.

However, three potential issues arise from this argument. First, 45% of the modeled values were more than twofold greater than these averaged empirical measurements, and 31% were at least an order of magnitude larger. The paper cites a study that included 2850 NO_x measurements at 26 sites in 10 different cropping systems and reported average peak (noon-time) and diel (adjusted for lower night-time temperature) fluxes of 4.0 and 1.9 kg of N ha^{-1} $year^{-1}$, respectively, if hourly summer fluxes are extrapolated to annual emissions. Even the agreement between the top-down and bottom-up estimates reported in Almaraz *et al*. is difficult to assess without an analysis of model uncertainty; and even when canopy exchange is considered, the bottom-up modeled flux for July and August is 32 to 47% greater than the top-down estimate based on airborne measurements collected in July and August.

Second, too much weight has been given to the hot moments of measured emissions when scaling up temporally. The best agreement between modeled and observed high-magnitude NO fluxes reported in Almaraz *et al*. occurs in the Imperial Valley, CA. The fluxes modeled by Almaraz *et al*. appear to agree with chamber-based measurements published in Oikawa *et al*. (*4*), which was reported by Almaraz *et al*. as 21 kg of N ha^{-1} $year^{-1}$ in table 1. Almaraz *et al*. derived this estimate by extrapolating an average flux of 66 ng of NO_x-N m^{-2} s^{-1} to represent annual emissions. This extrapolation is inappropriate because the NO measurements by Oikawa *et al*. were largely designed to characterize emission pulses following summer fertilization. Because of the aims of that research, the measurements are weighted toward high-emission periods. Furthermore, Oikawa *et al*. reported integrated emission factors of 1.8 to 6.6%, less than the emission factor if the annual average emission rate were 21 kg of N ha^{-1} $year^{-1}$ (~11%, if averaged across both small and large field measurements). Excluding or down-adjusting the Oikawa *et al*. "Observed NO" from table 1 limits the range of the empirical data and further challenges the quantitative coherence between modeled and observed values.

Third, the perception that state-wide N fertilizer consumption in agriculture has increased since the 1990s is not supported by the available data. California fertilizer N sales plateaued in the early 2000s (*5*). According to the Association of American Plant Food Control Officials, annualized mineral N fertilizer sales have declined to pre-2000 levels (*6*). The California Department of Food and Agriculture reports the sales of Home and Garden fertilizer blends and compost for organic agriculture under code 0 ("Identified by Grade"). Following reporting changes in 2012, 44% of total fertilizer sales have fallen in this category, resulting in a tonnage increase by more than 300,000 metric tons (*7*). However, Almaraz *et al*.

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did not assess nonagricultural fertilizer inputs, such as for Home and Garden fertilizer use, in urban areas. Future nitrogen modeling groups using these data should be aware of these reporting differences, as well as the need to differentiate among the uses and sources of nitrogen fertilizers (agricultural versus nonagricultural uses) to properly interpret the model outputs.

The impact of these inconsistencies is difficult to assess because Almaraz *et al*. did not present their N budget for California. The N rate and total emission data estimated in this study infer an agricultural area of 6,423,737 ha and a natural ecosystem area of 33,810,000 ha. The total annual N fertilizer inputs to agriculture would thereby amount to 848,000 metric tons. This estimate exceeds fertilizer N sales in California from 1980 to 2007, a period for which Almaraz *et al*. used crop-specific data to generate N inputs, by almost 50% (*6*). It also well exceeds total nitrogen fertilizer sales in 2016, even when additional sources of nitrogen fertilizers are included (*7*). It is unclear whether this estimate includes other sources of reactive N (that is, N from biological fixation and deposition), although the authors state that manure N was not included. Is it possible that the authors overestimated fertilizer N inputs? Or are the authors using the term "fertilizer" when they actually meant "reactive" N?

We ask that the authors upload their complete N budget and assumptions for water-filled pore space, along with an uncertainty analysis, for California as supplemental data. This will help the reader properly compare and otherwise address the apparent inconsistencies in emission factors and allow for an improved assessment of the role of agriculture as a source of NO_x in California.

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