Attachments to Letter AU (continued)

Attachment 7

From: Patricia Ann Lazicki <palazicki@UCDAVIS.EDU>
Sent: Thursday, May 28, 2020 2:05 PM
To: Ramy Colfer; Daniel J Geisseler
Cc: Richard Smith; Mike Menes
Subject: RE: Regression Models used to predict %N mineralization based on C:N ratio or %N

Hi Ramy,

Thanks for reaching out. Please see the equations below. The N concentrations are on a dry weight basis, including the liquid amendments. For liquids that the reported N% is usually wet weight, so the C:N ratio will be more appropriate.

I'd frame them more as mineralization potentials (under ideal conditions—ie, incorporated into warm, uniformly moist soil) than as overestimations.

I'm linking to a couple other studies as well, in case it's helpful to give a bit of context to our numbers.
Gale et al., 2006 measures links lab-measured mineralization to fertilizer replacement value in a field experiment, using similar materials in the Pacific Northwest. And a group out of Georgia recently published a study conducted under very similar conditions and measured similar mineralization potentials.

Please feel free to stay in contact if you have any other questions.

Best,

Patricia

From: Ramy Colfer <rcolfer@true.ag>
Sent: Thursday, May 28, 2020 11:30 AM
To: Daniel J Geisseler <djgeisseler@ucdavis.edu>
Cc: Richard Smith <rifsmith@ucanr.edu>; Mike Menes <MMenes@true.ag>; Patricia Ann Lazicki <palazicki@UCDAVIS.EDU>
Subject: RE: Regression Models used to predict %N mineralization based on C:N ratio or %N

Daniel,

Attachments to Letter BI: Colby Pereira (June 21, 2020)

36817 Foothill Road Soledad, CA 93960 Phone (831) 678-0799 Fax (831) 678-3551

January 15, 2019

Mr. John Robertson
Executive Officer,
Central Coast Regional Water Quality Control Board
895 Aerovista Place, Suite 101
San Luis Obispo, CA 93401

Via: Email to AgNOI@waterboards.ca.gov

RE: Comments to Ag Order 4.0 Options Tables

Dear Mr. Robertson:

Thank you for the opportunity to provide comments on the Ag Order 4.0 Options Tables.

I am a fourth-generation member of a farming family in the Salinas Valley. Our farm supports the four families of the owners and over 700 employees. One of my focus areas within our company is food safety. I graduated from Cal Poly, SLO in 2005 with a Bachelor of Science Degree in Agricultural Business, Farm & Ranch Management concentration, and a Minor in Crop Science. Shortly after I graduated, the ag industry was faced with an E. coli outbreak related to spinach in 2006. Most will tell you that this was when the food safety model we are all familiar with was codified and has been an integral piece of the food production world since. I was very involved with the formation of the California Leafy Green Marketing Agreement (LGMA) metrics, serving on technical committees tasked with developing metrics based on the best science available to us. I've continued to serve on the LGMA Technical Advisory Committee since its inception in 2007, where we continually amend these metrics as more research is conducted and more timely scientific data is available. Additionally, I've participated on the Leafy Greens Task Force, formed in 2018 as a body to examine recent foodborne illness outbreaks. Coupled with my almost 14 years of day to day food safety management on the farm, actively participating in these efforts as well as involvement with other organizations such as Grower-Shipper Association of Central California's Food Safety Committee, Center for Produce Safety and California Leafy Greens Research Board has provided me a solid technical foundation in food safety. Our company is committed to growing a safe supply of food, all while being the best stewards of the land we can be.

The main area that I wanted to provide comment on in this letter is with regard to Table 5: Riparian Habitat Management for Water Quality Protection. Vegetative Buffers have long been a struggle for compliance with current food safety mandates. We find ourselves placed in the middle of a "no-win" situation, as on one hand we have State and Federal regulatory bodies telling us that we must keep areas adjacent to production locations free of excess vegetation and on the other hand we have an Ag Order mandate for riparian buffer requirements, or proposing to mandate the expansion of vegetated buffers, and the prohibition of removal of existing native riparian vegetative cover. This sets growers like ourselves up for failure and puts us at odds with conflicting rules from regulatory bodies.

Specifically from Table 5, with regard to Phasing or Prioritization; in the basis for phases, Option 1 does not take into account any adjacent land use activities which have a direct impact on food safety risk assessments that we conduct daily. Option 2 assumes that all ranches are exactly the same with regard to items such as adjacent land use, habitat and environmental factors, thus allowing us zero flexibility to manage each ranch differently according to its potential for food safety risk.

Additionally from Table 5, with regard to Numeric Limits; the table does not account for vegetative management practices that encourage proactive food safety practices. A prescriptive number codified in the regulatory order, which takes no consideration for vegetative management practices for a healthy food safety program, ties our hands as growers and removes important tools from our toolkits, such as the removal of overgrown existing native riparian vegetative cover. Since we do not fully understand the relationship between vegetative cover in close to proximity to fresh produce fields and the potential for introduction of harmful pathogens via animals, it seems premature to mandate specific percentages of the vegetative cover classes, with a time schedule to implement before having additional research to examine the issue.

Lastly, from Table 5, with regard to Monitoring and Reporting; compliance to the Riparian Management Reporting for both Option 1 and Option 2 will automatically provide a basis for the shippers we grow for to impose mandatory buffered areas into our production fields. A few examples of previous mandatory buffers that have been imposed by shippers because of food safety concerns: "50' Buffer along north side of blocks (rodents/harborage)", "75' Buffer along west side of block (undisturbed area / leaves from trees)", "60' Buffer along north side of block (trees/foreign material)", "50' Buffer from ditch (vegetation/rodents)", "100' Buffer along northeast end of block (trees/ foreign material)".[1] With reporting of this type of habitat, we are effectively setting ourselves up for mandatory buffers without the ability to manage the area to prove minimized risk associated with the habitat.

The California Leafy Green Marketing Agreement (LGMA) is an example of a State food safety program with regulatory compliance directives. The LGMA is a food safety program that verifies science-based farming practices using government audits and requires 100% compliance. The program was designed with a set of checks and balances to ensure leafy greens farmers do all they can to protect public health by establishing a culture of food safety on the farm. LGMA auditors are employed by the California Department of Food and Agriculture and licensed by the U.S. Department of Agriculture.

[1] From confidential shipper conducted food safety ranch audits

They are financially independent of the LGMA and leafy greens industry, unbiased and required to report threats to public health to regulatory authorities.[2]

FDA's Food Safety Modernization Act (FSMA) is an example of a Federal regulatory program tasked with food safety oversight. FDA has finalized seven major rules to implement FSMA, recognizing that ensuring the safety of the food supply is a shared responsibility among many different points in the global supply chain for both human and animal food. The FSMA rules are designed to make clear specific actions that must be taken at each of these points to prevent contamination.[3] Failure to comply with produce standards under FSMA would result in FDA's consideration of different regulatory actions that could include: the issuance of advisory letters, court actions (such as seizure or injunction), and administrative actions, such as administrative detention to gain control of adulterated or misbranded products, mandatory recall of violative food, or suspension of a facility's food registration to prevent the shipment of food.[4]

Additionally, as growers, we are subject to a variety of other audit inspections throughout the year. These include, but are not limited to, Shipper Audits (conducted by shippers we contract produce to), Customer Audits (buyers who procure product from those shippers) and 3rd Party Audits (outside parties contracted to perform benchmarked and fully recognized audit systems covering both Good Agricultural Practices (GAP) and Good Manufacturing Practices (GMP) scopes as well as Food Safety Management Systems (FSMS).

The reason I mention all of this is because a commonality amongst every single one of the above-mentioned food safety programs or audit systems is that they contain guidance metrics and compliance verification regarding production locations, adjacent land uses, animal intrusion and habitat management. This is where vegetated buffers and food safety co-management gets complicated. The outline that follows is an attempt to explain why flexibility in practices, as they pertain to vegetated buffers and riparian habitat, is so critical for growers as we navigate between good environmental stewardship and producing safe, healthy crops for consumers.

---

2 California Leafy Green Marketing Agreement https://lgma.ca.gov/government-audits/

3 FDA Food Safety Modernization Act (FSMA) https://www.fda.gov/food/guidanceregulation/fsma/

4 Frequently Asked Questions on FSMA
https://www.fda.gov/Food/GuidanceRegulation/FSMA/ucm247559.htm

Importance of "Maintained and Manicured" Vegetation [5]

- Food safety personnel, shippers, auditors must be able to visually assess the area for the presence of animals.
  - Overgrown vegetation = NO visibility
  - Without proper visibility, must assume presence of animal activity since visually cannot be discounted.
  - Mowed, sparse vegetation allows proper inspection of area.
    - Examples: rodent holes are more identifiable, animal signs (tracks, feces and carcasses) are easier to detect.
  - Properly maintained vegetation also discourages animals from moving in and creating new habitat as conditions are not ideal (potential for hiding, food sources, wind, etc.)
- Animals, specifically rodents for this discussion, who look for harborage pose a health risk to nearby crops.
  - Field rodents are vectors for disease and contamination including E. coli and Salmonella, and threaten food safety.
  - In one project,[6] ten iceberg lettuce and four spinach fields in the Salinas Valley region of Monterey County were surveyed from May 2009 through June 2010.
    - Mice were the most abundant rodent species found at 50% of sample sites monitored, followed by voles at over 7% of sites. (p.10)
    - On average, three times (3X) as many rodents were found at "wild borders", versus "in-crop".
      - This suggests that these riparian buffer areas should be one of the focus areas to prevent rodent migration into adjacent fields.
  - High growing vegetation is an attractant to birds which are enticed to perch upon the overgrowth.
    - High presence of birds = increased risk for fecal matter in adjacent growing area. [7]
    - High presence of birds = increased risk for product damage (loss of young seedlings and quality defects later in plant growth from pecking) [8]
    - High presence of birds = increased risk for foreign matter in adjacent growing area (bird feathers)
      - I would like to provide a specific example here of an especially problematic foreign matter issue associated with

[5] Note: this outline is a compilation from various industry guidance documents, verbiage from specific growing contracts and assorted audit checklists.
[6] 2010 study titled: FOOD SAFETY AND RODENT CONTROL IN LEAFY GREEN CROPS, completed by Salmon, Terrell. P., Gorenzel, W.P., Newman, P.D. and Lima, L., Department of Wildlife, Fish and Conservation Biology, University of California San Diego County Cooperative Extension, Unpublished final report. California Department of Food and Agriculture, Sacramento, California. Contract No. 09-0220. June 30, 2010 pp. 1-38
[7] See Photo #3 in photo attachment
[8] See Photo #4 in photo attachment

birds. Crow populations are significant in the Salinas Valley and take advantage of dense habitat when they can find it. During certain seasons, when nut trees have produced nut crops (i.e. walnuts), the crows will remove nuts from the trees with their beaks, then fly back to their vegetated habitat, nuts in tow. During the gathering and/or flight process, those nuts are often dropped from the bird's beak. Unfortunately, the nuts can fall into adjacent farm ground and unharvested food crops. Finding nuts in a crop could ultimately result in a farmer having to make the decision to not harvest that crop (or areas of the crop), if the adulteration is great. If those nuts are not removed prior to harvest and end up getting packed with harvested product, companies are subject to a Class 1 Recall (as defined by FDA) [9] of any harvested product that could have come into contact with those nuts.

- The susceptibility of foreign matter to a produce field increases significantly with overgrown vegetation and the inability for effective monitoring.
    - Products that are mechanically harvested are at an even higher risk. Machines will cut and collect anything in its path when travelling through a field – along with the intended product, this could include a rodent, vegetation debris, bird feathers, etc. and aside from the leafy item itself, nobody wants to find any of the others on their salad plate!

Importance of Monitoring with Bait Stations and/or Traps

- Vegetable crops exposure to rodents can be high risk. Failure to follow required principals could lead to fecal and/or bacteria adulteration of food source.
    - Bait and Trapping Control Devices are placed at strategic locations where there is the potential for rodent harborage or rodent activity, including areas such as fresh rodent holes, heavy brush areas, creeks, riverbanks, catch ponds, etc.
    - Rodent bait is not used while crops are growing and/or being harvested in adjacent blocks.
    - Bait is placed in stations to keep it from being accessible to birds and other non-target animals.
    - Bait Stations and Trapping Devices are inspected and documented at least twice a month or more often, depending on rodent activity.
        - Traps are utilized for determining rodent populations.

[9] Class I: Dangerous or defective products that predictably could cause serious health problems or death. Examples include: food found to contain botulinum toxin, food with undeclared allergens, a label mix-up on a lifesaving drug, or a defective artificial heart valve. https://www.fda.gov/ForConsumers/ConsumerUpdates/ucm049070.htm

        - Captured rodents are properly disposed of away from fields or water sources.
    - Bait Stations and Trapping Devices are placed sufficiently away from water supply and direct discharge water sources to prevent cross contamination.
    - Bait Stations have specifically designed caps that prevent surface water from entering the bait reservoir.

Animal Intrusion into Growing Area

- Instances of animal intrusion (tracks, fecal matter, carcass) ultimately result in buffered areas and loss of product. [10]
    - Buffers can vary in size depending on level of intrusion [11]
        - Typically 5' minimum buffer on low hazard (single set of tracks, single incident of fecal matter, effected area less than 10' diameter)
        - Typically 10' minimum buffer on medium hazard (more than one set of tracks, more than once incidence of fecal matter, effected area more than 10' in diameter but less than ¼ acre)
        - Anything high hazard results in significant buffered areas (more than ¼ acre contaminated)

As you can see, there are many factors to consider on our ranches as we work to meet all demands set forth by what could be construed as "competing interests". Understanding the various food safety concerns that an overly robust vegetated buffer or riparian area can have on growers, I urge you to consider less prescriptive mandates in this area.

Just this past November, we saw Centers for Disease Control & Prevention (CDC), public health and regulatory officials in several states, Canada, and the FDA investigate a multistate outbreak of Shiga toxin-producing Escherichia coli O157:H7 (E. coli O157:H7) infections associated with romaine. [12] As part of the investigation, based on the traceback, FDA investigators, along with state officials and CDC analyzed samples taken from several locations and found E. coli O157:H7 that matched the outbreak strain in one sample from sediment in an ag water reservoir on a ranch in Santa Barbara. However, other legs of the traceback don't seem related to this ranch, and the investigation is still ongoing. To have witnessed the widespread effect from what could potentially be a single positive finding only bolsters the argument that the food safety practices and guidelines that have been developed using sound science must be adhered to, in order to minimize risk. For a week, harvested romaine product was being discarded and mature fields ready

[10] See Photo #1 and Photo #2 in photo attachment

[11] California Leafy Green Marketing Agreement "Commodity Specific Food Safety Guidelines for the Production and Harvest of Lettuce and Leafy Greens". Adopted September 28, 2018. "Figure 5. PRE-HARVEST and HARVEST Assessment – Animal Hazard/Fecal Matter Decision Tree" p52 , "TABLE 5. ANIMAL HAZARD IN FIELD (WILD OR DOMESTIC)" p53 and internal company Standard Operating Procedures

[12] https://www.cdc.gov/ecoli/2018/o157h7-11-18/updates.html and https://www.fda.gov/Food/RecallsOutbreaksEmergencies/Outbreaks/ucm626330.htm

for harvest were being walked away from (ineffectively removing any N from the field). Employees were laid off and even now, as romaine has been "cleared" and deemed safe to consume, market demand has not rebounded. Consumers were forced to seek alternatives in the fresh produce section of the supermarket and restaurants to replace the ever-popular Caesar Salad on their menus.

We need not look any further than the most recent foodborne illness outbreaks related to fresh produce to understand that even the most minor deviations from standard food safety operating procedures on the ranch can have a very major effect on an entire industry, the workers employed by that industry and most important, the consumer.

General consumer confidence for leafy greens and vegetables is of the highest importance to our society and the health of our nation. The aforementioned example was specific to romaine, but we must think more broadly in terms of our overall food safety programs as an issue with one commodity affects the entire fresh produce supply chain in terms of consumer confidence. It takes time to repair the damage done to a commodity following an outbreak. According to a USDA – Economic Research Study, it took 68 weeks after the 2006 spinach outbreak for spinach sales to surpass where they were before the outbreak. [13] The more frequently food safety related outbreaks occur; there is a higher chance that the reputation of leafy greens as an overall category will be tarnished.

The ability to manage our growing areas, with practices outlined earlier in this letter, are critical to our operation. It is through these proactive practices that we stand the best chance at preventing future food safety related incidents. Leafy green vegetables are an important part of a healthy diet and provide a multitude of benefit to the human body. [14] Just to highlight a few, eating a diet rich in leafy greens can offer numerous health benefits including reduced risk of obesity, heart disease, high blood pressure and mental decline. [15] We are committed to doing our part to provide consumers, of which our families are as well, the opportunity to consume fresh produce daily with a sense of confidence and security.

Again, thank you for your consideration of these comments. It is our family's hope that they be taken into account before the final order is put in place.

Respectfully,

Colby Pereira
Food Safety Manager
Anthony Costa & Sons

[13] https://www.ers.usda.gov/amber-waves/2010/march/consumers-response-to-the-2006-foodborne-illness-outbreak-linked-to-spinach/

[14] https://www.ars.usda.gov/plains-area/gfnd/gfhnrc/docs/news-2013/dark-green-leafy-vegetables/

[15] https://www.ncbi.nlm.nih.gov/pubmed/29263222

Photo Attachment

Animal Intrusion (tracks) and Bird Damage (fecal matter) – Flags = Buffered "Do Not Harvest" Area

Photo #1

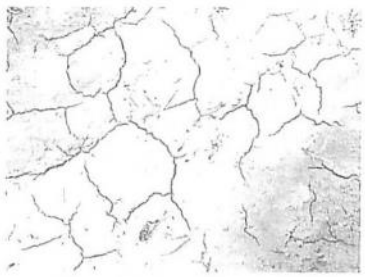


Photo #2

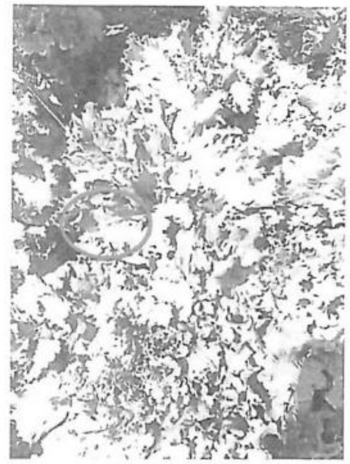


Photo #3

Product Damage

Photo #4

1

Your point is very well taken. We will include your comment in our presentation. We will propose using
the Lazicki et al. study mineralization rates but make clear that they are overestimates. It looks like we
need more organic fertility research Daniel!

I am hopeful that we can convince the Water Board to modify their approach to organic fertilizers but
not confident.

Thank you,
Ramy

From: Daniel Geisseler <djgeisseler@ucdavis.edu>
Sent: Thursday, May 28, 2020 11:11 AM
To: Ramy Colfer <rcolfer@true.ag>
Cc: Richard Smith (rifsmith@ucdavis.edu) <rifsmith@ucdavis.edu>; Mike Menes <MMenes@true.ag>;
palazicki@ucdavis.edu
Subject: Re: Regression Models used to predict %N mineralization based on C:N ratio or %N

Hi Ramy
Unfortunately, I don't have a better and easy way to calculate N mineralization rates.
I'm not opposed to using our values, but I would be more comfortable if a comment was added saying
that these are potential N mineralization rates and that the actual mineralization rates are lower during
the cold season and when the amendments are not incorporated. Under these conditions, growers
should be allowed to use lower values. Can you convince the Water Board to do that?

Patricia will send you the equation.

Best,

Daniel

On Thu, May 28, 2020 at 9:51 AM Ramy Colfer <rcolfer@true.ag> wrote:
Sorry for repeat, fixed email typo

From: Ramy Colfer
Sent: Thursday, May 28, 2020 9:48 AM
To: Daniel Geisseler <djgeisseler@ucdavis.edu>
Cc: palazicki@ucdavis.edu; Richard Smith (rifsmith@ucdavis.edu) <rifsmith@ucdavis.edu>; Mike Menes
<MMenes@true.ag>
Subject: RE: Regression Models used to predict %N mineralization based on C:N ratio

or %N

Thank Daniel for your email.

You make some very good points. Currently the CCRWQCB is proposing to treat organic fertilizer as
conventional fertilizer, meaning that all nitrogen applied in organic fertilizers would be considered
plant available and part of their TNA budget, which we know is not true. Even if the Lazicki et al. study
overestimates mineralization rates, it is based on very good science and at least organic growers won't
be faced with the incredibly draconian constraint of having all nitrogen applied (that is not CDFA
defined compost) going toward their nitrogen applied budget as the CCRWQCB has proposed.

If you have another proposed way we could calculate mineralization rates for a wide array of organic
soil amendments and organic fertilizers, I am absolutely all ears.

Much appreciated,
Ramy

From: Daniel Geisseler <djgeisseler@ucdavis.edu>
Sent: Thursday, May 28, 2020 9:34 AM
To: Ramy Colfer <rcolfer@true.ag>
Cc: palazicki@ucdavis.ed; Richard Smith (rifsmith@ucdavis.edu) <rifsmith@ucdavis.edu>
Subject: Re: Regression Models used to predict %N mineralization based on C:N ratio or %N

Hi Ramy
Thanks for your interest in our research. I just wanted to point out that the N mineralization rates we
observed were based on a study where the amendments were incorporated into soils kept at an
optimal moisture content and at a constant temperature of about 75 degree Fahrenheit.
In the field, the N mineralization rates can be quite different, for example when the organic
amendment is applied during the cooler months of the year, or when it is applied to the soil surface
and not incorporated (Richard has some interesting data on that). In addition, soil moisture is not
always optimal, for example outside the wetting zone under drip irrigation.
So the C to N ratio of the material is only one of several factors affecting N mineralization rates.
Temperature, soil moisture and contact with soil particles are other important factors that need to be

taken into account. Ignoring these factors can in some cases lead to a considerable overestimation of N
availability from organic amendments. Without adjustments for these factors, our values should
probably only be used for incorporated amendments applied to a surface (e.g sprinkler) irrigated
summer crop.

Best,

Daniel

On Thu, May 28, 2020 at 9:09 AM Ramy Colfer <rcolfer@true.ag> wrote:
Dear Dr. Lazicki,

I am interested in getting some additional information associated with the publication you recently
published in JEQ. I would like to request the regression models used to in Fig. 3A (Amendment C:N vs.
Mineral N) & Fig. 3B (Amendment %N vs. Mineral N). I wish to use the regression models to be used to
predict the amount of plant available N made available from different organic amendments and
organic fertilizers. We propose to use these regression models to modify the proposed rules in Ag
Order 4.0 by the CCRWQCB. The CCRWQCB has proposed to use mineralization rates when applying
compost but are currently excluding the use of mineralization rates for organic fertilizers.

I am happy to discuss the topic further if you would like. I have Cced Richard Smith and Daniel
Geisseler to keep them informed of our proposed communications with CCRWQCB. I have worked and
discussed this topic with Richard Smith in the past. I have attached the presentation that Richard
Smith, Jocelyn Bridson, and myself presented to the CCRWQCB during the last public comment period
as background information for you.

True is going to meet with the CCRWQCB in a couple of weeks. If you could share these regression
models with us in the coming week, we'd be very appreciative.

Sincerely,
Ramy Colfer, Ph.D.
VP, R&D and Ag. Services
True Organic Products
(831) 206-5609

--

Daniel Geisseler
Associate Cooperative Extension Specialist in Nutrient Management
University of California, Davis
Department of Land, Air and Water Resources
2146 PES Building
One Shields Avenue
Davis, CA, 95616

--

Daniel Geisseler
Associate Cooperative Extension Specialist in Nutrient Management
University of California, Davis
Department of Land, Air and Water Resources
2146 PES Building
One Shields Avenue
Davis, CA, 95616

Attachment 8

United States Department of Agriculture

National Center for Appropriate Technology NCAT

Nutrient Management Plan (590) for Organic Systems

Western State Implementation Guide

March 2014

National Center for Appropriate Technology (NCAT)
www.ncat.org

Oregon Tilth
www.tilth.org

Contents

Figure 2. Six-foot-tall fava bean cover crop, Fong Farms, Woodland, California 2006.

Funded by a grant from Western Sustainable Agriculture Research and Education (WSARE).

WESTERN SARE Sustainable Agriculture Research & Education

ACKNOWLEDGMENTS

Authors Rex Dufour (National Center for Appropriate Technology), Sarah Brown (Oregon Tilth), and Denise Troxell (NRCS)

Contributors and Reviewers Carrie Sendak (Oregon Tilth), Denise Troxell (OR-NRCS), Nick Andrews (Oregon State University), Giulio Ferruzzi (NRCS), Ben Bowell (Oregon Tilth), Jennifer Miller (Northwest Center for Alternatives to Pesticides), Rich Fasching (NRCS), Dennis Chessman (CA-NRCS)

We thank Dan Sullivan, Extension Soil Scientist, Oregon State University for assistance in presenting methods for estimating plant-available N release from organic inputs.

Editing and layout Karen Van Epen (NCAT), Robyn Metzger (NCAT)

All photos Rex Dufour (NCAT) unless other wise noted.

Figure 1. (On front cover) A farmer evaluates soil organic matter. Photo: David Lamm (NRCS)

Nutrient Management Plan (590) for Organic Systems
Implementation Guide

Purpose and Objectives

This document is an instruction guide for creating and implementing a nutrient management plan (NMP) on certified or transitioning organic lands.

An NMP for organically managed lands describes the amount, source, placement, form, and timing of the application of nutrients and soil amendments, generally by field, to meet crop nutrient needs while protecting water quality, improving soil health, and utilizing manure and other organically acceptable byproducts as nutrient sources.

Figure 3. Mowing vetch cover crop prior to transplant of processing tomatoes.

Developing an NMP entails:

1) Determining a crop's need for nitrogen, phosphorus, potassium, and other nutrients as appropriate
2) Crediting all significant sources of nutrients in the system such as the nitrogen contribution of cover crops, the mineralization of nitrogen based on input history (previous years' manure/ organic byproducts applications), and any nitrate that may be contained in irrigation water
3) Determining the target nutrient application rate (the difference between the crop need and the nutrient credits)
4) Evaluating likelihood of nutrients moving past the edge of a field or below the root zone (leaching and runoff)
5) Calculating the appropriate rate of manure/ organic product application based on this risk

Instructions

☞ **This icon indicates you should fill in data on the Implementation Requirement Worksheet in Appendix D, pages 26-30. Alternatively, you may use your state's NM planning tool to enter the crops grown.**

☞ **In the *Field and Crop Information* section, list the crops in the order they are grown. Check the box for the current crop. Make additional copies of the form as needed.**

3

Relevant National Organic Program (NOP) Regulations

National Organic Program (NOP) regulations section 205.203 should be referenced when assessing nutrient management in organic systems. The entire NOP regulations, as well as lists of approved and prohibited materials and other information can be found at the NOP website: www.ams.usda.gov/AMSv1.0/nop

Section 205.203 Soil Fertility and Crop Nutrient Management Practice Standard

(a) The producer must select and implement tillage and cultivation practices that maintain or improve the physical, chemical, and biological condition of soil and minimize soil erosion.

(b) The producer must manage crop nutrients and soil fertility through rotations, cover crops, and the application of plant and animal materials.

(c) The producer must manage plant and animal materials to maintain or improve soil organic matter content in a manner that does not contribute to contamination of crops, soil, or water by plant nutrients, pathogenic organisms, heavy metals, or residues of prohibited substances. Animal and plant materials include:

(1) Raw animal manure, which must be composted unless it is:

(i) Applied to land used for a crop not intended for human consumption;

(ii) Incorporated into the soil not less than 120 days prior to the harvest of a product whose edible portion has direct contact with the soil surface or soil particles; or

(iii) Incorporated into the soil not less than 90 days prior to the harvest of a product whose edible portion does not have direct contact with the soil surface or soil particles;

(2) Composted plant and animal materials produced through a process that:

(i) Established an initial C:N ratio of between 25:1 and 40:1; and

(ii) Maintained a temperature of between 131 °F and 170 °F for 3 days using an in-vessel or static aerated pile system; or

(iii) Maintained a temperature of between 131 °F and 170 °F for 15 days using a windrow composting system, during which period, the materials must be turned a minimum of five times.

(3) Uncomposted plant materials.

(d) A producer may manage crop nutrients and soil fertility to maintain or improve soil organic matter content in a manner that does not contribute to contamination of crops, soil, or water by plant nutrients, pathogenic organisms, heavy metals, or residues of prohibited substances by applying:

(1) A crop nutrient or soil amendment included on the National List of synthetic substances allowed for use in organic crop production;

(2) A mined substance of low solubility;

(3) A mined substance of high solubility: Provided, That, the substance is used in compliance with the conditions established on the National List of nonsynthetic materials prohibited for crop production;

(4) Ash obtained from the burning of a plant or animal material, except as prohibited in paragraph (e) of this section: Provided, That, the material burned has not been treated or combined with a prohibited substance or the ash is not included on the National List of nonsynthetic substances prohibited for use in organic crop production; and

(5) A plant or animal material that has been chemically altered by a manufacturing process: Provided, That, the material is included on the National List of synthetic substances allowed for use in organic crop production established in § 205.601.

(e) The producer must not use:

(1) Any fertilizer or composted plant and animal material that contains a synthetic substance not included on the National List of synthetic substances allowed for use in organic crop production;

(2) Sewage sludge (biosolids) as defined in 40 CFR part 503; and

(3) Burning as a means of disposal for crop residues produced on the operation: Except, That, burning may be used to suppress the spread of disease or to stimulate seed germination.

4

Section 1.
Determining Crop Need for Nitrogen, Phosphorus, and Potassium

This section outlines the process and resources to be used to calculate the demand of a particular crop for N, P, K, and other nutrients. Sources of information about crop nutrient needs can be found in Appendix C, Section 1.

It is common for organic farms to grow many different crops on relatively small acreages and for soil texture to vary across the farm. In order to develop a practical nutrient management plan for small plot acreages, it may be helpful to work with the grower to develop groupings of crops that have similar growth habits and nutrient needs, and manage the grouping's nutrient needs as a single entity. Crop rotations are required in organic systems. Growers should have a crop rotation plan as part of their organic system plan.

Additional information to guide soil sampling strategies and crop rotations on diverse organic vegetable farms can be found in Appendix C, Section 1. Some possible groupings include:

- **Nitrogen Needs (Ideal)**: Low/Medium/High N. See Table 1.
- **Plant Family**: For example, all solanaceous or all brassica crops
- **Timing**: Early season, mid-season, and late season plantings. Timing will impact nutrient availability from soil N mineralization. Early season crops will be unlikely to pull much from the soil and should be fertilized accordingly.
- **Crop Growth**: Short-season crops, long-season crops
- **Crop Type**: Grains, legumes, vegetables, fruits
- **Other**: Growers also take other considerations into account such as irrigation system layout, pest cycles, and soil type.

Table 1. Nitrogen requirement of vegetable crops based on seasonal nitrogen uptake

Low Total N Need <120 lb/acre	Medium Total N Need <120-200 lb/acre	High Total N Need >200 lb/acre
Baby greens	Carrot	Broccoli
Beans	Corn, Sweet	Cabbage
Cucumbers	Garlic	Cauliflower
Radish	Lettuce	Celery
Spinach	Melons	Potato
Squashes	Onion	
	Peppers	
	Tomatoes	

— Gaskell et al. 2006, Soil Fertility Management for Organic Crops

Western States grow a great variety of specialty crops. Your state cooperative extension office has horticulturalists and agronomists who can point you in the right direction for informational resources about the nutrient needs of less commonly planted crops. Some specialty crops may require higher amounts of particular micronutrients, which if not present will limit quality and yields. Other crops may be more sensitive to excess micronutrients, such as citrus sensitivity to boron. If no University recommendations are available for a crop, an acceptable strategy is to balance N, P and K applied with N, P and K expected to be removed by the crop.

☞ **Enter crop nutrient requirements and yields under *Recommended Nutrients to Meet Yield.***

Resources for Nutrient Requirements of Crops

California Cooperative Extension: www.ucanr.edu/sites/nm/Crop

Idaho Cooperative Extension and Idaho Nutrient Management: www.extension.uidaho.edu/nutrient

Nevada Cooperative Extension: See Section IV of eFotg, Nutrient Management 590 Job Sheet

NRCS Crop Nutrient Tool: http://plants.usda.gov/npk/main

NRCS Agricultural Waste Management Field Handbook provides data on N, P and K content of a wide variety of crops. ftp://ftp.wcc.nrcs.usda.gov/wntsc/AWM/handbook/ch6.pdf

Oregon State University Fertilizer/Nutrient Management Guidelines for Crops: http://extension.oregonstate.edu/catalog/details.php?sortnum=0134&name=Fertilizer+Guides&cat=Agriculture

5

Section 2. Developing Nutrient Credits

This section outlines some methods to calculate the nutrient contributions of:

- Soil Organic Matter (SOM) N mineralization
- Cover crops
- Organic amendments such as manure, compost, and some specialty fertilizers

The input history of a field is valuable, because many sources of organic N continue to mineralize over several years. As part of their organic recordkeeping requirements, organic farmers will have records of their soil fertility inputs, as well as yield data, and these can help to make a more comprehensive nutrient management plan. Transitioning farmers may have no records if they are early in their transition. Recent soil test information will provide information about what is present in the soil. Sources of information about organically acceptable fertilizers can be found in Appendix A, Tables 4, 5, and 6.

Soil Test Information and Resources

Soil test and leaf test results taken at different times of year provide valuable data for monitoring the performance of nutrient management plans. This is especially useful for refining N fertilizer rates. Pre-Side-dress Nitrate Tests (PSNTs) can ensure a grower that N levels are sufficient for the crop. PSNTs are generally done in the late spring and allow the grower to respond to early spring weather conditions that impact N mineralization. End-of-season soil nitrate tests can determine whether N fertilizer rates were higher than necessary. In most perennial crops, leaf tests are used to determine crop nutrient status. These naturally incorporate non-fertilizer nutrients taken up by the crop.

Mineralization from "native" SOM is generally not included in NRCS nutrient management plans because land-grant university recommendations for N fertilizer application rate assume typical soil N mineralization already (under conventional management). However under long-term management with consistently high organic-N inputs, soil N mineralization may supply a significant portion of crop N needs.

P and K should be managed by soil tests and attention to P and K removal through harvest of crop biomass. P and K fertilization decisions typically do not require consideration of target yield for the crop. Crops grown in soil with high P and K concentrations usually do not respond to P and K fertilizers. When large amounts of crop biomass are removed at harvest (i.e., alfalfa, or corn silage) significant amounts of P and K are removed from the soil. Other crops such as raspberries or blackberries do not remove large amounts of P and K.

Soil test results can come in either text or graphical formats. Discuss with the producer which format is preferred. University nutrient management guides provide more accurate crop-specific sufficiency levels than general ranges. Refer to your state's Conservation Practice Standard 590 for the required soil tests. A listing of accredited (NAPT-PAP certified labs) soil testing labs can be accessed at: www.naptprogram.org/pap/labs. Tests of plant tissue, water and organic materials must be conducted by labs that are certified by organizations noted in the state's 590 practice standard.

☞ **Enter soil test results under the *Soil Test Information* section.**

Nitrogen (N)

Nitrogen in organic systems is often tied up in organic matter, and not in a form that is immediately available to plants. Most organically acceptable sources of N have lower N concentrations than chemical fertilizers, and typically release less of their total N in the first cropping season. In contrast, PAN for chemical fertilizers such as urea, ammonium sulfate, and others, is 100%. That is, the N is available to the plant shortly after application as either nitrate (NO_3—this is the form most plants take up N), or ammonium (NH_4+). Possible sources of N for organic systems include compost, manures, cover crops, organic matter mineralization, organic approved fertilizers (fish meal, feather meal, etc.), and irrigation water.

In some locations, irrigation water can have significant levels of nitrates, so it's a good idea to have the irrigation water tested. The nitrate contribution from irrigation water should be taken into account when developing nutrient budgets, but there are many variables to consider: uptake by plant, irrigation water going beyond root system, and nitrate variability in water during the season. Table 12 provides conversions to calculate parts per million to lbs/acre inch.

6

PAN Plant-Available Nitrogen — *Nutrient Source: Soil N Mineralization*

This is the process whereby the stock of N that is "stored" in SOM decomposes to release plant-available nitrogen (PAN). Because organic farmers typically apply more organic matter than conventional farmers, PAN provided by soil N mineralization generally increases under organic management. The most recent organic inputs have the most influence on PAN release from soil organic matter. Most University nutrient management guides account for some background soil N mineralization, but not usually the high levels often found on established organic farms. The potential rate of soil N mineralization is a function of temperature, tillage, type of soil, and other factors. Any figures developed for PAN from soil N mineralization must be considered approximate.

While organic growers are required to record applications of materials, they might not have the nutrient analyses they need to do a calculation. Approximate numbers can be used in this situation.

How to Estimate Soil N Mineralization

1. Organic amendments (composts, manure, cover crops, fertilizers, etc.) applied to the soil will continue to provide PAN in subsequent years, but in ever-smaller quantities. Organic fertilizers such as blood meal, and feather meal, which are higher in N, will provide roughly 75% of their N in the first year and up to 50% of their N within a week of application (Gaskell, 2007). Assuming some materials have been applied, first calculate total N application per year. To determine the PAN for the current year, refer to Tables 7-9. The estimation rule for PAN available in years 1, 2, and 3 post-application are 8%, 5%, and 3% of total N applied, respectively. See Table 2, below.

2. If previous organic input history is not available, then credit 50 lbs N/acre if the field received substantial organic inputs (i.e. cover crops and manure or compost) for at least three consecutive years.

PAN Plant-Available Nitrogen — *Nutrient Source: Manures and Compost*

To determine PAN from manure and compost products you must first analyze the nitrogen content. Table 7, Appendix A, based on Oregon State University's Organic Fertilizer and Cover Crop Calculator (Andrews, et al.), identifies the PAN % or pounds after four weeks for manure, compost, and other organic amendments.

In order to obtain the full N benefits of manure, applications should be incorporated immediately. Significant amounts of ammonia-N can be lost within one to two days of application through volatilization (see Table 5 Appendix A). Significant amounts of N may also be lost if manure is applied too far in advance of the period of rapid N uptake by the crop, even if the manure is incorporated. For fall manure applications, up to half of the N may be lost by the time of greatest crop demand the following year (Magdoff, 2009).

Poultry litter is a popular organic amendment because of high N and P levels, but it may be high in arsenic, copper and zinc due to additives in their feed. Analysis of poultry litter should include Zn, Cu, and As to ensure that soil is not being contaminated.

Manure and Compost

Refer to page 4 for additional National Organic Program regulations related to the use of manure and compost on organic farms. It should also be noted that manure and compost inputs from non-organic sources are allowed as long as there is no risk of contamination.

Table 2. Example of PAN Credits.

If you applied 2,000 lbs/acre of broiler litter this year, and this litter is 4% N, then a total of 2,000 x .04 = 80 lbs N/acre was applied. This year, after ten weeks, the PAN from the 80 lbs applied will only be 45% or 36 lbs N (see Table 7). Next year, a year after application, 8% of the total N applied would be available; 80 lbs/acre x .08 = 6.4 lbs/acre. Two years after application, 5% of the N applied would be available (80 x .05 = 4lbs/acre and in year three, 3% would be available (80 x .03 = 2.4/lbs/acre).

PAN, Percent of Total N (general rule)	Years After Application	lbs PAN per acre
45% x 80 lbs/acre	current year	36
8% x 80 lbs/acre	1	6.4
5% x 80 lbs/acre	2	4
3% x 80 lbs/acre	3	2.4

7

PAN Plant-Available Nitrogen — *Nutrient Source: Cover Crops*

Cover crops and green manures provide both nitrogen and organic matter to the soil. The amount of N and organic matter provided will depend on the cover crop species, or mix, crop biomass, and growth stage when killed. These same factors will influence how rapidly the cover crop will decompose, and whether it will release, or immobilize PAN. Most PAN is released in four to six weeks after cover crop termination. For a comprehensive guide on this topic see 'Estimating Plant Available Nitrogen Release from Cover Crops, PNW 636'. In this publication we will review the shortcut method as outlined in that document.

PAN from any cover crop is minimal when the cover crop is very small. For solo cover crops, the best time to terminate the cover crop to maximize PAN depends on whether the cover crop is a legume or a non-legume.

➢ PAN from a robust stand of legumes (Figure 4) peaks at budding growth stage. PAN declines slowly as reproductive growth continues.

➢ PAN from cereal residues is positive early in the spring (through tillering, mid- to late March). As stem elongation proceeds (jointing), PAN from cereal residues declines. By the time the flag leaf (uppermost leaf) emerges (Feekes growth stage 8 or Zadoks 37), PAN from cereal crop residue is near zero. When cereal heads are visible, PAN from cereals is negative.

To maximize PAN, kill cereal cover crops early, but wait until bud stage to kill legumes. In cereal/legume mixtures, the best crop growth stage for maximum PAN benefit depends on the percentage of legume in the stand. Figure 4 illustrates how PAN changes over time in solo and mixed cover crop stands.

➢ When the cover crop is mostly legume, it behaves much the same as does a pure legume cover crop—the PAN from crop residue increases until cereal boot stage. (Feekes stage 10; Zadoks stage 45). After pure, or majority cereal stands reach jointing stage, PAN declines.

➢ When a cover crop has more cereal than legume (25 percent legume line in Figure 4), it follows a similar PAN curve as a solo cereal crop, but negative PAN is usually not seen until the cereal reaches boot stage (around mid-May).

Seeding legume/cereal mixes instead of a solo cereal crop allows greater flexibility in timing of cover crop termination without consequences of negative PAN. Cover crop mixes also combine the N fixation benefits of legumes with weed suppression and soil benefits of cereals. The percent N in cereals varies with field history. Fields that have a history of manure/compost application and/or legumes in rotation often have higher percent N in cereal than do fields with a history of only N fertilizer application.

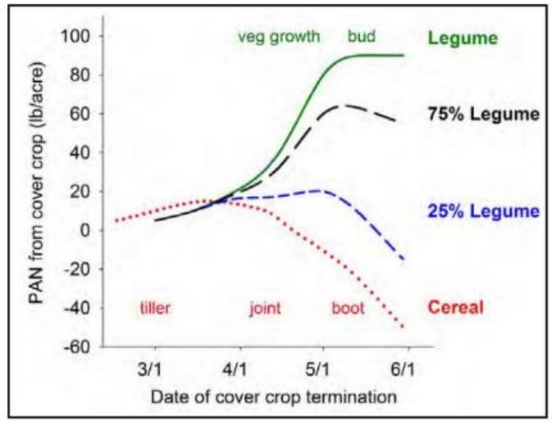


Figure 4. Effect of kill date on typical plant-available N (PAN) release from cereal, legume, or mixed stands. Based on compilation of field data from Willamette Valley cover crop trials. Figure from PNW 636, Estimating Plant-available Nitrogen Release from Cover Crops (Sullivan and Andrews, 2012), ©Oregon State University. Used by permission.

Biomass determination. The first step in estimating PAN from a cover crop is determining how much cover crop biomass is present. Visual estimates of cover crop biomass are not very accurate, especially for multi-species cover crop mixes. Cutting and weighing cover crop biomass is the preferred method to estimate PAN. Generally, cover crop roots are not included in this calculation. Harvest the above-ground cover crop with quadrats or a sickle bar mower (see Sullivan et al. 2012, Appendix C, Section 5 for additional sampling instructions). Any harvest method can be used that gives you a clean plant sample with a known harvest area.

To determine the PAN you'll need to estimate the percent dry matter (DM). You will need to either use a ballpark estimate of DM or oven dry the sample that has been collected. Mixed, vegetative cover crop DM will usually be between 12 and 18%. Very wet clover samples are usually 10 to 12 % DM. Cereals after head emergence are usually 20 to 25% DM.

8

Table 3. Estimated plant-available N (PAN) release from decomposition of crop residues in soil at 4 and 10 weeks

	Total N	C:N	4 wk	10 wk	4 wk	10 wk
	% N, dry wt		PAN, % of residue total N		lbs PAN/dry ton	
Cereals after head emergence	1	40	-40	<0	-8	<0
Cereals, tillering/jointing	2	20	5	20	2	8
50/50 Cereal/Legume Mix or Flowering Legumes	3	13	35	40	21	24
Legumes, vegetative	4	10	50	55	40	44

Adapted from OSU Extension publication EM9010-E (2010): Nutrient management guide for sweet corn (western Oregon).
Negative PAN values indicate immobilization of N by cover crop residue.
With a good stand and cover crop kill near bud or head emergence stage of plant development, the typical cover crop PAN credit is 50 lb N/acre for cereal legume mixtures, and up to 100 lb PAN/acre for legumes.

Shortcut method. Harvest and measure cover crop biomass and use typical values for cover crop DM and percent N (Table 3) to estimate PAN. See Table 9 for information and sample calculations about how to estimate cover crop contributions to PAN. More accurate estimations can be made with lab analysis (Sullivan & Andrews, 2012).

☞ **Enter figures for PAN available from previous applications of manure, compost and cover crops under *Nutrient Sources* section. Alternatively, use the rates established in your state's Nutrient Management Planning tool.**

Figure 5. Assessing the biomass of a cover crop.

Phosphorus (P)

In nature, most plants, except brassicas, receive much of their phosphorus through mycorrhizal interactions with the soil. This is particularly true in perennial systems. In organic systems, with regular cover crops, this interaction may be more prominent than in conventionally managed systems because organically managed soils can provide better conditions for mycorrhizal survival. If manures, or composted manures, have been regularly applied at rates to meet a crop's N needs, this may result in more P (and K) than the crop requires. After many years, this can result in over-accumulation of P and K, and any solution requires some combination of reduced applications and/or increased uptake and removal of these nutrients.

Potassium (K)

Potassium is available as a positive ion, or cation, and soils with a high cation exchange capacity (CEC) will generally have more capacity to store K. Most soils with high clay content, high organic matter levels (>3%), or both will have high CEC. There are thousands of pounds per acre of potassium in an average, non-sandy soil, but most of that is unavailable to plants—usually only 1-2% is available. Repeated applications of manures can lead to over-abundance of K (as well as P) in the soil.

Secondary Macronutrients and Micronutrients

Micronutrients are important to plant health, and can be limiting factors for both yield and quality in some production systems, or, when present in excessive amounts, can reduce availability of other nutrients or have a negative impact on plant health. Soil tests should provide information on the levels of the secondary macronutrients Calcium (Ca), Sulfur (S), and Magnesium (Mg), as well as micronutrients, such as Manganese (Mn), Iron (Fe), Zinc (Zn), Copper (Cu), and Boron (B). Documented soil deficiency, either through soil or tissue tests, is required per NOP regulations prior to applying any synthetic micronutrients.

9

Section 3.
Determining Nutrient Application Rates

Determining nutrient application rates consists of taking your crop's nutrient needs and subtracting any credits from the system. If the credits are less than the crop's nutrient need, the grower will need to apply supplemental nutrients in order to achieve the desired yield. Not all of the N applied this year will be available to the crop in the first year. A balanced nutrient program will supply sufficient PAN, P_2O_5, K_2O, and other nutrients while avoiding excesses. The example below demonstrates this approach.

☞ **Add up all the PAN nutrient credits from the previous applications of manure, compost, cover crops, as well as what can be expected from the irrigation water for this season's crop. This total should be subtracted from the total Nutrients Recommended for Yield.**

Example

Assume that after all the nutrient credits are subtracted, the values for the Nutrients Recommended for Yield are:

80 lbs PAN, 40 lbs P_2O_5, and 60 lbs K_2O per acre.

The manure analysis is 3% N, or 60 lbs total N per dry ton. According to Table 7, PAN Estimates for Uncomposted Manure, at 10 weeks PAN released is about 30% of the product, or 18 lbs PAN per dry ton. This material is 60% dry matter (therefore, 40% water), so to make consideration for this:

18 lbs PAN x 0.60 = 10.8 lbs PAN per "as is" ton

80 lbs PAN/ac ÷ 10.8 lbs PAN/Ton = 7.4 T/ac
This is the application rate that meets the crop's nitrogen need.

The manure also contains 20.6 lbs P_2O_5/dry ton (reported on a dry basis this time) and 37.8 lbs K_2O/dry ton according to the analysis.

To calculate how much P_2O_5 the 7.4 T/acre manure application rate will provide:

Application rate: 7.4 T/ac x .6 (60% dry matter) x 20.6 lbs P_2O_5/dry ton = 91.5 lbs P_2O_5/ac

91.5 lbs P_2O_5/ac is being applied with the 7.4 T/ac manure. Note that this is over twice as much P_2O_5 as is required by the crop. What are some other fertilizer options that would supply sufficient N, but less P_2O_5?

To calculate how much K_2O the 7.4 T/acre manure application rate will provide:

Application rate: 7.4 T/ac x .6 (60% dry matter) x 37.8 lbs K_2O/dry ton = 168 lbs K_2O/ac

168 lbs K_2O/ac is being applied with the 7.4 T/ac manure. This is nearly three times as much K_2O/ac as is required. What are some other fertilizer options that would supply sufficient N and P, but less K_2O/ac?

Refer to Tables 4-6 for additional organic fertilizer options.

Please note: *Calculations here use lbs N/ton or PAN/ton because litter and compost are commonly delivered by the ton.*

☞ **Enter planned NPK applications to the right of line 7 (Manure), line 8 (Compost), or line 9 (Specialty Fertilizers) in the appropriate nutrient column.**

10

Timing and Application Method

The timing, application, and incorporation methods of nutrients is most important relative to conserving N and P. Manure applications can provide significant N to the soil if they are incorporated within 12 hours of application. If the manure is broadcast and left on the surface, between one half and two thirds of the available N can be lost to volatilization after one week (see Table 5, Appendix A). Even in reduced-till systems, application close to the time of planting will decrease the likelihood of loss by runoff or erosion. Manures are generally best applied just before soil tillage, but diligence is needed to ensure that National Organic Program harvest-interval regulations related to manure applications are followed.

N mineralized from manures during a rainy period may leach N into the water table or, if eroded, may carry into surface waters. Fall applications of manure on annual row crops do risk considerable N loss, even if incorporated in regions with significant winter/spring precipitation. The ammonium is converted to nitrate, and then is subject to leaching and denitrification before the N is available for next year's crop (Magdoff & Van Es, 2009). Denitrification is the microbial conversion of nitrates to nitrous oxides, and mostly occurs in soils that become anaerobic due to high water tables, poor drainage, or surface sealing. But regardless of the mechanism of N loss, the loss of N to water or air should be minimized through applications of N appropriate to plant needs. The forms of N applied, as well as its time of application can influence the percentage N taken up by plants, and lost to leaching or the atmosphere. The goal of N management is to have minimal nitrate-N present in soil that could be leached through rainfall or irrigation or denitrified. This may allow for fall manure applications with low risk of N leaching due to winter rains. Manures mixed with large amounts of high-carbon bedding (straw or wood shavings) may immobilize N during the early stages of decomposition, and should be composted prior to application, or be applied long enough prior to planting to reduce the risk of immobilization.

Cover crop incorporation can be considered a fertilizer application and as Figure 6 shows, the peak PAN generally occurs after four weeks. That may be two or more weeks before the crop has its highest demand for N. Attempting to plant too soon after cover crop incorporation risks pest and pathogen problems associated with the decomposing cover crop.

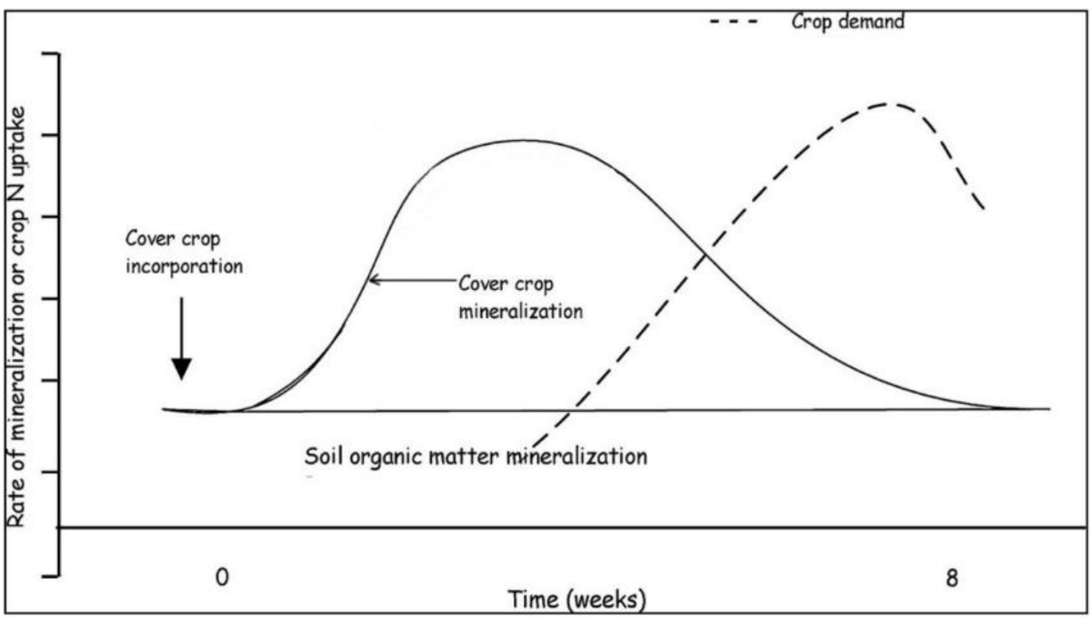


Figure 6. Timing of nitrogen (N) mineralization from cover crop residue in relation to crop N uptake (adapted from Gaskell et al., 2006). From Gaskell, M., and R. Smith. 2007. "Nitrogen Sources for Organic Vegetable Crops." HortTechnology October-December 2007 vol. 17 no. 4, 431-441) Note: Soil temperature plays an important role in the rate of N mineralization from soil organic matter.

11

Section 4.
Evaluating Risk of Leaching and Runoff

Risk factors associated with N leaving the field are not quite the same as P leaving the field, although erosion and runoff are risks common to both nutrients. These elements act differently in the environment—P does not volatilize and typically does not leach in significant quantities.

Soils with active biology and good aggregate stability will tend to absorb rainfall more easily. While this might reduce the likelihood of P runoff through soil erosion, leaching of N may still be a problem. Good aggregate stability also encourages airflow into the soil, reducing the risk of anaerobic conditions which cause denitrification and loss of soil N.

Assessing Risk of N Leaving the Field

Nitrates (NO_3), and ammonium (NH_4+) are both easily dissolved in water. As noted above in the 'Timing' section, application and incorporation of manures close to time of planting will help reduce risks from volatilization, leaching, and runoff. Fall manure applications risk significant N loss even if incorporated.

P Indices for the Pacific Northwest States

As of July 2013, all states are reviewing and updating their P-indices.

Risk Factors for P Leaving Field

- Soil texture (sandy vs. clay)
- Slope of land
- Amount of cover on the soil
- Amount and timing of nutrient application
- Amount and timing of rainfall events and irrigation
- Distance to perennial surface water
- High P levels in the soil

Mitigation Practices to Reduce P Loss from Field

- Keep crop residues on soil surface.
- Minimize duration of bare soil exposure to weather.
- Reduce length of slope using terraces, berms (vegetated with grass or hedgerows).
- Tailwater ponds reduce sediment in runoff.
- Reduce amount of P in soil surface over time by reducing the application rates for manure, compost or other products with high P content.
- No-till or minimum till may increase water soluble P loss from a field via heavy rainfall or irrigation events, even though no-till does reduce total P loss from erosion.

Figure 7. Organic management builds a soil full of aggregates and organic matter. Photo: Oregon Tilth

12

Section 5.
Calculating Nutrient Application Rates Based on Risk Analysis

In previous sections, you developed numbers for nutrient levels already in the ground from previous manure/ compost applications, cover crops, and irrigation water. You also developed numbers for nutrient levels required by the crop. If crop nutrient requirements exceed your nutrient availability estimates, then additional nutrients are required. Refer to Tables 4-6, Appendix A for a listing of organic nutrient sources and their N-P-K values. Review the P-Index for your state for detailed guidance.

☞ **Estimates for PAN and nutrient applications should be reviewed, and sources and amounts of nutrients to be applied should be revised as appropriate in light of P-Index policy. Add up lines 7, 8, and 9 under *Nutrient Sources* for "planned" and follow instructions in line 11, and the text in red at the bottom of the table.**

Record keeping

Organic farmers are required to maintain extensive records about their crops, crop rotations, inputs, and other practices used on their farms, including a soil fertility management plan. The 590 implementation requirement developed by following this guide can likely be used to meet many of the requirements related to soil fertility. Additionally, this information can be used to help organic growers track the trajectory of the SOM because organic rules state that the grower must maintain or improve SOM content.

Figure 8. A diversified organic farm manages nutrients in blocks to ensure the varying conditions and crops are appropriately considered. Photo: Sarah Brown (Oregon Tilth)

13

Appendix A. Tables

Table 4. Nutrient analysis (percent by weight) of common organic fertilizer materials (Gaskell et al., 2007)

Material	Nitrogen (% N)	Phosphorus (% P_2O_5)	Potassium (% K_2O)
Chilean nitrate	16	0	0
Blood meal	12	0	0
Feather meal	12	0	0
Fish meal/powder	10-11	6	2
Seabird & bat guano	9-12	3-8	1-2
Meat and bone meal	8	5	1
Soybean meal	7	2	1
Processed liquid fish residues*	4	2	2
Alfalfa meal	4	1	1
Pelleted chicken manure	2-4	1.5	1.5
Bone meal	2	15	0
Kelp	<1	0	4
Soft rock phosphate	0	15-30**	0
Potassium-magnesium sulfate	0	0	22
Cocoa shells	1	1	3
Cottonseed meal	6	2	2
Granite dust	0	0	5
Hoof & horn meal	11	2	0
Seaweed, ground	1	0.2	2
Muriate of potash (KCl)	0	0	60

* Note: all analyses are % by weight, as specified in state fertilizer laws. For liquids, product density (weight per gallon) should be used to calculate nutrient application rate: (g/ac)*(lb nutrient/g)=(lb nutrient/ac)

* *Soft rock phosphate provides only 1-3% of its P in acid soils, and little or no P in soils with pH over 7.

14

Table 5. Nutrient Content of Common Animal Manures and Manure Composts

This table includes general estimates of nutrient availability for manures and composts. These can vary widely depending on animal feed, management of grazing, the age of the manure, amount and type of bedding, and many other factors. Manure applications must be done in accordance with NOP 205.203 C.1-3. See page 4.

Production Guide for Organic Snap Beans for Processing. 2012. (A. Seaman, ed.) Cornell University Coop¬erative Extension. 50 p. http://nysipm.cornell.edu/organic_guide/bean.pdf

	Total N	P_2O_5	K_2O		N1[1]	N2[2]	P_2O_5	K_2O
	Nutrient content lbs/ton				Available nutrients lbs/ton in first season			
Dairy (with bedding)	9	4	10		6	2	3	9
Horse (with bedding)	14	4	14		6	3	3	13
Poultry (with litter)	56	45	34		45	16	36	31
Composted dairy manure	12	12	26		3	2	10	23
Composted poultry manure	17	39	23		6	5	31	21
Pelleted poultry manure[3]	80	104	48		40	40	83	43
Swine (no bedding)	10	9	8		8	3	7	7
	Nutrient content per 1000 gal				Available nutrients per 1000 gal in first season			
Swine finishing (liquid)	50	55	25		25 *	20 +	44	23
Dairy (liquid)	28	13	25		14 *	11 +	10	23

1. N1 is an estimate of the total available for plant uptake when manure is incorporated within 12 hours of applications.
2. N2 is an estimate of the total N available for plant uptake when manure is incorporated after 7 days.
3. Pelletized poultry manure compost.

* injected

\+ incorporated

15

Table 6. Pounds of fertilizer/acre needed to provide 20 to 100 pounds of N, P, K per acre.
This table is divided into three sections. Production Guide for Organic Snap Beans for Processing. 2012. (A. Seaman, ed.) Cornell University Cooperative Extension. 50 p. http://nysipm.cornell.edu/organic_guide/bean.pdf

Sources of Nitrogen	Pounds of fertilizer per acre to provide 20 to 100 pounds of N per acre				
	20# N	40	60	80	100
Blood meal 13% N	150	310	460	620	770
Soy meal 6% N (x 1.5*) also contains 2% P and 3% K_2O	500	1,000	1,500	2,000	2,500
Fish meal 9% N (also contains 6% P_2O_5)	220	440	670	890	1,100
Alfalfa meal 2.5% N also contains 2% P and 2% K_2O	800	1,600	2,400	3,200	4,000
Feather meal, 15% N (x 1.5*)	200	400	600	800	1,000
Chilean nitrate 16% N cannot exceed 20% of crop's need.	125	250	375	500	625

Sources of Phosphorus	Pounds of fertilizer per acre to provide 20 to 100 pounds of P_2O_5 per acre				
	20# P_20_5	40	60	80	100
Bonemeal 15% P_2O_5	130	270	400	530	670
Rock Phosphate 30% total P_2O_5 (x 4*)	270	530	800	1,100	1,300
Fish meal 6% P_2O_5 (also contains 9% N)	330	670	1,000	1,330	1,670

Sources of Potassium	Pounds of fertilizer per acre to provide 20 to 100 pounds of K_2O per acre				
	20# K_2O	40	60	80	100
Sul-Po-Mag 22% K_2O (also contains 11% Mg)	90	180	270	360	450
Wood ash (dry, fine, grey) 5% K_2O, also raises pH	400	800	1,200	1,600	2,000
Alfalfa meal 2% K_2O also contains 2.5% N	1,000	2,000	3,000	4,000	5,000
Greensand or Granite dust 1% K_2O (x 4*)	8,000	16,000	24,000	32,000	40,000
Potassium sulfate 50% K_2O	40	80	120	160	200

* Application rates for some materials are multiplied to adjust for their slow-to-very-slow release rates.

16

Table 7. Plant-Available Nitrogen (PAN) Estimates for Uncomposted Manure

This table is based on Oregon State University's organic fertilizer calculators, which compare the nutrient values and cost of cover crops, organic and synthetic fertilizers, and compost. One calculator is for larger acreages, providing numbers on a per-acre basis, and one is for smaller plots, providing numbers on a per-square-foot basis. These tools help calculate organic fertilizer needs and can be accessed at **http://smallfarms.oregonstate.edu/node/175833/done?sid=476**

Total N analysis (percent dry weight) of uncomposted fresh manure or other uncomposted organic fertilizer product	PAN Estimate		Total N analysis (lbs N per dry ton) of uncomposted fresh manure or other uncomposted organic fertilizer product	PAN Estimate	
Your value	After 4 weeks	After 10 weeks	Your value	After 4 weeks	After 10 weeks
Total N (% dry weight)	% PAN	% PAN	lb total N per dry ton	lbs PAN per dry ton	lbs PAN per dry ton
0.5	-23	-8	10	-2	-1
1.0	-15	0	20	-3	0
1.5	-8	8	30	-2	2
2.0	0	15	40	0	6
2.5	8	23	50	4	11
3.0	15	30	60	9	18
3.5	23	38	70	16	26
4.0	30	45	80	24	36
4.5	38	53	90	34	47
5.0	45	60	100	45	60
5.5	53	68	110	58	74
6.0	60	75	120	72	90
7.0	60	75	140	84	105
8.0	60	75	160	96	120
9.0	60	75	180	108	135
10.0	60	75	200	120	150
11.0	60	75	220	132	165
12.0	60	75	240	144	180

17

Table 8. Estimated Plant Available Nitrogen (PAN) from Finished Compost
This table is based on the PAN equation used with the Oregon State University Organic Fertilizer and Cover Crop Calculator, http://smallfarms.oregonstate.edu/calculator

Total N analysis of finished compost	PAN Estimate		Total N analysis of finished compost	PAN Estimate
Your value	10 wk		Your value	10 wk
Total N (% dry wt)	% PAN		lb total N per dry ton	lb PAN per dry ton
1.0	0		20	0
1.5	5		30	1.5
2.0	10		40	4.0
2.5	10		50	5.0
3.0	10		60	6.0

Source: OSU Fertilizer & Cover Crop Calculator (online; 2013)

Instructions

1. Find %N analysis of the compost in "Total N analysis" column.
2. PAN estimates are listed on the same row in units of "%PAN" or "lb PAN per dry ton."
3. To get PAN estimate for "as-is" moisture content, multiply Table PAN estimate (given for oven dry product; 100% dry matter) x %DM/100 in "as-is" manure.

Example

Lab analysis of a compost is 1.5% N or 30 lb N per ton (dry weight basis).
The "as-is" fertilizer product contains 40% dry matter and 60% moisture.
From the Table above, PAN released in soil after 10 wk is estimated to be 5% of product total N analysis (dry weight basis), or 2 lb PAN per dry ton.
At "as-is" dry matter content (40% in this example), the product provides about 0.8 lb PAN per "as-is" ton (calculated as 2 lb PAN per dry ton x 0.4).
Use this table only when you are sure that compost is "finished". Many poultry litter "composts" that smell of ammonia are not "finished" and they have PAN similar to fresh manure.

Additional Considerations

Compost analysis for mineral N (ammonium + nitrate-N) can provide useful data to guide application rate.
First year PAN release from compost is approximately equal to the sum of ammonium + nitrate-N applied.
Very little first-year PAN comes from mineralization of organic N in finished compost.
Composts usually contain less than 3% total N. "Composts" with more than 3% N are often not finished, they are dried manure.

18

Table 9. Estimated Plant Available Nitrogen (PAN) from Cover Crops
Table from PNW 636, Estimating Plant-available Nitrogen Release from Cover Crops (Sullivan and Andrews, 2012), ©Oregon State University. Used by permission.

Total N analysis of cover crop	PAN Estimate		Total N analysis of cover crop	PAN Estimate
Your value	After 10 weeks		Your value	After 10 weeks
Total N (% dry wt)	% PAN		lb total N per dry ton	lb PAN per dry ton
1.0	0		20	0
1.5	13		30	4
2.0	23		40	9
2.5	32		50	16
3.0	40		60	24
3.5	47		70	33
4.0	54		80	43

Rationale

Cover crop PAN can be estimated by a variety of methods.

A general approach to estimating cover crop PAN is provided in the SARE publication "Managing Cover Crops Profitably."

The SARE approach uses "rule of thumb" estimates of cover crop biomass and N concentration and PAN.

The approach described here is for growers who are willing to make some on-site measurements to estimate PAN from cover crops.

Approach

- A laboratory analysis for cover crop total N as a percentage in dry matter (DM) is a good predictor of a cover crop's capacity to release PAN for the summer crop.
- When cover crops contain a low N percentage (<1.5% N in DM), they provide little or no PAN.
- When cover crops contain a high N percentage (3.5% N in DM) they provide approximately 35 lb PAN per ton of dry matter.
- PAN release increases linearly, as cover crop N percentage (in DM) increases from 1.5 to 3.5%.
- Cover crops decompose rapidly, and release or immobilize PAN rapidly. Most PAN is released in 4 to 6 weeks after cover crop kill.
- Values for cover crop PAN listed here are most applicable to winter cover crop/summer vegetable crop rotations in western Oregon and Washington.
- The timing of PAN release will differ in regions outside of western WA and OR, but we expect a strong relationship between cover crop N% and PAN to be found in most locations.

Instructions

The table above is reproduced from PNW Extension publication 636, Estimating Plant-available Nitrogen Release from Cover Crops. To use these tables you will need to have either taken a lab analysis of your cover crop to identify the total %N or will be using an estimate based on Table 3.

1. Find %N analysis of the cover crop in "Total N analysis" column (far left). PAN estimates are listed on the same row in units of "%PAN" expressed as a percentage of total %N, or "lb PAN per dry ton."

Example

The fresh weight of a rye/vetch cover crop is about 16.5 tons/ac (33,000 lbs/ac). With 15% dry matter the cover crop produced 4,950 lbs dry matter/ac.

If total %N is analyzed or estimated to be 3%, percent PAN is approximately 40% (table 9). Multiply 4,950 lbs dry matter x (3/100) x (40/100) = 59 lbs PAN/ac.

19

Appendix B. Conversion Tables

Table 10. Converting pounds per acre to pounds per 1000 square feet

Pounds per acre	Pounds per 1000 square feet
50	1.1
75	1.7
100	2.3
125	2.9
150	3.4
175	4.0
200	4.6
225	5.2
250	5.7
275	6.3
300	6.9
325	7.5
350	8.0
375	8.6
400	9.2

One acre = 43,560 square feet

X lbs per acre = Y lbs per 1000 square feet

Example

Example 100 lbs per acre = 3.7 oz per 100 square feet

(100 lbs = 1,600 oz;

1,600 divided by 43,560 = .037 oz per sq. ft.; multiply by 100 to get 3.7 oz per 100 sq. ft.)

Table 11. Converting percentages of material per dry ton to pounds of material per dry ton

Percent of material dry weight basis	Pounds of material per dry ton
0.5	10
1	20
1.5	30
2	40
2.5	50
3	60
3.5	70
4	80
4.5	90
5	100
5.5	110
6	120
6.5	130
7	140
7.5	150
8	160
8.5	170
9	180
9.5	190
10	200
10.5	210
11	220
11.5	230
12	240

X percent of material per dry ton = Y lbs of material per dry ton

Example

.5% per dry ton = 10 lbs per dry ton

(.5% of material per dry ton =

.005 x 2,000 lbs per dry ton =

10 lbs of material per dry ton)

20

Table 12. Nutrient Management Conversion Factors					
Measurement	Unit	Symbol	Multiply by	To Obtain	Symbol
Volume	acre-inch	ac-in	27000	gallon	gal
	gallon	gal	8.35	pound	lb
Concentration	parts per million	ppm or mg/kg	0.002	pound/ton	lb/t
	parts per million	ppm or mg/L	0.00835	pound/1000 gallon	lb/1000 gal
	parts per million	ppm or mg/L	0.227	pound/acre-inch	lb/ac-in
	percent	%	20	pound/ton	lb/t
	percent	%	83.5	pound/1000 gallon	lb/1000 gal
	percent	%	2266	pound/acre-inch	lb/ac-in
	percent	%	10000	parts per million	ppm
	pound/ 1000 gallon	lb/1000 gal	27	pound/acre-inch	lb/ac-in
Dry or wet	percent dry matter (total solids)	% DM	0.01	solids fraction	DM
	percent moisture	% moisture	0.01	moisture fraction	MF
	manure, dry wt. basis		100/(%DM)	manure, "as-is" basis	
Nutrients	phosphorus	P	2.29	phosphate	P_2O_5
	potassium	K	1.20	potash	K_2O
	nitrogen in nitrate form	NO_3- N	1	nitrogen	N
	nitrogen in ammonium form	NH_4- N	1	nitrogen	N

21

Appendix C.
References to NRCS Publications and Other Resources

National Organic Program

www.ams.usda.gov/AMSv1.0/nop

Section 1. Determining Crop's Need for N, P, and K (page 5)

Baldwin et al. Crop Rotations on Organic Farms. www.cefs.ncsu.edu/resources/organicproductionguide/croprotationsfinaljan09.pdf

California Department of Food and Agriculture's (CDFA) Fertilizer Research & Education Program. www.cdfa.ca.gov/is/fflders/frep.html

Collins, D. Washington State University, Soil Testing: A Guide for Farms with Diverse Vegetable Crops. https://pubs.wsu.edu/ItemDetail.aspx?ProductID=15520&SeriesCode=&CategoryID=&Keyword=EM050E

Cornell Guides for Organic Fruits, Vegetables and Dairy. http://nysipm.cornell.edu/organic_guide/

Gaskell, M., R. Smith, J. Mitchell, S.T. Koike, C. Fouche, T. Hartz, W. Horwath, and L. Jackson. 2006. Soil Fertility Management for Organic Crops. University of California. 8 p. http://anrcatalog.ucdavis.edu/pdf/7249.pdf

Idaho Nutrient Management, www.extension.uidaho.edu/nutrient. Look under *Crop Nutrient Requirements.*

Oregon State University Nutrient Management Guidelines for Crops. http://extension.oregonstate.edu/catalog/details.php?search=nutrient+management&submit.x=0&submit.y=0

Minnesota Vegetable Nutrient Management Guide. www.extension.umn.edu/distribution/cropsystems/dc5886.html

Mohler, et al. 2009. Crop Rotation on Organic Farms. http://palspublishing.cals.cornell.edu/nra_order.taf?_function=view&ct_id=40

Section 2. Developing Nutrient Credits (pages 6-9)

Andrews, N., D. Sullivan, J. Julian, and K. Pool. Oregon State University Organic Fertilizer and Cover Crop Calculators. http://smallfarms.oregonstate.edu/node/175833/done?sid=476

Bowman, G., C. Shirley, and C. Cramer. 1998. Managing Cover Crops Profitably. SARE Publication. 212 p. www.sare.org/publications/covercrops/covercrops.pdf

California Department of Food and Agriculture's (CDFA) Fertilizer Research & Education Program. www.cdfa.ca.gov/is/fflders/frep.html

Cornell Guides for Organic Fruits, Vegetables and Dairy. http://nysipm.cornell.edu/organic_guide/

Gaskell, M., R. Smith, J. Mitchell, S.T. Koike, C. Fouche, T. Hartz, W. Horwath, and L. Jackson. 2006. Soil Fertility Management for Organic Crops. University of California. 8 p. http://anrcatalog.ucdavis.edu/pdf/7249.pdf

Idaho Nutrient Management. www.extension.uidaho.edu/nutrient, look under *Crop Nutrient Requirements*

Jackson, B.P., P.M. Bertsch, M.L. Cabrera, J.J. Camberato, J.C. Seaman, C.W. Wood. 2003. Trace element speciation in poultry litter. J Environ Qual. 2003 Mar-Apr., 32(2):535-40.

Magdoff, F., and H. Van Es. 2009. Building Soils for Better Crops. SARE. 292 p. www.sare.org/Learning-Center/Books/Building-Soils-for-Better-Crops-3rd-Edition

Miyao, G., P. Robbins, and M. Cain. 2001. Winter cover crops before late-season processing tomatoes for soil quality and production benefits. Final Report Summary, CDFA Fertilizer Research and Education Program (FREP).

Oregon State University Fertilizer/Nutrient Management Guidelines for Crops http://extension.oregonstate.edu/catalog/details.php?sortnum=0134&name=Fertilizer+Guides&cat=Agriculture

22

Schonbeck, M.W. 1988. Cover Cropping and Green Manuring on Small Farms in New England and New York: An Informal Survey. East Falmouth, MA. New Alchemy Research Report #10.

Sullivan, D.M. and N.D. Andrews. 2012. Estimating plant-available nitrogen release from cover crops. Pacific Northwest Extension Publication 636. Oregon State University Extension. Corvallis, OR. 23 p.

Zimmer, Gary F. 2000. The Biological Farmer. Acres, USA. 352 p.

Section 3. Determining Nutrient Application Rates (pages 10-11)

Andrews, N., D. Sullivan, J. Julian, and K. Pool. Oregon State University Organic Fertilizer and Cover Crop Calculators. http://smallfarms.oregonstate.edu/node/175833/done?sid=476

Gaskell, M., and R. Smith. 2007. Nitrogen Sources for Organic Vegetable Crops. HortTechnology October-December 2007 vol. 17 no. 4, 431-441)

Section 4. Evaluating Risk of Leaching and Runoff (page 12)

Magdoff, F., and H. Van Es. 2009. Building Soils for Better Crops. SARE. 292 p. www.sare.org/Learning-Center/Books/Building-Soils-for-Better-Crops-3rd-Edition

Section 5. Calculating Nutrient Application Rates Based on Risk Analysis (page 13)

Organic Fertilizer Association of California. www.organicfertilizerassociation.org

Available N, P and K in Organic Fertilizer

Available N in Organic Fertilizers (in Production Guide for Organic Snap Beans for Processing). 2012. Cornell University Cooperative Extension. 50 p. http://nysipm.cornell.edu/organic_guide/bean.pdf

NRCS Agricultural Waste Management Field Handbook. www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/technical/ecoscience/mnm/?&cid=stelprdb1045935

Sullivan, D.M. and N.D. Andrews. 2012. Estimating plant-available nitrogen release from cover crops. Pacific Northwest Extension Publication 636. Oregon State University Extension. Corvallis, OR. 23 p.

Plant-Available Nitrogen (PAN) from Cover Crops

Selected characteristics of important cover crops for California, in: Selecting the right cover crop gives multiple benefits, Ingels, C., et al. October 1994. http://ucce.ucdavis.edu/files/repository/calag/tab4805p45.jpg

Sullivan, D.M. and N.D. Andrews. 2012. Estimating plant-available nitrogen release from cover crops. Pacific Northwest Extension Publication 636. Oregon State University Extension. Corvallis, OR. 23 p.

Other Resources

eOrganic provides resources for people seeking information about organic agriculture. The website is part of eXtension, which draws on the expertise of staff at American land-grant universities and extension programs. www.eorganic.info

ATTRA's Sources of Organic Fertilizers and Amendments Database.
https://attra.ncat.org/attra-pub/org_fert/

Building Soils for Better Crops
and
Crop Rotation on Organic Farms: A Planning Manual. These and many other useful publications are free to download from the USDA's Sustainable Agriculture Research and Education SARE.
www.sare.org/Learning-Center/Books

Cornell's Soil Health Assessment Training Manual
Free to download at: http://soilhealth.cals.cornell.edu/extension/manual.htm

Manures for Organic Crop Production. George Kuepper. 2003. ATTRA publication IP127. www.attra.ncat.org/soils.html

Ohio State University's The Biology of Soil Compaction Fact Sheet (2009). ohioline.osu.edu/sag-fact/pdf/0010.pdf

Organic Center's Assessing Soil Quality in Organic Agriculture (2006). www.organic-center.org/reportfiles/SoilQualityReport.pdf

NRCS Soil Quality/Soil Health website.
www.soils.usda.gov/sqi

Soil Fertility in Organic Systems: A Guide for Gardeners and Small Acreage Farmers (2013). Pacific Northwest Extension Publication 646.
https://pubs.wsu.edu/ItemDetail.aspx?ProductID=15585&SeriesCode=&CategoryID=&Keyword=646

Web Soil Survey. http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm

Regional Organic Fertilizer Vendors

CALIFORNIA

Harmony Farm Supply
3244 Gravenstein Hwy North
Sebastopol, CA 95472
Mailing address: P.O. Box 460, Graton, CA 95444
(707) 823-9125

New Era Farm Service
2904 East Oakdale Avenue
Tulare, CA 93274
Telephone: (559) 686-3833
Fax: (559) 686-1453
www.newerafarmservice.com

Recology (www.thecompoststore.com) operates several composting sites:

Grover Environmental Products
3909 Gaffery Road
Vernalis, CA 95385
(866) 764-5765

Feather River Organics
3001 North Levee Road
Marysville, CA 95901

South Valley Organics
3675 Pacheco Pass Highway
Gilroy, CA 95020

Jepson Prairie Organics
6426 Hay Road
Vacaville, CA 95687
Mailing Address:
235 North First Street, Dixon, CA 95620
(800) 208-2370
Fax: (707) 678-5148

Sonoma Compost
550 Mecham Road
Petaluma, CA 94952
Phone: (707) 664-9113
Fax: (707) 664-1943
www.sonomacompost.com

Peaceful Valley Farm & Garden Supply
125 Clydesdale Court
P.O. Box 2209
Grass Valley, CA 95945
(888) 784-1722
www.groworganic.com

24

Z-Best Composting Facility
980 SR-25 (Hwy. 25 and Bolsa Road)
Gilroy, CA 95020
Phone: (408) 846-1573, (408) 263-2384
Fax: (408) 263-2393
www.z-best.com

IDAHO

OREGON

Concentrates, Inc. Organic Agriculture Specialists
5505 S.E. International Way
Milwaukie,OR 97222
(503) 234-7501, (800) 388-4870
Fax: (503) 234-7502
www. concentratesnw.com

Marion Ag Supply
Farm Store: 1-888-814-5727
503-633-4281
http://www.marionag.com/locations.htm

Naomi's Organic Farm Supply
2615 SE Schiller St.
Portland, OR 97206
(503) 517-8551
www. naomisorganic.blogspot.com

Nature's Needs
9570 NW 307th Avenue
North Plains, OR 97133
(503) 647-9489
Fax: (503) 647-9485

NW Greenlands

2200 NE Orchard Avenue
McMinnville, OR 97128
(503) 434-1671

2140 Turner Road
Salem, OR 97302
(503) 585-4510

8712 Aumsville Hwy SE
Salem, OR 97317
(503) 749-3117

NEVADA

25

Appendix D.
Implementation Requirement Worksheet

590 Organic NMP Implementation Requirement (IR)

Natural Resources Conservation Service **Jan-13**

NUTRIENT MANAGEMENT SPECIFICATION SHEET

Client:	**Tract:**	**Date:**
County:	**Fields:**	**Acres:**
Prepared by:		

DESIGN APPROVAL:								
Practice Code NO.	**PRACTICE**	**LEAD DISCIPLINE**	**CONTROLLING FACTOR**	**UNITS**	**JOB CLASS**			
					I	**II**	**III**	**IV**
590	**Nutrient Management**	**CED-EE& BSCD-Agron**	**Area**	**Acres**	**160**	**320**	**640**	**All**
This practice is defined as Job Class:								
Design Approved by: /s/					**Date:**			
Job Title:								

CLIENTS' ACKNOWLEDGEMENT STATEMENT:

The Client acknowledges that :

a. They have received a copy of the specification and understand the contents and requirements.

b.

The following information must be provided to NRCS by the client before this practice can be certified as applied :

❑ Nutrient management application records including soil, water plant and organic byproduct test results as specified in the plan;

❑ Documentation of quantities, analyses, sources, dates, and application methods of nutrients applied; documentation of weather conditions and soil moisture at the time of application, documentation of lapsed time to manure incorporation, rainfall, or irrigation.

❑ Periodic review of nutrient management including adjustments made; at a minimum this plan will be reviewed with each soil test cycle.

c. It shall be the responsibility of the client to obtain all necessary permits and/or rights, and to comply with all ordinances and laws pertaining to the application of this practice.

Accepted by: /s/____ **Date:** ____________________

CERTIFICATION

I have completed a review of the information provided by the client and certify this practice has been applied.

Certification by: /s/____ **Date:**____________________

Job Title: ______________________________________

The U.S. Department of Agriculture (USDA) prohibits discrimination in all its programs and activities on the basis of race, color, national origin, age, disability, and where applicable, sex, marital status, familial status, parental status, religion, sexual orientation, genetic information, political beliefs, reprisal, or because all or a part of an individual's income is derived from any public assistance program. (Not all prohibited bases apply to all programs.) Persons with disabilities who require alternative means for communication of program information (Braille, large print, audiotape, etc.) should contact USDA's TARGET Center at (202) 720-2600 (voice and TDD). To file a complaint of discrimination write to USDA, Director, Office of Civil Rights, 1400 Independence Avenue, S.W., Washington, D.C. 20250-9410 or call (800) 795-3272 (voice) or (202) 720-6382 (TDD). USDA is an equal opportunity provider and employer.

26

Organic Nutrient Management Specification

Natural Resources Conservation Service
NUTRIENT MANAGEMENT PLAN

Client: **Tract Number:** **Field Numbers:**

Select all purposes that apply

- ❑ Budget and supply adequate nutrients for plant production
- ❑ Minimize ag nonpoint source pollution of surface and groundwater resources (water quality)
- ❑ Properly utilize manure/organic materials as a plant nutrient source
- ❑ Maintain or improve soil condition
- ❑ Protect air quality by reducing nitrogen emissions (ammonia and NOx compounds) and the formation of particulates
- ❑ Attach an aerial photo or site map showing fields where nutrient management is being implemented

NOTE: You may have criteria to meet that are NOT handled in this job sheet, please refer to the standard.

FIELD AND CROP INFORMATION

List crop rotation. Identify current crop with a check.			
P Index:		Soil Map Unit:	
No Data			
Current	Crop	Yield	units
❑			
❑			
❑			
❑			
❑			
❑			

Soil Test Information*

Date:			**Laboratory Used:**		
Depth:			❑	**Certified Lab**	
N (NO_3 + NH_4)	**P**	**K**	**pH**	**OM%**	**Other:**
Use ppm for units, refer to last page of job sheet for conversions					
* For some crops (e.g., berries and tree fruit), Extension recommends using leaf analysis rather than soil tests.					

RECOMMENDED NUTRIENTS TO MEET YIELDS

	PAN (lbs/acre)	**P_2O_5 (lbs/acre)**	**K_2O (lbs/acre)**	**pH**
Reference:				
Notes on Adjustments:				

27

NUTRIENT SOURCES

Credits	PAN (lbs/acre)	P_2O_5 (lbs/acre)	K_2O (lbs/acre)	Other__ (lbs/acre)
1. Adjustment to soil N Mineralization				
2. Nitrate from irrigation water				
3. Nitrogen from previous cover crop				
4. Other source(s)				
5. TOTAL CREDITS				
6. FINAL NUTRIENTS RECOMMENDED FOR YIELD (FOR UPCOMING CROP)				
Nutrients to be applied to the field. Rate (lbs/acre)	**Planned**	**Planned**	**Planned**	**Planned**
7. Manure rate per acre**				
8. Compost* rate per acre**				
9. Specialty Organic Fertilizer Product: Analysis:				
10. Total Organic Inputs (add lines 7, 8, and 9)				
11. Nutrient status (subtract line 6 from line 10)				

If the number on line 11 is **positive**, this indicates over application.
If the number on line 11 is **negative**, this indicates under application.
If the number is 0, then planned applications meet and do not exceed crop requirements.
Include an explanation of these numbers, especially as they relate to the Phosphorus Index.

Describe Timing below (when organic applications are and are not going to occur) and Application Methods (surface applied with or without incorporation, time lapse until incorporation, applied at planting with tillage, injected, and other details relevant to how the organic amendments are applied).

Other Information/ Considerations to Nutrient Management Plan: 1) Calibration of application equipment is required. See Calibration tab for guidance on equipment calibration. Fillable calibration worksheets are available on Oregon's eFOTG, Section IV, Conservation Practices, WasteUtilization (633) Job Sheet and shall be attached to this nutrient management plan and reviewed with the producer.

*These values are developed, in part, from the soil tests.

Manure: Manure can be applied to organically grown crops, but with pre-harvest interval (PHI) restrictions. (See page 2, sections C. 1-3.) How the crop is grown and harvested with regard to soil contact will determine which pre-harvest interval is to be used.

***Compost**: Only agricultural products can be certified. Thus, soil and compost are not eligible for organic certification. However, see page 2, §205.203, C.2i-iii relating to how compost can be used and produced on organic farms.

28

	Additional information/requirements	
	❑	Location of sensitive areas
	❑	Soil test schedule including designation of critical fields
	Nutrient Application Bullets	
	❑	* Apply nutrient materials uniformly to application area(s).
	❑	* Nutrients shall not be applied to frozen, snow-covered, or saturated soil if the potential risk for runoff exists.
	❑	*Nutrient applications associated with irrigation systems shall be applied in a manner that prevents or minimizes resource impairment.
	❑	*Calibrate application equipment to ensure recommended rates are applied. See attached information on equipment calibration.
Operation and Maintenance		

The client is responsible for safe operation and maintenance (O&M) of the practice. O&M includes:

- Periodic plan review:
  At a minimum, plans will be reviewed and revised with each soil test cycle.
- Conduct additional manure analyses when there are significant changes in animal numbers and/or feed management.
- Calibrate application equipment to within __% of the recommended rate.
- Document the actual rate at which nutrients were applied.
- Handle all nutrient material with caution.
- Wear appropriate protective clothing.
- Clean up residual materials from equipment and recycle or dispose of properly.
- Recordkeeping: soil/water/organic materials analyses, quantities/analyses/sources of nutrients applied, dates and methods of application, weather conditions and soil moistures at the time of application, lapsed time to incorporation/rainfall/irrigation, crops planted, planting/harvest dates, crop yields, crop residues removed, and the specifics of plan reviews.

29

The Waste Utilization Worksheet below can be used to develop estimates for the lbs/acre of nutrients applied to a field in the form of "waste," an unfortunate term in this context. Proper calibration of application machinery is needed to know how much material has been applied, but the nutrient analysis of the material must also be known in order that the lbs nutrient/acre can be calculated.

633 OR-Specification

Natural Resources Conservation Service **May 2003**

WASTE UTILIZATION WORKSHEET

Spreader Equipment Calibration

(Using a Full Spreader Load)

Name:____ Date:________________

Operator:____________________________________ Spreader ID:___________

Perform the following operations to calibrate the solids spreader equipment:

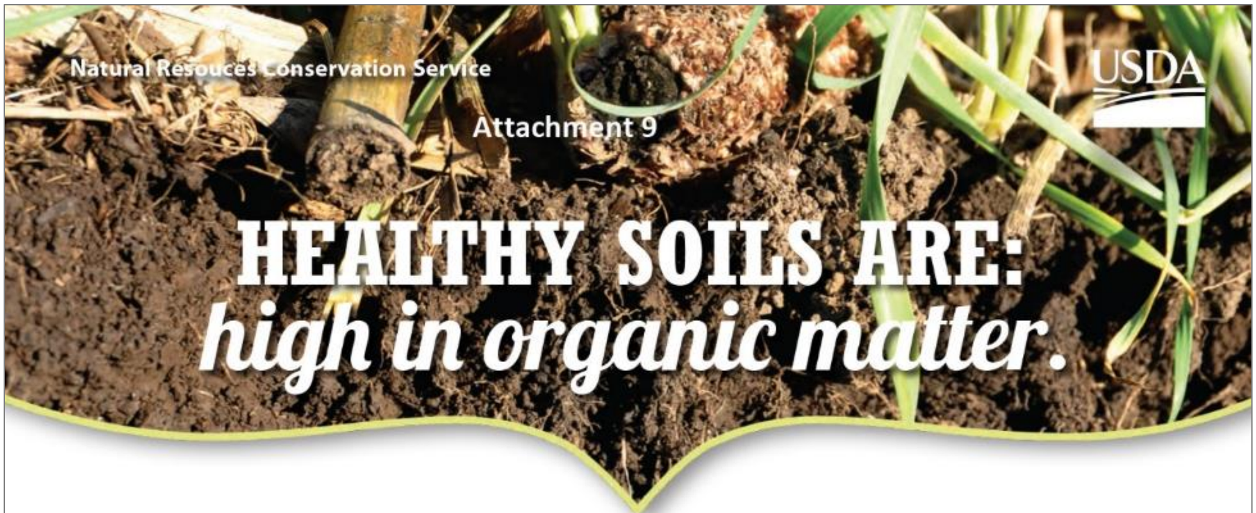
- ✓ Determine the weight of the waste material loaded in the spreader by using truck scales to weigh the spreader equipment when it is empty and full.
- ✓ Spread the loaded spreader on the field using consistent speed and spreader settings to cover the field uniformly. Spread in a rectangular pattern so the area calculation will be simple. Record engine rpm and gear settings used.
- ✓ Measure the length and width covered by the full load and compute the application rate in tons per acre using this worksheet.

Data and Calculations:

Steps	ID of Calibration Test					
	A	B	C	D	E	F
1. Date of calibration test-						
2. Engine RPM during spreading -						
3. Gear selected during spreading -						
4. Weight of empty spreader (lb) =						
5. Weight of loaded spreader (lb) =						
6. Weight of Waste in spreader (lb) - line 5 – line 4 =						
7. Length of spreading area (ft) =						
8. Width of spreading area (ft) =						
9. Area spread (sq ft) - line 7 x line 8 =						
10. Waste applied (lb/sq ft) – line 6 ÷ line 9=						
11. Convert to tons per acre - Line 10 x 21.78=						
12. Average Application Rate (tons per acre) – Sum of values in cells A11 through F11 divided by the total number of calibrations completed =						

Additional Notes:

30



HEALTHY SOILS ARE: *high in organic matter.*

Losing Organic Matter

Organic matter is vital to healthy soils, yet most modern agricultural operations are not managed in ways to retain high levels. Only half the original organic matter remains in most modern cultivated soils. In general, organic matter levels have fallen from 5-6 percent of the soil to less than 3 percent on most cropland soils.

Using tillage depletes organic matter. Each time the soil is tilled, oxygen is stirred into it, stimulating microbial action to decompose organic matter at an accelerated rate. As a matter of fact, when a woodland is cleared and planted or a prairie is plowed, most of the organic matter that was built over hundreds of years is lost within 10 years of tillage.

Combining frequent tillage with farming practices that leave little plant residue for soil microbes to eat (such as burning or removing crop residues) will lead to the depletion of organic matter.

unlock the SECRETS IN THE SOIL

ORGANIC MATTER *matters*. IN FACT, THERE MAY BE NO OTHER COMPONENT THAT'S MORE IMPORTANT TO A HEALTHY SOIL THAN ORGANIC MATTER.

The tiny fraction of soil composed of anything and everything that once lived—organic matter—is more than an indicator of healthy soils.

The carbon in organic matter is the main source of energy for the all-important soil microbes and is also the key for making nutrients available to plants. The list of positive influences high levels of organic matter have on healthy soils includes:

1. Provides a carbon and energy source for soil microbes
2. Stabilizes and holds soil particles together
3. Supplies, stores, and retains such nutrients as nitrogen, phosphorusand sulfur
4. Improves the soil's ability to store and move air and water
5. Contributes to lower soil bulk density and less compaction
6. Makes soil more friable, less sticky, and easier to work
7. Retains carbon from the atmosphere and other sources
8. Reduces the negative environmental effects of pesticides, heavy metals and other pollutants
9. Improves soil tilth in surface horizons
10. Increases water infiltration rates
11. Reduces crusting
12. Reduces water runoff
13. Encourages plant root development and penetration
14. Reduces soil erosion

HEALTHY SOILS ARE: *high in organic matter.*

Considering the long list of benefits organic matter has on soil health and crop production, increasing organic matter may well be the most important management step a producer can take to improve a farm's profitability and sustainability. In general, there are three ways to do that:

1. Increase the amount of plant and root production;
2. apply carbon-rich materials to the soil; and
3. use practices that slow rather than speed decomposition.

Cover crops, green manure crops, and perennial forage crops add organic matter, as do compost and manure. Growing crops and roots add biomass above and below the soil surface. However, not all that biomass is converted to soil organic matter—much of it is released as carbon dioxide and water. It can take 20,000 pounds of organic inputs such as crop residue to increase the actual soil organic matter from 4 percent to 5 percent.

Compost in particular breaks down more slowly and improves soil structure more quickly than other organic materials. Manure breaks down quickly to add nutrients for crops, but takes longer to improve the soil than compost.

COMPARING ACTIVE AND STABILIZED ORGANIC MATTER

	PORTION OF ALL ORGANIC MATTER	DECOMPOSITION TIME	FUNCTIONAL IMPORTANCE
ACTIVE	One-half to two-thirds	Up to several decades	Decomposes organic material to produce plant nutrients
STABILIZED	One-third to one-half	A century or more	Exceptional water holding capacity, soil structure benefits; reservoir for nutrients, including carbon

WANT TO LEARN MORE? VISIT WWW.NRCS.USDA.GOV

Active and Stabilized Organic Matter

Organic matter can be divided into two categories: active and stabilized. The portion made of fresh organic material and living organisms, as well as partially decomposed material that is slowly decomposing, is called "active organic matter."

Active organic matter and the microbes that feed on it are central to nutrient cycles in the soil. Nutrients, especially nitrogen, phosphorus, and sulfur, are held in this active organic matter until soil organisms release them for plant use.

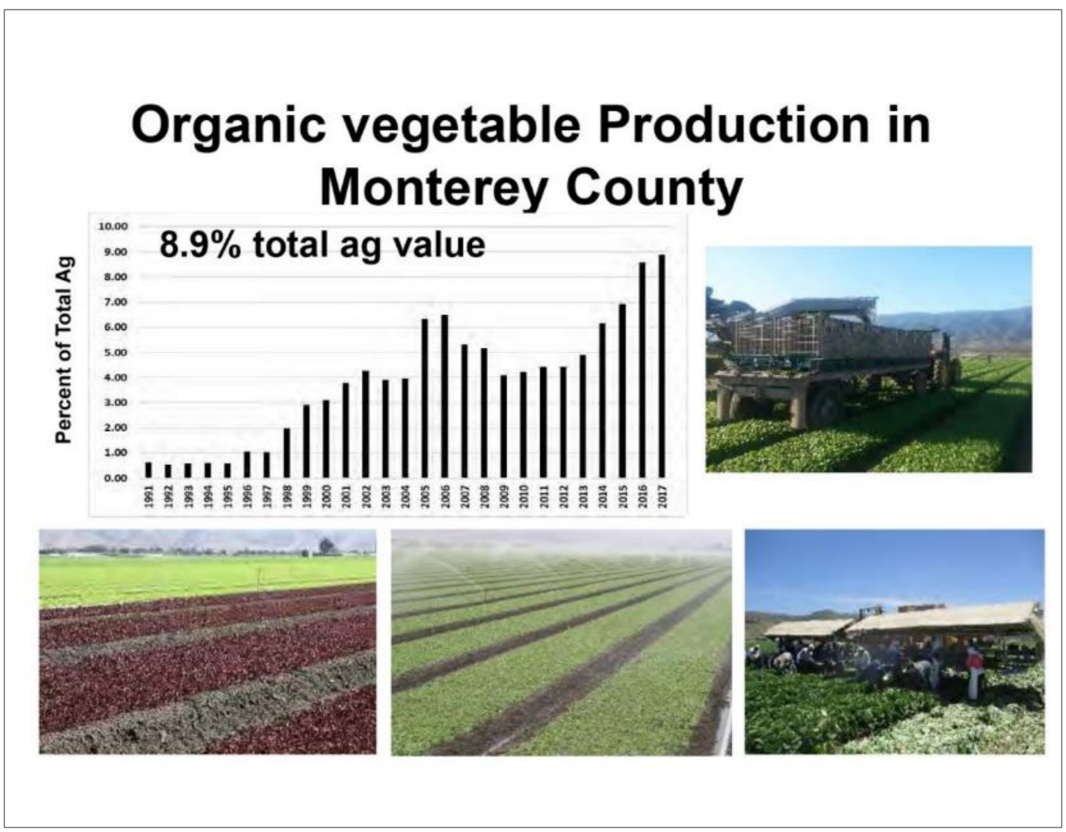
This accounts for there being much more nutrient volume in the soil than is available for plant use at any one time. For example, a soil with 3 percent organic matter contains about 3,000 pounds per acre of nitrogen, but only a small part of that (30-100 pounds) may become available to plants in any one year, depending on decomposition rates.

While active organic matter may decompose over a few decades, the stabilized portion of organic matter is made of larger, more complex compounds that are much more dif□cult for microbes to degrade. Much of the stabilized organic matter in the soil is highly decomposed plant and animal tissues that grew more than a century, and possibly several centuries, ago. This organic matter becomes carbon-rich humus that's resistant to further decay.

"Stabilized organic matter" or humus, acts like a sponge and can absorb six times its weight in water. It's also a reservoir for nutrient storage, sequestering carbon from the atmosphere and other sources.

Healthy soils need both active and stabilized organic matter to function well.

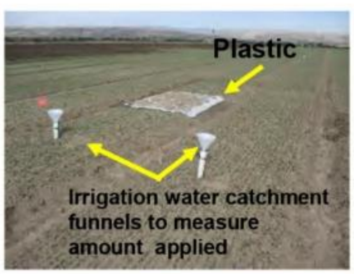

Attachment 10
Organic Nitrogen Mineralization Overview
Richard Smith, University of California Cooperative Extension
Monterey County

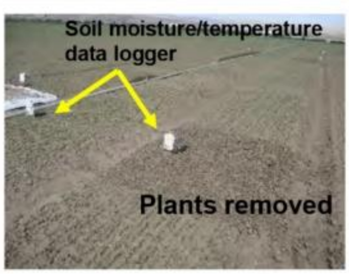

Organic vegetable Production in Monterey County
8.9% total ag value
Percent of Total Ag
10.00
9.00
8.00
7.00
6.00
5.00
4.00
3.00
2.00
1.00
0.00
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017

In-field Soil Organic Matter Mineralization Evaluations

- 20 evaluations were conducted with cooperating growers in commercial vegetable production fields
  - High density:
    - baby lettuce and spinach
  - Full term:
    - romaine and broccoli
- Replicated fertilized and non-fertilized plots were established in each field

In-field Soil Organic Matter Mineralization Evaluations





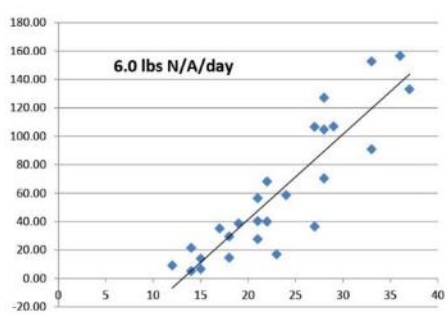
- In each unfertilized plot subplots included:
  1. Plants present
     - Estimate of soil N mineralized, plant removal, leaching
  2. No plants
     - Estimate of soil N mineralized, no plant removal, leaching
  3. No plants, covered with plastic
     - Estimate of soil N mineralized, no plant removal, no leaching

Summary of In-Field Nitrogen Mineralization Evaluations

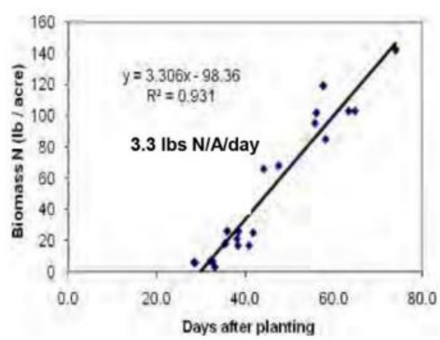
- **Estimates of N mineralization from the soil over the cropping cycle ranged from 0.3 to 3.3 lbs N/A/day; average = 1.6 lbs N/A/day**
- **Laboratory estimates ranged from 0.3 to 1.9 lbs N/A/day; average = 0.5 lbs N/A/day**
- **R^2 = 0.08 between the two estimates**
  - **Core collection issues in 2016; moisture conditions between lab and field varied; incubation temperatures varied; difficulties avoiding crop residue in production fields**

Effect of Nitrogen Fertilization

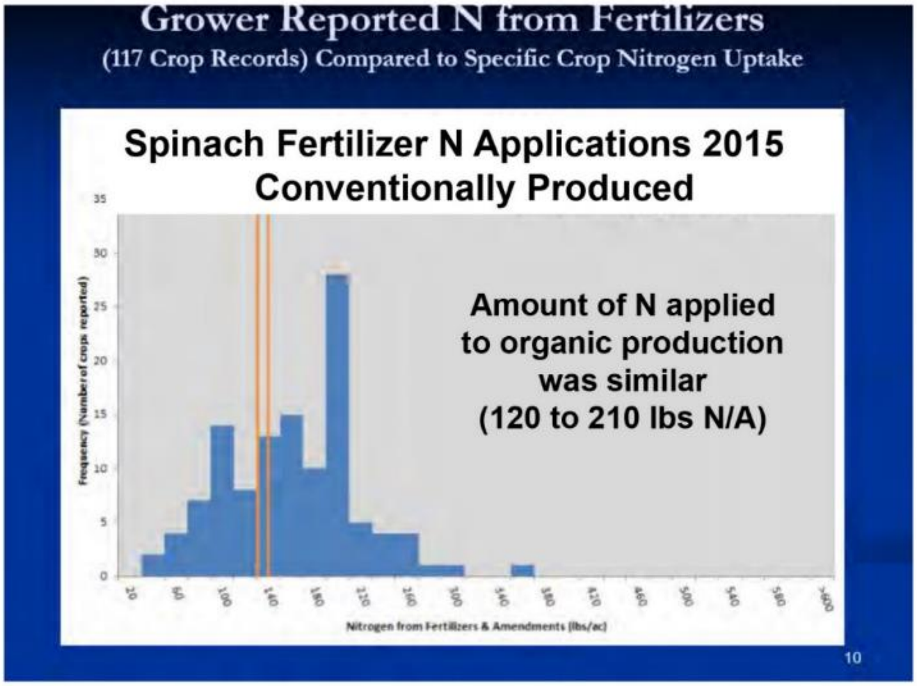
- **The yield of vegetables was improved by fertilization in 17 of the 20 field evaluations**

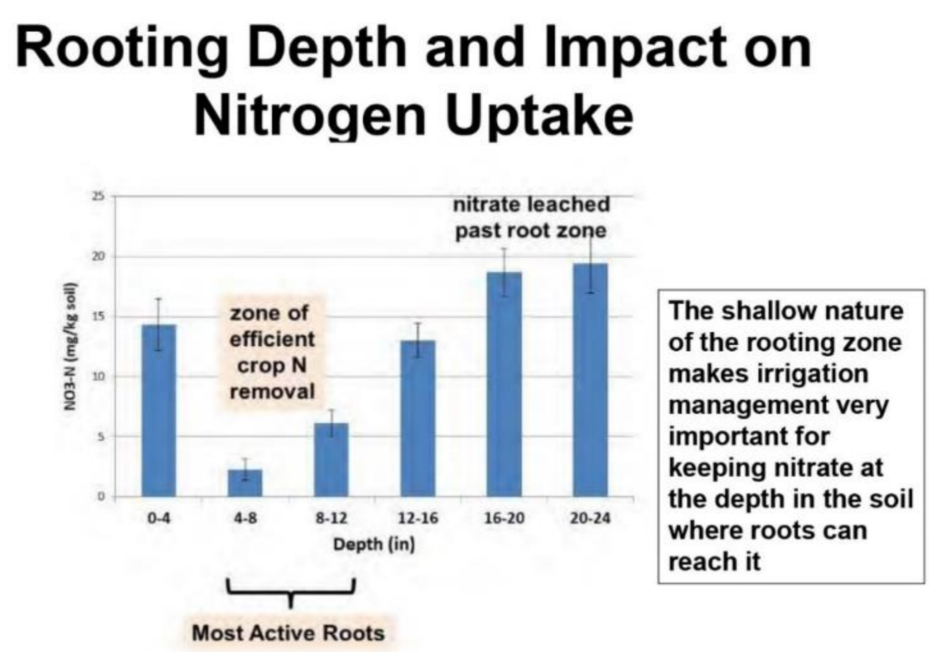


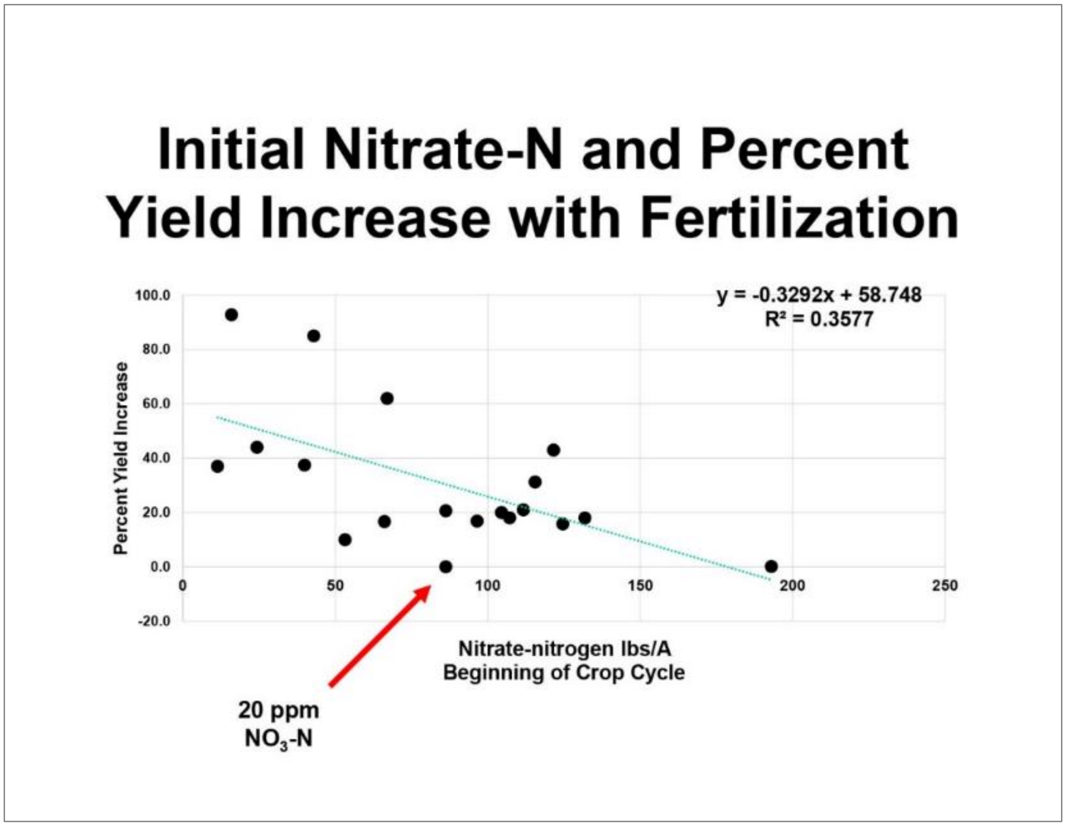
Spinach

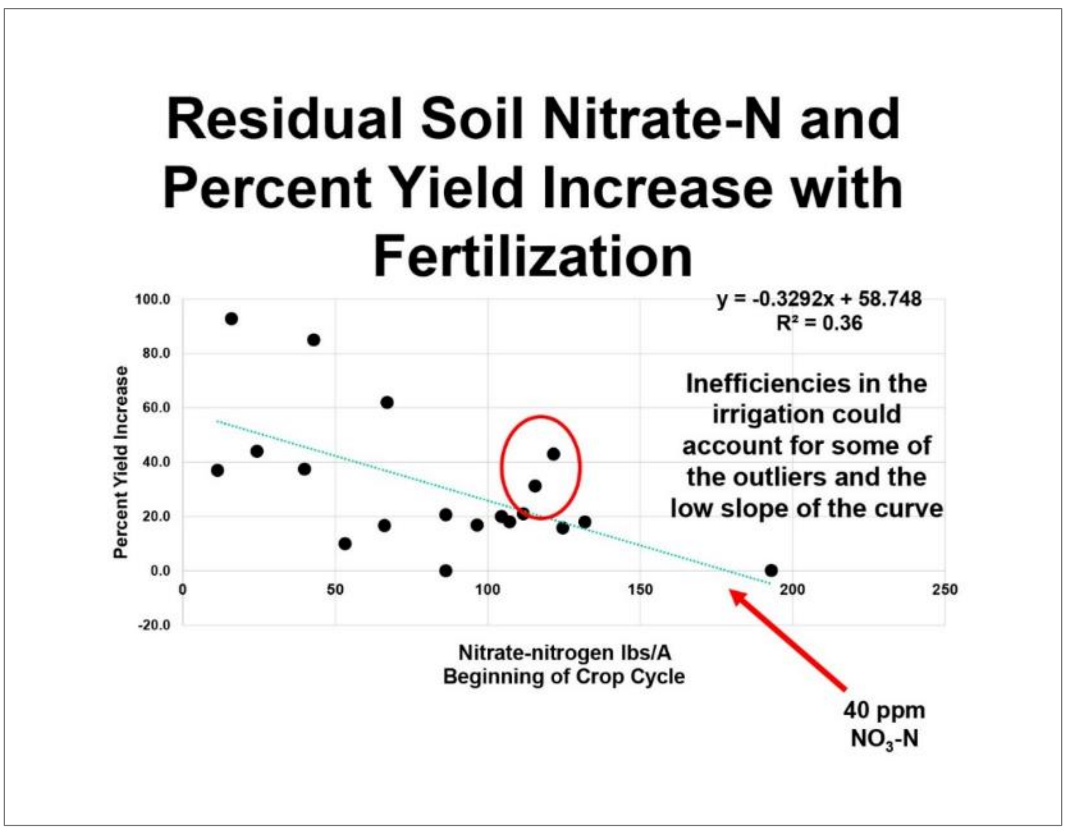


Lettuce


Grower Reported N from Fertilizers
(117 Crop Records) Compared to Specific Crop Nitrogen Uptake
Spinach Fertilizer N Applications 2015
Conventionally Produced
Amount of N applied
to organic production
was similar
(120 to 210 lbs N/A)
Frequency (Number of crops reported)
0
5
10
15
20
25
30
35
20
60
100
140
180
220
260
300
340
380
420
460
500
540
580
>600
Nitrogen from Fertilizers & Amendments (lbs/ac)
10


Rooting Depth and Impact on Nitrogen Uptake
nitrate leached
past root zone
zone of
efficient
crop N
removal
NO3-N (mg/kg soil)
0
5
10
15
20
25
0-4
4-8
8-12
12-16
16-20
20-24
Depth (in)
Most Active Roots
The shallow nature
of the rooting zone
makes irrigation
management very
important for
keeping nitrate at
the depth in the soil
where roots can
reach it


Initial Nitrate-N and Percent Yield Increase with Fertilization
y = -0.3292x + 58.748
R² = 0.3577
Percent Yield Increase
100.0
80.0
60.0
40.0
20.0
0.0
-20.0
0
50
100
150
200
250
Nitrate-nitrogen lbs/A
Beginning of Crop Cycle
20 ppm
NO3-N


Residual Soil Nitrate-N and Percent Yield Increase with Fertilization
y = -0.3292x + 58.748
R² = 0.36
Inefficiencies in the irrigation could account for some of the outliers and the low slope of the curve
Percent Yield Increase
100.0
80.0
60.0
40.0
20.0
0.0
-20.0
0
50
100
150
200
250
Nitrate-nitrogen lbs/A
Beginning of Crop Cycle
40 ppm
NO3-N

Nitrogen Fertility Trial 1

Planting lbs N/A	Topdress lbs N/A	Total lbs N/A	Initial NO_3-N	Fresh wt tons/A
80	80	160	21	6.9
40	80	120	21	6.9
0	0	0	21	6.4

Clay loam soil

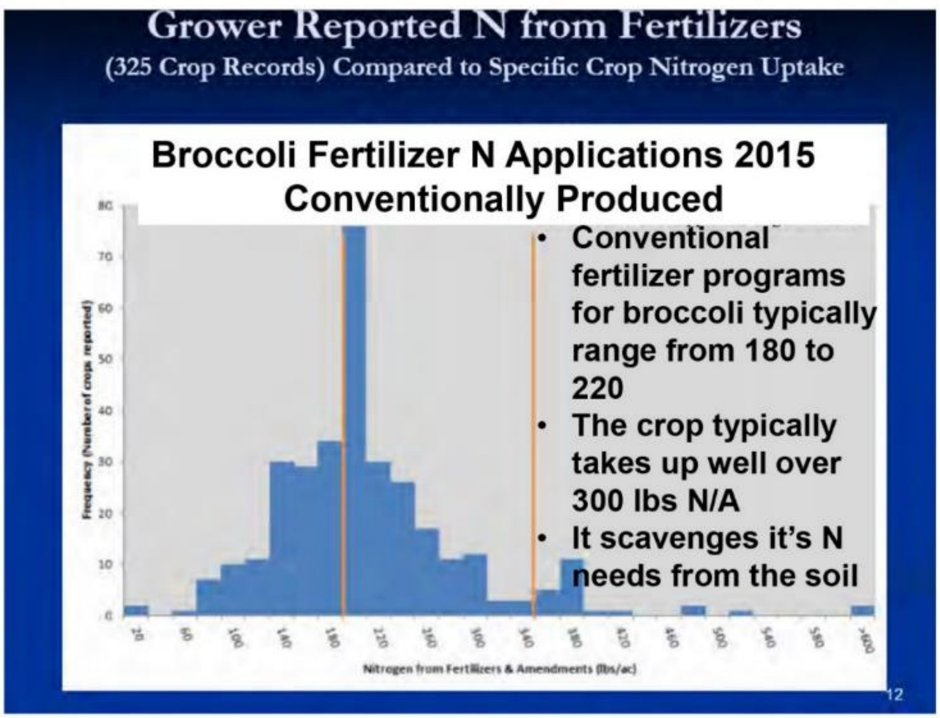
Nitrogen Fertility Trial 2

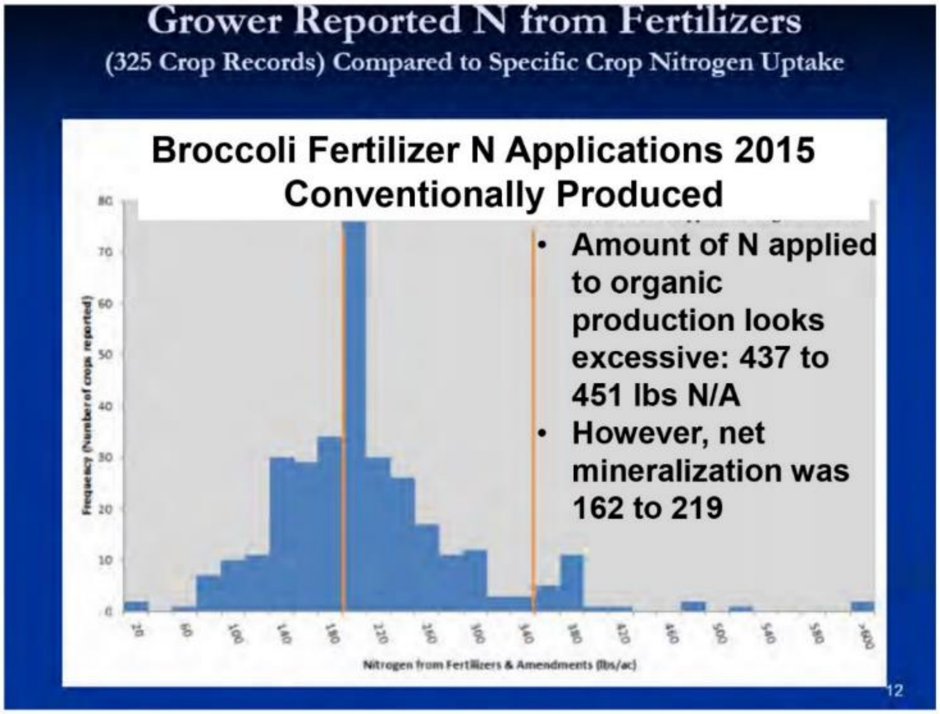
Planting lbs N/A	Topdress lbs N/A	Total lbs N/A	Initial NO_3-N	Fresh wt tons/A
160	0	160	27	7.7
120	0	120	27	6.8
0	120	120	27	5.7

Sandy loam soil

Soil Mineralization Summary

- Nitrate mineralization from soil organic matter alone, generally cannot provide sufficient N for fast-growing leafy greens
- Measurements of residual soil N at the beginning of the crop cycle can give an indication of the need for fertilization of the crop, but shallow rooted crops like spinach present specific challenges
- Leaching losses from the germination water can reduce the reliability of the early season nitrate evaluations





Fertilization of Broccoli

Soil Depth	Initial NO_3-N	No Fert NO_3-N	Fert NO_3-N
1’	30	1	9
2’	6	2	2
3’	5	5	13

Five drop on top applications; loamy sand soil

Soil Mineralization Summary

- Early season nitrate measurements for broccoli are less useful
- Fertilizer programs will vary depending on the amount of residual soil nitrate remaining in the soil given that broccoli typically is scavenging from the 2nd and 3rd foot to provide its N needs
- 2nd crop broccoli fertilizer programs can be more modest than 1st crop broccoli

In-field Fertilizer Mineralization Studies

Polypropylene Pouches with Fertilizer

- Pouches with fertilizer were placed into the soil at the beginning of the crop cycle
- 4-4-2 (blend of chicken manure, bone and meat meals) & 12-0-0 (feather meal)
- Pouches were buried & placed on soil surface to simulate application methods

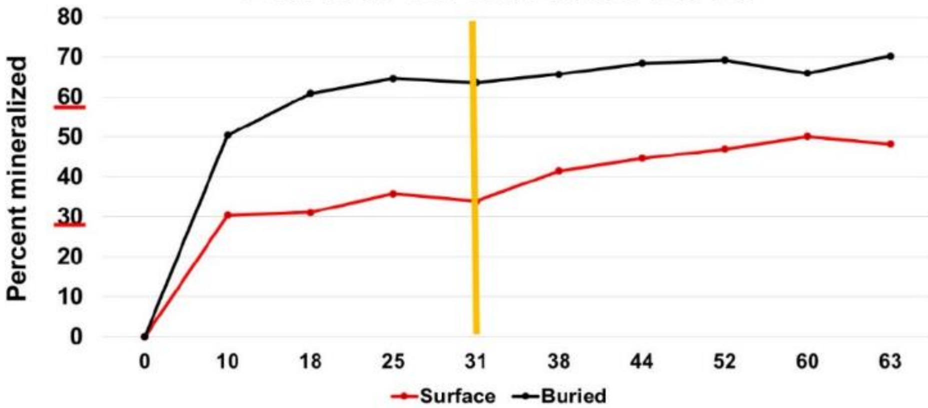
In-field Fertilizer Mineralization Studies

Buried in soil

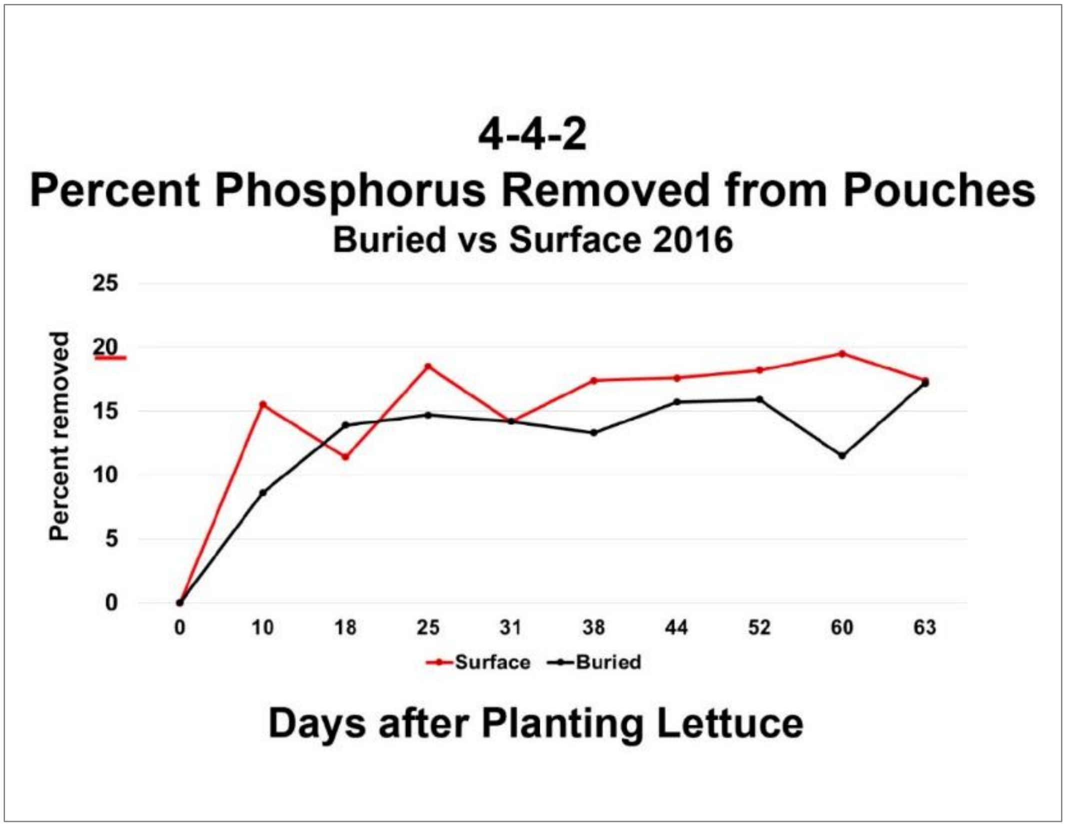
Place on top of soil

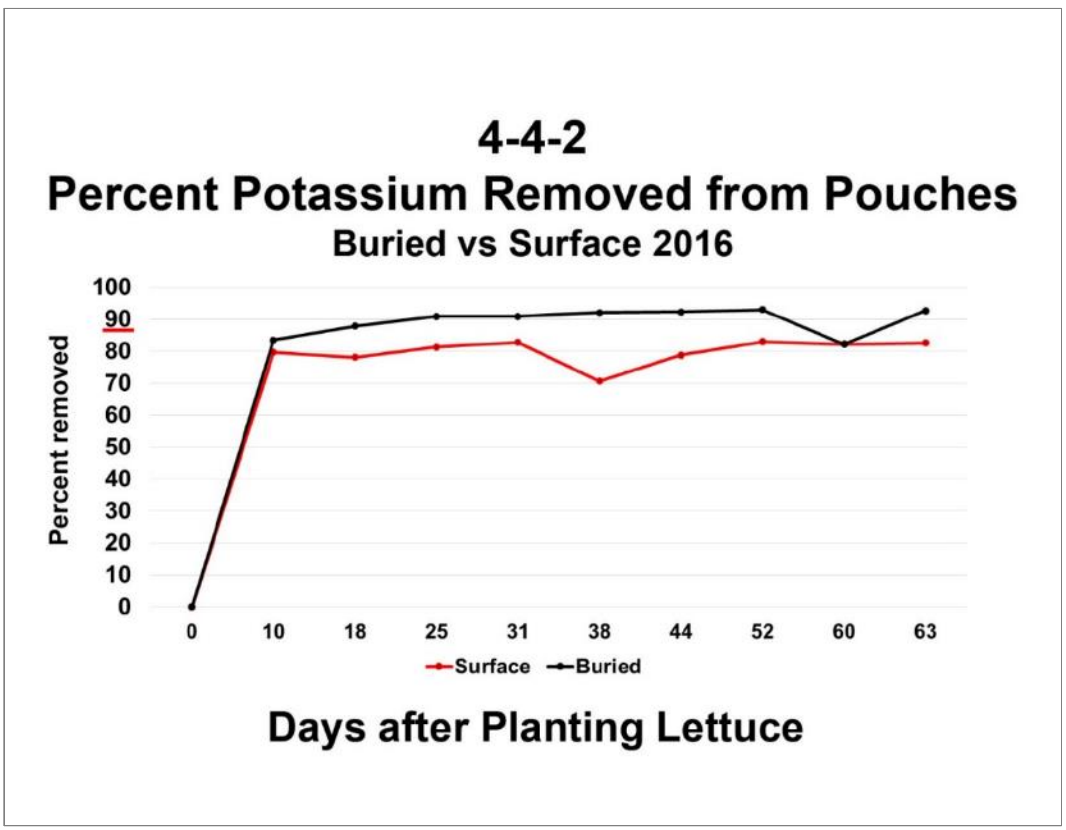
4 pouches collected weekly and analyzed for N, P & K over the crop cycle

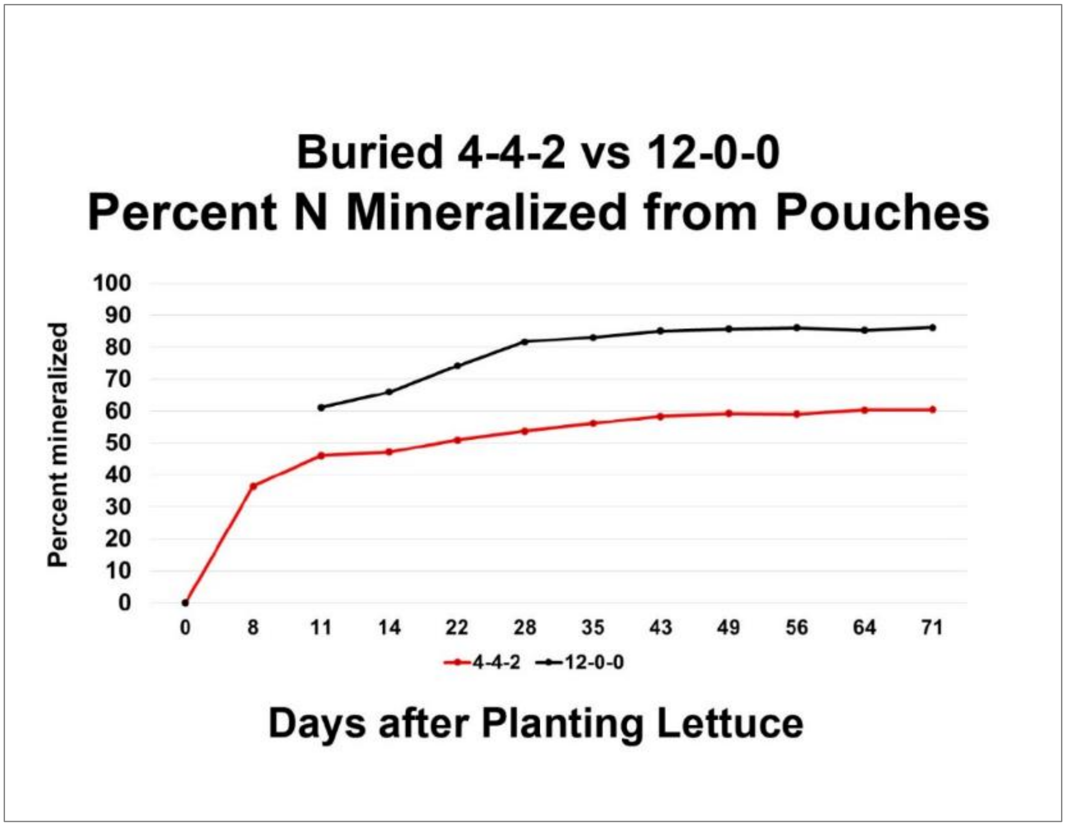
4-4-2
Percent N Mineralized from Pouches
Buried vs Surface 2016

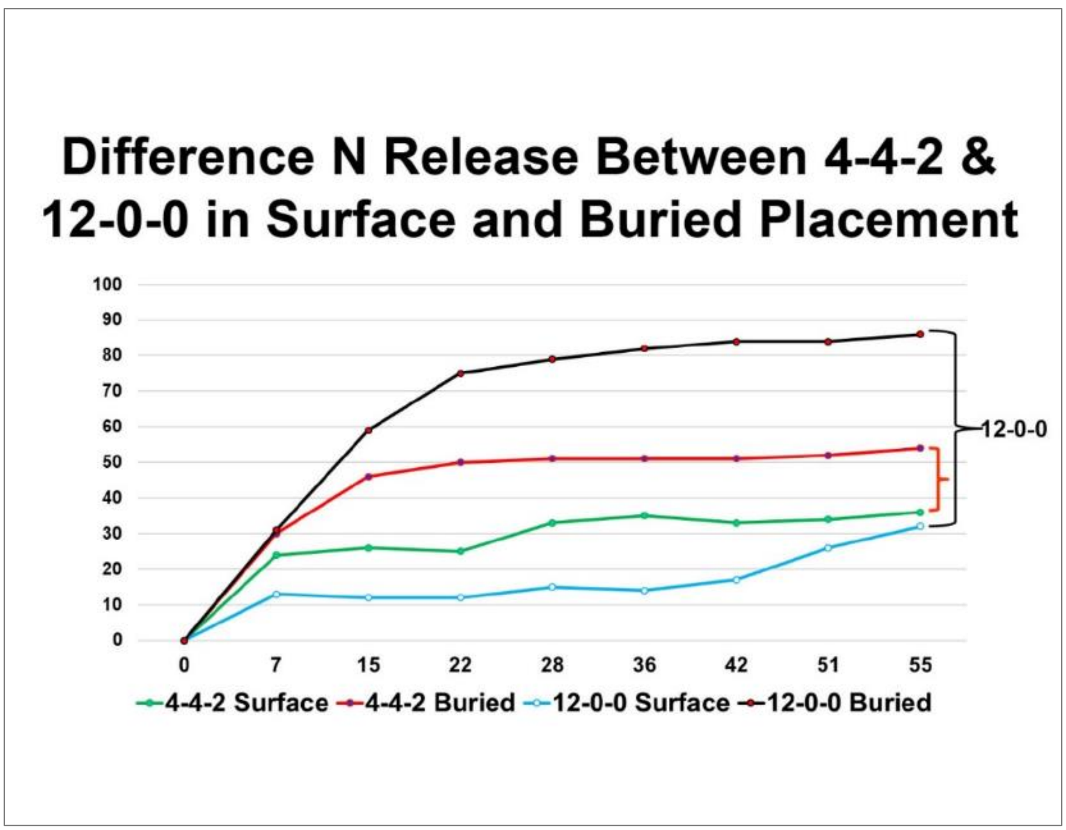


Days after Planting Lettuce


4-4-2
Percent Phosphorus Removed from Pouches
Buried vs Surface 2016
Percent removed
25
20
15
10
5
0
0
10
18
25
31
38
44
52
60
63
Surface
Buried
Days after Planting Lettuce


4-4-2
Percent Potassium Removed from Pouches
Buried vs Surface 2016
Percent removed
100
90
80
70
60
50
40
30
20
10
0
0
10
18
25
31
38
44
52
60
63
Surface
Buried
Days after Planting Lettuce


Buried 4-4-2 vs 12-0-0
Percent N Mineralized from Pouches
Percent mineralized
100
90
80
70
60
50
40
30
20
10
0
0
8
11
14
22
28
35
43
49
56
64
71
4-4-2
12-0-0
Days after Planting Lettuce


Difference N Release Between 4-4-2 &
12-0-0 in Surface and Buried Placement
100
90
80
70
60
50
40
30
20
10
0
0
7
15
22
28
36
42
51
55
12-0-0
4-4-2 Surface
4-4-2 Buried
12-0-0 Surface
12-0-0 Buried

Summary of Pouch Evaluations Buried vs Surface

- Placement of the material affects the speed of mineralization of N and may affect the rate of material needed for optimal growth
- Given the pH's of the soil, the phosphorus in 4-4-2 that comes from bone meal, is not available to the crop and remains in the soil as an insoluble mineral
- Potassium is rapidly released

Laboratory Incubations of Fertilizer Materials

Percent N Mineralized

Material	2 weeks	4 weeks	8 weeks
2.5-2.0-2.5	4.0	5.8	13.6
4-4-2	28.8	30.5	37.5
8-5-1	47.2	43.5	58.5
10-5-2	43.8	49.3	58.8
12-0-0	48.7	56.5	59.3

Lab evaluations generally had lower levels of N mineralization and it may be because they don't have issues with loss of material from the pouches

Fate of Unused Applied N

- Double or triple cropping may be leaving a significant amount of N from the unmineralized fertilizer in the soil
- What is the fate of this N?
- It is recalcitrant and adds to total N in the soil and probably continues to slowly mineralize
- In a survey of 20 pairs of organic and conventional fields we did not detect a build up of total N in organically managed fields

Comparison of 20 Pairs of Conventional and Organic Fields

Soil Constituent	Conventional	Organic
Organic Matter %	2.0	2.1
Total Nitrogen %	0.12	0.12
Total Carbon %	1.01	1.03
Phosphorous (Olsen) ppm	37	42
Phosphorous (Total) ppm	0.10	0.09

Carbon Content of Various Fertilizers

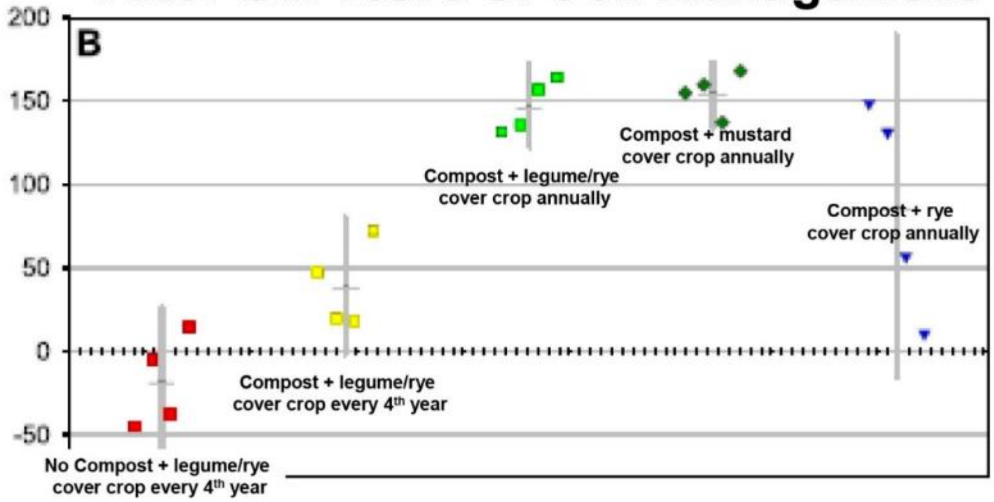
Fertilizer	% Carbon	Source
4-4-2	27.9	Poultry Manure, Meat and Bone Meal
12-0-0	46.1	Feather (+Meat and Bone) Meal
10-5-2	42.0	Meat, Bone and Feather meals & K_2SO_4
8-5-1	36.9	Meat, Bone, and Feather meals & poultry
7.5-5-7.5	37.2	Meat, Bone and Feather meals
2.5-2-2.5	25.2	Poultry manure
14-0-0	42.7	Hydrolyzed soybean

Input of Carbon

Material	Biomass lbs/A	Carbon content percent	Total carbon lbs/A
Compost	10,000[1]	29%	2,146
Cover crop	6,000	44%	2,640
4-4-2 2 baby crops @ 3000 each	5,400[2]	29%	1,566
8-5-1 1 broccoli crop	5,000[3]	41%	2,050

1 – 10,000 lbs/A @ 74% oven dry weight
2 – 6000 lbs/A (2 baby crops @ 3000 lbs/A each) @ 90% oven dry weight;
3 – 5650 lbs/A @ 90% oven dry weight

Changes in Microbial Biomass Carbon After Six Years of Soil Management



From Brennan et al, 2017

Differences Between Organic and Conventionally Managed Soils

- A wide range of soil parameters were analyzed and correlations between organic and conventional soils were conducted
- Differences between the two soil systems that were observed were:
  - Water Extractable Organic Nitrogen
  - Water Extractable Organic Carbon
  - FDA (an enzyme that indicates microbial activity and biomass)

Differences Between Organic and Conventionally Managed Soils

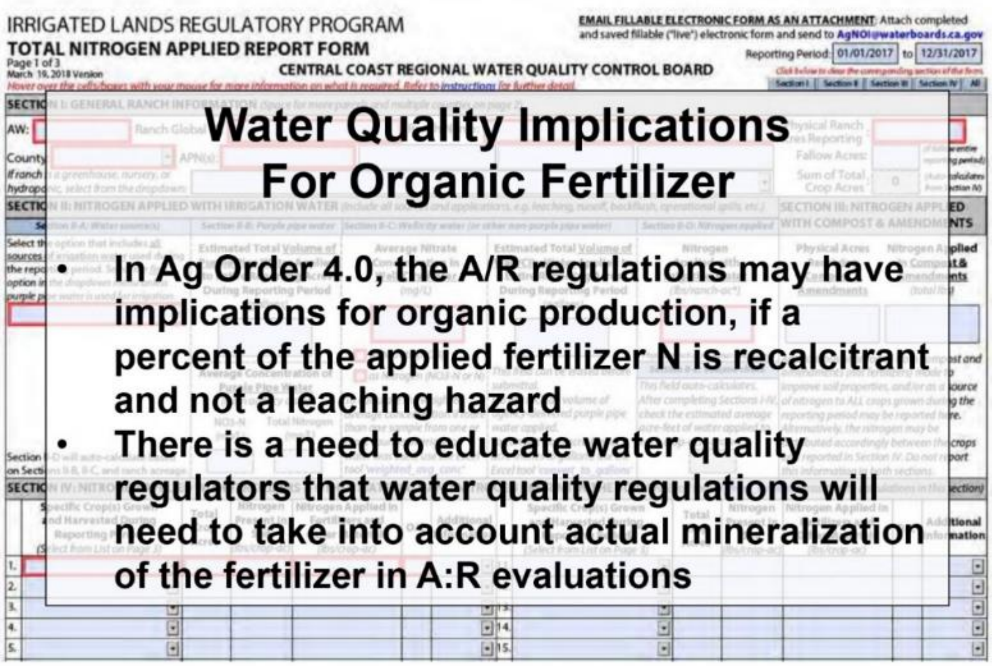
- The bottom line is that the carbon from the organic soils is not radically increasing soil organic matter
- The repeated application of the carbon from these fertilizers has a short-term effect on soil
- Without cover crops or compost, there is little movement towards long-lasting impacts on the soil

Differences Between Organic and Conventionally Managed Soils

- However, the effects that we measured may have tangible effects on the soil:
  - Improving water infiltration and movement in
  - Making the soils more suppressive to some soilborne diseases
  - Others

Organic Fertilizer Programs

- **The amount of N applied to the crops ranged from 1.2 to 5.7 times N uptake**
  - **A:U (<u>crop</u> <u>uptake</u>, not R – removal)**
- **54 – 70% of N in 4-4-2 mineralized over 60 days (2016 and 2017, respectively)**
- **Taking into account N mineralized from organic fertilizer over the crop cycle, the amount applied to crop uptake ranged from 0.4 to 2.8 times N uptake**



Water Quality Implications For Organic Fertilizer

- **In Ag Order 4.0, the A/R regulations may have implications for organic production, if a percent of the applied fertilizer N is recalcitrant and not a leaching hazard**
- **There is a need to educate water quality regulators that water quality regulations will need to take into account actual mineralization of the fertilizer in A:R evaluations**

Management Considerations

- Incorporated applications released faster and a higher percentage of the N that they contained than topical applications
- Higher analysis materials release more N and a greater percent of the N that they contained than lower analysis materials
- High density, baby vegetables only have two opportunities for applying N (preplant at 2 weeks after the first germ water)

Management Considerations

- Preplant tests of nitrate can be useful prior to planting and that is a key time to adjust fertilizer applications
- A nitrate test following the germination water may be a good measurement to avoid the effect of the leaching that occurs with the quantity of water applied to establish the crop

Management Considerations

- Preplant tests of nitrate can be useful prior to planting and that is a key time to adjust fertilizer applications
- A nitrate test following the germination water may be a good measurement to avoid the effect of the leaching that occurs with the quantity of water applied to establish the crop

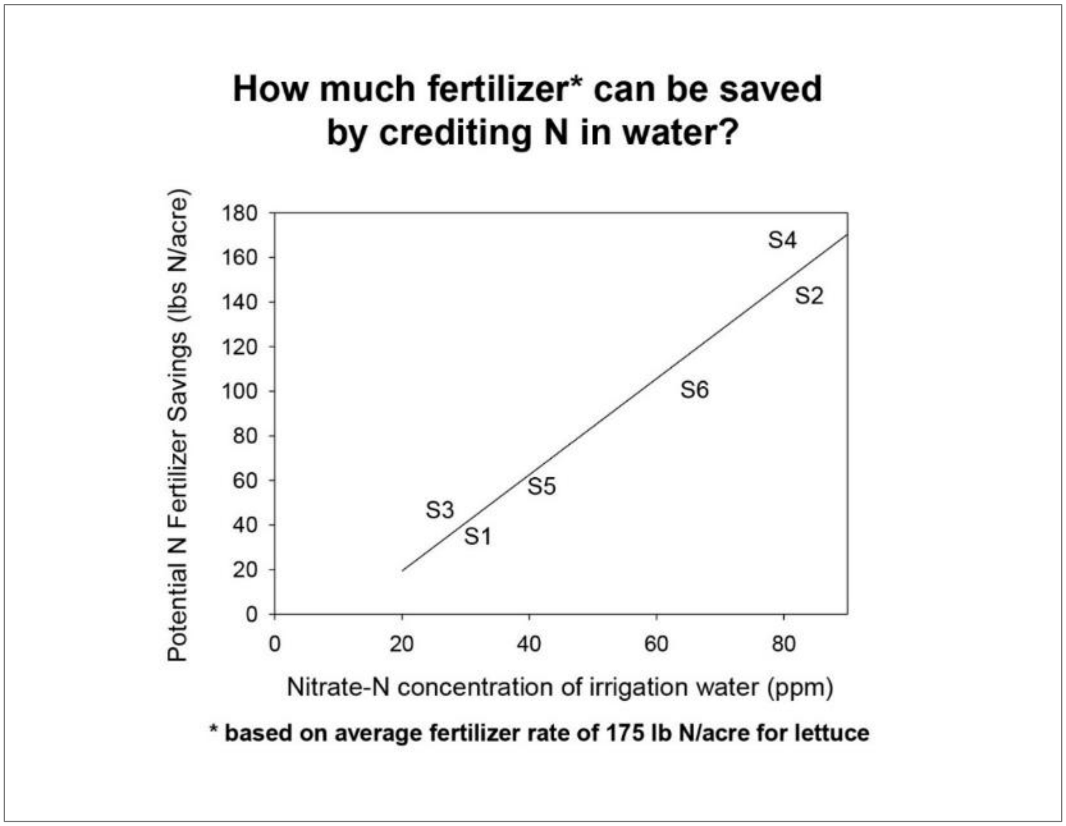
Another Important Factor:
Calculating N applied from irrigation water:

ppm NO_3-N x 0.23 = lbs N/acre inch

Example:

- Water nitrate-N concentration = 40 ppm
- 40 x 0.23 = 9 lbs N/acre inch

- If 1.5 inches of water applied:
- 1.5 x 9 lbs = 14 lbs N


How much fertilizer* can be saved
by crediting N in water?
Potential N Fertilizer Savings (lbs N/acre)
0
20
40
60
80
100
120
140
160
180
Nitrate-N concentration of irrigation water (ppm)
0
20
40
60
80
S1
S2
S3
S4
S5
S6
* based on average fertilizer rate of 175 lb N/acre for lettuce


Acknowledgements
• Cooperating growers
• Crop consultants, Wilbur Ellis, True Organics
• Other UC Researchers: Michael Cahn, Tim Hartz, Daniel Geisseler and Tricia Love
• Karina Mendez, Kacie Wynn, Bibiana Urbina, Jose Delgado and Ignacio Fregoso
• Funding provided by FREP

Attachment 11

Soil Carbon Sequestration Impacts on Global Climate Change and Food Security

R. Lal

The carbon sink capacity of the world's agricultural and degraded soils is 50 to 66% of the historic carbon loss of 42 to 78 gigatons of carbon. The rate of soil organic carbon sequestration with adoption of recommended technologies depends on soil texture and structure, rainfall, temperature, farming system, and soil management. Strategies to increase the soil carbon pool include soil restoration and woodland regeneration, no-till farming, cover crops, nutrient management, manuring and sludge application, improved grazing, water conservation and harvesting, efficient irrigation, agroforestry practices, and growing energy crops on spare lands. An increase of 1 ton of soil carbon pool of degraded cropland soils may increase crop yield by 20 to 40 kilograms per hectare (kg/ha) for wheat, 10 to 20 kg/ha for maize, and 0.5 to 1 kg/ha for cowpeas. As well as enhancing food security, carbon sequestration has the potential to offset fossil-fuel emissions by 0.4 to 1.2 gigatons of carbon per year, or 5 to 15% of the global fossil-fuel emissions.

The global soil carbon (C) pool of 2500 gigatons (Gt) includes about 1550 Gt of soil organic carbon (SOC) and 950 Gt of soil inorganic carbon (SIC). The soil C pool is 3.3 times the size of the atmospheric pool (760 Gt) and 4.5 times the size of the biotic pool (560 Gt, fig. S1). The SOC pool to 1-m depth ranges from 30 tons/ha in arid climates to 800 tons/ha in organic soils in cold regions, and a predominant range of 50 to 150 tons/ha. The SOC pool represents a dynamic equilibrium of gains and losses (Fig. 1). Conversion of natural to agricultural ecosystems causes depletion of the SOC pool by as much as 60% in soils of temperate regions and 75% or more in cultivated soils of the tropics. The depletion is exacerbated when the output of C exceeds the input and when soil degradation is severe. Some soils have lost as much as 20 to 80 tons C/ha, mostly emitted into the atmosphere. Severe depletion of the SOC pool degrades soil quality, reduces biomass productivity, and adversely impacts water quality, and the depletion may be exacerbated by projected global warming.

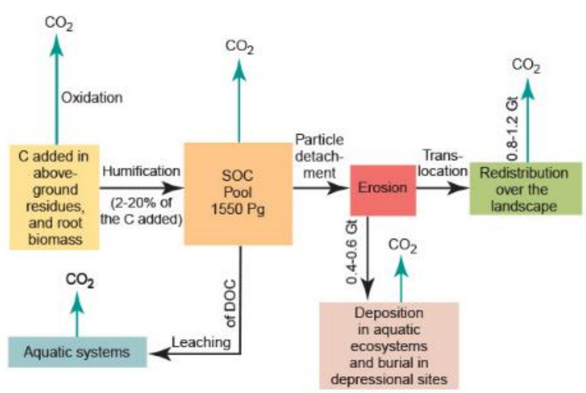


Fig. 1. Processes affecting soil organic carbon (SOC) dynamics. Arrows pointed upward indicate emissions of CO_2 into the atmosphere. There may also be emission of CH_4 under anaerobic conditions, although most well-drained soils are a sink of CH_4. DOC, dissolved organic carbon.

Terrestrial ecosystems contributed to atmospheric CO_2 enrichment during both the preindustrial and industrial eras (Table 1). During the preindustrial era, the total C emission from terrestrial ecosystems was supposedly about twice (320 Gt or 0.04 Gt C/year for 7800 years) that of the industrial era (160 Gt or 0.8 Gt C/year for 200 years) (*1*). Between 1850 and 1998, the emission from fossil-fuel combustion (270 ± 30 Gt) was about twice that from the terrestrial ecosystems (136 ± 55 Gt) (*2*). The latter includes 78 ± 12 Gt from soil, of which about one-third is attributed to soil degradation and accelerated erosion and two-thirds to mineralization (Table 1). The estimates of historic SOC loss range widely, from 44 to 537 Gt, with a common range of 55 to 78 Gt (*3*).

Soil Carbon Sequestration

Carbon sequestration implies transferring atmospheric CO_2 into long-lived pools and storing it securely so it is not immediately reemitted. Thus, soil C sequestration means increasing SOC and SIC stocks through judicious land use and recommended management practices (RMPs). The potential soil C sink capacity of managed ecosystems approximately equals the cumulative historic C loss estimated at 55 to 78 Gt. The attainable soil C sink capacity is only 50 to 66% of the potential capacity. The strategy of soil C sequestration is cost-effective and environmentally friendly (table S1).

The rate of increase in the SOC stock, through land-use change and adoption of RMPs, follows a sigmoid curve, attains the maximum 5 to 20 years after adoption of RMPs, and continues until SOC attains another equilibrium. Observed rates of SOC sequestration in agricultural and restored ecosystems depend on soil texture, profile characteristics, and climate, and range from 0 to 150 kg C/ha per year in dry and warm regions (*4*), and 100 to 1000 kg C/ha per year in humid and cool climates (*5*–*8*) (fig. S2). With continuous use of RMPs, these rates can be sustained for 20 to 50 years or until the soil sink capacity is filled (*8*, *9*). The SOC sequestration is caused by those management systems that add high amounts of biomass to the soil, cause minimal soil disturbance, conserve soil and water, improve soil structure, enhance activity and species diversity of soil fauna, and strengthen mechanisms of elemental cycling (Fig. 2, table S2). Common RMPs that lead to SOC sequestration are mulch

Carbon Management and Sequestration Center, The Ohio State University Columbus, OH 43210, USA. E-mail: lal.1@osu.edu

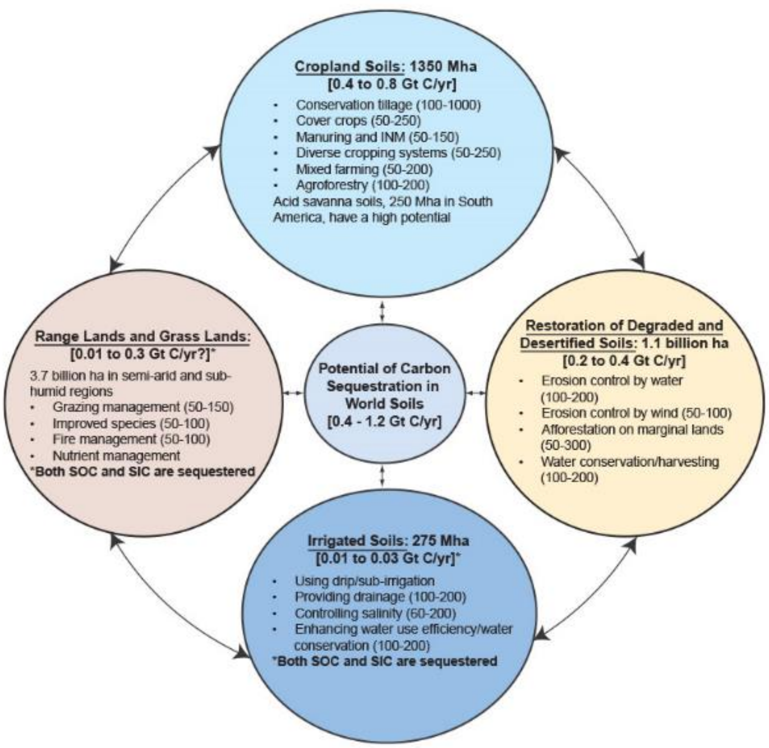


Fig. 2. Ecosystems with a high and attainable soil C sequestration potential are cropland, grazing/range land, degraded/desertified lands, and irrigated soils. Forest soils are included under afforestation of agriculturally marginal and otherwise degraded/desertified soils. Reforestation of previously forested sites have small additional soil C sequestration. The potential of C sequestration of range lands/grassland is not included in the global total because part of it is covered under other ecosystems, and there are large uncertainties. Rates of C sequestration given in parentheses are in kg C/ha per year, are not additive, and are low under on-farm conditions. [Rates are cited from (*2–9*, *15*, *25*, *37–39*) and other references cited in the supporting material.]

farming, conservation tillage, agroforestry and diverse cropping systems, cover crops (Fig. 3), and integrated nutrient management, including the use of manure, compost, biosolids, improved grazing, and forest management. The potential of SOC sequestration also lies in restoration of degraded soils and ecosystems (*10*) whose resilience capacity is intact. The rate of SIC sequestration as secondary carbonates is low (5 to 150 kg C/ha per year) and is accentuated by biogenic processes and leaching of carbonates into the groundwater (*11*, *12*), especially in soils irrigated with water containing low carbonates.

Soil Carbon Sequestration for Mitigating Climate Change

Estimates of the total potential of C sequestration in world soils vary widely from a low of 0.4 to 0.6 Gt C/year (*9*) to a high of 0.6 to 1.2 Gt C/year (*13*). Thus, the potential is finite in capacity and time. Nonetheless, soil C sequestration buys us time until the alternatives to fossil fuel take effect. Some issues related to this strategy are as follows:

1) Agricultural chemicals. Most RMPs involve C-based input. It takes 0.86 kg C/kg N, 0.17 kg C/kg P_2O_5, 0.12 kg C/kg K_2O, 0.36 kg C/kg lime, 4.7 kg C/kg of herbicides, 5.2 kg C/kg of fungicides, 4.9 kg C/kg of insecticides (*14*), and 150 kg C/ha for pumping groundwater for irrigation (*15*). Tillage operations emit 15 kg C/ha for moldboard plowing, 8 kg C/ha for chisel plowing and heavy tandem disking, 6 kg C/ha for light tandem disking, 11 kg C/ha for subsoiling, 4 kg C/ha for cultivation, and 2 kg/ha for rotary hoeing (*16*). Therefore, conversion from conventional till to no-till farming reduces emission by 30 to 35 kg C/ha per season (*16*). Similarly, a judicious use of C-based inputs is essential to enhancing use efficiency and minimizing losses. However, inputs are needed not for soil C sequestration per se, but for increasing food production and ensuring sustainable use of soil and water resources.

2) Nutrients required. Carbon is only one of the elemental constituents of humus. It is estimated that sequestration of 1 Gt of C in world soils would require 80 million tons (Mt) of N, 20 Mt of P, and 15 Mt of K. In comparison, the global fertilizer use in 2000 was 136 Mt (*17*). However, there are several sources of nutrients for C sequestration, including biological nitrogen fixation, recycling from subsoil, aerial deposition, use of biosolids, and crop residues. One ton of cereal residue contains 12 to 20 kg N, 1 to 4 kg P, 7 to 30 kg K, 4 to 8 kg Ca, and 2 to 4 kg Mg. Annually, 3 Gt of residues of grain crops are produced globally (table S3), which if recycled rather than removed for fuel and other uses, would improve soil quality and sequester C. Crop residue is also a potential source of energy by direct combustion, or for production of ethanol or H_2. It can be used either for biofuel production or to sequester C and improve soil quality. The economics of these two competing uses need to be assessed.

3) Soil erosion and deposition. The SOC is preferentially removed by wind- and water-borne sediments through erosional processes. Some of the SOC-enriched sediments are redistributed over the landscape, others are deposited in depressional sites, and some are carried into the aquatic ecosystems (Fig. 1). Although a part of the C translocated by erosion may be buried and redistributed (*18*), the rest is emitted into the atmosphere either as CO_2 by mineralization or as CH_4 by methanogenesis. Erosion-induced deposition and burial may be 0.4 to 0.6 Gt C/year compared with perhaps 0.8 to 1.2 Gt C/year emitted into the atmosphere (Fig. 1) (*19*). Quantification

of emission versus burial of C is a high priority. Yet, an effective soil erosion control is essential to sustainable use of agricultural soils and improving environment quality.

4) Extractive farming practices. The annual depletion rate of nutrients for sub-Saharan Africa (SSA) caused by low-input/subsistence farming is estimated to be 40 kg of NPK/ha of cultivated land since the mid-1960s (*20*). Mining SOC from soil for nutrients through organic-matter decomposition has an effect on the atmosphere similar to that of fossil-fuel combustion. Therefore, RMPs must enhance rather than deplete SOC pool and soil fertility, increase rather than maintain or decrease crop yield per unit use of fertilizer and other inputs, and improve rather than degrade soil quality.

5) Societal value and hidden benefits. Commodification of soil C is important for trading C credits. Carbon trading markets have existed since 2002, especially in European Union countries (*21*). The low current price ($1/ton CO_2) of SOC may increase with emission cap and regulation. For the concept of SOC credits trading to become routine as a part of the solution to mitigate climate change, the ability to measure short-term (3- to 5-year) changes in SOC pool exists (*22*), but the price of soil C must be based on both on-site and off-site societal benefits (table S4). Undervaluing soil C can lead to a "tragedy of the commons."

6) Hydrologic and carbon cycles. Because renewable freshwater is scarce, a projected increase in cereal production by 56% between 1997 and 2050 (*23*) must occur on the same or smaller land area and with the same or less water. Thus, linking the hydrologic and C cycles through conservation of water resources is crucial to improving agronomic yields and to soil C sequestration in dryland. The low SOC stock in rainfed agriculture can be enhanced through water conservation, water harvesting, and water-efficient farming systems. Enhancing SOC stock in dryland ecosystems through no-till farming is important to drought management: a truly win-win option (*24*).

7) Soil C sequestration and global warming. Global warming is a "century-scale" problem and a "global commons" issue. Soil C sequestration is a related but separate issue with its own merits of increasing productivity, improving water quality, and restoring degraded soils and ecosystems, irrespective of the global-warming debate. Offsetting fossil-fuel emissions by achievable SOC potential provides multiple biophysical and societal benefits (table S3). Furthermore, soil C sequestration is a bridge across three global issues—climate change, desertification, and biodiversity—and a natural link among three UN conventions.

8) Other greenhouse gases. Enhancing SOC stock increases the soil's capacity to oxidize CH_4, especially under no-till farming (*25*), but may also exacerbate emission of N_2O (*26*). Fluxes of CH_4 and N_2O may change the CO_2-mitigation potential of soil management practices and must be considered along with SOC sequestration.

Table 1. Estimates of pre- and postindustrial losses of carbon from soil and emission from fossil-fuel combustion. Data were compiled from diverse sources (*1–3*). Ruddiman (*1*) estimated the emission from land-use conversion during the postindustrial era at 0.8 Gt C/year for 200 years at 160 Gt C.

Source	Historic carbon emission (Gt)	
Preindustrial era		
Fossil-fuel combustion		0
Land-use conversion at 0.04 Gt C/year for 7800 years		320
Postindustrial era		
Fossil-fuel combustion (since 1850)		270 ± 30
Land-use conversion		136 ± 5
Soil cultivation	78 ±12	
Erosion	26 ± 9	
Mineralization	52 ± 8	

9) Soils of the tropics. Because of its severe depletion and degradation, the C sink capacity of soils of the tropics may be high, but the rate of sequestration can be low. The need for enhancing soil quality is also more urgent in soils of the tropics than in soils of high latitudes because of low crop yields. Yet, the challenge is greater because of weak institutions, limited infrastructure, and predominantly resource-poor agriculture systems. Soil-restorative farm policies must be implemented to mitigate soil-degradative trends.

10) Permanence. Soil C sequestration is a natural, cost-effective, and environment-friendly process. Once sequestered, C remains in the soil as long as restorative land use, no-till farming, and other RMPs are followed. Soil sink capacity and permanence are related to clay content and mineralogy, structural stability, landscape position, moisture and temperature regimes, and ability to form and retain stable microaggregates.

Soil Carbon Sequestration and Global Food Security

Global hotspots of soil degradation with a high priority for soil restoration and C sequestration include SSA, central and south Asia, China, the Andean region, the Caribbean, and the acid savannas of South America. Complete residue removal for fodder and fuel is a norm in south Asia and Africa. Thus, depletion of SOC stock from the root zone has adversely affected the soil productivity and environmental quality of these regions. Simply put, poor farmers have passed on their suffering to the land through extractive practices. They cultivate

Fig. 3. Important recommended management practices are no-till farming, cover crops, manuring and agroforestry. **(A)** Long-term no-till plots were established at the International Institute of Tropical Agriculture, Ibadan, in 1971 and continued through 1987. The adoption of no-till by small landholders in Africa and Asia has been constrained by removal of crop residue mulch for fodder and fuel, nonavailability of a proper seed drill that can cut through the residue, and prohibitively expensive herbicides. **(B)** Agroforestry, sowing wheat under the canopy of poplar, is widely practiced in Punjab, India. Other combinations of trees and crops and forages may be beneficial to sustainable use of soil-water resources and C sequestration in site-specific situations.

marginal soils with marginal inputs, produce marginal yields, and perpetuate marginal living and poverty. As a source of nutrients for growing crops, the SOC pool is a mean of production in subsistence farming systems of SSA, which accounts for only 2.5% of the fertilizer consumption and 2% of the world's irrigated land area, both essential to SOC sequestration. Benefits of RMPs cannot be realized in severely degraded soils depleted of their SOC stock—soil's life support system. An optimum level of SOC stock is needed to hold water and nutrients, decrease risks of erosion and degradation, improve soil structure and tilth, and provide energy to soil microorganisms. The SOC is a biomembrane that filters pollutants, reduces sediment load in rivers, decreases hypoxia in coastal ecosystems, degrades contaminants, and is a major sink for atmospheric CO_2 and CH_4. Fertilizer application is an important strategy of increasing crop yield in SSA (*27*), but its effectiveness is enhanced when used in conjunction with crop residue mulch (*28*) or trees (*20*). Increase in SOC stock increases crop yield even in high-input commercial agriculture (*29*), but especially in soils where it has been depleted (*30*). An increase of 1 ton of SOC increased wheat grain yield by 27 kg/ha in North Dakota, United States (*29*), and by 40 kg/ha in semi-arid pampas of Argentina (*31*), 6 kg/ha of wheat and 3 kg/ha of maize in alluvial soils of northern India (*32*), 17 kg/ha of maize in Thailand (*33*), and 10 kg/ha of maize and 1 kg/ha of cowpea in western Nigeria (*34*). High SOC stock is also needed to maintain consistent yields through improvements in water and nutrient holding capacity, soil structure, and biotic activity. The critical limit of SOC concentration for most soils of the tropics is 1.1% (*35*). Increasing SOC concentration from a low of 0.1 to 0.2% to a critical level of 1.1% is a major challenge for tropical ecosystems. Yet, a drastic reduction in the SOC pool in SSA and elsewhere must be reversed in order to advance food security. An 18-year experiment in Kenya showed that the yield of maize and beans was 1.4 tons/ha per year without external input and 6.0 tons/ha per year when stover was retained and fertilizer and manure were applied. The corresponding SOC stocks to 15-cm depth were 23.6 tons/ha and 28.7 tons/ha, respectively (*36*). This is the type of quantum jump in crop yields needed at the continental scale to ensure food security in SSA. The vicious cycle of declining productivity–depleting SOC stock–lower yields will have to be broken (Fig. 4) by improving soil quality through SOC sequestration in order to free much of humanity from perpetual poverty, malnutrition, hunger, and substandard living.

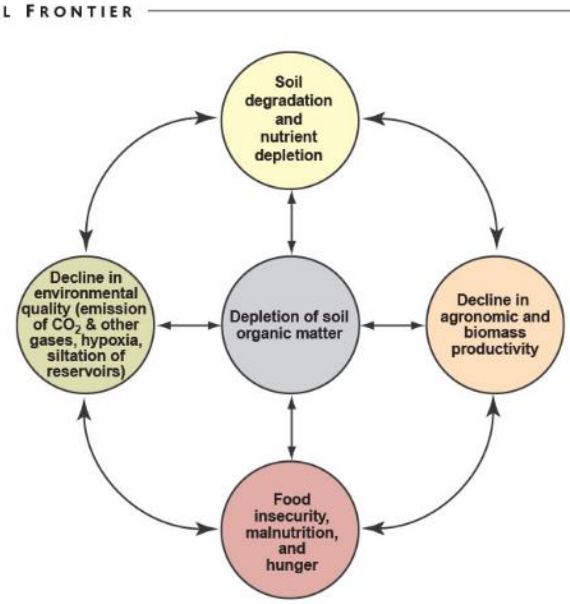


Fig. 4. Cereal yields in Africa have been stagnant since early 1970s and stand at a meager 1 ton/ha. The vicious cycle of deletion in soil organic matter–decline in crop yield–food insecurity–soil and environmental degradation can be broken by improving soil fertility through enhancement of the soil organic matter pool, which requires use of sustainable agricultural technologies for water and nutrients management, including no-till farming, composts and mulching, leguminous cover crops, water harvesting, agroforestry, and integrated farming systems, along with judicious use of chemicals. This strategy can break the tyranny of hunger.

Conclusions

Soil C sequestration is a strategy to achieve food security through improvement in soil quality. It is a by-product of the inevitable necessity of adopting RMPs for enhancing crop yields on a global scale. While reducing the rate of enrichment of atmospheric concentration of CO_2, soil C sequestration improves and sustains biomass/agronomic productivity. It has the potential to offset fossil-fuel emissions by 0.4 to 1.2 Gt C/year, or 5 to 15% of the global emissions.

Soil organic carbon is an extremely valuable natural resource. Irrespective of the climate debate, the SOC stock must be restored, enhanced, and improved. A C-management policy that includes regulation-based trading soil C must be developed. Likewise, a widespread adoption of RMPs by resource-poor farmers of the tropics is urgently warranted. The soil C sequestration potential of this win-win strategy is finite and realizable over a short period of 20 to 50 years. Yet, the close link between soil C sequestration and world food security on the one hand and climate change on the other can neither be overemphasized nor ignored.

References and Notes

1. W. Ruddiman, *Clim. Change* **61**, 261 (2003).
2. Intergovernmental Panel on Climate Change, *Land Use, Land Use Change and Forestry* (Cambridge Univ. Press, Cambridge, 2000), pp. 181–281.
3. R. Lal, *Adv. Agron.* **71**, 145 (2001).
4. R. D. Armstrong *et al.*, *Aust. J. Exp. Agric.* **43**, 141 (2003).
5. P. R. Grace *et al.*, *Aust. J. Exp. Agric.* **35**, 857 (1995).
6. J. Sa *et al.*, *Soil Sci. Soc. Am. J.* **65**, 1486 (2001).
7. C. A. Campbell *et al.*, *Can. J. Soil Sci.* **80**, 193 (2000).
8. T. O. West, W. M. Post, *Soil Sci. Soc. Am. J.* **66**, 1930 (2002).
9. D. R. Sauerbeck, *Nutr. Cycling Agroecosyst.* **60**, 253 (2001).
10. W. L. Silver *et al.*, *Restor. Ecol.* **8**, 394 (2000).
11. L. C. Nordt, L. P. Wilding, L. R. Drees, in *Global Climate Change and Pedogenic Carbonates*, R. Lal, J. M. Kimble, H. Eswaran, B. A. Stewart, Eds. (CRC/Lewis, Boca Raton, FL, 2001), pp. 43–63.
12. R. Levy, in *Chemistry of Irrigated Soils*, R. Levy, Ed. (Van Nostrand-Reinhold, New York, 1984), pp. 182–229.
13. R. Lal, *Crit. Rev. Plant Sci.* **22**, 151 (2003).
14. T. O. West, G. Marland, *Environ. Pollut.* **116**, 439 (2002).
15. R. F. Follett, *Soil Till. Res.* **61**, 77 (2001).
16. R. Lal, *Environ. Int.*, in press.
17. International Fertilizer Development Center (IFDC), *World Fertilizer Consumption* (IFDC, Muscle Shoals, AL, 2000).
18. S. V. Smith, W. H. Renwick, R. W. Buddemeier, C. J. Crossland, *Global Biogeochem. Cycles* **15**, 697 (2001).
19. R. Lal, *Environ. Int.* **29**, 437 (2003).
20. P. A. Sanchez, *Science* **295**, 2019 (2002).
21. E. Johnson, R. Heinen, *Environ. Int.* **30**, 279 (2004).
22. R. Lal *et al.*, *Assessment Methods for Soil Carbon* (CRC, Boca Raton, FL, 2001).
23. M. W. Rosegrant, S. A. Cline, *Science* **302**, 1917 (2003).
24. R. Lal *et al.*, *Science* **304**, 393 (2004).
25. J. Six *et al.*, *Agronomie* **22**, 755 (2002).
26. P. Smith *et al.*, *Nutr. Cycl. Agroecosyst.* **60**, 237 (2001).
27. C. Pieri, *Agron. Trop* **41**, 1 (1986).
28. C. F. Yamoah *et al.*, *Field Crops Res.* **75**, 53 (2002).

29. A. Bauer, A. L. Black, *Soil Sci. Soc. Am. J.* **58**, 185 (1994).
30. A. E. Johnston, *Soil Use Manage.* **2**, 97 (1986).
31. M. Diaz-Zorita *et al.*, *Soil Tillage Res.* **65**, 1 (2002).
32. M. Kanchikerimath, D. Singh, *Agric. Ecosyst. Environ.* **86**, 155 (2001).
33. S. Petchawee, W. Chaitep, in *Organic Matter Management in Upland Systems in Thailand* (Australian Center for International Agricultural Research, Canberra, Australia, 1995), pp. 21–26.
34. R. Lal, *Geoderma* **25**, 215 (1981).
35. J. A. Aune, R. Lal, *Trop. Agric.* **74**, 96 (1997).
36. J. J. Kapkiyai *et al.*, *Soil Biol. Biochem.* **31**, 1773 (1999).
37. R. Lal *et al.*, *Soil Sci.* **168**, 827 (2003).
38. R. J. Manlay *et al.*, *Agric. Ecosyst. Environ.* **88**, 215 (2002).
39. D. G. Wright *et al.*, *Commun. Soil Sci. Plant Anal.* **32**, 1803 (2001).
40. This work was sponsored by the Carbon Management and Sequestration Center, School of Natural Resources, Ohio Agricultural Research and Development Center, and the Climate Change CIRIT, The Ohio State University, Columbus, OH.

Supporting Online Material
www.sciencemag.org/cgi/content/full/304/5677/1623/DC1
SOM Text
Figs. S1 to S3
Tables S1 to S4
References

VIEWPOINT

Breaking the Sod: Humankind, History, and Soil

J. R. McNeill[1]* and Verena Winiwarter[2,3]

For most of history, few things have mattered more to human communities than their relations with soil, because soil provided most of their food and nutrients. Accordingly, some of the earliest written documents were agricultural manuals intended to organize, preserve, and impart soil knowledge. Indeed, ancient civilizations often worshipped the soil as the foundry of life itself. For the past century or two, nothing has mattered more for soils than their relations with human communities, because human action inadvertently ratcheted up rates of soil erosion and, both intentionally and unintentionally, rerouted nutrient flows.

Our distant ancestors found their food by hunting and foraging. They depended indirectly on soils to support plant growth, but they did not much alter soils by their actions, except where they routinely burned vegetation. With the transitions to agriculture (which probably happened independently at least seven times, beginning about 10,000 years ago), human dependence upon, and impact upon, soils became more direct and more obvious. Neolithic farmers, in southwest Asia and elsewhere, depleted soils of their nutrients by cultivating fields repeatedly, but they simultaneously enriched their soils once they learned to keep cattle, sheep, and goats, pasture them on nonarable land, and collect them (or merely their dung) upon croplands. They also worshipped deities that they connected not only to fertility in livestock and women, but also to soil productivity.

When a population lived amid the fields that sustained them, the net transfer of nutrients into or out of the fields remained minor, as after shorter or longer stays in human alimentary canals and tissues, nutrients returned to the soils whence they had come. Urban life changed that, systematically drawing nutrients from fields to cities, from whence wastes left via streams or rivers, en route to the sea. So civilization, with its systemic links between cities and hinterlands, over the past 5000 years has posed an ongoing challenge for farmers trying to maintain soil fertility.

Soil Erosion

In most settings, agriculture promoted soil erosion, although to highly varying degrees. On a global scale, soil erosion occurred in three main waves. The first arose as a consequence of the expansion of early river-basin civilizations, mainly in the second millennium B.C.E. Farmers left the valleys and alluvial soils of the Yellow River, Indus, Tigris-Euphrates, and lesser rivers (or from the Maya lowlands) and ascended forested slopes, where they exposed virgin soils to seasonal rains. The loess plateau of north China, for example, began to erode more quickly during this period, earning the Yellow River its name (*1*). Over the next 3000 years, farmers in Eurasia, Africa, and the Americas gradually converted a modest proportion of the world's forests into farmland or pasture and thereby increased rates of soil erosion, but the fertile soils of the world's grasslands were little affected.

That changed in the 16th to 19th centuries when, in a second great wave of soil erosion, stronger and sharper plowshares helped break the sod of the Eurasian steppe, the North American prairies, and the South American pampas. The exodus of Europeans to the Americas, Australia, New Zealand, Siberia, South Africa, Algeria, and elsewhere brought new lands un-

[1]Edmund A. Walsh School of Foreign Service, Intercultural Center 600, Box 571035, Georgetown University, 3700 O Street, N.W., Washington, DC 20057–1035, USA. [2]Institute for Soil Research, University of Natural Resources and Applied Life Sciences, A-1-80, Vienna, Austria. [3]Institute for Interdisciplinary Studies, A-1070, Vienna, Austria.

*To whom correspondence should be addressed. E-mail: mcneillj@georgetown.edu

Fig. 1. A 16th-century Italian fresco of a cultivated field.

Attachment 12

Food and Agriculture Organization of the United Nations

A woman crossing one of several streams which feed an irrigation canal used for climate-smart agriculture in Tanzania. ©FAO/Daniel Hayduk

Soils help to combat and adapt to climate change
by playing a key role in the carbon cycle

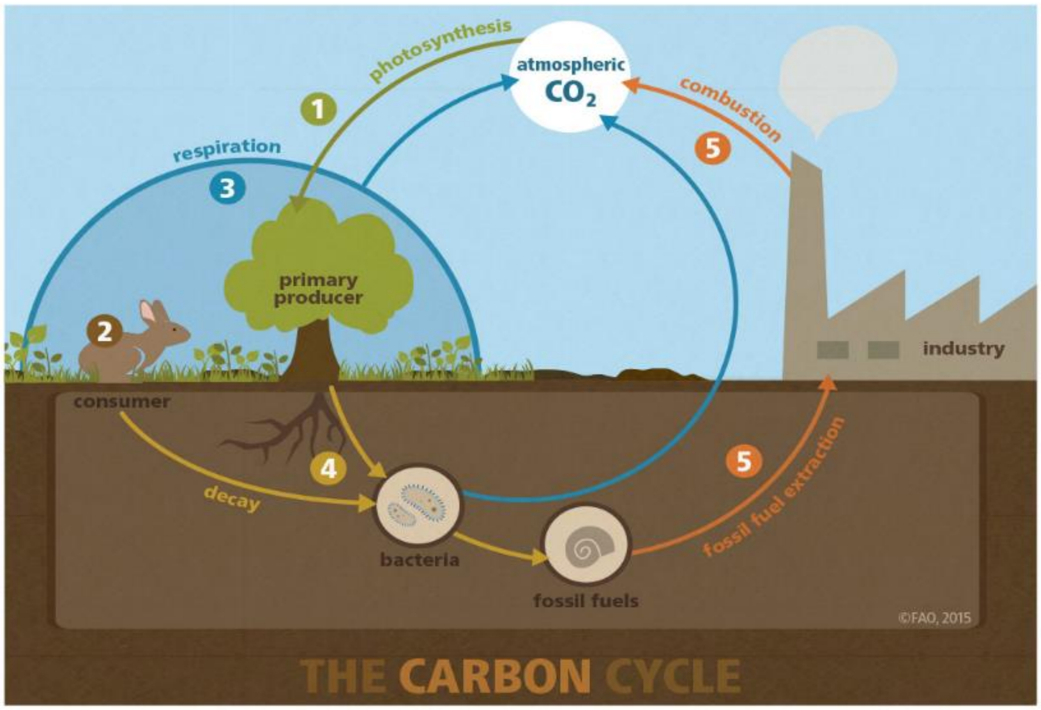
2015 International Year of Soils

Healthy soils provide the largest store of terrestrial carbon. When managed sustainably, soils can play an important role in climate change mitigation by storing carbon (carbon sequestration) and decreasing greenhouse gas emissions in the atmosphere. Conversely, if soils are managed poorly or cultivated through unsustainable agricultural practices, soil carbon can be released into the atmosphere in the form of carbon dioxide (CO_2), which can contribute to climate change. The steady conversion of grassland and forestland to cropland and grazing lands over the past several centuries has resulted in historic losses of soil carbon worldwide. However, by restoring degraded soils and adopting soil conservation practices, there is major potential to decrease the emission of greenhouse gases from agriculture, enhance carbon sequestration and build resilience to climate change.

Sustainable Satoyama–Satoumi landscape management in Japan builds resilience to climate change. ©FAO/Kazem Vafadari

SOILS AND THE CARBON CYCLE

The carbon cycle is the exchange of carbon (in various forms, e.g., carbon dioxide) between the atmosphere, ocean, terrestrial biosphere and geological deposits. Most of the carbon dioxide in the atmosphere comes from biological reactions that take place in the soil. Carbon sequestration occurs when carbon from the atmosphere is absorbed and stored in the soil. This is an important function because the more carbon that is stored in the soil, the less carbon dioxide there will be in the atmosphere contributing to climate change.



THE CARBON CYCLE

1. Plants use carbon dioxide from the atmosphere, water from the soil and sunlight to make their own food and grow in a process called photosynthesis. The carbon they absorb from the air becomes part of the plant.
2. Animals that feed on the plants pass the carbon compounds along the food chain.
3. Most of the carbon the animals consume is converted into carbon dioxide as they breathe (respiration), and is released back into the atmosphere.
4. When the animals and plants die, the dead organisms are eaten by decomposers in the soil (bacteria and fungi) and the carbon in their bodies is again returned to the atmosphere as carbon dioxide.
5. In some cases, the dead plants and animals are buried and turn into fossil fuels, such as coal and oil, over millions of years. Humans burn fossil fuels to create energy, which sends most of the carbon back into the atmosphere in the form of carbon dioxide.

2015 International Year of Soils

KEY CHALLENGES

Climate change represents a serious threat to global food security, not least because of its effects on soils. Changes in temperature and rainfall patterns can have a great impact on the organic matter and processes that take place in our soils, as well as the plants and crops that grow from them. In order to meet the related challenges of global food security and climate change, agriculture and land management practices must undergo fundamental transformations. Improved agriculture and soil management practices that increase soil organic carbon, such as agro-ecology, organic farming, conservation agriculture and agroforestry, bring multiple benefits. They produce fertile soils that are rich in organic matter (carbon), keep soil surfaces vegetated, require fewer chemical inputs, and promote crop rotations and biodiversity. These soils are also less susceptible to erosion and desertification, and will maintain vital ecosystem services such as the hydrological and nutrient cycles, which are essential to maintaining and increasing food production. FAO also promotes a unified approach, known as Climate-Smart Agriculture (CSA), to develop the technical, policy and investment conditions that support its member countries in achieving food security under climate change. CSA practices sustainably increase productivity and resilience to climate change (adaptation), while reducing and removing greenhouse gases whenever possible (mitigation).

Moringa seedlings at a tree nursery. The moringa tree can play an important role in mitigating climate change and increasing the incomes of poor farmers in Africa. ©FAO/Daniel Hayduk

A villager walking through a peat bog in Tunisia. ©FAO/Giulio Napolitano

FAO IN ACTION

The Organic Soils and Peatlands Climate Change Mitigation Initiative

Peatlands store tremendous amounts of carbon. However, when they are drained and used, mainly for agriculture, grazing and forestry, peatlands become significant sources of greenhouse gas emissions. Peatlands drainage and peat fires are responsible for almost 10 percent of greenhouse gas emissions from the Agriculture, Forestry and Other Land Use sector (AFOLU). The vital role peatlands play in avoiding and reducing greenhouse gas emissions, as well as in water regulation and unique biodiversity conservation, is insufficiently recognized. The Organic Soils and Peatlands Climate Change Mitigation Initiative is an informal network of organizations established to raise awareness about peatlands, promote strategic action for reducing greenhouse gas emissions from peatlands and organic soils, and safeguard their other vital ecosystem services. FAO and the Initiative identify three main strategies for reducing emissions from peatlands and organic soils: 1. secure undrained peatlands to prevent emissions; 2. rewet drained peatlands to reduce emissions; and 3. adapt management strategies for peatlands that cannot be rewetted.

2015 International Year of Soils

FAO IN ACTION

A view of terraced hills, which help soils retain water and prevent erosion.
©FAO/Giulio Napolitano

Three Rivers Sustainable Grazing Project in China

Restoring degraded grasslands through sustainable grassland management can lock more carbon in soils and biomass, increase the water-holding capacity of the soil and enhance grassland biodiversity. The Three Rivers Sustainable Grazing Project in the Qinghai province of China aims to restore degraded grazing land and sequester soil carbon, while at the same time increasing productivity, building resilience and improving livelihoods in smallholder herder communities. The pilot programme is helping local yak- and sheep-herding households adopt a combination of sustainable grassland management options related to grazing intensity, grass cultivation and animal husbandry. The average annual mitigation potential in the first 10 years of the project was estimated at 63 000 tonnes of CO_2 equivalent per year.

Climate-Smart Agriculture for smallholder farmers in Kenya and Tanzania

As part of its two pilot projects in Tanzania and Kenya, FAO's Mitigation of Climate Change in Agriculture (MICCA) programme selected and promoted the uptake of different practices based on experts and participatory assessments with farmers. Some 9 000 farmers in both countries, 40 percent of whom were women, received training on climate-smart agriculture, resulting in 736 energy-efficient cooking stoves being adopted to reduce deforestation. 79 tree nurseries were created, 417 000 tree seedlings were planted and 6 ha of terraces were established (on 204 farms) to conserve soil and water. Two biogas digesters were also installed to produce renewable energy from cow manure.

KEY FACTS

- Land-use conversions and drainage of organic soils for cultivation are responsible for about 10 percent of all greenhouse gas emissions.
- It is estimated that because of drainage, peatlands are currently the third-largest emitter of greenhouse gases in the AFOLU sector.
- It is estimated that soils can sequester around 20 PgC (petagrams of carbon) in 25 years, more than 10 percent of the anthropogenic emissions.
- Greenhouse gas emissions from agriculture, forestry and fisheries have nearly doubled over the past 50 years, and could increase an additional 30 percent by 2050 without greater efforts to reduce them.
- Emissions generated during the application of synthetic fertilizers accounted for 14 percent of agricultural emissions in 2012, and are the fastest growing emissions source in agriculture, having increased some 45 percent since 2001.
- Peatlands and organic soils contain nearly 30 percent of the world's soil carbon but only cover three percent of the earth's land area.
- The AFOLU sector is responsible for just under a quarter (~10–12 $GtCO_2eq/yr$) of anthropogenic greenhouse gas emissions, mainly from deforestation and agricultural emissions from livestock, soil and nutrient management.
- Soil carbon sequestration increases the ability of soils to hold soil moisture, withstand erosion and enrich ecosystem biodiversity, which helps cropping systems to better withstand droughts and floods.

Food and Agriculture Organization of the United Nations
Viale delle Terme di Caracalla
00153 Rome, Italy
Tel:(+39) 06 57051
Fax:(+39) 06 570 53152
e-mail: soils-2015@fao.org
web: www.fao.org

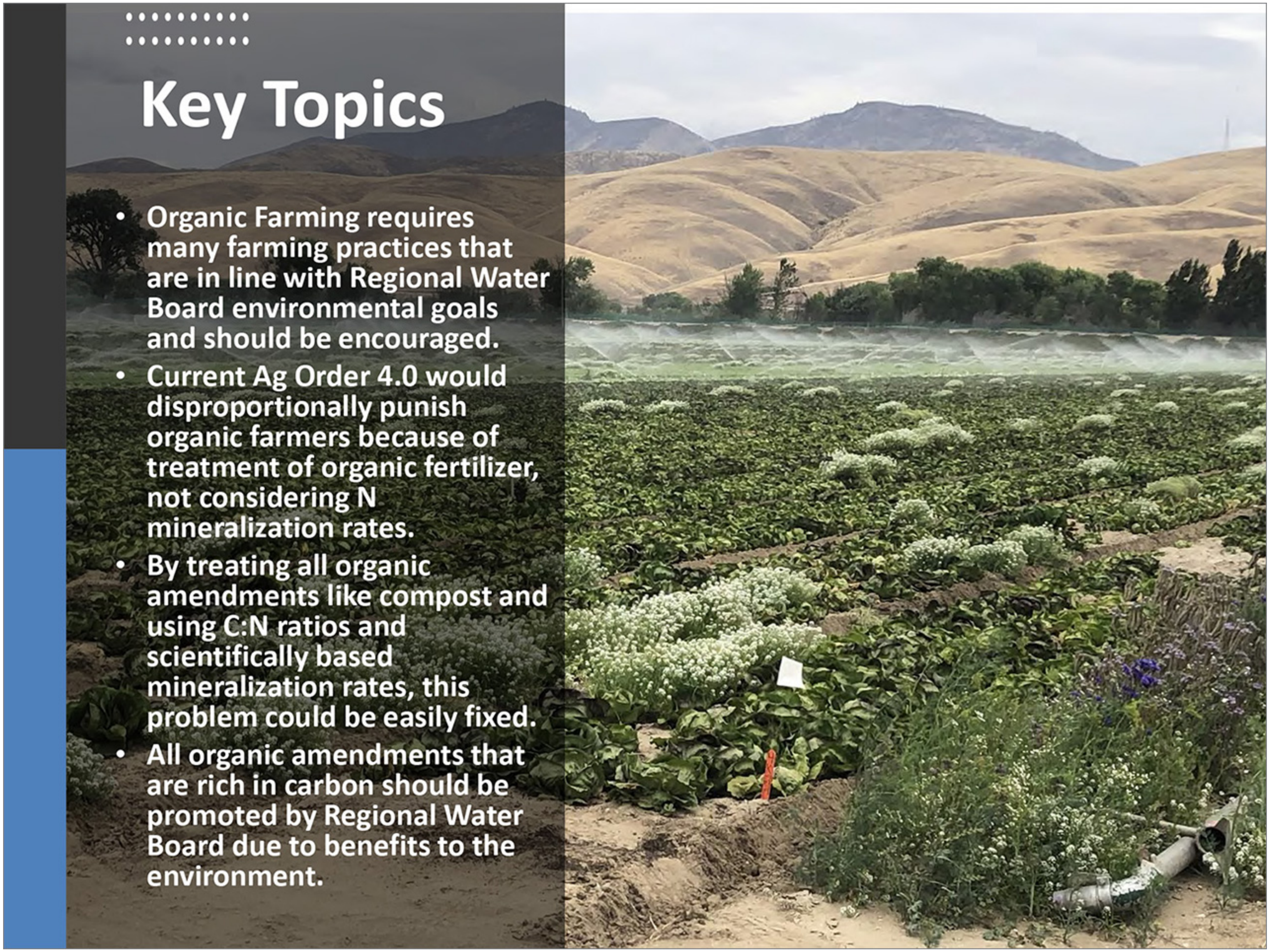
#IYS2015

fao.org/soils-2015


Attachment 13
THE WORLDWIDE
LEADER
IN ORGANIC
FERTILIZER
ORGANIC PRODUCTS. INC
Presenting to
CALIFORNIA
Water Boards
STATE WATER RESOURCES CONTROL BOARD
REGIONAL WATER QUALITY CONTROL BOARDS
www.true.ag


Made in the USA
Servicing our customer for more than 15 years
Earthbound Farm ORGANIC
GRIMMWAY FARMS
josie's organics
organicgirl
Driscoll's ONLY THE FINEST BERRIES
FOXY organic
Dole ORGANIC
Taylor Farms
Tasteful SELECTIONS
Zirkle FRUIT COMPANY
Cal-Organic FARMS
CALIFORNIA GIANT BERRY FARMS
Sunkist
TANIMURA & ANTLE
Rainier FRUIT COMPANY
LUNDBERG FAMILY FARMS
SUN PACIFIC
BONIPAK SINCE 1932
TRUE ORGANIC PRODUCTS, INC
www.true.ag


Ag Order 4.0
Impact to Organic Farming
• Organic in California
• Ag Order 4.0 Goals & Organic Farming Goals
• Ag Order 4.0 and the penalty for organic farmers
• Food Safety Considerations
• Solution – using C:N ratios to predict N mineralization rates
TRUE
ORGANIC PRODUCTS, INC
www.true.ag


Key Topics
• Organic Farming requires many farming practices that are in line with Regional Water Board environmental goals and should be encouraged.
• Current Ag Order 4.0 would disproportionally punish organic farmers because of treatment of organic fertilizer, not considering N mineralization rates.
• By treating all organic amendments like compost and using C:N ratios and scientifically based mineralization rates, this problem could be easily fixed.
• All organic amendments that are rich in carbon should be promoted by Regional Water Board due to benefits to the environment.

Organic Farming Sector in California Agriculture

Substantial and Still Growing

- Organic sales in the U.S. totaled a new record of $49.4 billion in 2017, up 6.4% from 2016 (OTA)
- Organic farms in the U.S. produced and sold $7.6 Billion worth of organic products in 2016, up 23% from 2015 (CCOF)
- California produced 38% of total US farm commodity value for organics, with $2.9 Billion in organic crops, poultry, livestock, and dairy products sold in 2017 (CCOF)

California County	Value of Organic Crops (2017)	Acres of Organic Farm Land (2017)
Monterey	$390,295,000	40,859
Santa Cruz	$109,058,000	6,702
San Benito	56,511,500	46,802
Ventura	197,386,000	8,851
Imperial	-	45,216

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Ag Order 4.0 – Goals

Protect and restore beneficial uses and achieve water quality objectives

- Minimizing nitrate discharges to groundwater
- Minimizing nutrient discharges to surface water
- Minimizing toxicity in surface water from pesticide discharges
- Protecting and restoring riparian and wetland habitat, and
- Minimizing sediment discharges to surface water

Consistent with Organic Food Production Act

"Organic agriculture practices ...are used to minimize pollution from air, soil and water."

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Ag Order 4.0 – Discharge Limit in older version

$A_{FER} + A_{IRR} - R$ = Target or Limit (lbs/ac/ranch/year) (depends on year)

- Application Limits A_{FER} cannot exceed Target or Limit
- Ranches that repeatedly exceed the numeric discharge limit per the time schedule may be limited or prohibited from applying A_{FER}.
  - A_{FER} is the amount of nitrogen applied in fertilizers, ***compost**, and **other amendments***
  - A_{IRR} is the amount of nitrogen applied through the irrigation water based on the groundwater nitrate concentration
  - $A_{FER} + A_{IRR}$ = the total amount of nitrogen applied
  - R is the amount of nitrogen removed through harvest, pruning, or other methods, plus the nitrogen sequestered in perennial crop permanent wood

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Ag Order 4.0 – Fertilizer Nitrogen Limits

Compliance Pathway 1:

$$A_{FER} + (C \times A_{COMP}) + A_{IRR} - R = \text{Nitrogen Discharge}$$

Compliance Pathway 2:

$$A_{FER} + (C \times A_{COMP}) = R$$

Fundamentally Incomplete

Conventional N ≠ Organic N

(A_{FER} for conventional is not the same as A_{FER} for organic)

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Ag Order 4.0 – Fertilizer Nitrogen Limits

Compliance Pathway 1:

$$A_{FER} + (C \times A_{COMP}) + A_{IRR} - R = \text{Nitrogen Discharge}$$

The Central Coast Water Board's standard compost discount factors (C) are defined below. Different compost discount factors are applied based on the carbon to nitrogen (C:N) ratio of the product.2

i. For C:N ratio > 11:1, C = 0.05. That is, 5 percent of the nitrogen in the compost will be counted in the A-R compliance calculation.

ii. For C:N ratio ≤ 11:1, C= 0.10. That is, 10 percent of the nitrogen in the compost will be counted in the A-R compliance calculation.

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Ag Order 4.0 – Fertilizer Nitrogen Limits

Compost Has Limits for Organic Farmers:

1. Compost is good at improving soil health at appropriate rates. However
   - Compost is not a good source of **plant available nitrogen** for many crops.
   - Many growers are concerned to use compost (especially manure-base compost) due to potential **food safety risks.**
2. Organic fertilizers meet organic grower needs
   - Comprehensive food safety program including kill-steps for all organic products (True is food safety leader, ISO 22000).
   - Organic fertilizers are made from manures and agricultural byproducts (feather meal, meat & bone meals) so they can deliver more plant available nitrogen. N mineralization rates are generally 40-60%, way below conventional fertilizers (100% plant available).

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Propose ***Organic Ammendments***

Inclusion of all Organic Ammendments
(i.e. compost, dry and liquid organic fertilizers)

Proposed Compliance Pathway 1:

$$A_{FER} + (M \times A_{ORG}) + A_{IRR} - R = \text{Nitrogen Discharge}$$

Proposed Compliance Pathway 2:

$$(M \times A_{ORG}) = R$$

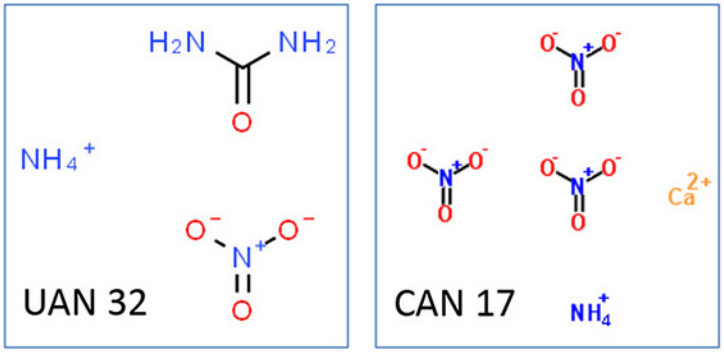
M based on predicted mineralization rates that is calculated using C:N ratio of organic amendments

TRUE ORGANIC PRODUCTS, INC

www.true.ag

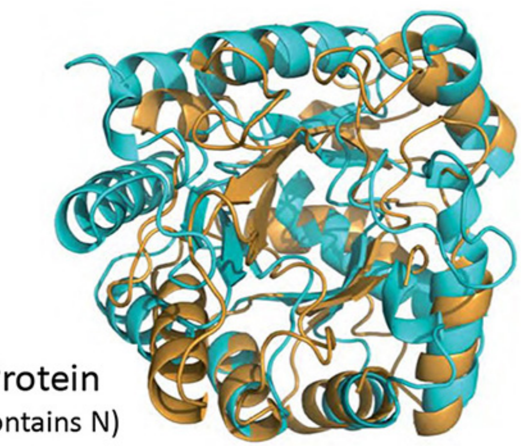
Organic Nitrogen vs Conventional Nitrogen

Fundamentally Different



UAN 32 CAN 17

100% Plant Available Nitrogen (PAN)
Thus 100% Total Nitrogen Applied



Protein
(Contains N)

Organic growers will have to grow crops with 40-60% less mineralized nitrogen than conventional growers

Ag Order 4.0 will disproportionally penalize organic farming

- N must be converted to PAN or "mineralized"
- Only a fraction of total nitrogen applied in organic fertilizer and organic amendments by organic growers is converted to mineralized nitrogen
- Mineralization rates of organic amendments and fertilizers vary greatly but are generally below 60% in laboratory studies.

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Received: 28 April 2019 | Accepted: 7 October 2019
DOI: 10.1002/jeq2.20030

Journal of Environmental Quality

TECHNICAL REPORTS
Waste Management

Nitrogen mineralization from organic amendments is variable but predictable

Patricia Lazicki[1] | Daniel Geisseler[1] | Margaret Lloyd[2]

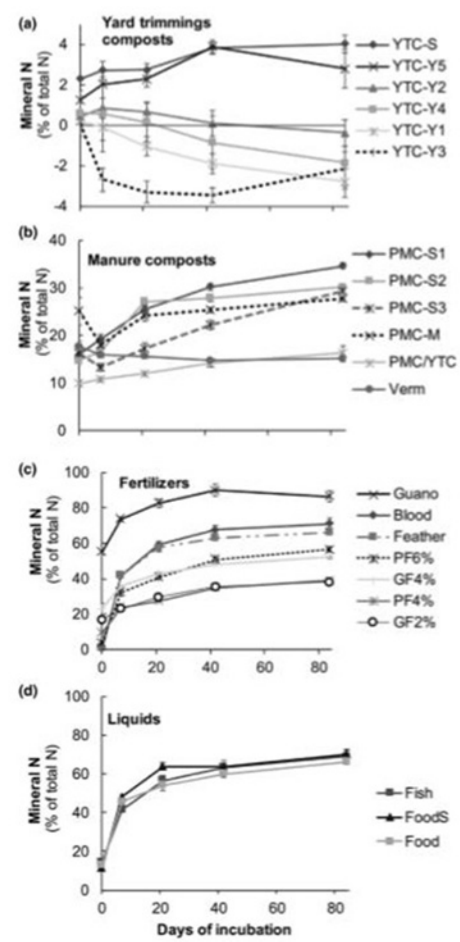
[1]Dep. of Land, Air and Water Resources, Univ. of California, Davis, One Shields Ave., Davis, CA 95616, USA
[2]Univ. of California Cooperative Extension, Capitol Corridor, 70 Cottonwood St., Woodland, CA 95695, USA

Correspondence
Patricia Lazicki, Dep. of Land, Air and Water Resources, Univ. of California, Davis, One Shields Ave., Davis, CA 95616, USA.
Email: palazicki@ucdavis.edu

Assigned to Associate Editor Mussie Habteselassie.

Abstract

To manage nitrogen (N) efficiently, organic growers must be able to predict the amount and timing of plant-available N from organic amendments. In this study, we measured N mineralization from a variety of organic amendments, including composted animal manures and plant material, pelleted and granular organic fertilizer formulations, slaughter waste products, and hydrolyzed liquid fertilizers. In a laboratory incubation, we measured net N mineralization from materials mixed with either organically or conventionally managed soil at 23°C and 60% water holding capacity after 0, 7, 21, 42, and 84 d. We found that net mineral N change in the amended soils after 84 d of incubation fell into four categories: immobilization to 5% of applied N for yard trimmings composts, 15–30% for poultry manure composts, 35–55% for granular fertilizers, and 60–90% for quick release products. However, across all amendments the amount of plant-available N after 84 d of incubation was well correlated with the carbon (C)/N ratio ($R^2 = 0.92$). Within amendment types, the C/N ratio predicted N mineralization for yard trimmings composts ($R^2 = 0.91$), manure composts ($R^2 = 0.81$), and specialty fertilizer and slaughter products ($R^2 = 0.88$) but not liquid products ($R^2 = 0.11$). Soil management history did not consistently affect net N mineralization but may have influenced timing.



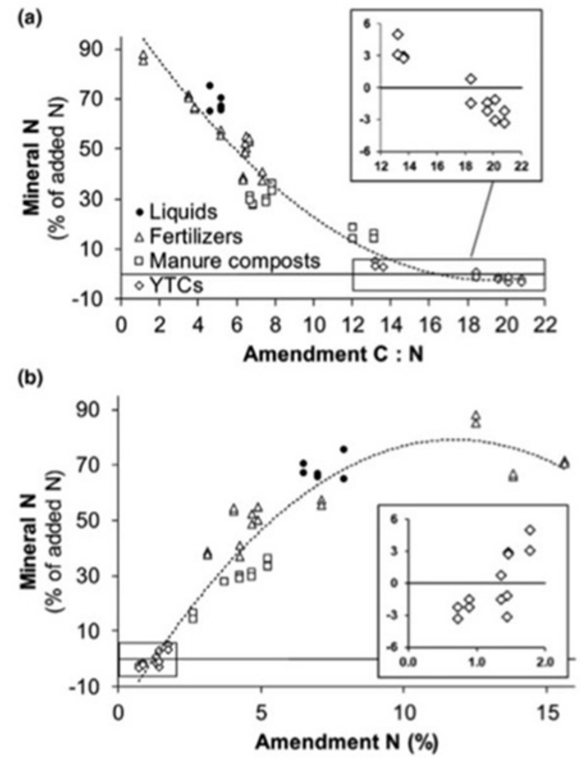


FIGURE 3 Relationship between the proportion of N in the mineral form after 84 d of incubation at 23°C in conventional (CONV) or organic (ORG) soil and (a) amendment C/N ratio and (b) amendment N concentration on a dry-weight basis. Insets represent the yard trimmings composts (YTCs). The R^2 value for the trendline in graph (a) is 0.927. The trendline in graph (b) has an R^2 value of 0.915

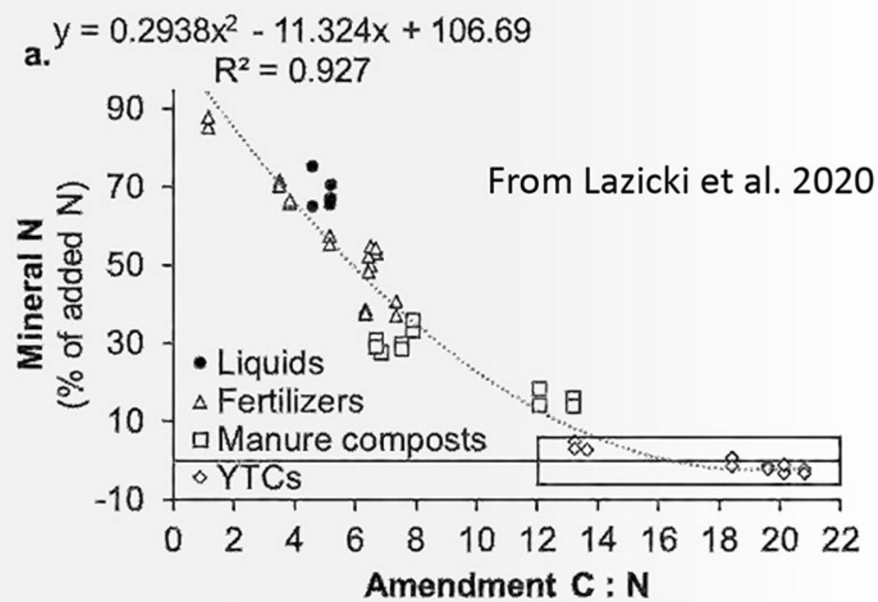
Mineral N Correlation

Proposed Compliance Pathway 1:

$A_{FERT} + (M \times A_{ORG}) + A_{IRR} - R$ = Nitrogen Discharge

Proposed Compliance Pathway 2:

$A_{FERT} + (M \times A_{ORG}) = R$



C to N ratio	Predicted N Mineralization Rate (M)
<1.5	1.000
1.5	0.904
2	0.852
2.5	0.802
3	0.754
3.5	0.707
4	0.661
4.5	0.617
5	0.574
5.5	0.533
6	0.493
6.5	0.455
7	0.418
7.5	0.383
8	0.349
8.5	0.317
9	0.286
9.5	0.256
10	0.228
10.5	0.202
11	0.177
11.5	0.153
12	0.131
12.5	0.111
13	0.091
13.5	0.074
14	0.058
14.5	0.043
15	0.030
15.5	0.018
16	0.007

Example: *Organic Ammendments*

How it works on some True Organic Fertilizers

Proposed Compliance Pathway 1:

$$A_{FER} + (M \times A_{ORG}) + A_{IRR} - R = \text{Nitrogen Discharge}$$

Proposed Compliance Pathway 2:

$$(M \times A_{ORG}) = R$$

Organic Fertilizer	Ingredients	C:N Ratio	Predicted N Mineralization Rate (M)
True 2.5-2-2.5	Poultry Manure	9.5	0.26
True 4-4-2	Poultry Manure, Meat & Bone Meal, Feather Meal	7	0.42
True 8-5-1	Meat & Bone Meal, Poultry Manure, Feather Meal	4.5	0.62
True 10-5-2	Meat & Bone Meal, Feather Meal	4	0.66

M based on predicted mineralization rates that is calculated using C:N ratio of organic amendments

TRUE ORGANIC PRODUCTS, INC

www.true.ag

UCCE Nutrient Management Specialist Unit
Land, Air & Water Resources
•djgeisseler@ucdavis.edu
• https://caes.ucdavis.edu/people/daniel-geisseler

University of California Cooperative Extension

Bio

Daniel Geisseler, an assistant Cooperative Extension specialist in the Department of Land, Air and Water Resources, specializes in nutrient management. Geisseler completed his Ph.D. in soil science at the University of California, Davis. He worked as a postdoctoral scientist at the University of Kassel in Germany and at UC Davis before joining our faculty in 2014.

Research interests:

Cropping systems, nutrient management, plant nutrition, soil nitrogen cycle, nitrogen use efficiency.

Comment from Daniel Geisseler in regards to using Lazicki et al. 2020 regression model to predict N mineralization rates:

"I'm not opposed to using our values, but I would be more comfortable if a comment was added saying that these are potential N mineralization rates and that the actual mineralization rates are lower during the cold season and when the amendments are not incorporated. Under these conditions, growers should be allowed to use lower values. Can you convince the Water Board to do that?"

Richard Smith
Farm Advisor, Vegetable Crop Production & Weed Science
Cooperative Extension Monterey County
1432 Abbott Street
Salinas, CA 93901
(831) 759-7357
rifsmith@ucanr.edu
http://cemonterey.ucanr.edu

UC CE University of California Cooperative Extension

Input of Carbon

Material	Biomass lbs/A	Carbon content percent	Total carbon lbs/A
Compost	10,000[1]	29%	2,146
Cover crop	6,000	44%	2,640
True 4-4-2 2 baby crops @ 3000 each	5,400[2]	29%	1,566
True 8-5-1 1 broccoli crop	5,000[3]	41%	2,050

1 – 10,000 lbs/A @ 74% oven dry weight
2 – 6000 lbs/A (2 baby crops @ 3000 lbs/A each) @ 90% oven dry weight;
3 – 5650 lbs/A @ 90% oven dry weight

Increased Carbon Aligns with Ag Order 4.0

The **carbon** in organic matter is the main source of energy for the all important soil microbes and is also the key for making nutrients available to plants. The list of positive influences high levels of organic matter have on healthy soils includes:

USDA NRCS
U.S. Department of Agriculture
Natural Resources Conservation Service

1. Provides a carbon and energy source for soil microbes
2. Stabilizes and holds soil particles together
3. Supplies, stores, and retains such nutrients as nitrogen, phosphorus and sulfur
4. Improves the soil's ability to store and move air and water
5. Contributes to lower soil bulk density and less compaction
6. Makes soil more friable, less sticky, and easier to work
7. Retains carbon from the atmosphere and other sources
8. Reduces the negative environmental effects of pesticides, heavy metals and other pollutants
9. Improves soil tilth in surface horizons
10. Increases water infiltration rates
11. Reduces crusting
12. Reduces water runoff
13. Encourages plant root development and penetration
14. Reduces soil erosion

TRUE ORGANIC PRODUCTS, INC

www.true.ag

Sustainability Program

Science AAAS

Soil Carbon Sequestration Impacts on Global Climate Change and Food Security

R. Lal

Science 11 Jun 2004:
Vol. 304, Issue 5677, pp. 1623-1627
DOI: 10.1126/science.1097396

Abstract

The carbon sink capacity of the world's agricultural and degraded soils is 50 to 66% of the historic carbon loss of 42 to 78 gigatons of carbon. The rate of soil organic carbon sequestration with adoption of recommended technologies depends on soil texture and structure, rainfall, temperature, farming system, and soil management. Strategies to increase the soil carbon pool include soil restoration and woodland regeneration, no-till farming, cover crops, nutrient management, manuring and sludge application, improved grazing, water conservation and harvesting, efficient irrigation, agroforestry practices, and growing energy crops on spare lands.

An increase of 1 ton of soil carbon pool of degraded cropland soils may increase crop yield by 20 to 40 kilograms per hectare (kg/ha) for wheat, 10 to 20 kg/ha for maize, and 0.5 to 1 kg/ha for cowpeas. As well as enhancing food security, carbon sequestration has the potential to offset fossil fuel emissions by 0.4 to 1.2 gigatons of carbon per year, or 5 to 15% of the global fossil-fuel emissions.

Sustainability Program

AGRICULTURE, CLIMATE, EARTH SCIENCES

Can Soil Help Combat Climate Change?

BY RENEE CHO |FEBRUARY 21, 2018

Comments

Photo: NestorT

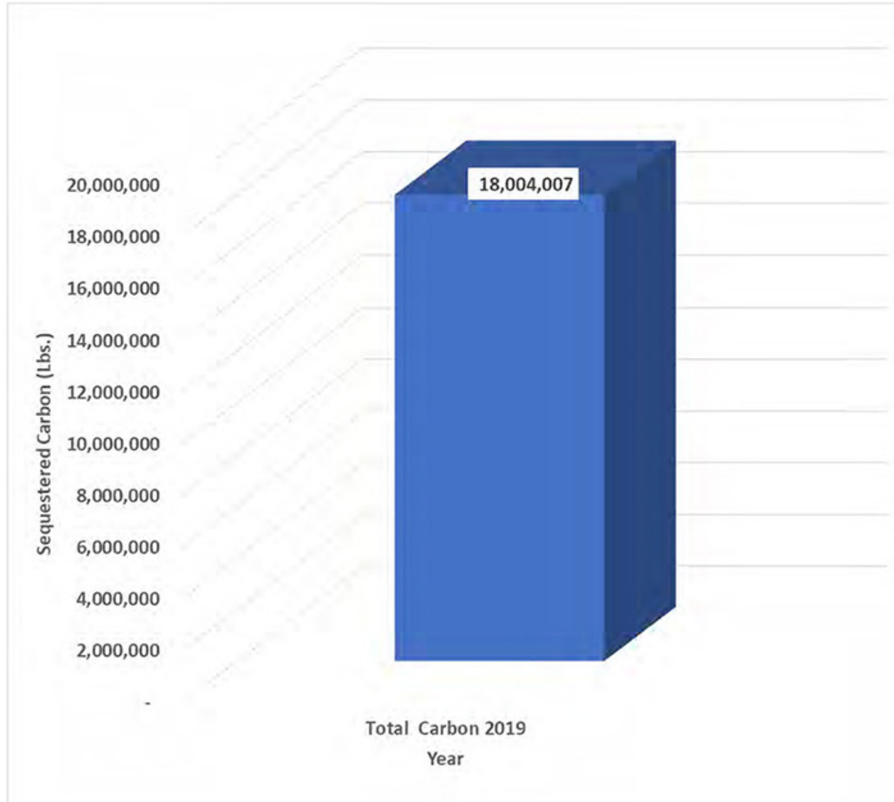
To avoid the most dangerous effects of climate change, the Paris Accord recommends limiting global warming to less than 2° C above pre-industrial levels. Achieving that will likely involve removing carbon dioxide from the atmosphere, according to the Intergovernmental Panel on Climate Change. But strategies like capturing and storing the carbon emissions from biofuel-burning power plants, or planting new forests to absorb carbon, can create their own problems. If used on a scale large enough to be effective, they would require too much land, water, or energy, or are too expensive.

Sequestering carbon in soil, however, is a relatively natural way of removing carbon dioxide from the atmosphere with fewer impacts on land and water, less need for energy, and lower costs. Better land management and agricultural practices could enhance the ability of soils to store carbon and help combat global warming.

Current Carbon Application

Carbon Applied by A Central Coast Grower during 2019 & 2020 using True Organic Products

All 2019, 2020 only for January to March 27 use



TRUE ORGANIC PRODUCTS, INC

www.true.ag

Soil Health Evaluation, Background History of Commercial Fields

Block	Crop rotation	Type of crop	Fertilizer applied	Farm History
Organic Field 1	One crop/yr	Brassica/ Broccoli	True 8-5-1, 3000 lb/acre	~15 yr Organic
Organic Field 2	One crop/yr	Brassica/ Broccoli	True 8-5-1, 3000 lb/acre	~15 yr Organic
Organic Field 3	Two crops/yr	Broccoli/ Celery/ Romaine/ Spinach	True 8-5-1, 6000 lb/acre	~20 yr Organic
Conventional Field 1	Two crops/yr	Broccoli/ Spinach/ Romaine	Standard Conventional NPK programs	Conventional

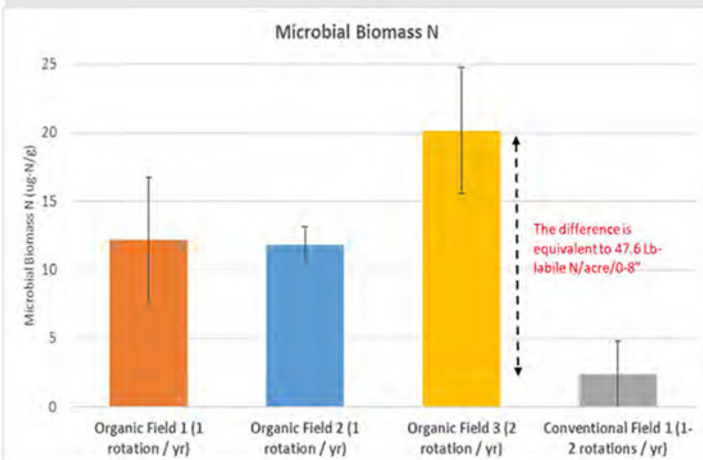
Soil sampling at pre-planting stage, early Mar 2019

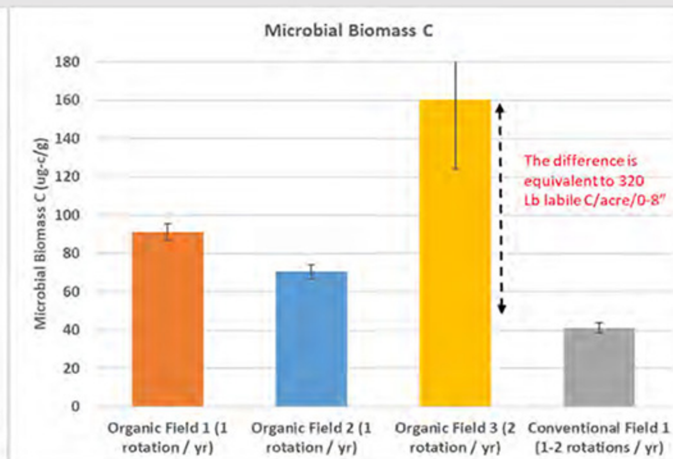
Microbial biomass C and N

Relevance to soil health:

i. Size of microbial biomass pool is a key indicator of soil biological status.

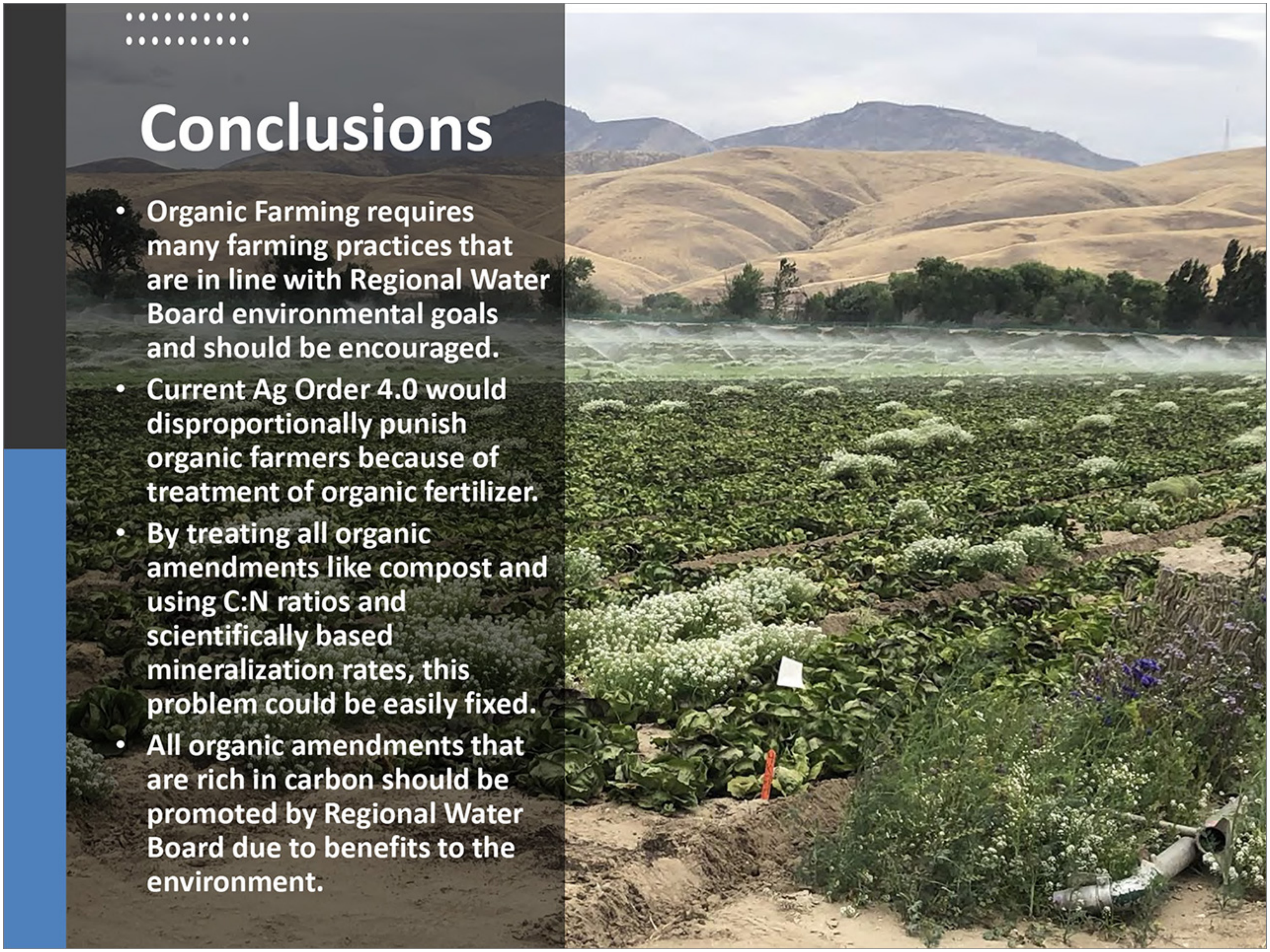
ii. As microbial pool turns over, the C and N in the biomass is released as available C and N, so the size of Microbial Biomass N is related to potential of partial supply of N for the crop.





Organic Amendments - Nitrogen that does not Mineralize

- The nitrogen in Organic Amendments that does not mineralize during a cropping cycle is thought to become **recalcitrant** and moves into the Soil Organic Matter pool (Gaskell et al., UC ANR publication # 7249).
- The more carbon that is added to the soil, the higher the soil organic matter level is increased and the more nitrogen can be stored in a non-mineral form that should not leach.


Conclusions
• Organic Farming requires many farming practices that are in line with Regional Water Board environmental goals and should be encouraged.
• Current Ag Order 4.0 would disproportionally punish organic farmers because of treatment of organic fertilizer.
• By treating all organic amendments like compost and using C:N ratios and scientifically based mineralization rates, this problem could be easily fixed.
• All organic amendments that are rich in carbon should be promoted by Regional Water Board due to benefits to the environment.


ORGANIC PRODUCTS. INC
MANUFACTURING PLANT
559.866.3001
CORPORATE HEADQUARTERS
831.375.4796
info@true.ag
Sales@true.ag
www.true.ag

Attachment 14

Reduced nitrate leaching and enhanced denitrifier activity and efficiency in organically fertilized soils

Sasha B. Kramer*†, John P. Reganold‡, Jerry D. Glover§, Brendan J. M. Bohannan*, and Harold A. Mooney*†

*Department of Biological Sciences, Stanford University, Stanford, CA 94305; ‡Department of Crop and Soil Sciences, Washington State University, Pullman, WA 99164; and §The Land Institute, 2440 East Water Well Road, Salina, KS 67401

Contributed by Harold A. Mooney, January 17, 2006

Conventional agriculture has improved in crop yield but at large costs to the environment, particularly off-site pollution from mineral N fertilizers. In response to environmental concerns, organic agriculture has become an increasingly popular option. One component of organic agriculture that remains in question is whether it can reduce agricultural N losses to groundwater and the atmosphere relative to conventional agriculture. Here we report reduced N pollution from organic and integrated farming systems compared with a conventional farming system. We evaluated differences in denitrification potential and a suite of other soil biological and chemical properties in soil samples taken from organic, integrated, and conventional treatments in an experimental apple orchard. Organically farmed soils exhibited higher potential denitrification rates, greater denitrification efficiency, higher organic matter, and greater microbial activity than conventionally farmed soils. The observed differences in denitrifier function were then assessed under field conditions after fertilization. N_2O emissions were not significantly different among treatments; however, N_2 emissions were highest in organic plots. Annual nitrate leaching was 4.4–5.6 times higher in conventional plots than in organic plots, with the integrated plots in between. This study demonstrates that organic and integrated fertilization practices support more active and efficient denitrifier communities, shift the balance of N_2 emissions and nitrate losses, and reduce environmentally damaging nitrate losses. Although this study specifically examines a perennial orchard system, the ecological and biogeochemical processes we evaluated are present in all agroecosystems, and the reductions in nitrate loss in this study could also be achievable in other cropping systems.

denitrification | nitrogen | organic agriculture | sustainable agriculture | integrated farming

The intensification of agricultural production over the past 60 years and the subsequent increase in global synthetic N inputs have resulted in substantial N pollution and ecological damage (1). The primary source of N pollution comes from N-based agricultural fertilizers, whose use is forecasted to double or almost triple by 2050 (2). The application of N fertilizers has resulted in N leakage from agricultural systems into groundwater, rivers, coastal waters, and the atmosphere (3). Nitrate leaching and N_2O emissions from agricultural soils are recognized as significant environmental threats by scientists, environmental groups, and agricultural policymakers (4, 5).

Nitrate leaching and runoff into rivers and estuarine ecosystems are responsible for algal blooms and eutrophication and also pose a public health risk (6, 7). For example, 9% of U.S. domestic wells sampled during 1993–2000 had nitrate concentrations exceeding the U.S. Environmental Protection Agency's (EPA) maximum contaminant level of 10 mg·liter^{-1} as N (8). In the Yakima River Basin of Washington State, where this study was conducted, 13% of the samples taken from small-watershed sites exceeded the EPA's maximum contaminant level, indicating a potential health risk to nearby residents with shallow wells (9).

N_2O, a greenhouse gas nearly 300 times more effective at radiative warming than carbon dioxide (10), is produced mainly during the microbially mediated process of denitrification. There has been a marked increase in atmospheric N_2O over the past 150 years, largely attributed to fertilized agriculture (11). In most unmanaged systems, the majority of the gas produced during denitrification is fully reduced N_2, a nonreactive and environmentally benign gas. However, a variable portion of the nitrate that enters the process will escape as N_2O before being fully reduced. In agricultural systems N_2O emissions are enhanced after fertilization (12). The proportion of gas escaping as N_2O [relative rate of N_2O emissions (rN_2O)] is highly dependent on environmental factors, with rN_2O being lowest in high C environments (13, 14).

Given the environmental problems associated with the production and use of synthetic fertilizer, there is a great need for researchers concerned with global climate change and nitrate pollution to evaluate reduction strategies (1, 3, 4). Fertilization with organic wastes and composts is a means of recycling terrestrially available N, thereby reducing both N inputs to the biosphere and dependence on fossil fuels needed to produce synthetic fertilizers (15). It has also been suggested that the use of organic fertilizers, alone or in combination with synthetic fertilizers, may mitigate N pollution from agricultural systems (16, 17).

Here we investigate the role of the soil denitrifier community in mediating the magnitude and composition of gaseous N emissions and the relative balance of N losses through denitrification and leaching after fertilization in organic, integrated, and conventional apple orchards. For 9 years before this study, the organic system followed a regimen of organically certified practices, including the exclusion of synthetic agrochemicals, and the integrated system used a combination of organic and conventional fertilizers and techniques.

This study is unique because it compares denitrifier function in soils from organic, integrated, and conventional plots and then examines the field implications of observed functional differences after a fertilization experiment. Although numerous studies have compared N cycling in response to organic and synthetic fertilizer amendments (Table 4, which is published as supporting information on the PNAS web site), none has simultaneously quantified gaseous and leaching N losses after fertilization to our knowledge. In addition, comparative fertilization studies are frequently complicated by differences in N input intensity among the systems. This study examines denitrification and leaching from organic, integrated, and conventional systems receiving the same amount of N inputs but in different forms.

Conflict of interest statement: No conflicts declared.

Abbreviations: rN_2O, relative rate of N_2O emissions; PLFA, phospholipid fatty acid; ha, hectare.

†To whom correspondence may be addressed. E-mail: sash@stanford.edu or hmooney@stanford.edu.

4522–4527 | PNAS | March 21, 2006 | vol. 103 | no. 12 www.pnas.org/cgi/doi/10.1073/pnas.0600359103

Table 1. Soil biological and chemical properties

Treatment	Total soil N, ppm	Organic matter, %	Microbial biomass c*	Microbial biomass N†	Nitrification potential‡	L-asparaginase§	β-Glucosidase¶	DP‖	Potential N_2O**	Potential rN_2O	DP:MBC
Organic	1,955^{a}	3.40^{a}	512.7^{a}	61.0^{a}	0.77^{a}	91.9^{a}	192.9^{a}	113.92^{a}	43.08^{a}	0.38^{a}	0.25^{c}
Integrated	1,755^{a}	3.10^{a}	420.8^{a,b}	39.7^{b}	0.63^{a,b}	63.7^{b}	134.4^{b}	40.39^{b}	30.82^{b}	0.78^{b}	0.10^{b}
Conventional	1,242^{b}	2.23^{b}	357.7^{b}	34.0^{b}	0.49^{b}	59.8^{b}	131.3^{b}	12.21^{c}	8.68^{c}	0.73^{b}	0.04^{a}

Different superscript letters within rows indicate significant differences at the 0.05 level (least significant difference). MBC, microbial biomass carbon; DP, denitrification potential.

*mg of C pcr kg of soil.

†mg of N per kg of soil.

‡g of NO_3^- per g of soil per h.

§mg of N per g of soil per 2 h.

¶mg of *p*-nitrophenol per g of soil per h.

‖μmol $N_2O + N_2$ per h per g of soil.

**μmol of N_2O per h per g of soil.

Results and Discussion

Initially we compared a suite of fundamental ecological characteristics of the soils from the different treatment plots. Organically farmed soils had higher total N, organic matter content, microbial biomass C and N, nitrification potential, and L-asparaginase and β-glucosidase activity (enzymes indicative of microbial N and C cycling potential) compared with conventionally farmed soils (Table 1). These data are in agreement with other studies that have demonstrated that organically farmed soils support more active microbial populations then their conventional counterparts (18–20). Microbial activity and soil organic matter are important for soil nutrient cycling, a valuable ecosystem service (21), particularly when external inputs are not reliably available. Increased microbial activity and soil organic matter in the organically farmed soils results from a combination of enhanced C inputs during fertilization and increased grass cover relative to the integrated and conventional systems where glyphosate was used to keep the tree strips clean.

We then analyzed differences in microbial community composition using phospholipid fatty acid (PLFA) analysis, a standard microbial technique that quantifies fatty acids from microbiol cell walls. This technique can be used in conjunction with principal-components analysis to visualize similarities and differences in microbial communities. We found significant compositional differences between the microbial communities in the organic, integrated, and conventional systems (Fig. 1). The PLFA results indicate that the observed differences in microbial activity among the treatment soils may result from a combination of differences in microbial community size and composition.

As the primary biological source of gaseous N emissions, the denitrifier community is of particular significance for those interested in mitigating N pollution. We assessed differences in denitrifier activity and efficiency among the systems using a denitrification potential assay. When carbon, nitrate, oxygen, and pH were adjusted to ideal denitrification conditions in the laboratory, we found that denitrifying communities were more active (higher overall gas emissions, denitrification potential) and more efficient (lower potential rN_2O) in the organically farmed soils than in the conventionally farmed soils, with the integrated resembling the conventional in terms of efficiency and falling between the two treatments in terms of activity (Table 1). In addition, when denitrification potential was normalized to microbial biomass by calculating the ratio, the organic treatment had the highest ratio of denitrification potential to microbial biomass (Table 1), indicating that a higher proportion of the microbial community is able to denitrify in the organically farmed soils. Groffman and Tiedje (22) found that this ratio was a good predictor of annual denitrification N losses. In the field, differences in microbial function are mediated by environmental factors, and both activity and efficiency will vary in response to local conditions.

To examine field denitrification rates and their significance to overall N losses, we measured the relative magnitude of N losses as gaseous N_2, N_2O, and nitrate leaching for 1 month after fertilization in subplots in the experimental apple orchard. The conventional subplots were fertilized with $Ca(NO_3)_2$; the integrated subplots were fertilized with equal parts composted chicken manure and $Ca(NO_3)_2$; and in the organic plots two separate subplots were established to test two organic fertilizers: composted chicken manure and alfalfa meal. Nitrogen inputs were equal across treatments.

For the month after fall and spring fertilization, cumulative N_2O emissions were roughly equivalent among the four fertilizer treatments and significantly above controls in the fall (Table 2) (for time series data see Figs. 4 and 5, which are published as supporting information on the PNAS web site). Nitrous oxide emission rates ranged from 1 to 9 ng of N_2O-N $cm^{-2} \cdot h^{-1}$, higher

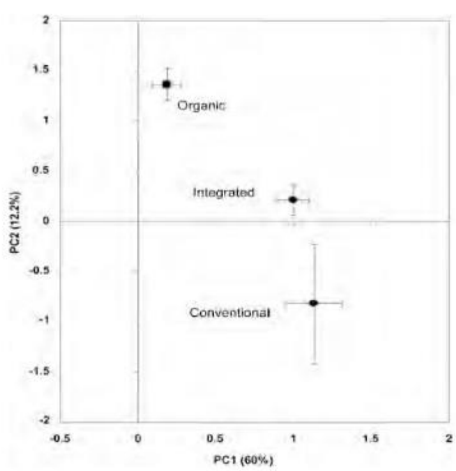


Fig. 1. PLFA analysis of soil samples from the organic, integrated, and conventional apple production systems. PLFA analysis, a standard microbial technique that can be used in conjunction with principal-components analysis to visualize similarities and differences in microbial communities, showed that there are significant differences between the microbial communities in the organic and the integrated and conventional systems. PLFA was performed in August 2002 on soils collected from 0- to 7.5-cm depth.

SUSTAINABILITY SCIENCE

ECOLOGY

Table 2. N_2O emissions and NO_3^- leaching

Treatment and subplots	NO_3 leaching (fall), μg of NO_3-N at 100 cm	N_2O (fall), g/ha N_2O-N	NO_3 leaching (spring), μg of NO_3-N at 100 cm	N_2O (spring), g/ha N_2O-N	Annual NO_3 leaching, μg of NO_3-N at 100 cm	Leaf N, %
Organic						
Compost	$9.66^{a,b}$	$88.57^{b,c}$	180.13^{a}	330.83^{b}	241.26^{a}	2.59^{a}
Alfalfa	$9.38^{a,b}$	55.65^{b}	234.11^{a}	316.10^{b}	309.84^{a}	2.68^{a}
Control	3.73^{a}	16.83^{a}	68.06^{a}	$282.28^{a,b}$	108.47^{a}	2.51^{a}
Integrated						
$CaNO_3$ + compost	14.08^{b}	124.57^{c}	608.26^{b}	327.25^{b}	772.83^{b}	2.40^{a}
Control	4.43^{a}	19.24^{a}	97.50^{a}	$269.03^{a,b}$	154.85^{a}	2.67^{a}
Conventional						
$CaNO_3$	13.08^{b}	125.87^{c}	$1,092.24^{b}$	325.98^{b}	$1,352.52^{c}$	2.55^{a}
Control	3.41^{a}	30.24^{a}	73.38^{a}	175.70^{a}	130.96^{a}	2.56^{a}

All data were log-transformed for analysis. Significant differences at the 0.05 level (least significant difference) are indicated by different letters within rows. Fall and spring measurements of N_2O and nitrate teaching are for 1 month after fertilization.

than observed rates in most unfertilized systems and similar to observed rates in most fertilized annual cropping systems (12, 23) (for comparison to other systems see Table 5, which is published as supporting information on the PNAS web site). Overall N_2O losses in the month after fertilization were <1% of the applied fertilizer N in both the fall and the spring.

On Nov. 16, 2002, and May 19, 2003, N_2 and N_2O emission rates were simultaneously assessed by using intact cores and acetylene to determine whether soils receiving organic fertilizer amendments showed enhanced N_2 emission rates. N_2 loss rates were significantly higher from both organic treatments compared with the conventional treatment on both dates, mirroring the differences in function seen in the laboratory. Rates of N_2O emissions were similar among all four fertilizer treatments on both dates, implying enhanced denitrification efficiency in the organic and, to a lesser extent, the integrated treatments relative to the conventional (Fig. 2 and Table 3). This difference in efficiency can be attributed to number of factors including (*i*) increased C inputs from grass roots and fertilizer in the organic treatments; (*ii*) higher soil C and N in the organic and integrated treatments than the conventional treatment; (*iii*) larger, more active microbial communities in the organic treatments; and (*iv*) the observed differences in the functioning of the denitrifier communities.

Nitrate leaching was determined by using ion exchange resin bags placed below the rooting zone. Cumulative nitrate leaching was highest from the conventionally fertilized treatment after spring fertilization, followed by the integrated and then the two organic treatments, where leaching rates were not significantly higher than controls. Spring nitrate losses were an order of

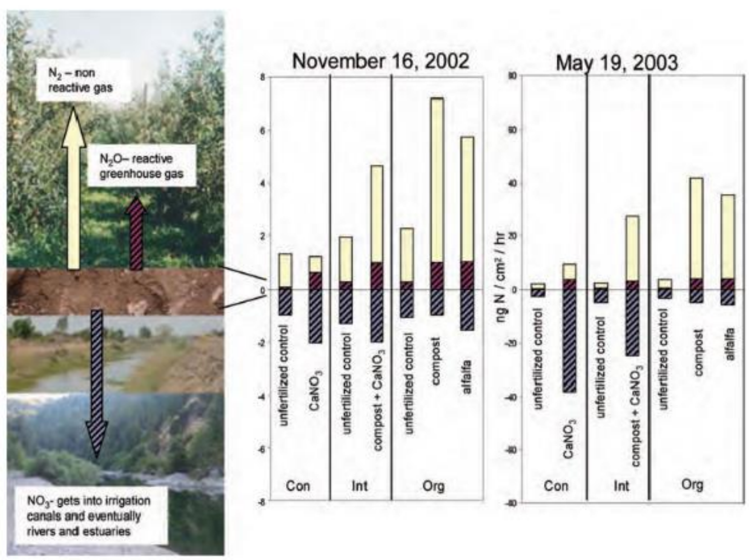


Fig. 2. Relative N loss rates from conventional (Con), integrated (Int), and organic (Org) orchard treatments on Nov. 16, 2002, and May 19, 2003. Nitrate leaching rates are based on the assumption that the 1-cm^2 resin bag absorbed nitrate from a soil column not >10 cm^2 and not <1 cm^2, giving a range for nitrate leaching. The values shown are the average of minimum and maximum leaching rates. See Table 3 for numeric values and statistical significance.

Table 3. Relative daily N loss rates and soil nitrate pools

Treatment and subplots	November 16, 2002				May 19, 2003			
	Nitrate leaching	N_2O emissions	N_2 emissions	Soil nitrate	Nitrate leaching	N_2O emissions	N_2 emissions	Soil nitrate
Organic								
Compost	0.021^{a}	0.024^{b}	0.148^{b}	2.18^{a}	0.118^{a}	0.096^{b}	0.906^{c}	2.18^{a}
Alfalfa	0.038^{a}	0.025^{b}	0.113^{b}	1.40^{a}	0.135^{a}	0.099^{b}	0.753^{c}	2.47^{a}
Control	0.026^{a}	0.006^{a}	$0.048^{a,b}$	1.06^{a}	0.075^{a}	0.014^{a}	0.077^{a}	1.82^{a}
Integrated								
$CaNO_3$ + compost	0.048^{a}	0.024^{b}	$0.088^{a,b}$	6.67^{b}	0.593^{b}	0.079^{b}	$0.577^{b,c}$	$3.18^{a,b}$
Control	0.031^{a}	$0.007^{a,b}$	0.040^{a}	1.14^{a}	0.112^{a}	0.013^{a}	0.049^{a}	0.58^{a}
Conventional								
$CaNO_3$	0.049^{a}	$0.015^{a,b}$	0.015^{a}	8.87^{c}	0.916^{b}	0.095^{b}	$0.133^{a,b}$	5.43^{b}
Control	0.024^{a}	0.002^{a}	$0.030^{a,b}$	1.20^{a}	0.064^{a}	0.004^{a}	0.049^{a}	1.30^{a}

All loss rates are expressed in ng of N per cm^2 per h. Soil nitrate is expressed as kg/ha N (for mean daily nitrate pools for 1 month after fall and spring fertilizations see Fig. 7, which is published as supporting information on the PNAS web site). Nitrate leaching rates are based on the assumption that the 1-cm^2 resin bag absorbed nitrate from a soil column not >10 cm^2 and not <1 cm^2, giving a range for nitrate leaching. The values shown are the average of minimum and maximum leaching rates. All data were log-transformed for analysis. Different superscript letters within rows represent significant differences at $P < 0.05$ (least significant difference).

magnitude higher than losses after fall fertilization (Table 2). This seasonal difference likely resulted from weekly irrigation during the growing season and from build-up of nitrate during the winter, when plant uptake was slow and rainfall was insignificant. Nitrate is a mobile anion that does not bind to soil particles. In this orchard nitrate leaching was highly correlated with soil nitrate pools that were significantly higher in the conventional treatment after fertilization (Fig. 3). Increased water flow and increased mineralization of soil organic matter in response to warm soils during the spring and summer greatly increase the potential for nitrate leaching.

Resin bags remained in the soil for 1 year, allowing for comparison of annual relative nitrate losses. Differences among the treatments in nitrate leaching were most pronounced on an annual basis, with leaching from conventionally fertilized plots 4.4 and 5.6 times higher than that in the two organic treatments, with the integrated treatment in between (Table 2). Lower annual nitrate leaching from the organic and integrated treatments can be explained by a combination of factors that resulted in reduced nitrate levels in the organic treatments and, to a lesser extent, the integrated treatment relative to the conventional treatment. Most importantly, the conventional treatment was

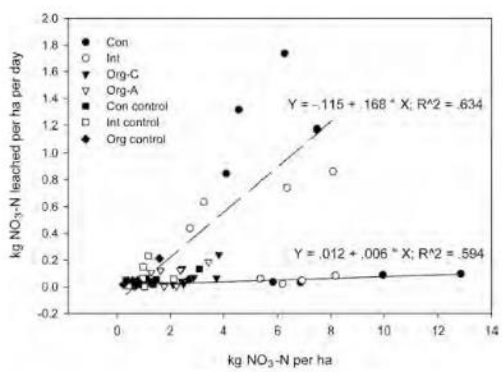


Fig. 3. Relationship between soil nitrate and daily nitrate leaching rate on Nov. 16, 2002 (solid line), and May 19, 2003 (dashed line). Different symbols represent various treatments. Both regressions are significant at $P < 0.0001$.

fertilized with $Ca(NO_3)_2$, whereas the majority of the N applied to the organic treatments and some of the N applied to the integrated treatment during fertilization was incorporated in organic matter, which must be mineralized and nitrified before contributing to the soil nitrate pool. Another factor reducing nitrate losses from the organic treatments was increased denitrification that enhanced gaseous N losses. Although increased grass cover can also reduce leaching losses, the similar amounts of nitrate leached from the control plots (no fertilizer added) in all three treatments (Table 2) indicate that the greater grass cover in organic treatments did not significantly influence nitrate losses.

We determined a conservative range for nitrate leaching rates based on the assumption that the 1-cm^2 resin bag absorbed nitrate from a soil column not >10 cm^2 and not <1 cm^2. Leaching rates in the conventional treatment were of similar magnitude to N_2 emissions rates in either of the two organic treatments at spring sampling (Fig. 2). In this orchard, organic fertilization practices shift the relative balance of gaseous and leaching losses such that proportionately more N is lost as N_2 from the organically farmed soils and as leachate from the conventionally farmed soils.

This study does not establish a direct causal link between enhanced gas emissions and reduced leaching. However, when gas emissions are high as on May 19, 2003, when denitrification as N_2 losses in the organic treatments were equal to 42% of the soil nitrate pool (Table 3), denitrifiers can effectively reduce the amount of nitrate in the soil profile susceptible to leaching. Given the significant correlation ($P < 0.0001$) between the soil nitrate pool and daily nitrate leaching rates on Nov. 16, 2002, and May 19, 2003 (Fig. 3), a substantial reduction in the soil nitrate pool would be expected to reduce nitrate leaching. It is also notable that enhanced gas emissions in the organic treatments occur without a problematic increase in N_2O emissions.

The notion that enhanced gaseous losses may reduce leaching losses is not new. Strategies aimed at mitigation of nitrate leaching often involve the use of wetlands, where rapid denitrification prevents excess nitrate losses, thereby protecting surface water and aquifers (24). Denitrifiers provide an important ecosystem service by removing excess nitrate from terrestrial and aquatic ecosystems (21). Our results suggest that organic, and to a lesser extent integrated, management practices can foster active and efficient denitrifier communities, thereby serving a similar purpose on working farmlands.

SUSTAINABILITY SCIENCE

ECOLOGY

Although not the focus of this study, crop yield is a primary concern for farmers. We assessed crop N status by evaluating leaf N levels at harvest. All four subplot fertilization treatments resulted in statistically similar leaf N levels (Table 2), which were within the critical nutrient range for optimal fruit growth and productivity (25), indicating that organic fertilization, alone or in combination with synthetic fertilizers, may provide a viable alternative to conventional fertilization practices. These results are supported by previous research, which showed that the three apple production systems, when receiving the same amount of N fertilizer, had similar cumulative yields in the first 6 years of the study (18).

Conclusions

This study is significant in that it examines both N_2O and N_2 emissions in the lab and in the field. Although several field studies have compared both gaseous loss pathways after inputs of organic or mineral fertilizer, none has examined the implications of enhanced N_2 from organic systems for nitrate leaching or overall agricultural N pollution to our knowledge. Most comparative studies of N losses from organic and conventional systems are complicated by differences in N application rates and timing among treatments (26) (Table 4). In our study equal amounts of N were simultaneously applied in different forms to each of the treatments. The study was designed to highlight the ecological mechanisms that underlie agricultural N losses from different management systems. Although the mechanisms and processes described in this study are ubiquitous in agroecosystems, actual N losses from organic and conventional farms will vary depending on the specific management and ecology of the system.

The results of our study indicate that use of organic fertilizers in orchards significantly reduces harmful nitrate leaching and enhances denitrifier activity and efficiency. The microbial processes described in these Washington apple orchards operate in all soil ecosystems, and, as such, the observed reductions in environmentally damaging nitrate losses are theoretically achievable in other cropping systems, such as vegetable and grain systems, where denitrifier activity is enhanced through C inputs as organic fertilizers, crop residues, or root exudates from cover crops.

Given the problems associated with global N enrichment caused by agricultural practices, the observed reduction in environmentally damaging nitrate losses from the organic and integrated systems in this study is of important practical significance for both public health and the environment. It is critical for scientists, farmers, and policymakers interested in addressing N pollution problems to look for agricultural systems where reduction in synthetic fertilizer use is possible. Apples and other high-value perennial food crops, which constitute ≈21% of the total value of U.S. food crops (27), are good candidates for consideration because of their reduced N demand.

Materials and Methods

Study Area. The experimental site covered 1.7 hectares (ha) of four replicate plots for each of three apple production systems in a randomized complete block design. The organic treatment followed the U.S. Department of Agriculture National Organic Program (www.ams.usda.gov/nop/NOP/NOPhomeNetscape.html, accessed November 6, 2005) and the Washington State Department of Agriculture Organic Food Program (http://agr.wa.gov/FoodAnimal/Organic/default.htm#OrganicFoodProgram, accessed November 6, 2005) certification guidelines. The conventional treatment followed practices reflecting the management of typical conventional, commercial apple orchards in Washington State. The integrated treatment combined soil, horticultural, and pest management practices from the organic and conventional systems. Details of the experimental design and farming practices are described elsewhere (18, 19). Details of fertilizer management between 1994 and 2003 are shown in Table 6, which is published as supporting information on the PNAS web site.

Soil Analyses. On Aug. 10, 2002, we collected soil samples (0- to 7.5-cm depth) from the 12 organic, integrated, and conventional plots and analyzed them in the laboratory within 3 days for potential denitrification, potential N_2O fluxes, and rN_2O by using the soil slurry method described by Cavigelli and Robertson (28). Nitrification potential was assessed by using the method described by Hart *et al.* (29). Potential assays were used to measure microbial function independent of environmental variability among the treatment soils and can be viewed as a long-term, integrative product of multiple physical and biological factors. We analyzed L-asparaginase and β-glucosidase as described by Tabatabai (30). PLFA analysis was performed according to the techniques described by Bossio *et al.* (31). Details of analytical procedures for soil organic matter, total N, microbial biomass C and N, and mineralizable N are described elsewhere (19).

Fertilization Subplots. Fertilizer recommendations and timing were determined in cooperation with apple orchard managers and professional horticultural consultants in the region to ensure that input rates were reasonably representative of farmer practices. Within each of the 12 plots of the experimental orchard, three 4-m^2 subplots were established and fertilized in different forms according to farm management system at a rate of 67.3 kg·ha^{-1} N on Oct. 22, 2002, and 44.9 kg·ha^{-1} N on May 1, 2003 (Fig. 6, which is published as supporting information on the PNAS web site). These split application rates are typical for young grafted trees in the Yakima Valley area, where the experimental orchard was located. The conventional subplots were fertilized with $Ca(NO_3)_2$; the integrated subplots were fertilized with equal parts composted chicken manure and $Ca(NO_3)_2$; and in the organic plots two separate subplots were established to test two organic fertilizers: composted chicken manure and alfalfa meal. Nitrogen inputs were held constant across treatments to facilitate a mechanistic interpretation of the results without the added complication of differences in input intensity. One unfertilized control subplot was also established in each of the 12 plots to determine baseline N cycling data.

Nitrogen Loss Measurements. We installed cation–anion exchange resin bags at 100-cm depths to estimate relative nitrate mobility in the soil profile as a proxy for nitrate leaching. Nitrate at 100-cm soil depth was used as an estimate of leaching because this depth was below most roots of the trees in the orchard. After removal, we returned resin bags to the laboratory and extracted nitrate by shaking bags in 50 ml of 2 M KCl. We analyzed KCl extracts for inorganic N by using an Alpkem RFA/2. Resin bags remained in the soil from Oct. 19, 2002, through Oct. 19, 2003; we replaced and analyzed them monthly.

We monitored N_2O emissions and soil nitrate availability 10 times from Oct. 19 through Nov. 22, 2002 (fall fertilization), and 5 times from May 1 through June 2, 2003 (spring fertilization). We used static chambers to measure fluxes of N_2O as described by Matson *et al.* (12). After removal of the chamber lid, soil temperature was measured and one soil core (0–15 cm) was removed from within the plot for analysis of inorganic N and soil moisture. Inorganic N was immediately extracted from the soil cores by placing a 10-g subsample of soil in 100 ml of 2 M KCl, shaking for 1 min, and then allowing to equilibrate overnight. The supernatant was removed and stored at 4°C until analysis by using an Alpkem RFA/2. Soil moisture was determined by taking a 10-g subsample from each core, weighing it fresh, and then drying it at 105°C for 2 days before reweighing it to determine gravimetric water content. Cumulative N_2O emissions in the month after fertilization were calculated by extrapolating measured rates during periods between sampling dates.

Overall denitrification and rN_2O were assessed according to Mosier and Klemedtsson (32) by using structurally intact paired cores on Nov. 16, 2002, and May 19, 2003. Two intact cores (0–15 cm) were obtained from each plot, and cores were immediately sealed with a septum-fitted lid for gas analysis. Within 2 h of collection, jars were flushed with air to remove any N_2O accumulated, and then one of the cores from each ring site was injected with CaC_2-generated acetylene to a final volume of 15–20% to block N_2O reductase. The remaining cores were not treated with acetylene, so they could be used to assess N_2O production and rN_2O. Gas samples were collected at 2, 6, and 18 h and stored in Wheaton vials until analysis with a gas chromatograph.

To determine the relative magnitude of the different N loss pathways for the organic, integrated, and conventional plots, N loss rates on Nov. 16, 2002, and May 19, 2003, were converted to ng of N $cm^{-2} h^{-1}$. Resin bags could not be used to precisely quantify nitrate leaching on a per-area basis because it was not possible to determine the exact diameter of the soil column that drains through the bag. For these calculations it was assumed that the 1-cm^2 resin bag absorbed resin from a soil column not >10 cm^2 and not <1 cm^2, giving a range for nitrate leaching (minimum and maximum, respectively).

Leaf Nitrogen. We collected 20 leaf samples at harvest in August 2003 from each of 10 trees per treatment block. Leaves from each treatment were pooled, air-dried, ground up, and analyzed by using an elemental analyzer from Carlo Erba Instruments (Milan).

Statistical Analyses. Soil property, N loss, and leaf measurements for treatments were statistically analyzed by using SAS software (SAS Institute, Cary, NC) for a randomized complete block design. The least significant difference mean separation procedure was used to determine differences at the 0.05 level of significance.

We thank A. Dolph, E. Dolph, and Stemilt Growers, Inc., for the use of their farm; T. Crews, J. Galloway, D. Huggins, L. Klein, R. Koenig, W. Schlesinger, and P. Vitousek for many insightful comments on earlier drafts of the manuscript; P. Matson for providing advice, laboratory space, and equipment for this project; and Z. Moskovitz, A. Sullivan, T. Gagnolet, R. Alldredge, G. Peck, and R. Rupp for technical assistance. This work was supported by the U.S. Department of Agriculture National Research Initiative Competitive Grants Program (Managed Ecosystems), the U.S. National Science Foundation Graduate Research Fellowship Program, The Land Institute Graduate Research Fellowship Program, and the Teresa Heinz Environmental Science and Policy Fellowship Program.

1. Alcamo, J., Ash, N. J., Butler, C. D., Callicott, J. B., Capistrano, D., Carpenter, S. R., Castilla, J. C., Chambers, R., Chopra, K., Cropper, A., *et al.* (2005) *Millenium Ecosystem Assessment: Ecosystems and Human Well-Being: A Framework for Assessment* (Island, Washington, DC).
2. Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W. H., Simberloff, D. & Swackhamer, D. (2001) *Science* **292,** 281–284.
3. Galloway, J., Aber, J., Erisman, J., Seitzinger, S., Howarth, R., Cowling, E. & Cosby, B. (2003) *BioScience* **53,** 341–356.
4. Tilman, D., Cassman, K., Matson, P., Naylor R. & Polasky, S. (2002) *Nature* **418,** 671–677.
5. Environmental Defense Fund (2001) *The Case for Reforming Farm Programs to Preserve the Environment and Help Family Farmers, Ranchers, and Foresters* (Environmental Defense Fund, Washington, DC).
6. Beman, J., Arrigo, K. & Matson, P. (2005) *Nature* **434,** 211–214.
7. Wolfe, A. H. & Patz, J. A. (2002) *Ambio* **31,** 120–125.
8. Nolan, B., Hitt, K. & Ruddy, B. (2002) *Environ. Sci. Technol.* **36,** 2138–2145.
9. Fuhrer, G. J., Morace, J. L., Johnson, H. M., Rinella, J. F., Ebbert, J. C., Embrey, S. S., Waite, I. R., Carpenter, K. D., Wise, D. R. & Hughes, C. A. (2004) *Water Quality in the Yakima River Basin, Washington, 1999–2000,* Circular 1237 (U.S. Geological Survey, Reston, VA).
10. Ramaswamy, V., Boucher, O. & Haigh, J. (2001) in *Climate Change 2001: The Scientific Basis, Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*, eds. Houghton, J. T., Meira Filho, L. G., Bruce, J., Lee, H., Callender, B. A., Haites, E., Harris, N. & Maskell, K. (Cambridge Univ. Press, Cambridge, U.K.), pp. 349–416.
11. Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S. & van Cleemput, O. (1998) *Nutrient Cycling Agroecosystems* **52,** 225–248.
12. Matson, P., Billow, S., Hall, S. & Zachariassen, J. (1996) *J. Geophys. Res.* **101,** 18533–18545.
13. Firestone, M., Firestone, R. & Tiedje, J. (1980) *Science* **208,** 749–751.
14. Weier, K. L., Doran, J. W., Power, J. F. & Walters, D. T. (1993) *Soil Sci. Soc. Am. J.* **57,** 66–72.
15. Ceotto, E. (2005) *Bioresource Tech.* **96,** 191–196.
16. Drinkwater, L., Wagoner, P. & Sarrantonio, M. (1998) *Nature* **396,** 262–265.
17. Nosengo, N. (2003) *Nature* **425,** 894–895.
18. Reganold, J., Glover, J., Andrews, P. & Hinman, H. (2001) *Nature* **410,** 926–930.
19. Glover, J., Reganold, J. & Andrews, P. (2000) *Agric. Ecosystem Env.* **80,** 29–45.
20. Mader, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P. & Niggli, U. (2002) *Science* **296,** 1694–1697.
21. Ewel, K. C. (1997) in *Nature's Services: Societal Dependence on Natural Ecosystems*, ed. Daily, G. (Island, Washington, DC), pp. 329–344.
22. Groffman, P. & Tiedje, J. (1989) *Soil Biol. Biochem.* **21,** 621–626.
23. Roelandt, C., Van Wesemael, B. & Rounsevell, M. (2005) *Global Change Biol.* **11,** 1701–1711.
24. Jordan, T., Correll, D. & Weller, D. (1993) *J. Environ. Qual.* **22,** 467–473.
25. Gavlak, R. G., Horneck, D. A. & Miller, R. O. (1994) *Plant, Soil, and Water Reference Methods for the Western Region*, Western Regional Extension Publication 125 (Univ. of Alaska Cooperative Extension Service, Fairbanks).
26. Kirchmann, H. & Bergstrom, L. (2001) *Commun. Soil Sci. Plant Anal.* **32,** 997–1028.
27. U.S. Department of Agriculture National Agricultural Statistics Service (2004) *Agricultural Statistics 2004* (U.S. Government Printing Office, Washington, DC).
28. Cavigelli, M. & Robertson, G. P. (2000) *Ecology* **81,** 1402–1414.
29. Hart, S. C., Stark, J. M., Davidson, E. A. & Firestone, M. K. (1994) in *Methods of Soil Analysis Part II: Microbiological and Biochemical Properties*, eds. Weaver, R. W., Angle, J. S. & Bottomley, P. J. (Soil Sci. Soc. Am., Madison, WI), pp. 985–1018.
30. Tabatabai, M. A. (1994) in *Methods of Soil Analysis Part II: Microbiological and Biochemical Properties*, eds. Weaver, R. W., Angle, J. S. & Bottomley, P. J. (Soil Sci. Soc. Am., Madison, WI), pp. 775–853.
31. Bossio, D., Scow, K., Gunapala, N. & Graham, K. (1998) *Microbial Ecol.* **36,** 1–12.
32. Mosier, A. R. & Klemedtsson, L. (1994) in *Methods of Soil Analysis Part II: Microbiological and Biochemical Properties*, eds. Weaver, R. W., Angle, J. S. & Bottomley, P. J. (Soil Sci. Soc. Am., Madison, WI), pp. 1047–1065.

SUSTAINABILITY SCIENCE

ECOLOGY

Attachment 15

Agriculture, Ecosystems and Environment 164 (2013) 32–52

Contents lists available at SciVerse ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Review

The potential of organic fertilizers and water management to reduce N_2O emissions in Mediterranean climate cropping systems. A review

Eduardo Aguilera[a,b,*], Luis Lassaletta[c,e], Alberto Sanz-Cobena[d], Josette Garnier[e], Antonio Vallejo[d]

[a] Spanish Society of Organic Farming (SEAE), Camí del Port, S/N. Edif ECA Pat Int 1º - km 1 (Ap 397) 46470 Catarroja, Valencia, Spain
[b] Universidad Pablo de Olavide, Ctra. de Utrera, km. 1, 41013, Sevilla, Spain
[c] Department of Ecology, Universidad Complutense de Madrid, c/José Antonio Novais s/n 28040 Madrid, Spain
[d] Escuela Técnica Superior de Ingenieros Agrónomos, Universidad Politécnica de Madrid, Ciudad Universitaria, 28040 Madrid, Spain
[e] UPMC/CNRS, UMR Sisyphe, Box 123, 4, Place Jussieu, 75005 Paris, France

ARTICLE INFO

Article history:
Received 21 September 2011
Received in revised form 5 July 2012
Accepted 11 September 2012

Keywords:
Nitrous oxide
Mediterranean cropping systems
Synthetic fertilizer
Organic fertilizer
Irrigated crops
Rainfed crops

ABSTRACT

Environmental problems related to the use of synthetic fertilizers and to organic waste management have led to increased interest in the use of organic materials as an alternative source of nutrients for crops, but this is also associated with N_2O emissions. There has been an increasing amount of research into the effects of using different types of fertilization on N_2O emissions under Mediterranean climatic conditions, but the findings have sometimes been rather contradictory. Available information also suggests that water management could exert a high influence on N_2O emissions. In this context, we have reviewed the current scientific knowledge, including an analysis of the effect of fertilizer type and water management on direct N_2O emissions.

A meta-analysis of compliant reviewed experiments revealed significantly lower N_2O emissions for organic as opposed to synthetic fertilizers (23% reduction). When organic materials were segregated in solid and liquid, only solid organic fertilizer emissions were significantly lower than those of synthetic fertilizers (28% reduction in cumulative emissions). The EF is similar to the IPCC factor in conventionally irrigated systems (0.98% N_2O-N N applied^{-1}), but one order of magnitude lower in rainfed systems (0.08%). Drip irrigation produces intermediate emission levels (0.66%). Differences are driven by Mediterranean agro-climatic characteristics, which include low soil organic matter (SOM) content and a distinctive rainfall and temperature pattern. Interactions between environmental and management factors and the microbial processes involved in N_2O emissions are discussed in detail.

Indirect emissions have not been fully accounted for, but when organic fertilizers are applied at similar N rates to synthetic fertilizers, they generally make smaller contributions to the leached NO_3^- pool. The most promising practices for reducing N_2O through organic fertilization include: (i) minimizing water applications; (ii) minimizing bare soil; (iii) improving waste management; and (iv) tightening N cycling through N immobilization. The mitigation potential may be limited by: (i) residual effect; (ii) the long-term effects of fertilizers on SOM; (iii) lower yield-scaled performance; and (iv) total N availability from organic sources. Knowledge gaps identified in the review included: (i) insufficient sampling periods; (ii) high background emissions; (iii) the need to provide N_2O EF and yield-scaled EF; (iv) the need for more research on specific cropping systems; and (v) the need for full GHG balances.

In conclusion, the available information suggests a potential of organic fertilizers and water-saving practices to mitigate N_2O emissions under Mediterranean climatic conditions, although further research is needed before it can be regarded as fully proven, understood and developed.

Contents

* Corresponding author at: Universidad Pablo de Olavide, Geography, Carretera de Utrera Km 1, 41013 Sevilla, Spain. Tel.: +34 954978136; fax: +34 954348359.
E-mail address: emagufer@upo.es (E. Aguilera).

0167-8809/$ – see front matter © 2012 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.agee.2012.09.006

1. Introduction

Nitrous oxide (N_2O) is a powerful greenhouse gas (GHG). It is 298 times stronger than CO_2 at the 100-year time horizon and in 2005, it accounted for 6.1% of combined GHG radiative forcing (Forster et al., 2007). According to the cited authors, the atmospheric N_2O concentration rose from 270 to 319 ppb in the period 1900–2005, after having previously remained relatively stable for the previous two millennia. Agricultural emissions represent about 60% of global anthropogenic N_2O emissions. They increased by 17% from 1990 to 2005 and are projected to increase by 35–60% up to 2030 (Smith et al., 2007).

Nitrogen fertilizer applications to soils, whether organic or synthetic, result in N_2O emissions, as this gas is a by-product of the transformation of N compounds added to the soil. N_2O fluxes from soil are mainly driven by microbial activity, through nitrification and denitrification processes (Firestone and Davidson, 1989). In spite of the existence of a large body of knowledge on the mechanisms that underlie these pathways, there is still an insufficient understanding of the finer details of the process, such as how the composition of organic N fertilizers affects denitrification, nitrification and emission rates (Vallejo et al., 2006).

Besides soil emissions after fertilizer applications (direct emissions), fertilizer-related N_2O production can also result from indirect emissions. Downstream of the cropping system, N_2O is produced when N compounds, and particularly leached nitrate (NO_3^-) and volatilized ammonia (NH_3), are subsequently transformed into N_2O (IPCC, 2006a). These indirect sources can represent a significant fraction of total agricultural N_2O emissions (Garnier et al., 2009). Upstream of the cropping system, N_2O and other GHG are emitted as by-products of fertilizer production, storage and transport (Snyder et al., 2009). Although these emissions are very dependent on the methods used to obtain fertilizers, half of synthetic N fertilizer-related GHG emissions could occur in the production phase, whereas the other half occurs from the soil (Tirado et al., 2010). In 2001, fertilizer production accounted for 1% of the global energy demand; 72% of this energy corresponded to N, and a further 16% to compound fertilizers containing N (Ramírez and Worrell, 2006).

There is increasing interest in the application of organic fertilizers to soils (e.g., Hargreaves et al., 2008; Petersen et al., 2003; Singh and Agrawal, 2008; Smil, 1999), as they can contribute to climate change mitigation through C sequestration (Diacono and Montemurro, 2010), at the same time helping to tackle problems associated with waste management and meeting the nutrient and organic matter needs of agricultural soils (Tirado et al., 2010). Nevertheless, the use of organic fertilizers also has a number of drawbacks, which include the energy costs associated with transport and the land spreading of the fertilizers (Wiens et al., 2008), potential pollution with heavy metals and other toxic substances (Petersen et al., 2003), the availability of organic N sources, and GHG emissions (Snyder et al., 2009).

The type and composition of fertilizers have been shown to affect direct N_2O emissions from cropped soils (Stehfest and Bouwman, 2006), although the differences between applying organic and synthetic fertilizers are still not clear. For example, Laegreid and Aastveit (2002) analyzed several databases and found higher direct N_2O emissions from manure than from mineral fertilizers. Although there is great uncertainty in the estimation of the effect of fertilization type on indirect N_2O emissions, overall emissions could actually be slightly lower for organic fertilizers, as calculated in a top–down analysis (Davidson, 2009).

These information gaps are especially relevant for Mediterranean-type cropping systems, where an increasing

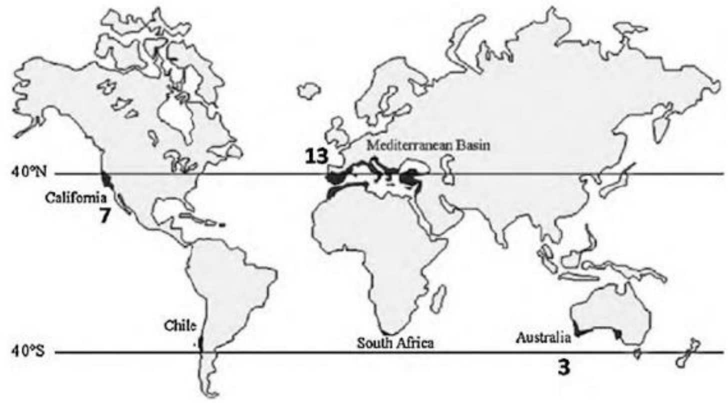


Fig. 1. Regions of the world with a Mediterranean climate and number of papers measuring field N_2O emissions in each region.

body of knowledge is being built. However, the data involved are rather dispersed in the existing literature and no compilations have yet been made. Most N_2O emission studies have been conducted under temperate climatic conditions, but ecological processes in regions with Mediterranean climates are affected in different ways to those subject to other climatic conditions, as has been shown in various fields of research including plant physiology (González-Fernández et al., 2010), N biogeochemistry (Breiner et al., 2007), limnology (Álvarez-Cobelas et al., 2005) and biodiversity (Barriga et al., 2010). Aschmann (1973) defined areas with Mediterranean climates as those in which at least 65% of annual rainfall occurs in winter and in which annual precipitation ranges from 275 to 900 mm. The average winter temperature is below 15 °C, but the number of hours per year with temperatures below 0 °C does not exceed 3% of the total. This climate is characterized by seasonal dryness and many of its subtypes could be classified as semi-arid. This climate type is found in five different parts of the world; they are generally on the west coasts of continents and between latitudes 32° and 40° north and south of the equator. These areas are: the Mediterranean basin; California; Central Chile; the Cape region of South Africa; and South and South-West Australia (Fig. 1). The diversity of soil types is very wide in areas with Mediterranean climates on account of their extensions, variety of geological origins and different land uses. The only common feature shared by these soils is their low organic matter content, while they also often contain low levels of mineral nutrients. The Mediterranean biome is highly biodiverse, but subject to extremely intense development pressure (Underwood et al., 2009). Large increases in yield have been achieved through the intensification of agricultural practices in areas with Mediterranean climates (e.g. Ryan et al., 2009). However, the associated agronomic practices are now undermining both soil and water quality (Zalidis et al., 2002) and are also affecting the natural biodiversity through, for example, N deposition in ecosystems (Ochoa-Hueso et al., 2011).

Research into the effects of N fertilization on N_2O fluxes under Mediterranean conditions has yielded somewhat contradictory results. Moreover, IPCC Tier 1 N_2O emission factors (EF) are mainly based on temperate climate data, and Leip et al. (2011) have recently emphasized the need of more specific EF considering particular parameters such as fertilizer type or climate. The development of Mediterranean climate-specific information on N_2O emission would therefore greatly improve the accuracy of national GHG inventories for these regions. In this context, we have reviewed and analyzed the currently available scientific information associated with fertilizer-related N_2O emissions under Mediterranean climatic conditions, with the following objectives:

(1) to compare and contrast direct N_2O emissions from the application of organic and synthetic fertilizers, describing the influence of agricultural practices and environmental factors
(2) to get an overview of indirect N_2O sources related to fertilizer use, both upstream and downstream of the cropping system
(3) to identify options for mitigating N_2O emissions, and their respective drawbacks, through the application of organic fertilizers in these agroecosystems and to detect the main gaps in currently available information.

2. Methods

A wide review was performed which included every paper containing data on N_2O emissions in Mediterranean agroecosystems. Articles were consulted on the ISI Web of Knowledge database by simultaneously typing the words "nitrous oxide" or "N_2O", "emission", and either the word "Mediterranean" or the name of a country with territory in areas with a Mediterranean climate, according to the map in Fig. 1. This search based on keywords was complemented with a search through the literature cited in the articles found.

N_2O emission data have been expressed as cumulative emissions and EF. Cumulative emission data refer to the sum of N_2O fluxes over the reported measurement period. For measurements covering more than 1 year, the values cited were converted to 1-year references. In some cases, cumulative emissions were estimated through the interpolation of emission values obtained from the figures provided in these papers. Figures were digitized using GetData v.2.24 software (Fedorov, 2002). N_2O EF refers to the proportion of fertilizer N that is released as N_2O-N during the measurement period after discounting emissions from an unfertilized control treatment (equation (1)). Studies covering less than a full growth season were excluded from EF analyses. In many cases, measurement period was extended for a full growth season or more, but was shorter than 1 year; therefore the annual EF may be underestimated. In order to explore different methodologies, we estimated an additional EF (EF_{annual}) and cumulative emissions by linearly extrapolating average N_2O emission levels during sampling period to the rest of the year.

$$EF = \frac{\text{kg } N_2O\text{-N(F)} - \text{kg } N_2O\text{-N(C)}}{\text{kg N fertilizer applied}} \quad (1)$$

where EF is the emission factor (N_2O-N emitted as % of fertilizer-N applied) and N_2O-N (F) and N_2O-N (C) are the reported cumulative N_2O emissions (kg N ha^{-1} yr^{-1}) from the fertilized and control (unfertilized) treatments, respectively. N fertilizer applied is the rate of N applied during the study (kg ha^{-1} yr^{-1}). For slow N release organic amendments (i.e. solid organic fertilizers), "available N" was preferentially chosen as the "N fertilizer applied" value instead of the total N content in the fertilizer; this was in line with the procedure followed by most of the authors whose work was reviewed. The "available N" approach takes into account the fact that only a fraction of the N contained in organic fertilizers mineralizes during the measurement period. The EF for organic fertilizers obtained applying this approach is therefore greater than that obtained with the total N approach recommended by the IPCC (2006a). However, this avoids any possible underestimation of the EF associated with the residual effect of solid organic fertilizers, which is particularly useful for short sampling periods (although this assumption is open to criticism, see Section 6.1). EF based on total N applied was also calculated for some analyses in order to compare the results obtained with the available N method and then discuss them.

In a first approach, the effect of fertilizer type on cumulative emissions of N_2O and EF was studied using a general matrix containing the results of all the publications reviewed. Fertilizer type was grouped at four levels: (i) synthetic, including urea, ammonium nitrate, ammonium sulfate and NPK compound fertilizers; (ii) organic (solid), including residues of cover crops (legumes and non-legumes), organic manure, composted municipal solid waste, composted cattle and sheep manure, and composted thick fractions of digested pig slurries; (iii) organic (liquid), including raw or digested pig slurries; (iv) organic + synthetic, including mixtures of organic and synthetic sources of N, except when the organic N was only represented by residues of the previous crop, in which case the treatment was classified as "synthetic".

The influence of water management on N_2O emissions was studied by classifying this factor in three categories: (i) rainfed systems, in which no irrigation water was applied; (ii) high-water systems, including furrow, sprinkler and micro-sprinkler irrigation; (iii) low-water systems, including surface and subsurface drip irrigation techniques.

We have observed a high degree of variability in the published N_2O emissions associated with organic and synthetic fertilization. Therefore, in a second approach, the dataset was further narrowed down by restricting the data used to pairwise comparisons of field emissions from organic and synthetic fertilizers in order to examine them meta-analytically. Weighted meta-analysis requires information on variance and the number of replications of each treatment; studies that did not report these data were therefore also excluded. These pairwise comparisons represent experiments where synthetic and organic fertilizers are compared under similar agro-climatic conditions and application rates. In many occasions, multiple organic fertilizers were compared to a common control (synthetic) group. Therefore, in order to avoid an over-estimation of the precision of the mean effect size (Borenstein et al., 2009; Hungate et al., 2009), the resulting pairs were combined into one composite effect size, which variance is given by equation (2).

$$\text{var}\left(\frac{1}{m}\sum_{i=1}^{m} Y_i\right) = \left(\frac{1}{m}\right)^2 \left(\sum_{i=1}^{m} V_i + \sum_{i \neq j}(r_{ij} \times \sqrt{V_i} \times \sqrt{V_j})\right) \quad (2)$$

where m represents the number of correlated pairs, Y_i and V_i are the effect size and the variance of pair i, and r is the correlation coefficient, which is equal to 0.5 because all pairs share a common control.

No EF standard deviations (SD_{EF}) were provided in any of the reviewed papers, so they were calculated from cumulative emission standard deviations and sample sizes, following equation (3).

$$SD_{EF} = \frac{\sqrt{(n_F - 1) \times SD_F^2 + (n_C - 1) \times SD_C^2 / (n_F + n_C - 2)}}{\text{kg N fertilizer applied}} \quad (3)$$

where n_F and n_C are the number of observations in the fertilized and control (unfertilized) treatments, respectively. SD_F and SD_C are the standard deviations in the fertilized and control treatments, respectively.

We chose the response ratio (RR) as the effect size unit for both cumulative emissions and EF data. This RR-value is the ratio between some measured quantity in the experimental (organic in our case, $\bar{X}_{Org}$) and control (synthetic, $\bar{X}_{Syn}$) groups (RR = $\bar{X}_{Org}/\bar{X}_{Syn}$). We used the natural log of RR (L_i) to perform the analysis (equation (4) for its mean, and equation (5) for its variance), because this transformation results in a much more normal sampling distribution in small samples (Hedges et al., 1999).

$$L_i = \ln(\bar{X}_{Org}) - \ln(\bar{X}_{Syn}) \quad (4)$$

$$\text{var}_i = \frac{(SD_{Org})^2}{n_{Org} \times \bar{X}_{Org}} + \frac{(SD_{Syn})^2}{n_{Syn} \times \bar{X}_{Syn}} \quad (5)$$

where SD is the standard deviation and n the number of observations in the organic and synthetic groups. Individual effect sizes were pooled together into a common effect size following a random effects model; this is appropriate for situations in which individual studies are not projected to share a common effect size. This weighted mean effect size and its related statistical parameters were calculated using Comprehensive Meta-Analysis software (Borenstein and Rothstein, 1999). The weighted mean L_i and their variances were transformed back in the presentation of the results.

In order to summarize all the data reviewed, information on cumulative emissions and EF grouped by other factors was also provided: N fertilizer rate, crop type and tillage. Some treatments were excluded from all the analyses: (i) rice paddies, because only one measurement was available (Skiba et al., 2009) and the conditions for N_2O production are very different from those of the rest of the crop types; (ii) experiments employing chemicals or additives such as nitrification inhibitors, because their use is limited on a global scale (Stehfest and Bouwman, 2006); (iii) treatments with N fertilizer rates of more than 300 kg N ha^{-1} yr^{-1} of available N, or 500 kg N ha^{-1} yr^{-1} of total N (solid organic fertilizers), as higher rates are not considered representative of typical Mediterranean agroecosystems. We only found one such treatment (in Heller et al., 2010); (iv) studies published before 2000, in order to assure a relatively homogeneous measurement methodology. This implied the exclusion of only one paper (Ryden and Lund, 1980a).

3. Direct N_2O emissions in Mediterranean cropping systems

3.1. General overview of the papers reviewed

Published field research into N_2O emissions from Mediterranean agricultural systems began three decades ago in the Santa Maria Valley, Santa Barbara, California (Ryden and Lund, 1980a), although this pioneering study was not followed by any others until very recently, when research results from the La Poveda research station in Central Spain were published (Vallejo et al., 2005). Over the short period from 2005 to mid-2011, a valuable body of knowledge has been acquired. 24 field experiments are now available

E. Aguilera et al. / Agriculture, Ecosystems and Environment 164 (2013) 32–52

Table 1
Characteristics of field experiments measuring N_2O emissions in Mediterranean cropping systems.

Reference	Study site	Crop type	Fertilizer[2]	Irrigation[3]	Duration (days)	Soil type	Observations
Barton et al. (2011)	Cunderdin, SW Australia	Lupin	Legume CC, none	Rainfed	350	Natric Haploxeralf and Typic Natrixeralf	
Barton et al. (2010)	Cunderdin, SW Australia	Canola	U, none	Rainfed	360	Natric Haploxeralf and Typic Natrixeralf	
Barton et al. (2008)	Cunderdin, SW Australia	Wheat	U, none	Rainfed	365	Natric Haploxeralf and Typic Natrixeralf	
Burger et al. (2005)	Russell Ranch (LTRAS-CIFS[1]), Davis, California, USA	Tomato	Compost, CC, synthetic fertilizers	Furrow		Typic Xerothent and Mollic Haploxeralf	Not included (no data on cumulative emissions)
De Gryze et al. (2010)	Central Valley, California, USA	Various rotations	Compost, legume CC, synthetic fertilizers	Furrow, rainfed			Not included (only simulated N_2O emission reported). Tillage comparisons
Garland et al. (2011)	Arbuckle, California, USA	Vineyard, legume mix (cover crop)	Legume CC, U-AN	Drip (vineyard), rainfed (CC)	194	Willows silty clay	Tillage comparisons
Heller et al. (2010)	Volcani, Israel	Maize	NPK, chicken manure	Drip	365	Typic Rhodoxeralf	One treatment with very high N fertilization (not included). Tillage comparisons
Kallenbach et al. (2010)	Russell Ranch, Davis, California, USA	Tomato	LCC, NPK, AS	Furrow, drip	365	Reiff loam and Yolo silt loam	
Kong et al. (2007)	Russell Ranch (LTRAS-CIFS), Davis, California, USA	Maize	Compost, legume CC, U-AS	Furrow	142	Typic Xerothent and Mollic Haploxeralf	Not included (same experiment as Kong et al., 2009)
Kong et al. (2009)	Russell Ranch (LTRAS-CIFS), Davis, California, USA	Maize	Compost, legume CC, U-AS	Furrow	142	Typic Xerothent and Mollic Haploxeralf	Tillage comparisons
Lee et al. (2009)	Yolo County, California, USA	Maize-Sunflower-Chickpea	U-AN, NPK	Furrow	912.5	Myers clay (Entic Chromoxererts)	Tillage comparisons
López-Fernández et al. (2007)	La Poveda, Madrid, Spain	Maize	MSW compost, sheep manure compost, PS, U, none	Furrow	200	Typic Xerofluvent	
Lugato et al. (2010)	Beano, Udine, Italy	Maize	Not specified	Irrigation	1095	Chromi-Endoskeletic Cambisol	Not included (only simulated N_2O emission reported)
Meijide et al. (2007)	La Poveda, Madrid, Spain	Maize	PS, DPS, DPS compost, MSW compost, U, none	Furrow	145	Typic Xerofluvent	Nitrification inhibitor used in some treatments (not included)
Meijide et al. (2009)	El Encín, Madrid, Spain	Barley	PS, DPS, SS compost, MSW, U, none	Rainfed	335	Calcic Haploxerepts (USDA) – Calcaric Cambisol (FAO)	
Menéndez et al. (2008)	El Malagón, Córdoba, Spain	Wheat-Sunflower-Faba bean	AN, none	Rainfed	22, 29	Vertisol (Typic Haploxerert)	Not included (short measurement period). Tillage comparisons
Petersen et al. (2006)	Reggio Emilia, Italy	Wheat-Alfalfa-Maize-Grass	Manure, legume CC, synthetic fertilizers	Irrigation	365		
Ryden and Lund (1980a)	Santa Maria Valley, Santa Barbara, California, USA	Horticultural crops	Manure, legume CC, NPK	Furrow	365		Not included (old study)
Sánchez-Martín et al. (2008b)	El Encín, Madrid, Spain	Melon	AN, none	Drip, furrow	140	Calcic Haploxerepts (USDA) – Calcaric Cambisol (FAO)	
Sánchez-Martín et al. (2010a)	El Encín, Madrid, Spain	Melon	DPS, none	Drip, furrow	150	Calcic Haploxerepts (USDA) – Calcaric Cambisol (FAO)	

E. Aguilera et al. / Agriculture, Ecosystems and Environment 164 (2013) 32–52 37

Table 1 (*Continued*)

Reference	Study site	Crop type	Fertilizer[2]	Irrigation[3]	Duration (days)	Soil type	Observations
Sánchez-Martín et al. (2010b)	El Encín, Madrid, Spain	Onion	Manure, DPS, none	Micro-sprinkler	365	Calcic Haploxerepts (USDA) – Calcaric Cambisol (FAO)	
Skiba et al. (2009)	Castellaro, Italy	Rice, Fennel and Maize	Not specified	Irrigation	365	Not specified	Not included (N_2O emission reported only for rice paddies)
Steenwerth and Belina (2008)	Monterey, California, USA	Vineyard cover crop	CC, none	Rainfed	365	Cumulic Haploxeroll, or Haplic Chenozem	Tillage comparisons
Steenwerth and Belina (2010)	Monterey, California, USA	Vineyard	U-AN, CC	Drip	33.3	Cumulic Haploxeroll, or Haplic Chenozem	Not included (short measurement period). Tillage comparisons
Townsend-Small et al. (2011)	Irvine and Popoma, California, USA	Maize, horticulture	AP (maize), AN (horticulture)	Rainfed	365	Sorrento loam and clay loam	
Vallejo et al. (2005)	La Poveda, Madrid, Spain	Maize	PS, none	Furrow	215	Typic Xerofluvent	Nitrification inhibitor used in some treatments (not included)
Vallejo et al. (2006)	El Encín, Madrid, Spain	Potato	PS, DPS, DPS compost, MSW, U, none	Furrow	150	Calcic Haploxerepts (USDA) – Calcaric Cambisol (FAO)	Nitrification inhibitor used in some treatments (not included)

[1] LTRAS-CIFS: Center for Integrated Farming Systems, formerly known as the long-term research on agricultural systems experiment.
[2] Organic fertilizers (solid). CC: cover crops; MSW: municipal solid wastes; SS: sewage sludge; DPS compost: composted thick fraction of digested pig slurry; Organic fertilizers (liquid); PS: raw pig slurry; DPS: digested pig slurry (thin fraction). Synthetic fertilizers. U: urea; AN: ammonium nitrate; NPK: compound fertilizers; AS: ammonium sulfate; AP: ammonium phosphate.
[3] Rainfed: no irrigation water is applied; high-water: furrow and micro-sprinkler irrigation; low-water: drip (surface or subsurface) irrigation.

in the scientific literature, which relate to 13 different study sites (Table 1). These relate to three of the five areas in the world that have Mediterranean climates. All the field measurements of N_2O emissions were performed using closed chambers, with a sampling frequency of usually between 1 day and 2 weeks, which usually increased after important management events such as fertilization, irrigation or significant rainfall. The measurement frequency was highest in the Australian studies (Barton et al., 2008, 2010, 2011), which used soil chambers connected to a fully automated system and collected samples every 180 min. These field data were complemented by 11 laboratory experiments that addressed N_2O emissions from Mediterranean soils, or under specific Mediterranean conditions, such as the addition of organic matter sources from Mediterranean cropping systems (olive residues) (García-Ruiz and Baggs, 2007).

3.2. Factors influencing N_2O emissions

3.2.1. General effects of fertilizer type and water management

Emissions were highest for slurries ("liquid organic fertilizers", 4.4 kg N_2O-N ha^{-1} yr^{-1} on average, Table 2, Fig. 2a), followed by organic–synthetic mixtures and synthetic fertilizers (respectively, 3.5 and 3.0 kg N_2O-N ha^{-1} yr^{-1}), whereas solid organic fertilizers and unfertilized treatments ("none" group) showed the lowest values (1.7 and 1.8 kg N_2O-N ha^{-1} yr^{-1}, respectively).

Despite the high degree of variability in the treatments and conditions included in this analysis, a clear influence of water management on cumulative emissions and EF could be observed in both cases (Fig. 3). Emission levels in rainfed treatments were one order of magnitude lower than those in conventionally irrigated ones (mean cumulative emissions were 0.4 and 4.0 kg N_2O-N ha^{-1} yr^{-1} and average EF values were 0.08% and 1.01% for rainfed and high-water irrigation treatments, respectively; Tables 2 and 3). Drip irrigation (low-water category) showed intermediate distributions in emission levels (the averaged cumulative emissions and EF in this category were 1.2 kg N_2O-N ha^{-1} yr^{-1} and 0.66%).

The high influence of irrigation can be explained by the fact that in Mediterranean agroecosystems this type of management activity is usually applied during the summer dry period, which leads to optimal moisture and temperature conditions for N_2O production (Section 3.3). In this way, average cumulative N_2O emissions for Mediterranean cropping systems irrigated with conventional techniques were normally slightly higher than the cumulative emissions for high N application rates (200–250 kg ha^{-1}) obtained from the global data compiled by Stehfest and Bouwman (2006), whereas average EFs were generally similar to the default IPCC factor of 1%. Other works have also shown that when semi-arid soils are irrigated, an increase in N_2O emissions can be expected (Horvath et al., 2010; Maraseni et al., 2010). In rainfed systems, different limitations on N_2O production are imposed throughout the cropping cycle, especially when the semi-arid conditions are marked. This result was very much in line with low N_2O emissions linked to fertilizer application reported for semi-arid agroecosystems under different climatic conditions (Dick et al., 2008; Galbally et al., 2008). What limited information is available about drip-irrigated systems suggests intermediate emission levels, which points to a high potential for N_2O mitigation when using this water-saving technique (Section 5.1). The N_2O mitigation effect of applying low-water irrigation techniques also applies to other world climate types, e.g., under temperate continental (Liu et al., 2011) and arid continental climates (Scheer et al., 2008).

3.2.2. Meta-analysis of the fertilizer type effect

N_2O emission levels from organic fertilizers were in many cases lower than those from synthetic fertilizers when compared on a pair-wise basis, even if the variability was sometimes very

Table 2
Number of observations (N), mean and standard deviation (SD) of cumulative N_2O emissions (kg N_2O-N ha^{-1} yr^{-1}) for some of the factors with a significant influence on N_2O emissions from agricultural fields.

	N	Cumulative emissions (experiment period)		Cumulative emissions (year estimate)		Mean N fertilizer rate (kg N ha^{-1} yr^{-1})	Mean experiment length (days)
		Mean	SD	Mean	SD		
Fertilizer type							
None	18	1.8	1.8	3.9	4.2	0	242
Organic + synthetic	11	3.5	3.3	6.6	8.0	171	234
Synthetic	22	3.0	2.6	5.2	5.9	159	288
Organic (solid)	17	1.7	1.9	3.1	4.3	147	263
Organic (liquid)	16	4.4	3.0	9.0	6.7	163	220
Water management type							
Rainfed	19	0.4	0.5	0.6	1.0	57	316
High-water	54	4.0	2.6	7.8	6.3	137	231
Low-water	11	1.2	1.0	2.0	1.7	128	254
N fertilizer rate (kg N ha^{-1} yr^{-1})							
0–75	25	1.4	1.7	2.9	3.9	11	248
75–150	16	1.2	1.1	1.2	1.1	113	356
150–225	29	5.3	2.2	11.2	6.0	178	192
225–500	6	2.9	3.0	3.8	3.0	265	291
Crop type							
Horticulture	21	1.9	1.5	3.4	3.4	100	281
Maize	26	4.5	3.2	9.0	7.4	176	207
Other	12	4.0	2.3	8.0	6.3	113	257
Winter cereals	8	0.3	0.1	0.3	0.1	91	343
Vineyard	5	0.4	0.3	0.4	0.2	35	297
Legume	5	0.7	0.9	0.7	1.8	0	220
None	7	3.2	1.5	5.8	2.7	121	221
Tillage type							
Minimum tillage	10	1.9	2.6	2.7	2.8	133	246
Standard tillage	9	1.1	1.4	1.6	1.4	111	232

N fertilizer rate refers to N applied during the experimental period. In the case of solid organic fertilizers this value corresponds to available N in the experimental period, not to total N applied.

high. The mean response ratio for cumulative emissions was 0.77 ($p < 0.05$), meaning that average emission levels were 23% lower for organic than for synthetic fertilizers (Fig. 4a). When organic fertilizers were segregated in solid and liquid ones, only organic solid fertilizer emissions were significantly lower than synthetic (RR = 0.72, $p < 0.05$). An EF comparison showed a 23% reduction for organic fertilizers (RR = 0.77, $p = 0.08$), but the differences were only significant for solid materials (RR = 0.8, $p < 0.05$) (Fig. 4b).

Table 3
Number of observations (N), mean and standard deviation (SD) of emission factor (EF, %N_2O-N over N applied) for some of the factors that have a significant influence on N_2O emissions from agricultural fields.

	N	EF (experient period)		EF (year estimative)		Mean N fertilizer rate (kg N ha^{-1} yr^{-1})	Mean experiment length (days)
		Mean	SD	Mean	SD		
Fertilizer type							
Organic + synthetic	3	1.22	0.62	3.04	1.57	175	147
Synthetic	11	0.82	0.73	1.71	1.79	142	251
Organic (solid)	9	0.54	0.48	0.97	1.17	147	261
Organic (liquid)	16	0.91	0.70	1.75	1.34	163	220
Water management type							
Rainfed	7	0.08	0.04	0.09	0.05	114	343
High-water	30	1.01	0.63	2.02	1.48	162	213
Low-water	2	0.66	0.33	1.65	0.75	175	145
N fertilizer rate (kg N ha^{-1} yr^{-1})							
0–75	1	0.06		0.06		75	360
75–150	12	0.36	0.31	0.37	0.30	115	353
150–225	26	1.06	0.66	2.31	1.42	175	173
Crop type							
Horticulture	10	0.67	0.31	1.10	0.96	136	277
Maize	12	1.33	0.70	2.74	1.34	177	180
Other	6	1.08	0.71	2.61	1.75	158	185
Winter cereals	6	0.09	0.05	0.10	0.05	121	340
None	5	1.08	0.35	0.91	0.64	170	200

N fertilizer rate refers to N applied during the experimental period. In the case of solid organic fertilizers this value corresponds to available N in the experimental period, not to total N applied.

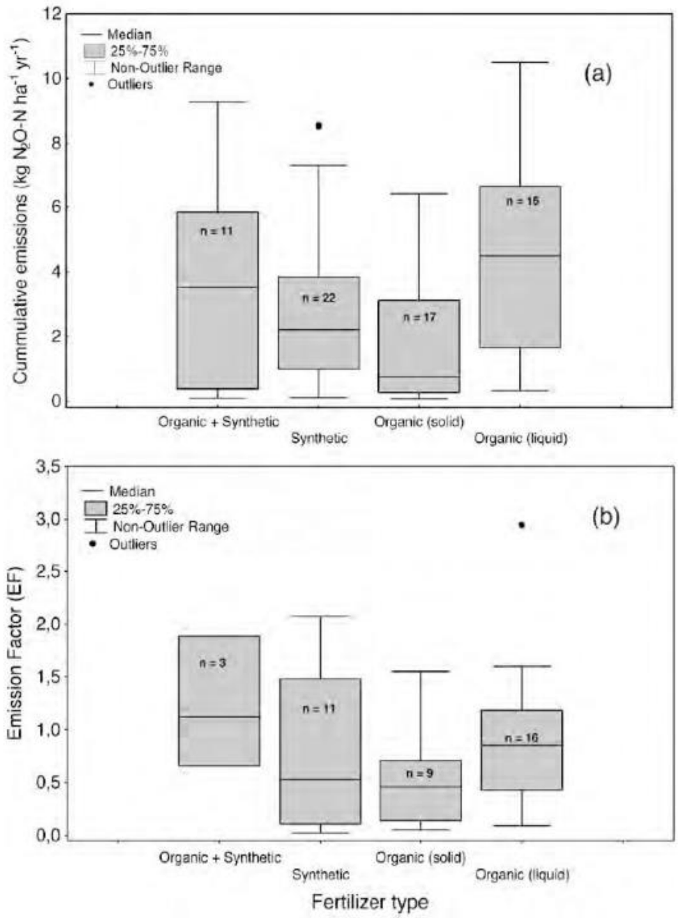


Fig. 2. N_2O emissions from Mediterranean cropping systems according to fertilizer type, expressed as: (a) cumulative emissions during experimental period, (b) emission factor. Numbers in the boxes indicate sample sizes.

When solid organic fertilizer EF was calculated on a total applied N basis, the specific response ratio for this category dropped from 0.8 to 0.37; in other words, from a 20% reduction in N_2O emissions in comparison with synthetic fertilizers, the reduction reached up to 63% with total applied N ($p < 0.001$) (Fig. 4c). These results show that the choice between total applied N and available N is highly relevant for EF calculation (Section 6.1).

The smaller N_2O emissions from solid compared to liquid organic materials is probably related to their lower concentrations of ammonium (NH_4^+) and their low rate of N mineralization, which usually prevent very high soil mineral N contents being reached. This reduction also seems to be influenced by the semi-arid features which are very common in Mediterranean soils, in which C and N contents are usually low (Section 3.3.1). Relatively high emission levels for liquid organic fertilizers (pig slurries) may be influenced by the highly mineralized nature of the N contained in these materials. NH_4^+ levels could actually represent as much as 81% and 78%, respectively, of the total N in raw and digested pig slurries (Meijide et al., 2009). Low N_2O emissions for solid organic as opposed to synthetic or liquid organic fertilizers have also been reported in temperate areas (Gregorich et al., 2005).

3.2.3. *Other factors influencing N_2O emissions*

3.2.3.1. N application rate. To our knowledge, there were no studies comparing EF as affected by N application rates under Mediterranean conditions. The IPCC approach proposes a linear relationship between N_2O fluxes and fertilizer application rates. However, many experimental data suggest that this relationship may be non-linear, with EF being lowest at low N fertilizer rates and highest at high rates. Accordingly, the global data compiled by Stehfest and Bouwman (2006) suggest a N-shaped curve for N_2O responses to N fertilizer applications, while other studies point to an exponential curve (i.e. Cardenas et al., 2010; Hoben et al., 2011).

In our review, a clear relationship between fertilizer dose and N_2O emissions could be observed in EF (Table 3), but not in cumulative emissions (Table 2), where high-emitting, irrigated control

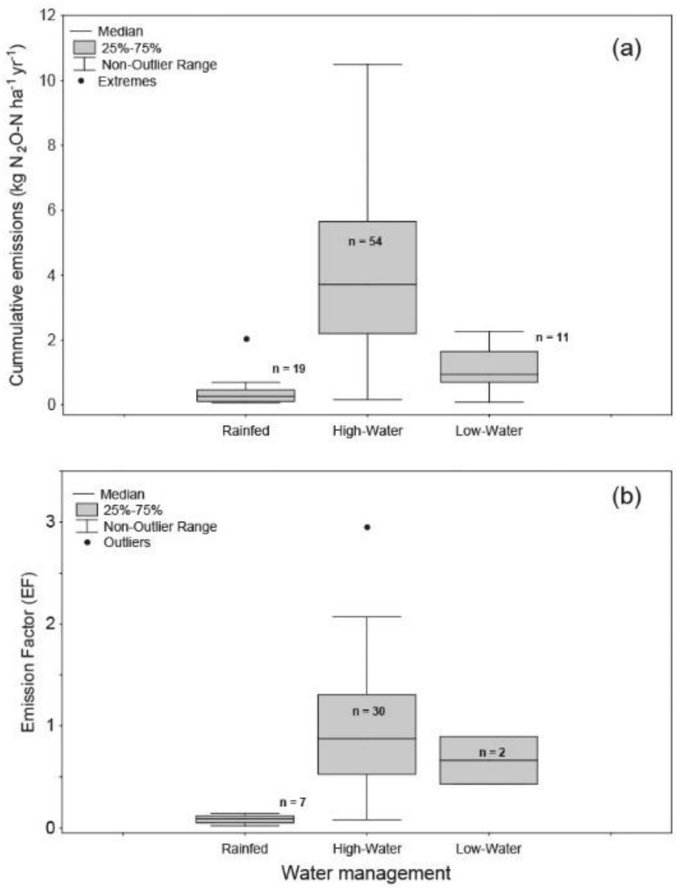


Fig. 3. N_2O emissions from Mediterranean cropping systems according to water management type. Rainfed: no irrigation; high-water: conventional irrigation systems (furrow and sprinkler); low-water: drip irrigation. Emissions are expressed as: (a) cumulative emissions, (b) emission factor. Numbers in the boxes indicate sample sizes.

treatments in the lowest dose group (0–75 kg N), and low sample size in the highest dose one (225–500 kg N) may have hidden the underlying trend. 1.8 kg N_2O-N ha^{-1} yr^{-1} was emitted on average from non-fertilized agricultural soils (Table 2). These "background emissions", which occurred on unfertilized control plots, averaged 63.5% of the fertilized treatment emissions ($N = 42$). Similar results were obtained by Kroeze and Seitzinger (1998) in their top–down analysis; they estimated that in the Mediterranean basins of Europe, 210.7 Mt of background N_2O was emitted from agricultural soils, while only 132.9 Mt of N_2 was associated with the use of manure and synthetic fertilizers. Background emissions are discussed in Section 6.2.

3.2.3.2. Crop type. Although none of the studies analyzed compared emissions from different crop types in the same year, cumulative N_2O emissions varied in a range of from 1 to 4 in function of the crop type assessed in the database analyzed (Tables 2 and 3). The main reason for this was the fact that the influence of crop type is closely related to other important factors that control N_2O production such as the N fertilizer rate and irrigation regime. For example, winter cereals and legumes exhibit exceptionally low N_2O emission levels. Both crops are grown under similar conditions: as rainfed crops during the cool rainy season, with low N fertilizing rates for winter cereals and no fertilizer applications for legumes. Low N_2O emissions have also been observed for cereals in temperate systems (Dobbie et al., 1999). Low N_2O fluxes in vineyards could similarly be related to very low N and water inputs. These systems are rainfed or drip-irrigated. In the latter case, the water applied is limited to the soil next to the vine, representing roughly one-third of the total surface area (Garland et al., 2011). The highest emissions were registered for maize. N fertilizer application rates and water inputs are normally very high for maize fields, which are cultivated under conventional irrigation techniques and high summer temperatures.

3.2.3.3. Tillage. The lower mean cumulative emissions registered for tilled as opposed to no-tilled or minimum-tilled treatments, shown in Table 2, were corroborated by the results of most of the

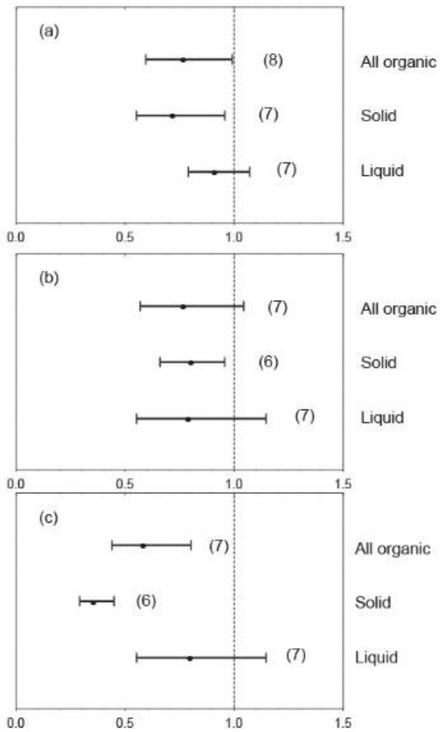


Fig. 4. Effect of fertilizer type (organic vs. synthetic) on (a) cumulative N_2O emissions; (b) N_2O emission factor (EF) calculated considering available N in solid organic fertilizers; (c) EF calculated considering total N in solid organic fertilizers. The effect is expressed as the response ratio ($RR = \bar{X}_{Org}/\bar{X}_{Syn}$) and categorized into solid and liquid organic fertilizers. Mean values and 95% confidence intervals of the back-transformed response ratios are shown (number of comparisons in parentheses). Emissions are significantly different if confidence intervals do not overlap. 1. Number of aggregated paired comparisons are indicated in parentheses (see Section 2 for independence criteria).

studies that specifically addressed this question (Garland et al., 2011; Kong et al., 2009; Lee et al., 2006; Menéndez et al., 2008; Steenwerth and Belina, 2008, 2010), although in some cases the differences were not significant (Garland et al., 2011). There was only one exception to this general trend (Heller et al., 2010), in which emissions were twice as high for the tilled compared to the no-tilled treatment. Tillage events themselves showed no clear effect on N_2O emissions. Thus, whereas in a vineyard soil N_2O pulses occurred after tillage events (Steenwerth and Belina, 2008), in a laboratory experiment there was no N_2O flux response nor any response in the denitrification rate to a simulated tillage event (Calderón et al., 2000). Higher emission rates under minimum or no-tilled treatments were mainly attributed to less aeration, which could have enhanced denitrification, particularly as most of the emissions occurred after irrigation events (Lee et al., 2009; Kong et al., 2009). Studies specifically measuring denitrification show heterogeneous responses of this microbial activity to different tillage practices, ranging from lower (Menéndez et al., 2008) to higher (Melero et al., 2011) denitrification activity under no-tillage, despite both studies were performed under similar conditions (rainfed systems on Vertisol soils). Researchers have also reported higher N_2O emissions from no-tilled plots in both humid and dry climates of the world (Six et al., 2004). In the long term, however, this relationship could disappear or even reverse (Omonode et al., 2011; Six et al., 2004).

3.2.3.4. Soil mineral N. Soil mineral N can generally be found either in the form of NH_4^+ or NO_3^-. These two compounds are respectively the substrates of nitrification and denitrification and therefore they both can stimulate N_2O emissions. In accordance with this, some authors have reported a positive correlation between N_2O flux and NO_3^- concentration in the soil (Lee et al., 2006; López-Fernández et al., 2007; Sánchez-Martín et al., 2010b). In some cases, this positive correlation only existed during certain periods (e.g. after the onset of irrigation, López-Fernández et al., 2007) or for certain values of NO_3^- content (e.g. >6.5 mg N kg soil^{-1} for temperate climates, Vilain et al., 2010). This last result may explain why many authors have not found any correlation between N_2O emissions and soil NO_3^- (Garland et al., 2011; Sánchez-Martín et al., 2008b; Steenwerth and Belina, 2008; Vallejo et al., 2005, 2006).

Soil NH_4^+ concentration has been significantly and positively correlated with N_2O emissions in many cases (Garland et al., 2011; Heller et al., 2010; Meijide et al., 2007, 2009; Petersen et al., 2006; Sánchez-Martín et al., 2008b; Vallejo et al., 2005). A significant relationship between the NH_4^+ content of the applied fertilizer and N_2O emissions has also been reported (Meijide et al., 2009). In many other cases, however, the correlation is not significant (Barton et al., 2008, 2010, 2011; Steenwerth and Belina, 2008; Vallejo et al., 2004, 2006). There have even been cases in which negative correlations have been found; this occurred in two different experiments involving no-tilled treatments (Garland et al., 2011; Lee et al., 2006). These somewhat contradictory findings suggest that other factors could be limiting N_2O production in cases in which no evident relationship between emissions and soil mineral N has been found. A lack of synchrony between N_2O flux measurement and soil sampling could also have been responsible for some of these results (Barton et al., 2008).

3.2.3.5. Soil organic carbon (SOC) and dissolved organic carbon (DOC). In the studies analyzed, SOC concentration was within a narrow range around 1% of soil mass (7–11.3 g C kg soil^{-1}); as a result, we could not study its relationship with N_2O emissions. DOC is a more dynamic parameter than SOC and its relationship with N_2O emissions was studied in some of the papers reviewed, although they did not show a homogeneous response. Thus, whereas some studies reported a positive correspondence between the two parameters (López-Fernández et al., 2007; Sánchez-Martín et al., 2010b; Vallejo et al., 2006), this relationship was only verified during the irrigation period, and sometimes only at the beginning of this period (Vallejo et al., 2006), while in others, it was absent throughout the experiment (Meijide et al., 2007). In a laboratory experiment, Sánchez-Martín et al. (2008a) showed that labile C (glucose) added to a low-carbon, basic Mediterranean soil strongly reduced N_2O emissions when a mineral N source was applied. Similar results were obtained with temperate soils and added acetate (Laverman et al., 2010). This effect was not, however, so evident in a high-carbon, acid Scottish soil, where emissions were reduced under high-water conditions (90% water filled pore space) but not under low-water conditions (40% WFPS) (Sánchez-Martín et al., 2008a). On the other hand, DOC also has an influence on the denitrification rate and N_2O/N_2 ratio, and high-carbon content does not necessarily mean that this substrate is biodegradable for denitrification (see Section 3.3.1).

3.2.3.6. Water filled pore space (WFPS). The general association between water inputs and N_2O emissions described in Section 3.2.1 is not so clear in the relationship between the WFPS and

N_2O fluxes presented in the individual studies. In some cases, N_2O emissions were reported to have increased linearly, with WFPS values increasing from 20% to 70% (Lee et al., 2006), whereas in many other cases this relationship was not found (Garland et al., 2011; Sánchez-Martín et al., 2008b; Steenwerth and Belina, 2008; Vallejo et al., 2005, 2006); this apparently contradicts results reported for temperate climates (Vilain et al., 2010). The lack of correlation between WFPS and N_2O emissions at a temporal scale of days or weeks suggests that WFPS only imposes the upper and lower limits to the microbial processes of nitrification and denitrification; in consequence, within a certain WFPS range, N_2O emissions could be largely unlinked to soil water content (Vilain et al., 2010). Accordingly, increases in WFPS were significantly related to N_2O emission pulses from Mediterranean soils when changes in soil water content were very marked, such as when rainfall or irrigation occurred after a dry period (Barton et al., 2008, 2010, 2011; Garland et al., 2011; López-Fernández et al., 2007; Meijide et al., 2009; Steenwerth and Belina, 2008, see Section 3.3 in this paper).

3.2.3.7. Temperature. The annual temperature range across different sites (13.2–20.9 °C) was too narrow to divide into categories, so the information was qualitatively reviewed. Temperature is a key factor for microbial activity and is therefore considered to have a pronounced effect on N_2O emissions. Moreover, soil temperature indirectly affects N_2O production through its influence on soil water evaporation rate, and subsequently on WFPS. In accordance, the studies reviewed here suggest that this factor, along with water and N inputs, was responsible for the large differences in N_2O emissions that were observed between winter-cropped (usually rainfed) and summer-cropped (irrigated) Mediterranean soils. A positive, significant correlation between temperature and N_2O flux was generally identified (Heller et al., 2010; Lee et al., 2009; Meijide et al., 2007; Petersen et al., 2006; Vallejo et al., 2006). In some cases, such a relationship was not evident throughout the whole experimental period, but only under certain circumstances. For example, low temperatures (below 10–12 °C) were associated with very low (Lee et al., 2009) or even negative (Meijide et al., 2009; Sánchez-Martín et al., 2010b) N_2O fluxes. In the study by López-Fernández et al. (2007), temperature only influenced N_2O flux during the irrigation period; this was probably due to restrictions on N_2O production imposed by low-water availability in the pre-irrigation period. Sánchez-Martín et al. (2010a) observed rapid soil desiccation after slurry application due to high temperatures, which delayed nitrification until the onset of irrigation. Other studies found no correlation between N_2O fluxes and temperature during the experimental period (e.g. Sánchez-Martín et al., 2008a; Steenwerth and Belina, 2008). In other climates, Horvath et al. (2010) found a link between N_2O emissions and soil temperatures up to 20 °C. Schaufler et al. (2010) reported a non-linear increase in N_2O emissions with temperature for a large set of soil cores studied under laboratory conditions.

3.2.3.8. pH. The magnitude and origin of N_2O emissions can be markedly affected by soil pH. At low pH, N_2O reduction is usually inhibited, which results in increased emissions (Baggs et al., 2010; Sánchez-Martín et al., 2008a), although this effect may only be apparent in the long term (Baggs et al., 2010). In the experiments reviewed, the soils had pH values ranging from 5.2 to 8.1 (with an average of 7.7 for all the treatments). Only two trials were performed in slightly acid soils: one in Australia (pH 6, Barton et al., 2008, 2010, 2011) and one in California (pH 5.2–7.2, Garland et al., 2011). The N_2O emissions in those trials were lower than average, but they all related to extensive systems with low or zero water and N inputs.

3.2.3.9. Texture. Soil texture affects soil N_2O production through its influence on soil aeration which, in turn, conditions nitrification and denitrification processes. For sediment, denitrification was shown to increase with the proportion of fine-textured sediment, <50 μm (Garnier et al., 2009). In this study, most of the soils analyzed were medium-textured, with only one study site corresponding to a fine-textured soil (Garland et al., 2011) and two corresponding to light-textured soils, one of which was in Australia (Barton et al., 2008, 2010, 2011) while the other was in California (Steenwerth and Belina, 2008). N_2O emissions were lower than average in the three mentioned sites, but the differences cannot be attributed to soil texture due to the low number of studies.

3.3. The influence of fertilization type and irrigation management on biochemical processes driving the annual pattern of N_2O emissions

The production of N_2O, whether in soils, wastewater treatment plants, sediments or water bodies, mainly results from biological transformations of nitrogenous compounds (Wrage et al., 2001). As Firestone and Davidson (1989) reported, soil aerobic microorganisms can nitrify soil NH_4^+ to NO_3^-, with N_2O being emitted as a by-product of this transformation. This NO_3^- can then be sequentially reduced to nitric oxide (NO), N_2O and finally N_2 by anaerobic denitrifiers; N_2O emissions occur when the reduction is incomplete. Finally, nitrifier denitrification has been proposed as a third pathway for N_2O production in the soil (Kool et al., 2011; Wrage et al., 2001). Through this pathway, autotrophic nitrifiers oxidize NH_3 to nitrite (NO_2^-) and then reduce NO_2^- to NO, N_2O and N_2 (Wrage et al., 2001).

In real agroecosystems, there is rapid shifting between the different pathways in line with changes in environmental factors and management activities. The typical Mediterranean climate pattern includes a very marked drought period during summer and usually has mild temperatures and an erratic distribution of rainfall over the rest of the year; this contributes to the existence of several wetting and drying cycles. When the soil is rewetted, without reaching complete anoxia, microbial activity is recovered, leading to a pronounced peak in N_2O and CO_2 emissions; this is what has been called the "pulse" or "Birch" effect (Birch, 1958; Beare et al., 2009; Davidson et al., 1993). This sub-optimal activity is enhanced by the availability of large quantities of C and N substrates that have been accumulated in the soil due to the death of soil microorganisms during the previous dry period. A large proportion of the annual N_2O fluxes in Mediterranean cropping systems is comprised of pulses that occur after rainfall or, when present, irrigation events, especially when the soil was previously dry. N_2O pulses after rainfall events are also common in temperate climates, where they may be driven by denitrification (Davidson et al., 1993; Vilain et al., 2010). Pulses may also be driven by aerobic processes (nitrification) when the climate conditions are semi-arid (Galbally et al., 2008). Pulses driven by nitrification are, however, usually of lower intensity, in absolute terms, than those driven by denitrification (see Section 3.3.2).

On the other hand, all of these biochemical processes can also occur in the soil simultaneously due to the high complexity of soil structure, where adjacent microsites can show very different levels for such soil parameters as WFPS, NH_4^+, NO_3^- or C accumulation. This spatial and temporal heterogeneity has already been specifically studied in Mediterranean cropping systems. Kong et al. (2010) studied the abundance of ammonia oxidizing bacteria (AOB), denitrifiers and total bacterial communities in different soil microenvironments under different long-term management regimes. They showed that despite the fact that AOB and denitrifier community abundances were affected by management practices, they were largely decoupled from N cycling. Nitrifier and denitrifier communities were larger and fluctuated more in microaggregates than in particulate organic matter (POM) and silt-and-clay fractions. These findings suggest that microaggregates are potential hotspots for N_2O production in the soil (Kong et al., 2010).

Although nitrification and denitrification are the processes responsible for N_2O production in the soil, they may not be correlated with N_2O flux. Denitrification activity could actually shift from being a source to a sink for N_2O, depending on its efficiency for N_2O reduction to N_2. This efficiency has sometimes been shown to be enhanced under Mediterranean conditions, thus potentially decreasing N_2O emissions. This process may occur at relatively low WFPS (Menéndez et al., 2008), but it is usually enhanced when N_2O diffusivity is low due to very high WFPS (Lee et al., 2009; Sánchez-Martín et al., 2010b; Vallejo et al., 2005) and when soil NO_3^- content is low (Ryden and Lund, 1980b; Sánchez-Martín et al., 2008a, 2010b). Labile organic C sources could also increase denitrification efficiency (Section 3.3.1). Whether driven by this or by another process, a net N_2O uptake has been observed on some occasions (Barton et al., 2008, 2011; Garland et al., 2011; Meijide et al., 2009; Sánchez-Martín et al., 2010b). N_2O uptake events occurred at the end of spring, coinciding with low WFPS and low mineral N levels and/or high DOC (Barton et al., 2008; Garland et al., 2011). The information related to the different biochemical pathways for N_2O production that were influenced by seasonal changes in environmental factors is reviewed in the following sections and grouped according to fertilizer type and water management regime.

3.3.1. *Fertilizer type*

Organic fertilizers are very heterogeneous materials, whose properties can vary widely depending on their origin and processing. As a general trend, however, adding organic matter to the soil provides the labile C substrates needed for denitrification, which is further enhanced by the creation of anaerobic microsites, even when soil WFPS is <55% (García-Ruiz and Baggs, 2007). The positive effect of a range of different organic fertilizers on the denitrification rate has been verified by López-Fernández et al. (2007), Meijide et al. (2007) and Vallejo et al. (2006), in irrigated arable plots in Central Spain, in which a larger proportion of N_2O fluxes were driven by denitrification in soils amended with organic matter than in synthetic N fertilized soils. On the contrary, in drip-irrigated treatments (Sánchez-Martín et al., 2008b), digested NH_4^+-enriched pig slurry-amended plots produced proportionally less N_2O by denitrification (56%) than the unfertilized control (92%).

DOC (see Section 3.2.3) is a soil parameter which is affected by the type of fertilizer employed. Applying mineral N promotes the consumption of DOC by soil microbial biomass (Sánchez-Martín et al., 2008b) and applying urea is usually related to a temporal (1–2 months) increase in DOC (Meijide et al., 2007; Vallejo et al., 2006). Complex organic materials like composts release labile C compounds during their mineralization process, even though their DOC promoting effect could be delayed until 3–4 months after application (Meijide et al., 2007). DOC concentrations have been correlated with the denitrification rate in numerous cases (López-Fernández et al., 2007; Meijide et al., 2007; Vallejo et al., 2006) and especially during the irrigation period. High DOC during irrigation promoted anoxia, which favored denitrification, but not necessarily N_2O emissions (López-Fernández et al., 2007; Meijide et al., 2007). This implies that labile C could also reduce the N_2O/N_2 ratio. Indeed, enhanced denitrification efficiency due to the availability of labile C resulted in a reduction in net N_2O emissions in a Cumulic Haploxeroll in California (Steenwerth and Belina, 2010).

A second possible explanation for N_2O reduction by organic fertilizers was pointed to by Dick et al. (2008) for semi-arid cropping systems in Mali, where lower N_2O emissions were measured in plots receiving mixtures of urea and organic manure as opposed to those that only received urea. These authors suggested that N could be immobilized more efficiently by the existing microbial biomass when easily available C and N are simultaneously added to a soil lacking C and N. For example, adding different organic residues to Mediterranean soils fertilized with synthetic N reduced denitrification losses as a percentage of applied N (Coskan et al., 2002), which could perhaps be explained by more efficient N immobilization. However, this immobilization would not necessarily explain the results obtained in Mediterranean systems, where denitrification is usually promoted by organic fertilizers.

The nitrifier denitrification pathway is favored at low concentrations of available C and sub-anoxia (Wrage et al., 2001, see next section); these are common characteristics of Mediterranean soils. The addition of organic C sources to the soil could therefore influence this pathway. An increase in C availability could reduce the contribution of this pathway to overall N_2O emissions, as compared to that of other pathways, and this could help to explain the reduction in cumulative N_2O emissions observed in our analyses.

3.3.2. *Rainfed systems*

Low N_2O emission levels in Mediterranean rainfed cropping systems are conditioned by the typical agro-climatic features of these systems. During the winter season, N_2O production is usually limited by temperature but also by other factors such as low levels of WFPS (Barton et al., 2008, 2010), soil organic matter (Sánchez-Martín et al., 2008a) or mineral N (Lee et al., 2009). A strong coupling between N mineralization and immobilization has been reported at low soil temperatures (Barton et al., 2010). Moreover, the N fertilization rate in rainfed systems under these climatic conditions is generally low (Kroeze and Seitzinger, 1998; Ryan et al., 2009) due to crop growth limitations driven by climate. This practice could also contribute to low N_2O emissions.

In late spring, during maturation of winter crops, high temperatures favor microbial processes, but N_2O production may be limited by low NH_4^+ content and low WFPS (Meijide et al., 2009).

During summer, dry soil conditions usually prevent N_2O emissions, except when significant rainfall occurs (Barton et al., 2008, 2010, 2011). Rainfall events after the summer are usually related to significant N_2O pulses due to N mineralization and the subsequent accumulation of mineral N in the soil during this period (Meijide et al., 2009). This seasonal pattern of soil N dynamics also occurs in Mediterranean natural ecosystems (Ochoa-Hueso et al., 2011).

In Mediterranean rainfed systems, and especially in those cultivated in well-aerated soils, low rainfall leads to low WFPS and therefore to a high redox potential which is unsuitable for denitrification (Lugato et al., 2010). According to most of the studies reviewed, this makes nitrification the most usual pathway for N_2O production in low organic matter, non-irrigated Mediterranean soils (Barton et al., 2008, 2011; Lugato et al., 2010; Meijide et al., 2009; Menéndez et al., 2008). This hypothesis is also supported by the positive relationship between N_2O fluxes and soil NH_4^+ levels (Meijide et al., 2009). N_2O pulses due to nitrification are relatively small (Sánchez-Martín et al., 2010a), which is in line with the typically low cumulative N_2O fluxes found in Mediterranean rainfed systems.

N_2O emissions due to denitrification can be common in rainfed systems after heavy rainfalls or when the rainy season is especially wet (Meijide et al., 2009; Sánchez-Martín et al., 2010b). Complete anoxia promoting the reduction of N_2O to N_2 is unlikely or very transient in Mediterranean rainfed systems. Therefore, wetter-than-average cropping years usually lead to higher N_2O emissions than drier ones (Sánchez-Martín et al., 2010b), which may also happen under temperate climatic conditions (Laville et al., 2011).

Kool et al. (2011) showed that the nitrifier denitrification pathway for N_2O production could be responsible for a significant fraction of N_2O emissions in agricultural soils, especially under moisture conditions that are sub-optimal for denitrification. Following this logic, this pathway could be important in N_2O production under Mediterranean conditions (Mondini et al., 2007; Sánchez-Martín et al., 2008a), particularly in rainfed and drip-irrigated systems, where the moisture content required for denitrification is not often reached. For example, Sánchez-Martín et al. (2008a) hypothesized that most N_2O was produced by nitrifier denitrification in a Mediterranean soil incubated under laboratory conditions at 40% WFPS and identified it as a significant source at 90% WFPS. Nitrifier denitrification has also been proposed as a possible pathway for N_2O uptake when conditions are not suitable for anaerobic denitrification (Chapuis-Lardy et al., 2007; Meijide et al., 2009). If nitrifier denitrification plays an important role in N_2O production and consumption in Mediterranean soils, this would imply that a significant fraction of N_2O emissions currently attributed to denitrification or nitrification could actually be driven by this biochemical pathway. Such uncertainty points to the need for more research in order to accurately understand the biochemical processes underlying N_2O emissions in Mediterranean agroecosystems.

3.3.3. Irrigated systems

Irrigation is associated to the intensification in N inputs, and in Mediterranean cropping systems it usually takes place in late spring and summer, when temperatures are highest. Therefore, the most relevant physical properties of the soil (temperature and water and N availability) are optimal for N_2O production during the cropping season. During the winter fallow period, the conditions are similar to those described for rainfed systems, though more residual N can be available in the soil due to the higher application rates of N fertilizers. In summer-irrigated Mediterranean systems, it is therefore usual for significant N_2O fluxes to occur throughout the annual cycle.

In the case of high-water irrigation techniques, such as furrow irrigation, near-saturation conditions are transiently reached for one or more days after irrigation. These conditions usually lead to a very high initial N_2O pulse after the first irrigation event (López-Fernández et al., 2007; Sánchez-Martín et al., 2008b, 2010a; Vallejo et al., 2005). This can then be followed by two or more large pulses in the course of the irrigation period (Kallenbach et al., 2010; Vallejo et al., 2006). Denitrification is usually the prevailing N_2O production pathway under this type of irrigation (Sánchez-Martín et al., 2008b); this is usually a minor pathway before the onset of irrigation and then becomes the main source during the irrigation period (López-Fernández et al., 2007; Vallejo et al., 2005), accounting for up to 99% of N_2O fluxes (Sánchez-Martín et al., 2010a). Nitrification is likely to occur in these systems when a high concentration of NH_4^+ is reached in the soil; this is sometimes the case when synthetic or liquid organic fertilizers are applied (Meijide et al., 2007; Sánchez-Martín et al., 2010b,c; Vallejo et al., 2006), providing aerobic conditions are met, which can be the case in high-water irrigated systems, where WFPS fluctuations are very large.

N_2O fluxes are generally lower in drip-irrigated soils than in those receiving high-water applications, and the emission patterns also differ. Drip irrigation promotes a small but steady flux of N_2O throughout the cropping season (Kallenbach et al., 2010), which could be accompanied by small N_2O pulses after each irrigation event (Sánchez-Martín et al., 2008b, 2010a) instead of one or various large pulses of the sort typically associated with furrow irrigation systems. As in rainfed systems, low-water availability in drip-irrigated soils results in nitrification becoming the most important source of N_2O (Kallenbach et al., 2010). This assumption is supported by papers that report a lack of relationship between N_2O fluxes and NO_3^- concentrations (Garland et al., 2011; Steenwerth and Belina, 2008) and large increases in the size of the soil NO_3^- pool after the NH_4^+ peak (Sánchez-Martín et al., 2008b). Indeed, N_2O production by denitrification is prevented by low-water availability, as the soil rarely exceeds 60% WFPS, either with subsurface (Kallenbach et al., 2010) or surface (Sánchez-Martín et al., 2008b, 2010a) drip irrigation techniques. However, according to the latter works, the spatial distribution of soil humidity, and subsequently of N_2O fluxes, stress the need for a stratified sampling of N_2O fluxes and soil parameters. For example, wet areas near drippers may lead to a N_2O source, but a WFPS of >80% can be locally reached in dripping points, which may promote denitrification to N_2.

4. Indirect sources of N_2O emissions

4.1. Upstream emissions

Emissions during the production, manufacturing and transport of fertilizers play a key role in total fertilizer emissions. It is not reasonable to consider these processes from a specifically Mediterranean perspective given the wide range of conditions under which fertilizers are produced in these areas. Nonetheless, there are general differences in the emission of GHG during the production of synthetic and organic fertilizers that are worth noting.

4.1.1. Synthetic fertilizers

The production of synthetic fertilizers requires a high consumption of fossil energy to reduce N_2 to NH_3. According to the IPCC (2006b), average CO_2 emissions due to NH_3 production in European plants range between 2.55 and 3.57 kg CO_2 per kg of fixed N, depending on the technology employed. In comparison, N_2O emissions from the soil calculated using IPCC EF are equivalent to 4.68 kg CO_2 per kg of applied N. In Mediterranean rainfed and drip-irrigated cropping systems, where N_2O EF is lower than the IPCC default EF, these pre-farm GHG emissions related to fertilizer production may actually be much greater than the on-farm N_2O emissions. For example, Biswas et al. (2008) performed a life cycle assessment (LCA) of rainfed wheat production in the Mediterranean climate region of Western Australia. They estimated that to produce one ton of wheat, 103.87 kg CO_2-eq was emitted as a result of urea production, whereas N_2O emissions from the field represented 26.98 or 175 kg CO_2-eq, according to whether region-specific (Barton et al., 2008) or IPCC (2006b) N_2O EF was employed.

4.1.2. Organic fertilizers

The use of organic materials as fertilizers requires the management of organic wastes. When the residual organic matter is not produced in the field, it needs to be handled, stored, transported, and sometimes transformed into more stable and easier to handle compounds. This management process is associated with GHG emissions. In the EU-27, N_2O emissions during the housing and storage of animal manure are estimated to be only slightly lower than those associated with their land application (Oenema et al., 2009). GHG emissions related to the production of organic fertilizers from organic waste should be accounted for by comparison with the emissions associated with conventional residue management (e.g., Kim and Kim, 2010; Prapaspongsa et al., 2010). A careful and site-specific assessment of GHG emissions is required during waste management in order to quantify upstream GHG emissions by organic fertilizers.

Legumes are virtually the only organic source of newly fixed N. N_2O emissions during N fixation by legumes are generally taken to be zero or negligible (IPCC, 2006a) and this has been verified under Mediterranean conditions (Barton et al., 2011). Nonetheless, N_2O emissions that occur during legume crop growth and indirect GHG emissions related to their cultivation should be taken into account in full GHG comparisons when legumes are used as green manure. Furthermore, land occupation associated with biological N fixation calls into question the possibility of a considerable substitution of Haber-Bosch-produced N. Indeed, some authors have shown that the internalization of energy and nutrient fluxes in sustainable agriculture may also have a "land cost" that is externalized in fossil fuel-based systems (Guzmán Casado and González de Molina, 2009). Even so, there is still great potential for reducing the N surplus and increasing N fixation in Mediterranean agroecosystems without needing to occupy any extra land (Section 5), even if the extent to which sources of organic N can replace synthetic ones still remains unclear.

4.1.3. Transport

The use of organic fertilizers (i.e. slurries, manures) requires large amounts of energy due to their weight. However, their production sources tend to be more local to the end user than synthetic fertilizers, which are generally produced in a few large manufacturing plants. Even without taking into account transport costs for synthetic fertilizers, Wiens et al. (2008) estimated that the distance that liquid pig manure was transported could be increased to 8.4 and 12.3 km, respectively, before the energy cost per kg of available N associated with this manure was equivalent to that of anhydrous ammonia or urea N. Nearby land could therefore receive the resulting manure at an appropriate rate and without high transport costs, as long as the concentration of livestock is not very high (Section 5.3).

4.2. Downstream emissions

NO_3^- leaching and NH_3 and oxidized N compounds (NO_x) volatilization are considered the main processes responsible for fertilizer-associated N_2O emissions outside the cropping system and are the only ones classified as "indirect fertilizer N_2O emissions" in the IPCC guidelines for GHG inventories (IPCC, 2006a). Downstream indirect emissions were estimated to represent about 13–17% of direct emissions in one temperate river basin (Garnier et al., 2009).

4.2.1. NO_3^- leaching

This is a source of major concern in many areas in Mediterranean countries, such as Spain (Lassaletta et al., 2009, 2010; Peña-Haro et al., 2010), because of its eutrophication potential and the negative impact on drinking water from surface or ground waters. In Mediterranean cropping systems, NO_3^- leaching can occur either in irrigated fields (Allaire-Leung et al., 2001) or be related to rainfall events during the rainy season (Angás et al., 2006), and it can be responsible for the loss of up to 25% of applied synthetic N in a normal winter fallow period (Sánchez-Martín et al., 2010b).

Very large N surpluses can occur from organically fertilized soils, resulting in NO_3^- leaching and related aquifer pollution. For example, the application of high rates of slurries has caused high N losses, in the form of NO_3^-, in intensive livestock production areas in NE Spain (Peñuelas et al., 2009). Even so, when organic and synthetic fertilizers are compared at similar N application rates, NO_3^- leaching is generally significantly lower with organic fertilizers (Antoniadis et al., 2010; Díez et al., 1997, 2000; Celik, 2009; Sánchez-Martín et al., 2010a). Some authors have even found N leaching to be lower in soils fertilized with compost than in unfertilized plots (Tejada and Gonzalez, 2006). In general, low (Sánchez-Martín et al., 2010a) or null (Díez et al., 2004) NO_3^- leaching reductions are associated with liquid, highly mineralized organic fertilizers, such as pig slurries. On the other hand, other approaches to nutrient management based on organic matter cycling, such as cover cropping, have also proved capable of strongly reducing NO_3^- leaching in Mediterranean environments (Salmerón et al., 2010; Steenwerth and Belina, 2008; Wyland et al., 1996).

The lower N leaching associated with organic amendments could be driven by a decrease in soluble N in the soil due to the increased performance and efficiency of denitrifiers (Kramer et al., 2006; Steenwerth and Belina, 2010), the immobilization of NO_3^- by a larger microbial biomass (Burger and Jackson, 2003), or the capture of N in the SOM which is built up by the addition of organic matter. For example, Kong et al. (2007) reported that an additional 590 kg N ha^{-1} had been stored in the soil after 11 years of organic management, along with the sequestration of 5.7 Mg C. Applying organic matter to the soil could also reduce the subsoil NO_3^- pool by enhancing the activity of denitrifying microorganisms in the subsoil or groundwater (Sánchez-Martín et al., 2010a), as organic C is the main limiting factor for denitrification in the subsoil (Haag and Kaupenjohann, 2001). In an experiment performed under Mediterranean conditions, DOC leaching of 2.3–4.8 kg C ha^{-1} helped to complete the reduction of 2.1–4.5 kg NO_3^--N ha^{-1} to N_2 (Sánchez-Martín et al., 2010a), implying capacities for subsoil NO_3^- removal of 100%, 13.2%, 10.4% and 6.7% for organic manure, control, digested pig slurry and urea treatments, respectively. These results should, however, be interpreted with care since incomplete subsoil denitrification could also release N_2O, which could be transported by drainage water due to its high solubility (Van Cleemput, 1998).

4.2.2. NH_3 volatilization

We did not find any field studies that compared NH_3 volatilization after organic and synthetic fertilization under Mediterranean climatic conditions, although some separate data are available. Various different synthetic fertilizers applied to wheat under simulated Mediterranean conditions were reported to have released 12–38% of their N (mostly as NH_3) (Buresh et al., 1990), whereas at arid and semi-arid sites in Syria, slightly lower gaseous N losses, of 11–18% (again mostly as NH_3), were recorded after urea application (Abdel Monem et al., 2010). The only micrometeorological studies conducted that specifically measured NH_3 losses reported: (i) 10.1% NH_3-N losses from urea (Sanz-Cobena et al., 2008); (ii) 5% losses after green manuring (Rana and Mastrorilli, 1998) and (iii) 20% of the Total Ammonium Nitrogen applied with a pig slurry spread at the soil surface (Sanz et al., 2010). The first two values, for synthetic and organic fertilization, are below the 20% default EF value established by the CORINAIR Emission Inventory Guidebook for regions with spring temperatures >13.8 °C (CORINAIR, 2006). Contrastingly, measured NH_3 losses, from synthetic and liquid organic manures, will be in accordance with the values proposed by the IPCC (10% and 20% for synthetic and organic fertilizers, respectively). Existing discrepancies could be associated with local climatic, soil and management conditions, e.g. dry conditions during the experimental period, the presence of vermiculites as the main clay mineral, and the application of 10 mm of irrigation immediately after fertilizing, all possibly favor large decreases in the availability of exchangeable NH_4^+, which can be potentially lost as NH_3 (Sanz-Cobena et al., 2008).

4.2.3. N_2O emissions

There is very little information about which fraction of the N lost from the cropping system in the form of NO_3^-, NH_4^+ or NH_3 is finally transformed into N_2O in Mediterranean environments. The available data suggest that this fraction could be very significant but variable. Measurements performed in the Douro Estuary in Portugal (Teixeira et al., 2010) revealed that 0.5–47% of the N gases produced were in the form of N_2O, and that emissions were correlated with sediment organic matter. On the plain of the River Po, in Northern Italy, springs were found to be supersaturated with N_2O and were subject to a significant degassing process. As a result this area had a very high potential as a source of N_2O and other GHG gases (Laini et al., 2011). In one stream in the Doñana National Park, SW Spain, NO_3^- pollution which originated in nearby agricultural fields, mostly from synthetic N sources, was associated with N_2O production, but also with that of CH_4 and CO_2 (Tortosa et al., 2011).

5. Mitigation options with organic fertilizers

5.1. Water-saving agricultural systems

The significant reduction in N_2O fluxes in rain-fed and drip-irrigated systems as opposed to conventionally irrigated systems (Sections 3.2.1 and 3.3) implies a high potential to mitigate N_2O emissions through the optimization of water use. However, the reduction in N_2O fluxes achieved by applying drip irrigation may only occur when a source of organic matter is applied (Kallenbach et al., 2010). Drip-irrigated systems foster water and N_2O emission savings while maintaining yields (Tognetti et al., 2003; Kallenbach et al., 2010), whereas in rainfed systems, yield-scaled emissions may be affected by a lower productivity (Section 6.3). However, a full GHG accounting should also consider the higher fossil energy consumption in irrigated systems (Alonso and Guzmán, 2010).

5.2. Minimization of bare soil

Bare fallows in herbaceous crop rotations and bare soils in woody perennial systems are usually maintained in Mediterranean environments in order to increase water and nutrient availability for commercial crops. This assumption has been challenged by research data, which show that bare fallows may not contribute to overall productivity as much as legume cover crops (López-Bellido et al., 2000; Martín-Rueda et al., 2007), as has also been reported in other dry environments (Rinnofner et al., 2008). Bare fallows and other bare soils may therefore represent stages, or areas, of the cropping systems capable of releasing large quantities of reactive N compounds (which are responsible for both direct and indirect N_2O emissions) without contributing to overall productivity.

Fallow and bare soil emissions can be avoided by system intensification, in which cash crops substitute bare fallows, and also through the cultivation of cover crops, either in crop rotations or in perennial systems. Cover crops have a large potential for increasing N retention in cropping systems and thereby reducing indirect N_2O emissions, mainly through (i) N immobilization by catch crops (e.g., McSwiney et al., 2010; Gabriel and Quemada, 2011), (ii) biological N fixation with legume green manures (e.g. Rinnofner et al., 2008) and (iii) soil protection against erosion (Boellstorff and Benito, 2005; Gómez et al., 2009). Their effect on direct N_2O emissions may vary according to the specific case. Legume cropping in the Mediterranean semi-arid environment of Western Australia yielded similar emission rates as bare soils (Barton et al., 2011). In California, Kallenbach et al. (2010) recorded higher N_2O emissions from a legume cover cropped treatment than from bare soil. These authors suggested that non-legume cover crops could help to reduce N_2O emissions due to their higher C:N ratio and deeper roots, which could extract soil N more efficiently. However, in order to maintain the benefits of biological N fixation, we would propose trials with mixtures of legumes and non-legumes and also their combination with low-quality organic residues (Section 5.4).

5.3. Improved waste management

The high population densities in most Mediterranean areas suggest that urban wastes could represent a significant source of organic matter for agricultural fields. Municipal solid waste and sewage sludge, especially if composted, usually show good agronomic performance, and they also promote an increase in soil organic carbon (Diacono and Montemurro, 2010). As we have already seen in this review, the use of these materials as fertilizers can also help to reduce direct and indirect N_2O emissions. In spite of these advantages, heavy metals and other toxic compounds may call into question their safe application to soils, which points to the need to appropriately separate urban organic wastes at source.

In the case of livestock farming, the continued specialization of livestock production units leads to increasing problems with the safe recycling of manure nutrients (Petersen et al., 2007), and encourages their application to nearby soils in high doses, which boosts N losses, and particularly N_2O ones. In some parts of Israel, for example, organic manures are applied at rates of over 1000 kg N ha^{-1}, which increases emissions to 34.4 kg N_2O-N ha^{-1} (Heller et al., 2010). The minimization of N surpluses that cause both pollution and dependence can be achieved by the circulation of materials between livestock and cropping systems, as demonstrated in other regions (Nekomoto et al., 2006). Biogas production is another integrated approach to waste management that can help to reduce GHG emissions.

5.4. Tightening the N cycle through N immobilization

Typical woody crops cultivated under Mediterranean climatic conditions include vines, olives, almonds, walnuts, citrus and other fruit trees. They occupy large areas and produce high quantities of pruning residues, which are normally burned in the field, resulting in emissions of trace GHG and also stored C. Proposals have recently been made for their use as energy sources (e.g. Di Giacomo and Taglieri, 2009; Kroodsma and Field, 2006). An alternative, or complementary option, is to incorporate these residues into the soil; this could greatly enhance soil carbon storage and biodiversity (Holtz and Caesar-Ton That, 2004) and would also protect the soil against erosion (Rodríguez-Lizana et al., 2008), while fostering a reduction in N losses. Indeed, N retention through N immobilization during fallow periods cannot only be achieved with catch crops (Section 5.2), but also through the addition of low-quality organic residues (Muhammad et al., 2011; Sakala et al., 2000). To be more specific, lignin and polyphenol rich materials, such as pruning residues, have been associated with reductions in N_2O fluxes over a broad range of conditions due to their strong N immobilization effect (Frimpong and Baggs, 2010; García et al., 1997; Gomes et al., 2009). In a laboratory experiment, García-Ruiz and Baggs (2007) demonstrated that N fertilizer application did not increase N_2O emissions if the soil had been mixed with olive leaves; this was related to the high lignin (11%) and polyphenol (2%) contents of olive residues.

The major concern here, is that N immobilization may negatively affect crop production (Frimpong and Baggs, 2010; Soumare et al., 2002), especially during the first months after application (Soumare et al., 2002). Potential reductions in crop productivity due to N immobilization could, however, be remedied by appropriately combining and timing the application of N sources with different mineralization kinetics. In this sense, mineralization kinetic parameters could be used to evaluate the most suitable N release pattern for organic fertilizers (Marinari et al., 2010). Successful examples of soil protecting practices in three different Mediterranean orchards, including the addition of pruning residues, led to a significant increase in fruit yield compared with conventional management strategies (Montanaro et al., 2009; Sofo et al., 2010).

6. Information gaps

We found a number of knowledge gaps that we would recommend addressing in future field research.

6.1. Length of the experiment

The studies included in this review were carried out on average for 243 days. The average experiment length was consistently higher in rainfed systems than in irrigated ones, where cropping period is usually shorter. Measurement periods of less than 1 year do not account for total annual emissions, e.g. those of the residual effects of fertilizers, which could lead to a possible underestimation of EF. An estimation of yearly cumulative emissions and EF based on simple modeling of measured emission levels resulted in a great increase in emissions in irrigated groups (Tables 2 and 3). This procedure, however, could overestimate yearly emissions because N_2O flux is usually highest during cropping period, when fertilizers and water are applied.

Significant emissions during the post-harvest fallow period have been recorded in Mediterranean cropping systems. In the summer fallow period of rainfed systems, emissions are relatively low, most of the time, due to the dry soil conditions (Steenwerth and Belina, 2008), and isolated summer rainfall only stimulates N_2O emissions very transiently due to rapid soil desiccation (Sánchez-Martín et al., 2010a). Large or multiple rainfall events during summer can, however, increase summer fallow emissions to 55% of yearly emissions (Barton et al., 2008, 2010, 2011). Significant quantities of N_2O may also be lost during the winter fallow periods of summer-cropped, irrigated fields (Burger et al., 2005; Kallenbach et al., 2010; Lee et al., 2009; Sánchez-Martín et al., 2010a).

As previously mentioned, there is particular concern about the residual effect of organic fertilizers, given their typically slow and extended N release, which can prolong their N_2O emission period (Jones et al., 2007) and would justify the use of "available N" rather than "total N" when calculating their EF (e.g., Vallejo et al., 2006, see Section 2 of this paper). For example, Meijide et al. (2009) found various N_2O emission peaks during the post-harvest period in organic-fertilized soils as opposed to only one peak in the control and synthetic treatments. Conversely, releasing labile C compounds during the mineralization of organic fertilizers in soils poor in organic matter could help to prevent N_2O emissions (Sánchez-Martín et al., 2010a). These findings challenge the assumption that N_2O fluxes would tend to be extended by the residual effect of organic fertilizers and underline the need for longer sampling periods and more long-term studies.

Moving beyond short term residual effect, Li et al. (2005) argued that techniques that promote C sequestration could enhance N_2O emissions in the long term due to the increase in soil organic carbon (SOC). Under Mediterranean conditions, long-term experiments comparing organic and synthetic fertilization have only been performed at one site: Russell Ranch in Davis, California (Burger et al., 2005; Kong et al., 2007, 2009). None of these studies reported increases in N_2O emissions, despite the fact that SOC pool increased significantly after up to 11 years of organic management. We therefore hypothesize that under Mediterranean conditions there may be a SOC content threshold above which fertilizer-related N_2O emissions would start to increase. In spite of this, at relatively low SOC levels around 1%, such as those studied by Kong et al. (2007, 2009), organic fertilizers would help to reduce N_2O fluxes.

6.2. Background emissions

Unfertilized control treatment N_2O emissions on average represented 63.5% of fertilized treatment emissions in the reviewed studies. This implies that a very large fraction of soil N_2O emissions cannot be explained by the fertilizer EF approach. The pulsing effect probably contributes to these high background emissions. Changes in management practices, such as irrigation methods, may also induce distinct changes in emissions from control and fertilized treatments, which would also affect N_2O EF. For example, Sánchez-Martín et al. (2010b) reported a sharp reduction in N_2O emissions with drip irrigation, as opposed to furrow irrigation. The reduction was, however, much greater in the control treatment; it produced greater EF values for drip-irrigated treatments, while cumulative emissions were actually lower than for furrow-irrigated treatments.

These data suggest that background emissions are influenced by management and consequently require specific accounting, as they are susceptible to improvement. The calculation of EF, by dividing cumulative N_2O emissions by the applied N rate, without subtracting unfertilized control emissions (e.g. Dobbie and Smith, 2003), is an approach that includes background emissions in estimations based on applied fertilizer. This method would be less useful, however, if these background emissions were affected by other management operations. We therefore recommend complementing EF data with cumulative emission data for every treatment, including unfertilized controls, when presenting research results.

6.3. N_2O EF and yield-scaled EF

EF was only provided in 52% of the reviewed studies that measured field N_2O emissions. In the other cases, it was not possible to calculate EF because unfertilized treatments were absent. Despite its limitations, EF provides a useful simplified tool for upscaling the emissions of a given region based on fertilization rates, provided that background emissions are also accounted for. Although N_2O emissions vary greatly, the EF approach is fairly well supported by field data (Stehfest and Bouwman, 2006; Petersen et al., 2006). However, the results analyzed in this review suggest that it needs region-specific modulations for such factors as fertilizer type and irrigation type.

Yield-scaled EF is also a very informative parameter for understanding site-specific trade-offs between fertilization type, N_2O fluxes and yield performance. Full GHG accounting methodologies such as LCA could greatly benefit from this information, as they are usually product based. When applied to a single type of fertilizer, yield-scaled EF usually reveals that N_2O emissions are smaller at intermediate N application rates (Hoben et al., 2011; Van Groenigen et al., 2010). When comparing management schemes, yield-scaled EF shows that environmental benefits may disappear if they are associated to lower yields (De Backer et al., 2009). In the Mediterranean context, Meijide et al. (2009) found that the performance of organic and synthetic fertilizers slightly differed according to whether they were evaluated on an applied N basis or on a yield basis. The authors also warned about the annual variability of crop yield, which should be taken into account when applying yield-scaled index.

Other studies comparing yields associated with organic and synthetic fertilization in Mediterranean cropping systems have obtained heterogeneous results, although most of the authors consulted reported similar yields for the two types of fertilization (Altieri and Esposito, 2008; Bilalis et al., 2010a,b; Caporali and Onnis, 1992; Clark et al., 1999; Díez et al., 1997, 2000; Deria et al., 2003; Drinkwater et al., 1995; Efthimiadou et al., 2009; Herencia et al., 2007; Lithourgidis et al., 2007; Madejón et al., 2001; Meijide et al., 2007; Montanaro et al., 2009; Montemurro et al., 2005, 2008; Montemurro, 2010; Morra et al., 2010; Pardo et al., 2009; Vallejo et al., 2006). Higher yields for organic fertilization have also been reported (Campiglia et al., 2011; Curuk et al., 2004; Deria et al., 2003; Karamanos et al., 2004; Madejón et al., 2001; Melero et al., 2006; Montemurro et al., 2008; Sofo et al., 2010), while other authors discovered yield reductions related to organic as opposed synthetic fertilizers (Annicchiarico et al., 2010; Denison et al., 2004; Deria et al., 2003; García-Martín et al., 2007; Kavargiris et al., 2009; Montemurro et al., 2006, 2007; Montemurro, 2009, 2010;

Morra et al., 2010). The disparity in the relative yield performance of organic and synthetic fertilization is understandable given the variety of types of organic fertilizers, management techniques and agro-climatic conditions. This complexity further emphasizes the need for yield data in specific N_2O emission studies.

6.4. Cropping systems that require more research

Different crop types have been unevenly studied. Maize, open-air horticultural crops and winter cereals have been studied under a fairly wide range of conditions and we now have a rough picture of the behavior of N_2O emissions in these systems, but information about other very important Mediterranean crop types is almost nonexistent. Despite the importance of many of the woody perennial crops grown in this biome, vines are the only crop in this category for which N_2O emissions have been measured. Greenhouse horticulture must also be studied, as its particular environmental conditions, including high humidity and temperature throughout the year, would probably affect the pattern of N_2O emissions.

Organic management systems also require specific research, as they not only exclude synthetic fertilization, but also the use of other synthetic compounds which may either increase or reduce N_2O emissions (Kinney et al., 2005; Spokas et al., 2006). Organic farming is a very important management option in Mediterranean areas. For example, the two European countries with the largest surface areas under organic farming are Spain and Italy (Willer and Kilcher, 2011), whereas California is the state with the largest organic acreage in the USA (USDA, 2011). Lower N_2O fluxes have been reported for organic fields under Mediterranean conditions (Burger et al., 2005; Kong et al., 2007, 2009; Petersen et al., 2006), but the available data is very limited and yield-scaled performance may be reduced by lower yields (Kong et al., 2009).

6.5. Full accounting of GHG emissions

Management recommendations based only on direct emissions may not meet abatement objectives if they are based on techniques that increase emissions at any other point in the life cycle of the fertilizers in question. There is a distinct absence of comparisons of full upstream GHG emissions between synthetic and organic fertilizers for Mediterranean cropping systems. Research into downstream emissions has mainly focused on NO_3^- leaching, as NH_3 volatilization has hardly been studied at all. Furthermore, there is very little data on which fraction of this N finally forms N_2O, although the available evidence suggests that it may be very significant (Section 4.2). The simultaneous estimation of the emissions of all of the GHG involved in the GWP of cropping systems can be facilitated by modeling approaches. For example, the DAYCENT model has been shown to be capable of accurately predicting soil GHG emissions for a series of Mediterranean cropping systems (De Gryze et al., 2010), and DNDC is another interesting option (Lugato et al., 2010). At the farm or final product level, LCA methodologies should be adjusted to Mediterranean environments using a specific EF (Biswas et al., 2008, 2011).

7. Concluding remarks

The data reviewed suggest that organic fertilizers and water-saving techniques could reduce agricultural N_2O emissions under Mediterranean climatic conditions. However, the number of experimental sites at which emissions from organic and synthetic fertilizers have been compared is very limited. In the first part of our analysis, which included the majority of the published data, cumulative N_2O emissions were lower for solid organic fertilizers than for synthetic and liquid organic fertilizers. In a more detailed approach, a meta-analysis of compliant studies revealed that direct N_2O emissions after the application of organic fertilizers were lower than emissions after the addition of synthetic fertilizers (an average 23% was observed for cumulative emissions and EF). When organic fertilizers were segregated in solid and liquid materials, only solid fertilizer emissions were significantly lower than synthetic. When total N instead of available N was used for EF calculations, solid organic fertilizers achieved a 67% reduction in EF compared to synthetic fertilizers. A slower mineral N release from organic fertilizers could prevent high soil N levels prone to N_2O losses. Moreover, the differences observed seem to be related to the semi-arid features that are very common in Mediterranean soils, where C and N contents are usually low.

High-water irrigated systems showed the largest losses of fertilizer N as N_2O, whereas these losses were reduced under water-saving irrigation techniques (i.e. surface or subsurface drip irrigation), and were minimal in rainfed systems, in which the N_2O emission response to N fertilizers was reduced by one order of magnitude compared to conventional irrigation and Tier 1 IPCC EF.

Indirect N_2O emissions have not been fully accounted for, but sectorial information suggests a large reduction in N_2O emissions for organic fertilizers. Upstream of the cropping system, substantial reductions in fossil energy and N_2O emissions can be achieved, given that organic fertilizers usually employ waste materials that have not been produced specifically for this purpose. Downstream of the cropping system, Mediterranean data suggest that indirect N_2O emission savings could be achieved by using organic fertilizers on account of their reduced NO_3^- exports.

Options to enhance N_2O mitigation by organic fertilizers include: (i) water management strategies, comprising drip irrigation and rainfed systems; (ii) the minimization of fallow and bare soil emissions through the intensification of crop rotation and the use of cover crops; (iii) improved waste management to reduce indirect emissions, including waste separation at origin, decentralized livestock farming and biogas production; (iv) tightening the N cycle through N immobilization with woody residues in order to minimize emissions during fallow and low crop demand periods.

Identified limitations to the mitigation of N_2O emissions by organic fertilizers include: (i) the residual effect as organic fertilizers could prolong the N_2O emission period due to slower N release; (ii) the long-term addition of organic fertilizers to soil could enhance N_2O emissions through an increase in SOM content; (iii) yield-scaled performance, since organic fertilization could be linked to lower yields, although yield reduction is not the most common response to the substitution of synthetic N sources by organic ones in Mediterranean systems; (iv) the availability of total N from organic sources may be constrained by the land cost of legume cultivation. This land cost would be minimized if N fixing crops were cultivated during inter-cropping periods and N losses were reduced, but the extent to which organic fertilizers can replace synthetic ones is so far unknown.

We found a number of knowledge gaps that should be addressed by future field research: (i) experiment length is usually <12 months, which can lead to the underestimation of N_2O EF due to failure to account for the residual and long-term effects of fertilizers; (ii) background N_2O emissions sometimes represent a large percentage of fertilized treatment emissions; (iii) N_2O EF and yield-scaled EF were not always provided in the studies reviewed; (iv) some types of cropping system need to be studied in greater depth, including rainfed, drip-irrigated, woody perennial, greenhouse horticulture and organic farming; (v) more full cropping system GWP estimations are needed, including indirect emissions and other GHG.

Overall, this review has demonstrated that there is still potential to mitigate N_2O emissions in Mediterranean agriculture through the use of both organic fertilizers and low-water management systems. In the first case, the potential lies in comparatively lower N_2O fluxes of organic fertilizers with respect to the use of synthetic fertilizers and in the reduction of indirect emissions both upstream and downstream of the cropping system. In the second, the restriction of water availability that occurs in rainfed and drip-irrigated cropping systems has been shown to effectively reduce N_2O emissions by limiting the microbial processes responsible for N_2O production. Further research is needed to bridge the knowledge gaps in current information and to develop strategies that can fully exploit this N_2O reduction potential without burdening environmental and yield performance.

Acknowledgments

The authors are grateful to the Spanish Minister of Science and Innovation and the Autonomous Community of Madrid for their economic support through Projects AGL2009-08412-AGR and Agrisost (S2009/AGR-1630), respectively.

Thanks are due to the FIRE (Fédération Ile de France de Recherche en Environnement, CNRS and UPMC) and to the Research in Paris program (Paris City), within the framework in which Dr. Luis Lassaletta worked in France. Discussions with Adrian Leip contributed to the analysis. We are also grateful to Benjamín S. Gimeno, Javier Castrillo and Olga Martínez for their assistance, suggestions and comments.

References

Abdel Monem, M., Lindsay, W., Sommer, R., Ryan, J., 2010. Loss of nitrogen from urea applied to rainfed wheat in varying rainfall zones in northern Syria. Nutr. Cycl. Agroecosys. 86, 357–366.

Allaire-Leung, S.E., Wu, L., Mitchell, J.P., Sanden, B.L., 2001. Nitrate leaching and soil nitrate content as affected by irrigation uniformity in a carrot field. Agr. Water Manage. 48, 37–50.

Alonso, A.M., Guzmán, G.J., 2010. Comparison of the efficiency and use of energy in organic and conventional farming in Spanish agricultural systems. J. Sustain. Agr. 34, 312–338.

Altieri, R., Esposito, A., 2008. Olive orchard amended with two experimental olive mill wastes mixtures: effects on soil organic carbon, plant growth and yield. Bioresour. Technol. 99, 8390–8393.

Álvarez-Cobelas, M., Rojo, C., Angeler, D.G., 2005. Mediterranean limnology: current status, gaps and the future. J. Limnol. 64, 13–29.

Angás, P., Lampurlanés, J., Cantero-Martínez, C., 2006. Tillage and N fertilization: effects on N dynamics and Barley yield under semiarid Mediterranean conditions. Soil Till. Res. 87, 59–71.

Annicchiarico, P., Chiapparino, E., Perenzin, M., 2010. Response of common wheat varieties to organic and conventional production systems across Italian locations, and implications for selection. Field Crop Res. 116, 230–238.

Antoniadis, V., Tsadilas, C.D., Dalias, P., 2010. Evaluation of sewage sludge as a soil amendment in relation to nitrate leaching. Agrochimica 54, 91–102.

Aschmann, H., 1973. Distribution and peculiarity of Mediterranean ecosystems. In: Di Castri, F., Mooney, H.A. (Eds.), Mediterranean Type Ecosystems. Springer-Verlag, Berlin, pp. 11–19.

Baggs, E.M., Smales, C.L., Bateman, E.J., 2010. Changing pH shifts the microbial source as well as the magnitude of N_2O emission from soil. Biol. Fert. Soils 46, 793–805.

Barriga, J., Lassaletta, L., Moreno, A.G., 2010. Ground-living spiders assemblages from Mediterranean habitats under different management conditions. J. Arachnol. 38, 258–269.

Barton, L., Butterbach-Bahl, K., Kiese, R., Murphy, D.V., 2011. Nitrous oxide fluxes from a grain-legume crop (narrow-leafed lupin) grown in a semiarid climate. Global Change Biol. 17, 1153–1166.

Barton, L., Kiese, R., Gatter, D., Butterbach-Bahl, K., Buck, R., Hinz, C., Murphy, D.V., 2008. Nitrous oxide emissions from a cropped soil in a semi-arid climate. Global Change Biol. 14, 177–192.

Barton, L., Murphy, D.V., Kiese, R., Butterbach-Bahl, K., 2010. Soil nitrous oxide and methane fluxes are low from a bioenergy crop (canola) grown in a semi-arid climate. GCB Bioenergy 2, 1–15.

Beare, M.H., Gregorich, E.G., St-Georges, P., 2009. Compaction effects on CO_2 and N_2O production during drying and rewetting of soil. Soil Biol. Biochem. 41, 611–621.

Bilalis, D., Karkanis, A., Triantafyllidis, V., Ladavos, A., Bizos, D., Patsiali, S., Efthimiadou, A., Papatheohari, Y., 2010a. Effects of organic and inorganic fertilization on growth, yield and nicotine content of flue-cured and oriental tobacco (*Nicotiana tabacum* L.) seedlings grown in organic and conventional float system. J. Food Agric. Environ. 8, 585–589.

Bilalis, D., Patsiali, S., Karkanis, A., Konstantas, A., Makris, M., Efthimiadou, A., 2010b. Effects of cultural system (organic and conventional) on growth and fiber quality of two cotton (*Gossypium hirsutum* L.) varieties. Renew. Agr. Food Syst. 25, 228–235.

Birch, H.F., 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10 (1), 19–31.

Biswas, W.K., Barton, L., Carter, D., 2008. Global warming potential of wheat production in Western Australia: a life cycle assessment. Water Environ. J. 22, 206–216.

Biswas, W.K., Barton, L., Carter, D., 2011. Biodiesel production in a semiarid environment: a life cycle assessment approach. Environ. Sci. Technol. 45, 3069–3074.

Boellstorff, D., Benito, G., 2005. Impacts of set-aside policy on the risk of soil erosion in central Spain. Agric. Ecosyst. Environ. 107, 231–243.

Borenstein, M., Rothstein, H., 1999. Comprehensive Meta-analysis: A Computer Program for Research Synthesis. Biostat, Englewood, NJ.

Borenstein, M., Hedges, L.V., Higgins, J.P.T., Rothstein, H.R., 2009. Introduction to Meta-analysis. John Wiley & Sons, Ltd.

Breiner, J., Gimeno, B.S., Fenn, M., 2007. Calculation of theoretical and empirical nutrient N critical loads in the mixed conifer ecosystems of southern California. ScientificWorldJournal 7, 198–205.

Buresh, R., Vlek, P., Harmsen, K., 1990. Fate of fertilizer nitrogen applied to wheat under simulated Mediterranean environmental conditions. Nutr. Cycl. Agroecosys. 23, 25–36.

Burger, M., Jackson, L.E., 2003. Microbial immobilization of ammonium and nitrate in relation to ammonification and nitrification rates in organic and conventional cropping systems. Soil Biol. Biochem. 35, 29–36.

Burger, M., Jackson, L.E., Lundquist, E.J., Louie, D.T., Miller, R.L., Rolston, D.E., Scow, K.M., 2005. Microbial responses and nitrous oxide emissions during wetting and drying of organically and conventionally managed soil under tomatoes. Biol. Fert. Soils 42, 109–118.

Calderón, F.J., Jackson, L.E., Scow, K.M., Rolston, D.E., 2000. Microbial responses to simulated tillage in cultivated and uncultivated soils. Soil Biol. Biochem. 32, 1547–1559.

Campiglia, E., Mancinelli, R., Radicetti, E., Marinari, S., 2011. Legume cover crops and mulches: effects on nitrate leaching and nitrogen input in a pepper crop (*Capsicum annuum* L.). Nutr. Cycl. Agroecosys. 89, 399–412.

Caporali, F., Onnis, A., 1992. Validity of rotation as an effective agroecological principle for a sustainable agriculture. Agric. Ecosyst. Environ. 41, 101–113.

Cardenas, L.M., Thorman, R., Ashlee, N., Butler, M., Chadwick, D., Chambers, B., Cuttle, S., Donovan, N., Kingston, H., Lane, S., Dhanoa, M.S., Scholefield, D., 2010. Quantifying annual N_2O emission fluxes from grazed grassland under a range of inorganic fertiliser nitrogen inputs. Agric. Ecosyst. Environ. 136, 218–226.

Celik, I., 2009. Effects of long-term organic amendments and mycorrhyza application on saturated hydraulic conductivity and nitrate leaching in a Typic Xerofluvent. Fresen. Environ. Bull. 18, 2312–2322.

Chapuis-Lardy, L., Wrage, N., Metay, A., Chotte, J.L., Bernoux, M., 2007. Soils, a sink for N_2O? A review. Global Change Biol. 13, 1–17.

Clark, M.S., Horwath, W.R., Shennan, C., Scow, K.M., Lantni, W.T., Ferris, H., 1999. Nitrogen, weeds and water as yield-limiting factors in conventional, low-input, and organic tomato systems. Agr. Ecosyst. Environ. 73, 257–270.

CORINAIR, 2006. http://reports.eea.europa.eu/EMEPCORINAIR4/en/B1010vs4.0.pdf

Coskan, A., Gok, M., Onac, I., Inal, I., Saglamtimur, T., 2002. The effect of wheat straw, corn straw and tobacco residues on denitrification losses in a field planted with wheat. Turk. J. Agric. For. 26, 349–353.

Curuk, S., Sermenli, T., Mavi, K., Evrendilek, F., 2004. Yield and fruit quality of watermelon (*Citrullus lanatus* (Thumb.) Matsum. & Nakai.) and melon (*Cucumis melo* L.) under protected organic and conventional farming systems in a Mediterranean region of Turkey. Biol. Agric. Hortic. 22, 173–183.

Davidson, E.A., 2009. The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. Nat. Geosci. 2, 659–662.

Davidson, E.A., Matson, P.A., Vitousek, P.M., Riley, R., Dunkin, K., Garciamendez, G., Maass, J.M., 1993. Processes regulating soil emissions of NO and N_2O in a seasonally dry tropical forest. Ecology 74, 130–139.

De Backer, E., Aertsens, J., Vergucht, S., Steurbaut, W., 2009. Assessing the ecological soundness of organic and conventional agriculture by means of life cycle assessment (LCA): a case study of leek production. Brit. Food J. 111, 1028–1061.

De Gryze, S., Wolf, A., Kaffka, S.R., Mitchell, J., Rolston, D.E., Temple, S.R., Lee, J., Six, J., 2010. Simulating greenhouse gas budgets of four California cropping systems under conventional and alternative management. Ecol. Appl. 20, 1805–1819.

Denison, R.F., Bryant, D.C., Kearney, T.E., 2004. Crop yields over the first nine years of LTRAS, a long-term comparison of field crop systems in a Mediterranean climate. Field Crop Res. 86, 267–277.

Deria, A.M., Bell, R.W., O'Hara, G.W., 2003. Organic wheat production and soil nutrient status in a Mediterranean climatic zone. J. Sustain. Agr. 21, 21–47.

Di Giacomo, G., Taglieri, L., 2009. Renewable energy benefits with conversion of woody residues to pellets. Energy 34, 724–731.

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401–422.

Dick, J., Kaya, B., Soutoura, M., Skiba, U., Smith, R., Niang, A., Tabo, R., 2008. The contribution of agricultural practices to nitrous oxide emissions in semi-arid Mali. Soil Use Manage. 24, 292–301.

Díez, J.A., Román, R., Caballero, R., Caballero, A., 1997. Nitrate leaching from soils under a maize-wheat-maize sequence, two irrigation schedules and three types of fertilisers. Agric. Ecosyst. Environ. 65, 189–199.

Díez, J.A., Caballero, R., Román, R., Tarquis, A., Cartagena, M.C., Vallejo, A., 2000. Integrated fertilizer and irrigation management to reduce nitrate leaching in Central Spain. J. Environ. Qual. 29, 1539–1547.

Díez, J.A., Hernaiz, P., Munoz, M.J., de la Torre, A., Vallejo, A., 2004. Impact of pig slurry on soil properties, water salinization, nitrate leaching and crop yield in a four-year experiment in Central Spain. Soil Use Manage. 20, 444–450.

Dobbie, K.E., McTaggart, I.P., Smith, K.A., 1999. Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. J. Geophys. Res. -Atmos. 104, 26891–26899.

Dobbie, K.E., Smith, K.A., 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. Global Change Biol. 9, 204–218.

Drinkwater, L.E., Letourneau, D.K., Workneh, F., Vanbruggen, A.H.C., Shennan, C., 1995. Fundamental differences between conventional and organic tomato agroecosystems in California. Ecol. Appl. 5, 1098–1112.

Efthimiadou, A., Bilalis, D., Karkanis, A., Froud-Williams, B., Eleftherochorinos, I., 2009. Effects of cultural system (organic and conventional) on growth, photosynthesis and yield components of sweet corn (*Zea mays* L.) under semi-arid environment. Not. Bot. Horti. Agrobo. 37, 104–111.

Fedorov, S., 2002. GetData Graph Digitizer version 2.24. Available at www.getdata-graph-digitizer.com

Firestone, M.K., Davidson, E.A., 1989. Microbiological basis of NO and N_2O production and consumption in soils. In: Andreae, M.O., Schimel, D.S. (Eds.), Exchanges of Trace Gases Between Terrestrial Ecosystems and the Atmosphere. John Wiley & Sons, New York.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In: Press, C.U. (Ed.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M., Manning, Z., Chen, M., Marquis, K.B., Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, United Kingdom and New York, NY, USA.

Frimpong, K.A., Baggs, E.M., 2010. Do combined applications of crop residues and inorganic fertilizer lower emission of N_2O from soil? Soil Use Manage. 26, 412–424.

Gabriel, J.L., Quemada, M., 2011. Replacing bare fallow with cover crops in a maize cropping system: yield, N uptake and fertiliser fate. Eur. J. Agron. 34, 133–143.

Galbally, I.E., Kirstine, W.V., Meyer, C.P., Wang, Y.P., 2008. Soil-atmosphere trace gas exchange in semiarid and arid zones. J. Environ. Qual. 37, 599–607.

García, C., Vallejo, A., Díez, J.A., García, L., Cartagena, M.C., 1997. Nitrogen use efficiency with the application of controlled release fertilizers coated with kraft pine lignin. Soil Sci. Plant Nutr. 43, 443–449.

García-Martín, A., López-Bellido, R.J., Coleto, J.M., 2007. Fertilisation and weed control effects on yield and weeds in durum wheat grown under rain-fed conditions in a Mediterranean climate. Weed Res. 47, 140–148.

García-Ruiz, R., Baggs, E.M., 2007. N_2O emission from soil following combined application of fertiliser-N and ground weed residues. Plant Soil 299, 263–274.

Garland, G.M., Suddick, E., Burger, M., Horwath, W.R., Six, J., 2011. Direct N_2O emissions following transition from conventional till to no-till in a cover cropped Mediterranean vineyard (*Vitis vinifera*). Agric. Ecosyst. Environ. 141, 234–239.

Garnier, J., Billen, G., Vilain, G., Martinez, A., Silvestre, M., Mounier, E., Toche, F., 2009. Nitrous oxide (N_2O) in the Seine river and basin: observations and budgets. Agric. Ecosyst. Environ. 133, 223–233.

Gomes, J., Bayer, C., Costa, F.D., Piccolo, M.D., Zanatta, J.A., Vieira, F.C.B., Six, J., 2009. Soil nitrous oxide emissions in long-term cover crops-based rotations under subtropical climate. Soil Till. Res. 106, 36–44.

Gómez, J.A., Guzmán, M.G., Giráldez, J.V., Fereres, E., 2009. The influence of cover crops and tillage on water and sediment yield, and on nutrient, and organic matter losses in an olive orchard on a sandy loam soil. Soil Till. Res. 106, 137–144.

González-Fernández, I., Bermejo, V., Elvira, S., Sanz, J., Gimeno, B.S., Alonso, R., 2010. Modelling annual pasture dynamics: application to stomatal ozone deposition. Atmos. Environ. 44, 2507–2517.

Gregorich, E.G., Rochette, P., VandenBygaart, A.J., Angers, D.A., 2005. Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil Till. Res. 83, 53–72.

Guzmán Casado, G.I., González de Molina, M., 2009. Preindustrial agriculture versus organic agriculture: the land cost of sustainability. Land Use Policy 26, 502–510.

Haag, D., Kaupenjohann, M., 2001. Landscape fate of nitrate fluxes and emissions in Central Europe – a critical review of concepts, data, and models for transport and retention. Agric. Ecosyst. Environ. 86, 1–21.

Hargreaves, J.C., Adl, M.S., Warman, P.R., 2008. A review of the use of composted municipal solid waste in agriculture. Agric. Ecosyst. Environ. 123, 1–14.

Hedges, L.V., Gurevitch, J., Curtis, P.S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150–1156.

Heller, H., Bar-Tal, A., Tamir, G., Bloom, P., Venterea, R.T., Chen, D., Zhang, Y., Clapp, C.E., Fine, P., 2010. Effects of manure and cultivation on carbon dioxide and nitrous oxide emissions from a corn field under mediterranean conditions. J. Environ. Qual. 39, 437–448.

Herencia, J.F., Ruiz-Porras, J.C., Melero, S., Garcia-Galavis, P.A., Morillo, E., Maqueda, C., 2007. Comparison between organic and mineral fertilization for soil fertility levels, crop macronutrient concentrations, and yield. Agron. J. 99, 973–983.

Hoben, J.P., Gehl, R.J., Millar, N., Grace, P.R., Robertson, G.P., 2011. Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. Global Change Biol. 17, 1140–1152.

Holtz, B.A., Caesar-Ton That, T.C., 2004. Wood chipping almond brush to reduce air pollution as a sustainable alternative to burning that enhances soil quality and microbial diversity. In: Lartey, R.T., Caesar, A.J. (Eds.), Emerging Concepts in Plant Health Management. Research Signpost, Kerala, India, pp. 159–185.

Horvath, L., Grosz, B., Machon, A., Tuba, Z., Nagy, Z., Czobel, S.Z., Balogh, J., Peli, E., Foti, S.Z., Weidinger, T., Pinter, K., Fuehrer, E., 2010. Estimation of nitrous oxide emission from Hungarian semi-arid sandy and loess grasslands; effect of soil parameters, grazing, irrigation and use of fertilizer. Agric. Ecosyst. Environ. 139, 255–263.

Hungate, B.A., van Groenigen, K.J., Six, J., Jastrow, J.D., Lue, Y.Q., de Graaff, M.A., van Kessel, C., Osenberg, C.W., 2009. Assessing the effect of elevated carbon dioxide on soil carbon: a comparison of four meta-analyses. Global Change Biol. 15, 2020–2034.

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories. Vol. 4. Agriculture, Forestry and Other Land Use. Intergovernmental Panel on Climate Change, IGES, Japan.

IPCC, 2006. Guidelines for National Greenhouse Gas Inventories.Vol. 3. Industrial Processes and Product Use. Intergovernmental Panel on Climate Change, IGES, Japan.

Jones, S.K., Rees, R.M., Skiba, U.M., Ball, B.C., 2007. Influence of organic and mineral N fertiliser on N_2O fluxes from a temperate grassland. Agric. Ecosyst. Environ. 121, 74–83.

Kallenbach, C.M., Rolston, D.E., Horwath, W.R., 2010. Cover cropping affects soil N_2O and CO_2 emissions differently depending on type of irrigation. Agric. Ecosyst. Environ. 137, 251–260.

Karamanos, A.J., Bilalis, D., Sidiras, N., 2004. Effects of reduced tillage and fertilization practices on soil characteristics, plant water status, growth and yield of upland cotton. J. Agron. Crop Sci. 190, 262–276.

Kavargiris, S.E., Mamolos, A.P., Tsatsarelis, C.A., Nikolaidou, A.E., Kalburtji, K.L., 2009. Energy resources' utilization in organic and conventional vineyards: energy flow, greenhouse gas emissions and biofuel production. Biomass Bioenerg. 33, 1239–1250.

Kim, M.H., Kim, J.W., 2010. Comparison through a LCA evaluation analysis of food waste disposal options from the perspective of global warming and resource recovery. Sci. Total Environ. 408, 3998–4006.

Kinney, C.A., Mandernack, K.W., Mosier, A.R., 2005. Laboratory investigations into the effects of the pesticides mancozeb, chlorothalonil, and prosulfuron on nitrous oxide and nitric oxide production in fertilized soil. Soil Biol. Biochem. 37, 837–850.

Kool, D.M., Dolfing, J., Wrage, N., Van Groenigen, J.W., 2011. Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. Soil Biol. Biochem. 43, 174–178.

Kong, A.Y.Y., Fonte, S.J., van Kessel, C., Six, J., 2007. Soil aggregates control N cycling efficiency in long-term conventional and alternative cropping systems. Nutr. Cycl. Agroecosys. 79, 45–58.

Kong, A.Y.Y., Fonte, S.J., van Kessel, C., Six, J., 2009. Transitioning from standard to minimum tillage: trade-offs between soil organic matter stabilization, nitrous oxide emissions, and N availability in irrigated cropping systems. Soil Till. Res. 104, 256–262.

Kong, A.Y.Y., Hristova, K., Scow, K.M., Six, J., 2010. Impacts of different N management regimes on nitrifier and denitrifier communities and N cycling in soil microenvironments. Soil Biol. Biochem. 42, 1523–1533.

Kramer, S.B., Reganold, J.P., Glover, J.D., Bohannan, B.J.M., Mooney, H.A., 2006. Reduced nitrate leaching and enhanced denitrifier activity and efficiency in organically fertilized soils. Proc. Natl. Acad. Sci. U.S.A. 103, 4522–4527.

Kroeze, C., Seitzinger, S.P., 1998. The impact of land use on N_2O emissions from watersheds draining into the Northeastern Atlantic Ocean and European Seas. In: Nitrogen, the Confer-N-S, pp. 149–158.

Kroodsma, D.A., Field, C.B., 2006. Carbon sequestration in California agriculture, 1980-2000. Ecol. Appl. 16, 1975–1985.

Laegreid, M., Aastveit, A.H., 2002. N_2O emission from fertilizer use. In: Non-CO_2 Greenhouse Gases: Scientific Understanding, Control Options and Policy Aspects, pp. 233–238.

Lassaletta, L., Garcia-Gomez, H., Gimeno, B.S., Rovira, J.V., 2009. Agriculture-induced increase in nitrate concentrations in stream waters of a large Mediterranean catchment over 25 years (1981-2005). Sci. Total Environ. 407, 6034–6043.

Lassaletta, L., Garcia-Gomez, H., Gimeno, B.S., Rovira, J.V., 2010. Headwater streams: neglected ecosystems in the EU water framework directive. Implications for nitrogen pollution control. Environ. Sci. Policy 13, 423–433.

Laverman, A.M., Garnier, J.A., Mounier, E.M., Roose-Amsaleg, C.L., 2010. Nitrous oxide production kinetics during nitrate reduction in river sediments. Water Res. 44, 1753–1764.

Laini, A., Bartoli, M., Castaldi, S., Viaroli, P., Capri, E., Trevisan, M., 2011. Greenhouse gases (CO_2, CH_4 and N_2O) in lowland springs within an agricultural impacted watershed (Po River Plain, northern Italy). Chem. Ecol. 27, 177–187.

Laville, P., Lehuger, S., Loubet, B., Chaumartin, F., Cellier, P., 2011. Effect of management, climate and soil conditions on N_2O and NO emissions from an arable crop rotation using high temporal resolution measurements. Agr. Forest Meteorol. 151, 228–240.

Lee, J., Six, J., King, A.P., Van Kessel, C., Rolston, D.E., 2006. Tillage and field scale controls on greenhouse gas emissions. J. Environ. Qual. 35, 714–725.

Lee, J., Hopmans, J.W., van Kessel, C., King, A.P., Evatt, K.J., Louie, D., Rolston, D.E., Six, J., 2009. Tillage and seasonal emissions of CO_2, N_2O and NO across a seed bed and at the field scale in a Mediterranean climate. Agric. Ecosyst. Environ. 129, 378–390.

Leip, A., Busto, M., Winiwarter, W., 2011. Developing spatially stratified N_2O emission factors for Europe. Environ. Pollut. 159, 3223–3232.

Li, C., Frolking, S., Butterbach-Bahl, K., 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Climatic Change 72, 321–338.

Lithourgidis, A.S., Matsi, T., Barbayiannis, N., Dordas, C.A., 2007. Effect of liquid cattle manure on corn yield, composition, and soil properties. Agron. J. 99, 1041–1047.

Liu, C.Y., Wang, K., Meng, S.X., Zheng, X.H., Zhou, Z.X., Han, S.H., Chen, D.L., Yang, Z.P., 2011. Effects of irrigation, fertilization and crop straw management on nitrous oxide and nitric oxide emissions from a wheat-maize rotation field in northern China. Agric. Ecosyst. Environ. 140, 226–233.

López-Bellido, L., López-Bellido, R.J., Castillo, J.E., López-Bellido, F.J., 2000. Effects of tillage, crop rotation, and nitrogen fertilization on wheat under rainfed Mediterranean conditions. Agron. J. 92, 1054–1063.

López-Fernández, S., Díez, J.A., Hernaiz, P., Arce, A., García-Torres, L., Vallejo, A., 2007. Effects of fertiliser type and the presence or absence of plants on nitrous oxide emissions from irrigated soils. Nutr. Cycl. Agroecosys. 78, 279–289.

Lugato, E., Zuliani, M., Alberti, G., Vedove, G.D., Gioli, B., Miglietta, F., Peressotti, A., 2010. Application of DNDC biogeochemistry model to estimate greenhouse gas emissions from Italian agricultural areas at high spatial resolution. Agric. Ecosyst. Environ. 139, 546–556.

Madejón, E., López, R., Murillo, J.M., Cabrera, F., 2001. Agricultural use of three (sugar-beet) vinasse composts: effect on crops and chemical properties of a Cambisol soil in the Guadalquivir river valley (SW Spain). Agric. Ecosyst. Environ. 84, 55–65.

Maraseni, T.N., Cockfield, G., Maroulis, J., 2010. An assessment of greenhouse gas emissions: implications for the Australian cotton industry. J. Agric. Sci. 148, 501–510.

Marinari, S., Lagomarsino, A., Moscatelli, M.C., Di Tizio, A., Campiglia, E., 2010. Soil carbon and nitrogen mineralization kinetics in organic and conventional three-year cropping systems. Soil Till. Res. 109, 161–168.

Martín-Rueda, I., Muñoz-Guerra, L.M., Yunta, F., Esteban, E., Tenorio, J.L., Lucena, J.J., 2007. Tillage and crop rotation effects on barley yield and soil nutrients on a Calciortidic Haploxeralf. Soil Till. Res. 92, 1–9.

McSwiney, C.P., Snapp, S.S., Gentry, L.E., 2010. Use of N immobilization to tighten the N cycle in conventional agroecosystems. Ecol. Appl. 20, 648–662.

Meijide, A., Díez, J.A., Sánchez-Martín, L., López-Fernández, S., Vallejo, A., 2007. Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. Agric. Ecosyst. Environ. 121, 383–394.

Meijide, A., Garcia-Torres, L., Arce, A., Vallejo, A., 2009. Nitrogen oxide emissions affected by organic fertilization in a non-irrigated Mediterranean barley field. Agric. Ecosyst. Environ. 132, 106–115.

Melero, S., Porras, J.C.R., Herencia, J.F., Madejon, E., 2006. Chemical and biochemical properties in a silty loam soil under conventional and organic management. Soil Till. Res. 90, 162–170.

Melero, S., Pérez-de-Mora, A., Murillo, J.M., Buegger, F., Kleinedam, K., Kublik, S., Vanderlinden, K., Moreno, F., Schloter, M., 2011. Denitrification in a vertisol under long-term tillage and no-tillage management in dryland agricultural systems: key genes and potential rates. Appl. Soil Ecol. 47, 221–225.

Menéndez, S., Lopez-Bellido, R.J., Benitez-Vega, J., González-Murua, C., López-Bellido, L., Estavillo, J.M., 2008. Long-term effect of tillage, crop rotation and N fertilization to wheat on gaseous emissions under rainfed Mediterranean conditions. Eur. J. Agron. 28, 559–569.

Mondini, C., Cayuela, M.L., Sinicco, T., Cordaro, F., Roig, A., Sanchez-Monedero, M.A., 2007. Greenhouse gas emissions and carbon sink capacity of amended soils evaluated under laboratory conditions. Soil Biol. Biochem. 39, 1366–1374.

Montanaro, G., Celano, G., Dichio, B., Xiloyannis, C., 2009. Effects of soil-protecting agricultural practices on soil organic carbon and productivity in fruit tree orchards. Land Degrad. Dev. 21, 132–138.

Montemurro, F., 2010. Are organic N fertilizing strategies able to improve lettuce yield, use of nitrogen and N status? J. Plant Nutr. 33, 1980–1997.

Montemurro, F., 2009. Different nitrogen fertilization sources, soil tillage, and crop rotations in winter wheat: effect on yield, quality, and nitrogen utilization. J. Plant Nutr. 32, 1–18.

Montemurro, F., Canali, S., Convertini, G., Ferri, D., Tittarelli, F., Vitti, C., 2008. Anaerobic digestates application on fodder crops: effects on plant and soil. Agrochimica 52, 297–312.

Montemurro, F., Maiorana, M., Convertini, G., Ferri, D., 2007. Effect of different N fertilisation strategies on yield, plant N status, N uptake and soil characteristics in a maize-barley rotation. Agrochimica 51, 40–54.

Montemurro, F., Maiorana, M., Convertini, G., Ferri, D., 2006. Compost organic amendments in fodder crops: effects on yield, nitrogen utilization and soil characteristics. Compost. Sci. Util. 14, 114–123.

Montemurro, F., Maiorana, M., Convertini, G., Fornaro, F., 2005. Improvement of soil properties and nitrogen utilisation of sunflower by amending municipal solid waste compost. Agron. Sustain. Dev. 25, 369–375.

Morra, L., Pagano, L., Iovieno, P., Baldantoni, D., Alfani, A., 2010. Soil and vegetable crop response to addition of different levels of municipal waste compost under Mediterranean greenhouse conditions. Agron. Sustain. Dev. 30, 701–709.

Muhammad, W., Vaughan, S.M., Dalal, R.C., Menzies, N.W., 2011. Crop residues and fertilizer nitrogen influence residue decomposition and nitrous oxide emission from a Vertisol. Biol. Fert. Soils 47, 15–23.

Nekomoto, K., Hoshiba, S., Tamura, Y., Kato, H., Uchida, T., Matsumoto, K., Morita, S., 2006. Reduction of nitrogen surplus by material circulation between dairy farms and arable crop farms in the mixed production area, Japan. Int. Congr. Ser. 1293, 272–275.

Ochoa-Hueso, R., Allen, E.B., Branquinho, C., Cruz, C., Dias, T., Fenn, M.E., Manrique, E., Esther Perez-Corona, M., Sheppard, L.J., Stock, W.D., 2011. Nitrogen deposition effects on Mediterranean-type ecosystems: An ecological assessment. Environ. Pollut. 159, 2265–2279.

Oenema, O., Witzke, H.P., Klimont, Z., Lesschen, J.P., Velthof, G.L., 2009. Integrated assessment of promising measures to decrease nitrogen losses from agriculture in EU-27. Agric. Ecosyst. Environ. 133, 280–288.

Omonode, R.A., Smith, D.R., Gal, A., Vyn, T.J., 2011. Soil nitrous oxide emissions in corn following three decades of tillage and rotation treatments. Soil Sci. Soc. Am. J. 75 (1), 152–163.

Pardo, G., Cavero, J., Aibar, J., Zaragoza, C., 2009. Nutrient evolution in soil and cereal yield under different fertilization type in dryland. Nutr. Cycl. Agroecosys. 84, 267–279.

Peña-Haro, S., Llopis-Albert, C., Pulido-Velazquez, M., Pulido-Velazquez, D., 2010. Fertilizer standards for controlling groundwater nitrate pollution from agriculture: El Salobral-Los Llanos case study, Spain. J. Hydrol. 392, 174–187.

Peñuelas, J., Sardans, J., Alcaniz, J.M., Poch, J.M., 2009. Increased eutrophication and nutrient imbalances in the agricultural soil of NE Catalonia, Spain. J. Environ. Biol. 30, 841–846.

Petersen, S.O., Henriksen, K., Mortensen, G.K., Krogh, P.H., Brandt, K.K., Sorensen, J., Madsen, T., Petersen, J., Gron, C., 2003. Recycling of sewage sludge and household compost to arable land: fate and effects of organic contaminants, and impact on soil fertility. Soil Till. Res. 72, 139–152.

Petersen, S.O., Regina, K., Pollinger, A., Rigler, E., Valli, L., Yamulki, S., Esala, M., Fabbri, C., Syvasalo, E., Vinther, F.P., 2006. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. Agric. Ecosyst. Environ. 112, 200–206.

Petersen, S.O., Sommer, S.G., Béline, F., Burton, C., Dach, J., Dourmad, J.Y., Leip, A., Misselbrook, T., Nicholson, F., Poulsen, H.D., Provolo, G., Sørensen, P., Vinnerás, B., Weiske, A., Bernal, M.P., Böhm, R., Juhász, C., Mihelic, R., 2007. Recycling of livestock manure in a whole-farm perspective. Livestock Science 112, 180–191.

Prapaspongsa, T., Poulsen, T.G., Hansen, J.A., Christensen, P., 2010. Energy production, nutrient recovery and greenhouse gas emission potentials from integrated pig manure management systems. Waste Manage. Res. 28, 411–422.

Ramírez, C.A., Worrell, E., 2006. Feeding fossil fuels to the soil: an analysis of energy embedded and technological learning in the fertilizer industry. Resour. Conserv. Recycl. 46, 75–93.

Rana, G., Mastrorilli, M., 1998. Ammonia emissions from fields treated with green manure in a Mediterranean climate. Agr. Forest Meteorol. 90, 265–274.

Rinnofner, T., Friedel, J.K., de Kruijff, R., Pietsch, G., Freyer, B., 2008. Effect of catch crops on N dynamics and following crops in organic farming. Agron. Sustain. Dev. 28, 551–558.

Rodríguez-Lizana, A., Espejo-Perez, A.J., Gonzalez-Fernandez, P., Ordonez-Fernandez, R., 2008. Pruning residues as an alternative to traditional tillage to reduce erosion and pollutant dispersion in olive groves. Water, Air, Soil Pollut. 193, 165–173.

Ryan, J., Ibrikci, H., Sommer, R., McNeill, A., 2009. Nitrogen in rainfed and irrigated cropping systems in the mediterranean region. Adv. Agron. 104, 53–136.

Ryden, J.C., Lund, L.J., 1980a. Nitrous-oxide evolution from irrigated land. J. Environ. Qual. 9, 387–393.

Ryden, J.C., Lund, L.J., 1980b. Nature and extent of directly measured denitrification losses from some irrigated vegetable crop production units. Soil Sci. Soc. Am. J. 44, 505–511.

Sakala, W.D., Cadisch, G., Giller, K.E., 2000. Interactions between residues of maize and pigeonpea and mineral N fertilizers during decomposition and N mineralization. Soil Biol. Biochem. 32, 679–688.

Salmerón, M., Cavero, J., Quilez, D., Isla, R., 2010. Winter cover crops affect monoculture maize yield and nitrogen leaching under irrigated Mediterranean conditions. Agron. J. 102, 1700–1709.

Sánchez-Martín, L., Vallejo, A., Dick, J., Skiba, U.M., 2008a. The influence of soluble carbon and fertilizer nitrogen on nitric oxide and nitrous oxide emissions from two contrasting agricultural soils. Soil Biol. Biochem. 40, 142–151.

Sánchez-Martín, L., Arce, A., Benito, A., García-Torres, L., Vallejo, A., 2008b. Influence of drip and furrow irrigation systems on nitrogen oxide emissions from a horticultural crop. Soil Biol. Biochem. 40, 1698–1706.

Sánchez-Martín, L., Meijide, A., García-Torres, L., Vallejo, A., 2010a. Combination of drip irrigation and organic fertilizer for mitigating emissions of nitrogen oxides in semiarid climate. Agric. Ecosyst. Environ. 137, 99–107.

Sánchez-Martín, L., Sanz-Cobena, A., Meijide, A., Quemada, M., Vallejo, A., 2010b. The importance of the fallow period for N_2O and CH_4 fluxes and nitrate leaching in a Mediterranean irrigated agroecosystem. Eur. J. Soil Sci. 61, 710–720.

Sánchez-Martín, L., Dick, J., Bocary, K., Vallejo, A., Skiba, U.M., 2010c. Residual effect of organic carbon as a tool for mitigating nitrogen oxides emissions in semi-arid climate. Plant Soil 326, 137–145.

Sanz, A., Misselbrook, T.H., Sanz, M.J., Vallejo, A., 2010. Use of an inverse dispersion technique for estimating ammonia emission from surface-applied slurry. Agric. Ecosyst. Environ. 44, 999–1002.

Sanz-Cobena, A., Misselbrook, T.H., Arce, A., Mingot, J.I., Díez, J.A., Vallejo, A., 2008. An inhibitor of urease activity effectively reduces ammonia emissions from soil treated with urea under Mediterranean conditions. Agric. Ecosyst. Environ. 126, 243–249.

Schaufler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton, M.A., Zechmeister-Boltenstern, S., 2010. Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. Eur. J. Soil Sci. 61, 683–696.

Scheer, C., Wassmann, R., Klenzler, K., Lbragimov, N., Eschanov, R., 2008. Nitrous oxide emissions from fertilized irrigated cotton (*Gossypium hirsutum* L.) in the Aral Sea Basin, Uzbekistan: influence of nitrogen applications and irrigation practices. Soil Biol. Biochem. 40, 290–301.

Singh, R.P., Agrawal, M., 2008. Potential benefits and risks of land application of sewage sludge. Waste Manage. 28, 347–358.

Six, J., Ogle, S.M., Breidt, F.J., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long term. Global Change Biol. 10, 155–160.

Skiba, U., Drewer, J., Tang, Y.S., van Dijk, N., Helfter, C., Nemitz, E., Famulari, D., Cape, J.N., Jones, S.K., Twigg, M., Pihlatie, M., Vesala, T., Larsen, K.S., Carter, M.S., Ambus, P., Ibrom, A., Beier, C., Hensen, A., Frumau, A., Erisman, J.W., Brüggemann, N., Gasche, R., Butterbach-Bahl, K., Neftel, A., Spirig, C., Horvath, L., Freibauer, A., Cellier, P., Laville, P., Loubet, B., Magliulo, E., Bertolini, T., Seufert, G., Andersson, M., Manca, G., Laurila, T., Aurela, M., Lohila, A., Zechmeister-Boltenstern, S., Kitzler, B., Schaufler, G., Siemens, J., Kindler, R., Flechard, C., Sutton, M.A., 2009. Biosphere-atmosphere exchange of reactive nitrogen and greenhouse gases at the NitroEurope core flux measurement sites: measurement strategy and first data sets. Agric. Ecosyst. Environ. 133, 139–149.

Smil, V., 1999. Crop residues: agriculture's largest harvest – crop residues incorporate more than half of the world agricultural phytomass. Bioscience 49, 299–308.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., 2007. Agriculture. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric. Ecosyst. Environ. 133, 247–266.

Sofo, A., Palese, A.M., Casacchia, T., Celano, G., Ricciuti, P., Curci, M., Crecchio, C., Xiloyannis, C., 2010. Genetic, functional, and metabolic responses of soil microbiota in a sustainable olive orchard. Soil Sci. 175, 81–88.

Soumare, M.D., Mnkeni, P.N.S., Khouma, M., 2002. Effects of *Casuarina equisetifolia* composted litter and ramial-wood chips on tomato growth and soil properties in Niayes, Senegal. Biol. Agric. Hort. 20, 111–123.

Spokas, K., Wang, D., Venterea, R., Sadowsky, A., 2006. Mechanisms of N_2O production following chloropicrin fumigation. Appl. Soil Ecol. 31, 101–109.

Steenwerth, K., Belina, K.M., 2008. Cover crops and cultivation: impacts on soil N dynamics and microbiological function in a Mediterranean vineyard agroecosystem. Appl. Soil Ecol. 40, 370–380.

Steenwerth, K.L., Belina, K.M., 2010. Vineyard weed management practices influence nitrate leaching and nitrous oxide emissions. Agric. Ecosyst. Environ. 138, 127–131.

Stehfest, E., Bouwman, L., 2006. N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosys. 74, 207–228.

Teixeira, C., Magalhaes, C., Boaventura, R.A.R., Bordalo, A.A., 2010. Potential rates and environmental controls of denitrification and nitrous oxide production in a temperate urbanized estuary. Mar. Environ. Res. 70, 336–342.

Tejada, M., Gonzalez, J.L., 2006. Crushed cotton gin compost effects on soil biological properties, nutrient leaching losses, and maize yield. Agron. J. 98, 749–759.

Tirado, R., Gopikrishna, S.R., Krishnan, R., Smith, P., 2010. Greenhouse gas emissions and mitigation potential from fertilizer manufacture and application in India. Int. J. Agric. Sustain. 8, 176–185.

Tognetti, R., Palladino, M., Minnocci, A., Delfine, S., Alvino, A., 2003. The response of sugar beet to drip and low-pressure sprinkler irrigation in southern Italy. Agr. Water Manage. 60, 135–155.

Tortosa, G., Correa, D., Sánchez-Raya, A.J., Delgado, A., Sánchez-Monedero, M.A., Bedmar, E.J., 2011. Effects of nitrate contamination and seasonal variation on the denitrification and greenhouse gas production in La Rocina Stream (Doñana National Park, SW Spain). Ecol. Eng. 37, 539–548.

Townsend-Small, A., Pataki, D.E., Czimczik, C.I., Tyler, S.C., 2011. Nitrous oxide emissions and isotopic composition in urban and agricultural systems in southern California. J. Geophys. Res. -Biogeo. 116, G01013.

Underwood, E.C., Klausmeyer, K.R., Cox, R.L., Busby, S.M., Morrison, S.A., Shaw, M.R., 2009. Expanding the global network of protected areas to save the imperiled Mediterranean biome. Conserv. Biol. 23, 43–52.

USDA, 2011. U.S. certified organic farmland acreage, livestock numbers, and farm operations. USDA Economic Research Service Data Sets. Retrieved June, 2011, from http://www.ers.usda.gov/Data/Organic/

Vallejo, A., Díez, J.A., López-Valdivia, L.M., Cartagena, M.C., Tarquis, A., Hernaiz, P., 2004. Denitrification from an irrigated soil fertilized with pig slurry under Mediterranean conditions. Biol. Fert. Soils 40, 93–100.

Vallejo, A., García-Torres, L., Díez, J.A., Arce, A., López-Fernández, S., 2005. Comparison of N losses (NO_3^-, N_2O, NO) from surface applied, injected or amended (DCD) pig slurry of an irrigated soil in a Mediterranean climate. Plant Soil 272, 313–325.

Vallejo, A., Skiba, U.M., García-Torres, L., Arce, A., López-Fernández, S., Sánchez-Martín, L., 2006. Nitrogen oxides emission from soils bearing a potato crop as influenced by fertilization with treated pig slurries and composts. Soil Biol. Biochem. 38, 2782–2793.

Van Cleemput, O., 1998. Subsoils: chemo- and biological denitrification, N_2O and N_2 emissions. Nutr. Cycl. Agroecosys. 52, 187–194.

Van Groenigen, J.W., Velthof, G.L., Oenema, O., Van Groenigen, K.J., Van Kessel, C., 2010. Towards an agronomic assessment of N_2O emissions: a case study for arable crops. Eur. J. Soil Sci. 61, 903–913.

Vilain, G., Garnier, J., Tallec, G., Cellier, P., 2010. Effect of slope position and land use on nitrous oxide (N_2O) emissions (Seine Basin, France). Agr. Forest Meteorol. 150, 1192–1202.

Wiens, M.J., Entz, M.H., Wilson, C., Ominski, K.H., 2008. Energy requirements for transport and surface application of liquid pig manure in Manitoba, Canada. Agr. Syst. 98, 74–81.

Willer, H., Kilcher, L. (Eds.), 2011. TheWorld of Organic Agriculture – Statistics and Emerging Trends 2011. IFOAM–FiBL, Bonn–Frick.

Wrage, N., Velthof, G.L., van Beusichem, M.L., Oenema, O., 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33, 1723–1732.

Wyland, L.J., Jackson, L.E., Chaney, W.E., Klonsky, K., Koike, S.T., Kimple, B., 1996. Winter cover crops in a vegetable cropping system: impacts on nitrate leaching, soil water, crop yield, pests and management costs. Agric. Ecosyst. Environ. 59, 1–17.

Zalidis, G., Stamatiadis, S., Takavakoglou, V., Eskridge, K., Misopolinos, N., 2002. Impacts of agricultural practices on soil and water quality in the Mediterranean region and proposed assessment methodology. Agric. Ecosyst. Environ. 88, 137–146.