

# Increasing importance of deposition of reduced nitrogen in the United States

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**Rapid development of agriculture and fossil fuel combustion greatly increased US reactive nitrogen emissions to the atmosphere in the second half of the 20th century, resulting in excess nitrogen deposition to natural ecosystems. Recent efforts to lower nitrogen oxides emissions have substantially decreased nitrate wet deposition. Levels of wet ammonium deposition, by contrast, have increased in many regions. Together these changes have altered the balance between oxidized and reduced nitrogen deposition. Across most of the United States, wet deposition has transitioned from being nitrate-dominated in the 1980s to ammonium-dominated in recent years. Ammonia has historically not been routinely measured because there are no specific regulatory requirements for its measurement. Recent expansion in ammonia observations, however, along with ongoing measurements of nitric acid and fine particle ammonium and nitrate, permit new insight into the balance of oxidized and reduced nitrogen in the total (wet + dry) US nitrogen deposition budget. Observations from 37 sites reveal that reduced nitrogen contributes, on average, ~65% of the total inorganic nitrogen deposition budget. Dry deposition of ammonia plays an especially key role in nitrogen deposition, contributing from 19% to 65% in different regions. Future progress toward reducing US nitrogen deposition will be increasingly difficult without a reduction in ammonia emissions.**

ammonia | dry deposition | wet deposition | nitrogen oxides | agriculture

Beginning in the mid-20th century, emissions of anthropogenic reactive nitrogen ( $N_r$ ) to the atmosphere accelerated rapidly due to increased fossil fuel combustion and intensive agricultural activities (1–4). Once emitted to the atmosphere,  $N_r$  compounds are deposited to terrestrial and aquatic ecosystems through dry and wet processes. Although nitrogen is an essential and often limiting element for ecosystems, increases in  $N_r$  deposition resulting from increased emissions have raised concerns around the world due to its adverse environmental impacts, including decreased biological diversity, increased soil acidification, and lake eutrophication (5–9). Critical loads (CLs) have been widely used to quantify levels of  $N_r$  deposition that ecosystems can sustain without significant harmful effects (10, 11). Twenty-four of the 45 national parks in the contiguous United States were estimated to receive  $N_r$  deposition in 2013 exceeding the local CL (12). Atmospheric  $N_r$  (an important ingredient of ozone and fine particle formation) has also been linked with climate change and human health degradation (8, 13, 14).

Atmospheric  $N_r$  sources are dominated by emissions of nitrogen oxides ( $NO_x = NO + NO_2$ ) and ammonia ( $NH_3$ ) (8).  $NO_x$  is produced by a wide range of high temperature processes including lightning and the combustion of fossil fuels by vehicles, electric power generating units, and other industrial and natural combustion sources.  $NO_x$  is oxidized in the atmosphere and converted to a variety of forms, including nitric acid, with a short timescale (typically 1 d or less). Reis et al. (15) attributed more than 80% of the  $NH_3$  emissions in the United States to the

agricultural sector, including emissions from livestock waste and volatilization of N-based fertilizer.

During the last two decades, US  $NO_x$  emissions have steadily declined due to effective regulations designed to decrease  $NO_x$  contributions to ozone, fine particles, and acid deposition (16). Data from the National Emissions Inventory (NEI) Air Pollutant Emissions Trends (<https://www.epa.gov/air-emissions-inventories/air-pollutant-emissions-trends-data>) indicate that  $NO_x$  emissions decreased by nearly 41% from 1990 to 2010. Further reductions are expected in coming years due to additional policy actions (e.g., the Cross-State Air Pollution Rule, reductions in mobile source emissions, and 5-y reviews of the National Ambient Air Quality Standards). Meanwhile,  $NH_3$  emissions have been reported to increase by 11% between 1990 and 2010 (17), with contributing factors including regional growth in livestock numbers and increased application of  $NO_x$  controls such as selective catalytic reduction. Unlike  $NO_x$ ,  $NH_3$  emissions in the United States are not regulated. In 2006, US anthropogenic  $NH_3$  emissions were estimated at 2.8 Tg N/y; they are projected to increase to between 3.3 and 4.2 Tg N/y by 2050, mainly due to increases in N fertilizer application and livestock growth (12).

## Significance

Human activities have greatly increased emissions of reactive forms of nitrogen to the atmosphere. This perturbation to the nitrogen cycle has produced large increases of nitrogen deposition to sensitive ecosystems. Over recent decades, attention has focused on wet and dry deposition of nitrate stemming from fossil fuel combustion emissions of nitrogen oxides. Successful decreases in nitrogen oxides emissions in the United States have substantially decreased nitrate deposition. By contrast, emissions of ammonia, an unregulated air pollutant, and resulting deposition of ammonium have grown. Expanded observations demonstrate that deposition of reactive nitrogen in the United States has shifted from a nitrate-dominated to an ammonium-dominated condition. Recognition of this shift is critical to formulating effective future policies to protect ecosystems from excess nitrogen deposition.

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Nitrate wet deposition decreases were largest in the Northeastern United States, an area where large NO<sub>x</sub> emissions reductions were implemented. Lehmann and Gay (26) examined trends in nitrate concentrations in US wet deposition in detail for a period ending in 2009 and also highlight large reductions in the Northeastern United States.

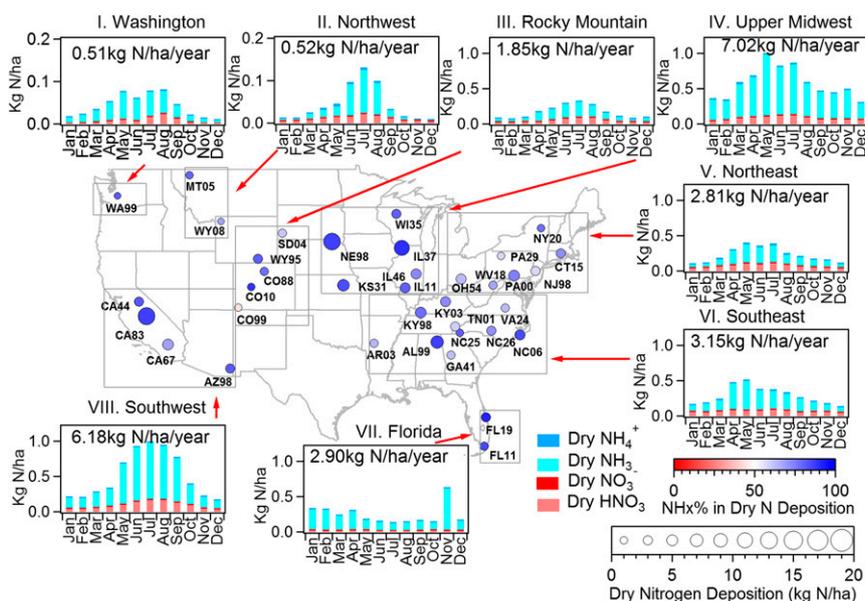
Thirty-seven of 45 states experienced increased ammonium wet deposition over the last two decades; for these states, the average increase was 22% (Fig. 2). Increases in ammonium wet deposition were especially common in the northern plains states; relatively large increases were also seen in North Carolina, Kentucky, Maryland, and New Jersey. Substantial increases in ammonium ion concentrations in precipitation in the Central and Western United States were previously reported through 2004 by Lehmann et al. (27). The increasing NH<sub>4</sub><sup>+</sup> wet deposition is broadly consistent with the estimates of increasing NH<sub>3</sub> emissions since the 1990s (17).

**Oxidized vs. Reduced Dry Inorganic N Deposition.** Gas phase nitric acid and ammonia and particulate ammonium and nitrate are potentially important contributors to dry inorganic N deposition. Limited historical measurements, especially for ammonia, prevent an analysis of long-term trends of oxidized vs. reduced dry inorganic nitrogen deposition like those presented above for wet deposition. Recent efforts to measure gas phase ammonia concentrations more routinely by the NADP Ammonia Monitoring Network (AMoN) and Interagency Monitoring of Protected Visual Environments (IMPROVE) NH<sub>x</sub> networks, however, allow comparison of the current balance between oxidized and reduced inorganic N dry deposition. We focus here on characterizing spatial patterns for the period 2011–2013. Fig. 3 illustrates (by circle size) the current magnitude of dry inorganic N deposition across the United States. Significant spatial variability is seen from site to site, reflecting difference in species concentrations. Estimated annual sums of dry deposition by gaseous ammonia and nitric acid and particulate ammonium and nitrate range from 0.49 (WY08) to 13.4 kg N/ha per year (NE98). Reduced N contributes more than 50% of the total calculated dry inorganic N deposition at all sites except Mesa Verde National

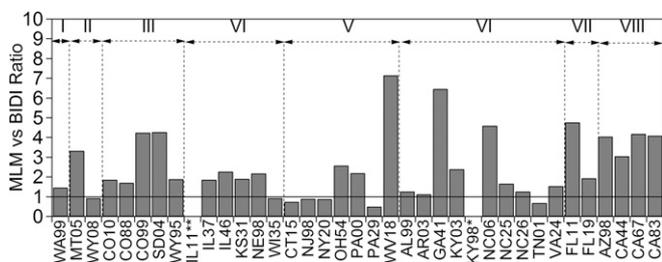
Park (CO99; 44%) in southwest Colorado. This remote arid site is expected to have relatively small agricultural impacts (28) but greater influence of NO<sub>x</sub> emitted from nearby oil and gas development (29) and the large, coal-fired Four Corners and San Juan power plants. The highest fractional and absolute reduced N contributions are seen, not surprisingly, in areas with substantial agricultural activity, including sites in Illinois (IL37 exhibits the highest reduced N fraction at 90%), Nebraska, and the Central Valley of California.

To examine overall dry deposition patterns, sites were grouped into eight regions (by proximity and similar trends) as follows (Table S1 and Fig. 3): Washington (I), Northwest (II; Montana and northern Wyoming), Rocky Mountain (III; western South Dakota and southern Wyoming, CO), Upper Midwest (IV; Wisconsin, Illinois, eastern Kansas, and eastern Nebraska), Northeast (V; New York, Connecticut, New Jersey, Pennsylvania, Ohio, and West Virginia), Southeast (VI; Kentucky, Virginia, Tennessee, North Carolina, Georgia, Alabama, and Arkansas), Florida (VII), and Southwest (VIII; California and southern Arizona). The lowest regional average dry N deposition flux was found in the Washington region (0.51 kg N/ha per year) and the highest in the Upper Midwest (7.02 kg N/ha per year), one of the nation's primary food production areas with large NH<sub>3</sub> emissions from livestock and fertilizer use.

In most regions, dry ammonia and nitric acid deposition display strong seasonal patterns, with higher values in summer and lower values in winter. These seasonal patterns are driven mostly by seasonal concentration patterns rather than changes in deposition velocity. Ammonia emissions increase with warmer summertime temperatures due to enhanced volatilization (30, 31). Active summertime photochemistry speeds conversion of NO<sub>x</sub> to nitric acid, whereas warmer summertime temperatures reduce formation of particulate ammonium nitrate, leaving more nitric acid and ammonia in the gas phase (32). Interestingly, dry NH<sub>3</sub> deposition is elevated during the winter in the Upper Midwest compared with other regions. Higher winter ammonia concentrations in this region might reflect trapping of cold season ammonia emissions (from livestock and/or winter fertilizer application) near the surface by a shallow boundary layer (28). Dry N deposition exhibits a winter seasonal maximum in Florida. Increased summertime precipitation



**Fig. 3.** Spatial and temporal trends in dry inorganic N deposition at 37 locations across the United States. Included are deposition of gaseous nitric acid and ammonia and PM<sub>2.5</sub> ammonium and nitrate. Fractional reduced N contributions are represented by circle color. The total deposition from these four species is indicated by circle size. The bar charts depict monthly average contributions of individual dry reduced and oxidized N deposition pathways for eight selected regions. The average total dry inorganic N deposition fluxes in different regions are shown by the number in each figure.



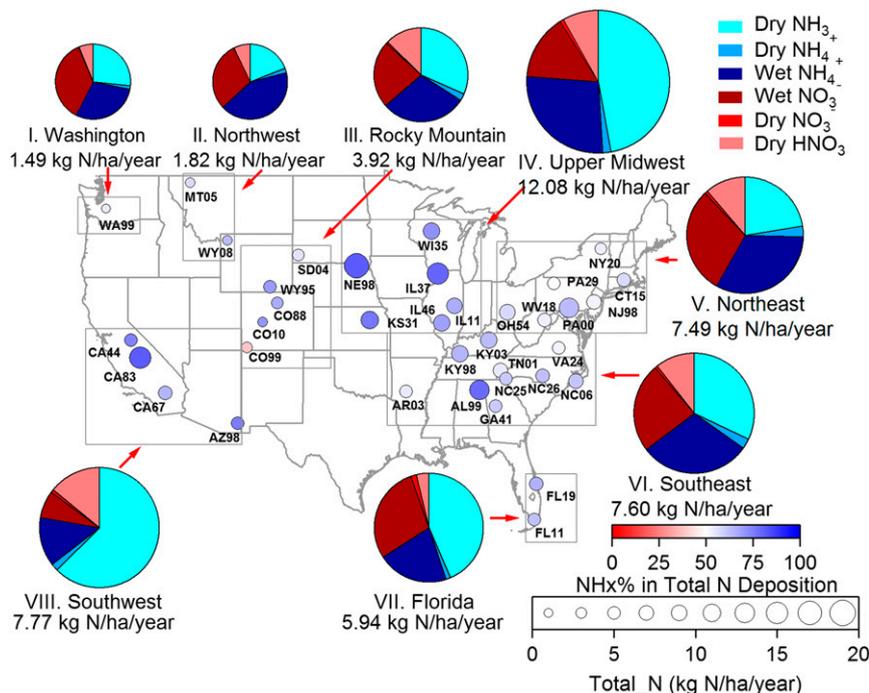
**Fig. 4.** Ratio of annual  $\text{NH}_3$  dry deposition rates estimated using the MLM vs. bidirectional approaches. Regions are indicated at top of graph. \*Due to a lack of meteorological data, the bidirectional flux model is not parameterized appropriately for site KY98. \*\*Due to vegetation type, the bidirectional flux model is not parameterized appropriately for site IL11.

here suppresses summertime atmospheric concentrations and therefore dry deposition of reduced and oxidized N species. In Florida wet N deposition contributed more than 75% of total (wet + dry) inorganic N deposition during summer when there was more precipitation (Fig. S1); dry deposition of reduced N was the dominant input during the drier winter season.

At the annual scale, ammonia dry deposition rates estimated using the multilayer model (MLM) approach are larger than those derived from the bidirectional model by a factor of 1.90 (median MLM/bidirectional flux ratio of 35 sites listed in Fig. 4). A reduction in  $\text{NH}_3$  dry deposition rates, relative to the unidirectional flux framework, was also observed on implementation of  $\text{NH}_3$  bidirectionality in the Community Multiscale Air Quality Model (33). MLM vs. bidirectional model differences vary across regions but generally result from stomatal and ground compensation points, as well as the effects of surface acidity, represented in the bidirectional framework. The net result of these processes is to reduce the atmosphere-surface  $\text{NH}_3$  concentration gradient,

and therefore the flux, relative to the unidirectional MLM deposition velocity approach, which assumes a zero surface concentration. Model differences are generally greatest in summer (Fig. S2) when temperature driven stomatal and soil compensation points are at a maximum. On average, the bidirectional and MLM approaches yield comparable net fluxes during winter when compensation points are lowest and surfaces are more acidic. Further discussion of the MLM vs. the bidirectional model is included in *SI Methods*.

The relatively large overall differences between the MLM and bidirectional  $\text{NH}_3$  flux estimates warrant brief discussion of the significant uncertainties that persist in modeling dry deposition of reactive nitrogen. MLM  $\text{HNO}_3$  deposition velocities, on which the  $\text{NH}_3$  deposition velocity used here is based, may contain up to  $\pm 25\%$  uncertainty related to error in the measurements that drive the model and underlying process parameterizations (34). Models of  $\text{HNO}_3$  deposition velocity also differ substantially themselves. For example, a multisite evaluation of the MLM and Big Leaf Model used in the Canadian Air and Precipitation Monitoring Network (CAPMoN) showed a median bias in hourly  $\text{HNO}_3$  deposition velocities of  $-35\%$ , with MLM yielding lower values (35). Regarding bidirectional  $\text{NH}_3$  models, comparisons to average measured  $\text{NH}_3$  fluxes over nonagricultural ecosystems generally demonstrate agreement within  $\pm 30\%$  using site-specific process parameterizations (36–39), although differences can be much larger under specific meteorological and surface conditions. Uncertainty may also be significantly larger when applying generalized parameterizations as done here. In that regard, recent versions of bidirectional models (40–42) have not yet been rigorously compared with each other or against flux measurements for natural ecosystems in North America. Although our analyses use commonly used approaches for both  $\text{HNO}_3$  deposition velocity and bidirectional  $\text{NH}_3$  flux, the abovementioned uncertainties are included to emphasize that the dry deposition component of the N deposition budget is significantly more uncertain



**Fig. 5.** Spatial trends in total reactive inorganic N deposition across the United States from July 2011 to June 2013. Fractional reduced N contributions to total N deposition (dry + wet) at the 37 sites are represented by circle color. The total inorganic nitrogen deposition is indicated by circle size. The pie charts show average fractional contributions of individual reduced and oxidized N deposition pathways for the eight regions, with each pie area proportional to the average total inorganic nitrogen deposition (also listed under each pie).

than the wet fraction, a point that should be considered in the interpretation of our results.

**Fractional Reduced N Contributions to the Total Inorganic N Deposition Budget.** With wet and dry deposition estimates available for 37 locations, the total wet plus dry nitrogen deposition budgets can be estimated across the United States (Fig. 5). Fractional deposition contributions by each wet and dry deposition pathway for each of the eight regions are also illustrated in Fig. 5. Reduced N deposition fractions in the eight regions range from 58% (Washington, I) to 78% (Southwest, VIII), with dry  $\text{NH}_3$  deposition alone contributing between 19% (Northwest, II) and 63% (Southwest, VIII). Fractional reduced N contributions at individual sites range from 42% at CO99 (Mesa Verde National Park) to 84% at CA83 in California's Central Valley. Ammonia dry deposition fractions ranged from 11% (PA27) to 74% (CA83). The spatial patterns of reduced N deposition fraction generally reflect spatial variations in agricultural activity including animal husbandry. Assuming that the biases between the MLM deposition velocity and bidirectional flux approaches shown in Fig. 4 are generally representative, a full assessment using the bidirectional approach would, at many sites, reduce overall deposition rates and the relative fraction of  $\text{NH}_3$  dry deposition. However, the general pattern observed in Fig. 5 remains consistent;  $\text{NH}_x$  still contributes the majority of inorganic N deposition at the national scale.

The site-specific circle sizes in Fig. 5 indicate the combined wet plus dry inorganic N deposition fluxes. Some regions exhibit majority dry deposition [e.g., dry deposition contributions of 58% and 79% in the Upper Midwest (IV) and Southwest (VIII), respectively], whereas others are more strongly influenced by wet deposition [e.g., wet deposition contributions of 66% and 72% in the Washington (I) and Southeast (VI) regions, respectively]. The largest deposition fluxes at individual sites tend to be observed at locations where fractional reduced N contributions are large. The maximum regional average inorganic N deposition flux (12.1 kg N/ha per year) was observed in the Upper Midwest region (IV); relatively large deposition fluxes were also observed for California and the eastern United States. These spatial patterns are similar to those identified in recent model simulations (43).

**Implications and Summary.** Increases in agricultural emissions of ammonia and the success of regulatory policies in decreasing  $\text{NO}_x$  emissions over the last two decades are changing the face of US reactive nitrogen deposition. Although US wet inorganic N deposition was once dominated by nitrate, wet inorganic N deposition now comes mostly from ammonium at nearly 70% of US monitoring sites. Although estimates of dry deposition fluxes of inorganic N inherently contain more uncertainty, dry and total (wet plus dry) inorganic N deposition fluxes also appear to be dominated by reduced N in most parts of the country. Decreases in wet and dry deposition fluxes of oxidized inorganic N species are expected to continue into the future as the United States continues to lower  $\text{NO}_x$  emissions. Current projections of ammonia emissions growth, meanwhile, suggest that reduced N deposition levels will grow in the future. In addition to the adverse impacts of reduced N deposition on ecosystem health, ammonia is an important precursor to fine particle formation. Fine particles decrease visibility (44) and negatively impact human health and increase health care costs (45, 46). Reductions in US ammonia emissions from agricultural and nonagricultural sources, whether by regulation or voluntary actions (e.g., agricultural producer adoption of best management practices), would yield a variety of positive

benefits for ecosystems and society. Increased study of atmospheric ammonia concentrations and improved measures of ammonia dry deposition fluxes are needed to design optimal strategies for achieving such benefits.

## Methods

Weekly precipitation concentrations of  $\text{NH}_4^+$  and  $\text{NO}_3^-$  were obtained from the NADP National Trends Network (NTN; [nadp.isws.illinois.edu/ntn/](http://nadp.isws.illinois.edu/ntn/)). Weekly gaseous  $\text{HNO}_3$  concentrations and particulate  $\text{NH}_4^+$  and  $\text{NO}_3^-$  concentrations were obtained from the Clean Air Status and Trends Network (CASTNET; <https://www.epa.gov/castnet>). Biweekly concentrations of gaseous  $\text{NH}_3$  were taken from the NADP AMoN ([nadp.isws.illinois.edu/AMoN/](http://nadp.isws.illinois.edu/AMoN/)). To gain greater spatial coverage of airborne  $\text{NH}_3$  concentrations, especially in the western United States,  $\text{NH}_x$  ( $\text{NH}_3 + \text{NH}_4^+$ ) measurements from a pilot IMPROVE  $\text{NH}_x$  monitoring network (28) were also used. More detailed information about these observation networks can be found in Table S2.

Wet deposition data were obtained from NTN sites for the periods 1990–1992 and 2010–2012. The number of sites analyzed changed due to network development over this period. From 1990 to 1992, there were 195 sites; 238 sites were available for the 2010–2012 period. Sites were not included if data were unavailable for  $\geq 1$  y in either period examined.

Oxidized and reduced N gas and particle concentrations were obtained for 37 sites (Table S1) where NTN and CASTNET monitoring stations were collocated with AMoN and/or IMPROVE  $\text{NH}_x$  sites. At 30 of these locations, 2 y of measurements (July 2011 to June 2013) were available. The remaining sites had data availability of at least 1 y. Concentrations of all species that contribute to the N deposition budget are not measured at these sites. Important missing compounds include inorganic (e.g.,  $\text{NO}_2$ ) and organic N (e.g., alkyl nitrates, peroxyacetyl nitrate, and amines) species. Wet deposition of organic N is also not routinely measured and therefore not considered in this analysis.

Wet N deposition was determined from the amount of total precipitation and the aqueous concentrations of  $\text{NH}_4^+$  and  $\text{NO}_3^-$ . Dry N deposition was calculated for each species as the product of the N species concentration and a deposition velocity. Deposition velocities of gaseous  $\text{HNO}_3$  and particulate  $\text{NH}_4^+$  and  $\text{NO}_3^-$  were provided by CASTNET for each of its measurement sites based on the MLM (47), with input of on-site meteorology and local site characteristics. Gaps in the meteorological data were addressed by using the CASTNET substitution method (48). The deposition velocity of  $\text{NH}_3$  is difficult to determine due to the bidirectional nature of the dry  $\text{NH}_3$  flux that depends strongly on local conditions (40). To estimate dry  $\text{NH}_3$  deposition here, its deposition velocity was calculated as 70% of the  $\text{HNO}_3$  deposition velocity following previous estimates (49–51). A review of field observations suggests that the  $\text{NH}_3$  deposition velocity is at least half and perhaps as high as the  $\text{HNO}_3$  deposition velocity. Our choice of 70% agrees well with the findings of Neiryneck et al. (52) and Nemitz et al. (39).

This  $\text{NH}_3$  deposition velocity approach is a simple approximation of unidirectional air-surface exchange, ignoring important bidirectional exchange processes that influence the magnitude and direction of the flux. To assess the potential importance of these bidirectional exchange processes and their impact on annual reactive N deposition budgets,  $\text{NH}_3$  fluxes derived from the unidirectional approach were compared with fluxes estimated using a two-layer bidirectional flux model (53). The bidirectional model uses hourly CASTNET meteorology and 2-wk integrated AMoN  $\text{NH}_3$  concentrations to estimate  $\text{NH}_3$  exchange with soil and vegetation, as well as net fluxes above the vegetation. Ammonia compensation points and leaf surface resistances were parameterized following the recommendations of Massad et al. (40) for natural vegetation. Development of this modeling framework is ongoing. Thus, the comparison is constrained to the dominant natural vegetation type at each site for which the Massad et al. (40) parameterizations are applicable, which excludes fertilized and nitrogen fixing crops and some other surfaces specified by CASTNET, including water, sand, and rock. Details of the bidirectional model and comparison are included in *SI Methods*.

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