



ASSOCIATION OF  
COMPOST  
PRODUCERS  
"We Build Healthy Soil"



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February 12, 2016

Karen Ross, Secretary  
California Department of Food and Agriculture  
1220 N Street  
Sacramento, CA 95814

**Subject: Draft Report for the Environmental Farming Act Science Advisory Panel,  
Agronomic rates of compost application for California croplands and  
Rangelands to support a CDFA Healthy Soils Incentives Program (Version  
1.0 – 1/5/2016)**

Dear Ryan,

The Association of Compost Producers (ACP) is pleased to offer these comments on the Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and Rangelands to support a CDFA Healthy Soils Incentives Program (Version 1.0 – 1/5/2016) ("Draft"). ACP is a non-profit trade association founded in 1995 representing over 100 public agency and private company compost producer members in California. ACP's mission is *to increase the quality, value and amount of compost being produced and used in California.*

In general, we are in broad agreement with the importance of providing incentives for the use of compost to build healthy soils in California. Our members fully understand that there are environmentally beneficial reasons for building healthy soil in California, e.g. carbon sequestration, water conservation and water quality. However, these potential benefits are currently external to the immediate economic benefits of crop production. So the implementation of the of the CDFA Healthy Soils Incentives Program will help to engage growers to use compost to help them experience directly *both* the economic *and* the environmental benefits of using compost on their working lands. The opportunity to sequester on average 0.42 MTCO<sub>2</sub>e/ton of compost feedstock used to manufacture and apply on working lands,<sup>1</sup> will be a very significant way for Californian's to not only reduce greenhouse gases, by building soil organic matter (SOM), but will also deliver the many other benefits that compost provides to California's environment.

<sup>1</sup> Method for Estimating Greenhouse Gas Emission Reductions from Compost From Commercial Organic Waste, November 14, 2011, California Air Resources Board, CalEPA., [http://www.arb.ca.gov/cc/protocols/localgov/pubs/compost\\_method.pdf](http://www.arb.ca.gov/cc/protocols/localgov/pubs/compost_method.pdf)

The Draft is an excellent start to promoting the extensive use of compost for building SOM. It will begin to lay a foundation for implementing the Healthy Soils Initiative "Short Term Action Plan", of providing "...incentives for improved soil management practices..."<sup>2</sup> However, this program should be continuously evaluated and upgraded as knowledge is gained and applied to using compost for building healthy soil. That is, we shouldn't take this Draft as the last word to provide guidance for delivering the incentives, rather it is the first jump off point from which further healthy soil metrics, research and best management practices can provide an ongoing springboard for compost use innovation, outreach and healthy soil market development.

Some specific recommendations for further development in this Draft include:

- **Compost "agronomic rates" have a limited definition and utility** – The concept and practice of "agronomic rates" as used in the Draft is based on standard practice recommended "rates" using synthetic fertilizer, i.e. predominantly only nitrogen compounds in the absence of carbon compounds. The sequestration, modification and transformation of those nutrients within high organic content soils is being extensively studied via many research programs nationwide, and locally in the UC Davis ASI California Nitrogen Assessment (CNA).<sup>3</sup> Without going into detail here, new data is showing that a number of parameters related to nitrogen dynamics, e.g. whether it is sequestered in living biological microorganisms or tissues (in the form of protein molecules), whether it's converted to the potent greenhouse gas, N<sub>2</sub>O, whether or not it leaches into the groundwater or runs off in surface water, etc., are all affected by the total amount and type of carbon compounds and materials that are present in the SOM in the soil. In general, the more SOM, above 3%, or more by weight, with no apparent limit up to as much as 50%, will literally change the fate of nitrogen in the soil and water as it feeds the plants. So because "agronomic rate," as it's currently construed, do not take this new information about the total carbon and the C:N ratio dynamics into account, *we see the rates proposed in the Draft as a minimum, and could likely be greatly increased.* The limiting factor will be the availability of compost, not the concern about adding too much nitrogen. This should be further monitored and studied as you implement the program.
- **No mention of "soil biology" and dependence on physical, chemical constituents:** A corollary to the above discussion about rates, is the need for a deeper understanding and associated best management practices to include the role of soil biology as it affects those rates, when using compost. While this is certainly beyond the scope of developing efficient incentives for using more compost, nevertheless, the granting of the incentives to growers is a perfect opportunity to educate both growers and compost producers about the value and mechanisms as to why building healthy soil with compost is so important, and why it should become part of all future cultivation practices. These incentives will convert more growers to the sustained use of compost, and thereby the sustained building of California's soils, as they fully understand and tangibly realize its many benefits. One of the main mechanisms of healthy soil is derives from the fact that it's living... it's not merely dead dirt. It's not simply a physical and chemical environmental to hold plants onto the ground while they are watered and fed with nutrients. Rather, healthy soil is a living ecosystem. So with the delivery of the financial incentive payments, it's a perfect opportunity to extend the knowledge of the use, mechanisms and benefits to the grower. This will provide a deeper understanding and appreciation to the growers. It might include conveying that biology as

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<sup>2</sup> [www.cdfa.ca.gov/EnvironmentalStewardship/pdfs/ShortTermActions.pdf](http://www.cdfa.ca.gov/EnvironmentalStewardship/pdfs/ShortTermActions.pdf)

<sup>3</sup> [asi.ucdavis.edu/programs/sarep/research-initiatives/are/nutrient-mgmt/california-nitrogen-assessment](http://asi.ucdavis.edu/programs/sarep/research-initiatives/are/nutrient-mgmt/california-nitrogen-assessment)

what makes healthy soil, and optimizes the three independent variables of any soil-water-plant cultivation system. Examples of this here, provide also a basis for creating simple field-based metrics and monitoring protocols for measuring and tracking the development of healthy soil ongoing:

- **Soil health & richness** – can be defined as the spectrum & dynamics of:
  - carbon compounds - quick carbon (food-sugars & starches), slow (food), geologic/inorganic carbon (biochar, CaCO<sub>3</sub>, etc.)
  - nitrogen compounds - ammonia, urea, nitrates, proteins/amino acids, even nitrous oxide, a powerful SLCP
  - other "fertilizers" and macro and micro nutrients
  - inorganic elements and compounds (key metals and trace minerals)
  - gases ("arable soil" or "tilth" esp. O<sub>2</sub>, N<sub>2</sub>, CO<sub>2</sub>, N<sub>2</sub>O, many others)
  - biology (bacteria, fungi, nematodes, etc. and biological succession as a basis of the above macro and micro nutrient dynamics)
- **Water health & abundance:** water use efficiency by conserving water via holding it in the root zone (in the cellulosic carbon compounds in the compost), and water quality (esp. soluble nitrogen compounds) by holding water and not letting it run off, and/or filtering it through the healthy soil horizon. (for an extensive reference list and discussion of this see CalRecycle's "Compost Mulch Water Conservation Lit Review-Jan. 2015" and "Organics-Water Conservation Reference Summary - 3.27.15" included along with this letter.
- **Plant health & productivity:** agronomic rate will depend, ultimately, on the individual cultivar. This was not included in the incentive program Draft. But since these "agronomic rates" are a lower limit (as discussed above), this important consideration can be added soon in future programs. As stated in the EFA SAP meeting (January 15, 2016), ACP is currently developing some simple "Healthy Soil Metrics" that growers can use in the field to both inform growers about the details of healthy soil as well as provide a simple and efficient system for measuring and using compost to achieve multiple values, the primary being one of crop productivity and food, fiber or feed product quality.
- **Use only quality "compost":** Unfortunately, without using compost that has been tested, the grower will not know the qualities of it, and whether it is appropriate for us on their particular crop or rangeland. So the program should begin with the main standard for compost, as defined in CalRecycle regulations, but should extend beyond that to include the Seal of Testing Assurance (STA) program developed and implemented by the US Composting Council.<sup>4</sup>
- **Engage with compost producers ongoing:** Since the Draft is particularly oriented to compost production and use, we urge CDFA to work ongoing with the ACP members, and the rest of the compost producer industry, to ensure that 1) quantities and qualities of compost are being produced that best serve this incentive program, and 2) industry works ongoing with growers to ensure that the objectives of the Healthy Soil Initiative are being met in perpetuity

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<sup>4</sup> <http://compostingcouncil.org/seal-of-testing-assurance/>

**Association of Compost Producers**  
*"We Build Healthy Soil"*

Again, thank you very much for the opportunity to provide this important input of the compost producer industry experience and perspectives on the use of compost to implement the CDFA Healthy Soils Incentives Program. We remain ready and willing to work with the CDFA implementation team on an ongoing basis to enhance both California soils to better improve our atmosphere and water environment while ever better supporting our world leading agricultural industry.

Sincerely,

A handwritten signature in black ink, appearing to read "Dan Noble". The signature is fluid and cursive, with a long horizontal stroke at the end.

Dan Noble, ACP Exec. Dir.  
Jeff Ziegenbein, ACP President



February 17, 2016

Amrith Gunasekara  
Science Advisor to the Secretary  
California Department of Food and Agriculture  
1220 N Street  
Sacramento, CA 95814

Subject: Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program

Dear Dr. Gunasekara:

The California Association of Sanitation Agencies (CASA) respectfully submits comments on the draft white paper establishing application rates for compost on croplands and rangeland. We understand the announced deadline for submitting comments has passed, but request that our input be considered as part of the ongoing dialogue to implement the Healthy Soils Initiative.

CASA is an association of local agencies, engaged in advancing the recycling of wastewater into usable water, maximizing beneficial use of biosolids, generating renewable energy, and producing other valuable resources. Through these efforts we help create a clean and sustainable environment for Californians. CASA members are actively engaged as partners with the state to fulfill a number of mandates and initiatives intended to mitigate climate change impacts. These include: (1) providing 50% of the state's energy needs from renewable sources; (2) reducing carbon dioxide equivalent emissions to 1990 levels; (3) reducing the carbon intensity of transportation fuel used in the state by 10%; (4) recycling 75% of the solid waste generated in the state; (5) reducing the release of short lived climate pollutants; and (6) advancing the objectives of the Healthy Soils Initiative through the recycling of biosolids to agricultural land.

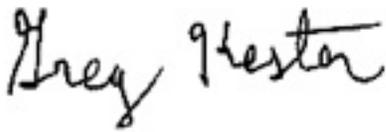
CASA fully supports the creation of financial incentives to expand the use of compost and other organic soil amendments in support of the Healthy Soils Initiative. We do have several questions and comments relative to the proposed application rates for compost on cropland and rangeland in California. They include:

1. Are the recommended application rates advisory or mandatory?
2. Will the application rates apply to biosolids compost?
3. Why are the application rates limited to 15% of the nitrogen need of the crop grown? Is it expected that synthetic fertilizer will be applied in order to make up the deficiency? If so, why not follow the biosolids model and allow for application of compost up to the nitrogen need of the crop to be grown?

4. Is there no concern about high C:N ratio compost which may cause short term nitrogen deficiency in crops?
5. Why are rates so limited for rangeland? There is much work from Utah, Colorado and elsewhere documenting the benefits of biosolids to overgrazed rangeland and it would appear that California could also benefit from such application.

We very much look forward to continued proactive work with CDFA on this and other issues related to the Healthy Soils Initiative. Please contact me at [gkester@casaweb.org](mailto:gkester@casaweb.org) or at 916-844-5262 with any response or for further clarification.

Sincerely,

A handwritten signature in black ink that reads "Greg Kester". The signature is written in a cursive, slightly slanted style.

Greg Kester  
Director of Renewable Resource Programs

cc: Secretary Karen Ross – CDFA  
Deputy Secretary Jenny Lester Moffitt – CDFA  
Bobbi Larson – Executive Director CASA  
Howard Levenson - CalRecycle

February 12, 2016

Dear Environmental Farming Act Science Advisory Panel, and California Department of Food and Agriculture:

I write to comment on the document *Agronomic rates of compost application for California croplands and rangelands* that was produced in support of the CDFA Healthy Soils Incentives Program. I am a professor of rangeland ecology and management at the University of California with thirty years of experience on California rangelands. I am concerned with the lack of scientific information about potential impacts to biodiversity, and would like to make some suggestions to reduce the risk to California's rangelands. I have reviewed the rangeland-related document and some of its supporting literature carefully.

California's rangelands are 40% of the state's land area, including more than 40 million acres of grassland and woodland. The replacement of the native grassland, brought about by the introduction of aggressive non-natives, cultivation, and livestock production during colonization, has created a grassland that is different in structure and timing than the original grasslands. In many places, perennial bunchgrasses have been replaced or reduced. In drier areas, there may originally have been a higher proportion of broadleaved species and native annuals before contact (Holstein 2011). An estimated 20 million acres of California rangelands are in private ownership (California Department of Forestry and Fire Protection–Fire and Resource Assessment Program 2003), around two-thirds of these are grazed by livestock (Forero et al. 1990; Huntsinger et al. 2010), and livestock grazing has been a widespread use for around 200 years depending on the region (Burcham 1982). These “novel” rangelands have been recognized worldwide as significant for biodiversity conservation as part of the California floristic province biodiversity hot spot (Myers et al. 2000). They provide extensive viewshed, including wildflower displays, watershed, food production, and wildlife habitat, including habitat for pollinators critical to California agriculture (Chaplin-Kramer et al. 2011).

I believe that California rangelands are worthy of great caution. California has highly heterogeneous soils and topography, creating unique patterns of vegetation in many places, and phenomena such as vernal pools. The diversity of California rangelands is to some degree a function of differing soils and nutrient conditions throughout the grassland. This diversity of plant communities is of course important to California's abundant wildlife. Grasses seem to benefit disproportionately from fertilization and this can lead to changes in vegetation composition and structure and in wildlife habitat. Among other things, the risk to wildflower displays and pollinators is obvious (Weiss 1999). Finally, if the proposed addition of compost increases biomass production, as it seems likely to do, without grazing this may increase fire risk, and wildfires release massive amounts of carbon.

The compost program looks promising, and benefits in terms of carbon sequestration are well supported in published studies carried out in California. However, there is not one published paper from California on diversity impacts used to develop the application recommendations. A personal communication based on a California study in review comes from Ryals et al, whose work I greatly respect, but looking up the latest paper, from 2015, nothing is said about impacts on plant diversity. (I am unable to review the personal comment as it is not included in the supporting documents.) As to the papers, again, none is based on research in California, and some are from work in extremely different climatic regimes. Borrowing from significantly different areas provides an insufficient basis for widespread application in California environments. The compost research itself in California is mostly from two spots. Great caution should be used in extending these few papers to all of California's unique rangelands.

Because of the potential benefits of the program for landowners and carbon sequestration, I have some suggestions for it. Sites where compost is applied should be monitored to evaluate the impacts and benefits to soil and plants and ecosystem services from rangelands. Second, until there is sufficient completed work to support it, applications should be limited to rangeland soils that have been previously tilled or subjected to major soil disturbance. Because of a history of failed homesteads, these areas are not uncommon in California rangelands. There may be other situations that could be evaluated for the initial, testing period of this program that will reduce potential risk to diversity. Application sites should be chosen carefully, to avoid potential impacts to wildlife, including pollinators, and rare plant communities. Carbon budgets that include the emissions from transportation, processing, and application should be developed to evaluate whether or not net carbon benefits are achieved through this substantial public investment. Further, application should not be made unless there is a reasonable assurance that livestock grazing will be present to reduce fuels. I urge a cautiously optimistic approach.

Nothing has been more promoted in the management literature of late than an adaptive approach to management. To adapt, information about results must be collected. To proceed responsibly, this has to be part of the process. The California rangeland community has avidly pursued management that creates a spectrum of ecosystem services. Wildlife, biodiversity, oaks, watershed, food production, and watershed have all benefited from a multifunctional approach. Compost application needs to be developed in concert with maintaining the full spectrum of environmental benefits from California rangelands.

Sincerely,



Lynn Huntsinger

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February 12, 2016

TO: Karen Ross, Secretary, California Department of Food and Agriculture

RE: Draft Report for the Environmental Farming Act Science Advisory Panel,  
Agronomic rates of compost application for California croplands and rangelands to  
support a CDFA Healthy Soils Incentives Program (Version 1.0 – 1/5/2016)

Dear Secretary Ross,

We appreciate the opportunity to comment on CDFA's *Draft Agronomic Rates of Compost Application for California Croplands and Rangelands*, which was presented to the Environmental Farming Act Science Advisory Panel to support a CDFA Healthy Soils Incentives Program. We are following up on comments provided during the workshop.

Firstly, we are expressing our strong support for CDFA to put incentives in place to build carbon in our agricultural soils through compost applications. We commend CDFA for their leadership with regards to the Healthy Soil Initiative and we would like to express our support for the agency's effort to create a cost share incentive program for the use of compost in our agricultural crop and range land systems to build soil organic matter.

We do, however, have several concerns about some of the information provided in the report and workshop discussion. In particular, we feel the report provides mixed messages regarding the use of compost, primarily in its questionable portrayal as a significant source of N migration into surface or ground water. While there may be overall concerns regarding total N applied to enhance the fertility of the soils, the addition of compost discussed is a very small contributor (5-10% of total N), with little discussion of the benefits of increased organic matter and microbes that will aid in the stabilization of all nutrients, significantly increasing plant availability and minimizing their migration in the environment. With 90-95% of the nutrient load on agricultural lands coming from other (often synthetic) sources, nutrient migration should not be identified as a limiting factor to the expanded use of finished compost. Conversely, with the increased tillage of the soil provided by the use of compost, fertilizer application can be curtailed, making it a part of the solution to efficient nutrient management.

Fundamentally, we understand a desire to take a conservative approach in moving this concept forward, and looking at potential environmental impacts is a part of such assessment. We believe that the draft study has failed to recognize some of the essential benefits of compost application on the basis of a lack of qualified study work during the literature review. It would be helpful to the stakeholder community to better understand where CDFA believes that information gaps exist – what additional, perhaps California-specific study work needs to be conducted to best support maximizing the benefits of compost application to our native soils. Even

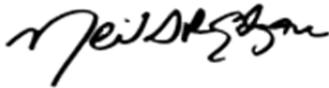
more helpful would be a prioritized list of the outstanding issues so that efforts may be undertaken to facilitate immediate action to resolve these voids and fully realize the climate change goals of the Healthy Soils Initiative through a more robust program.

While we agree that there is “too much variation in the scientific data within both “croplands” and “compost” to define a single application rate”, we believe the current methodology of solely using the C/N ratio is overly simplistic and does not adequately explain the expected nutrient availability or release from the applied materials. Furthermore, the study unreasonably limits the proposed application rates to well below what is considered typical or recommended usage for actual field applications in current practice.

We have fully review comments provided on this matter by Dr. Jeff Creque of the Carbon Cycle Institute and wholly support his analysis and recommendations, in addition to those provided in this letter.

We look forward to continued discussion in the development of the Healthy Soils Initiative and will continue efforts to help secure Greenhouse Gas Reduction Fund allocations proposed in the Governor’s Budget to support this worthy program.

Sincerely,

A handwritten signature in black ink, appearing to read "Neil S.R. Edgar". The signature is fluid and cursive, with the first name "Neil" being the most prominent part.

Neil S.R. Edgar  
Executive Director

# Carbon Cycle Institute

**TO:** Karen Ross, Secretary, California Department of Food and Agriculture

**RE:** Draft Report for the Environmental Farming Act Science Advisory Panel, *Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program (Version 1.0 – 1/5/2016)*

Dear Secretary Ross:

The Carbon Cycle Institute appreciates the opportunity to comment upon the Draft Report, *Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program (Draft)*. We strongly support CDFA plans to implement a cost-share incentive program for compost applications to working lands in California.

Together, soils and vegetation constitute the Terrestrial Carbon Pool, the only carbon pool on Earth that can be readily managed to store additional carbon with beneficial results for human ecology and the biosphere as a whole. Increasing the carbon content of soils leads to greater agricultural sustainability and resilience, increases soil water holding capacity, and helps ensure food security, especially in the context of our rapidly destabilizing climate system (Lal 2015) and California's vulnerability to recurrent drought.

As noted by the International Panel on Climate Change (IPCC 2014):

“.... climate change resulting from CO<sub>2</sub> emissions is irreversible on a multi-century to millennial time scale, ***except in the case of a large net removal of CO<sub>2</sub> from the atmosphere over a sustained period.***”

The opportunity to sequester significant quantities of atmospheric carbon dioxide (CO<sub>2</sub>) as soil organic carbon (SOC) in working lands of the state –with all the attendant ancillary benefits of doing so- must be realized if California is to meet its greenhouse gas (GHG) reduction goals.

## **Compost and Composting**

Compost is a biochemically stable product resulting from the managed, aerobic, thermophilic decomposition of organic (carbon-based) materials, suitable for beneficial application to soils (CalRecycle 2006). Compost feedstocks include many materials which otherwise represent significant sources of methane (CH<sub>4</sub>) due to their anaerobic decomposition under business as usual scenarios, including manures from livestock waste holding ponds and organic wastes diverted from landfills. Similarly, many organic materials end up as black carbon, nitrogen oxides and CO<sub>2</sub> when open burning is used as a disposal technique.

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Compost produced through a managed, aerobic process represents a particularly potent GHG reduction strategy by: 1) avoiding production of short-lived climate pollutants (CH<sub>4</sub>, N<sub>2</sub>O and black carbon) associated with alternative waste management strategies; 2) optimizing conservation of photosynthetically fixed carbon within compost feedstocks via the controlled decomposition ecology of the compost environment; 3) enabling direct application of beneficial high-carbon amendments to soils; 4) displacing synthetic sources of nutrients and avoiding their attendant water quality and GHG costs, and 5) minimizing agroecosystem nutrient losses, including nitrous oxide (N<sub>2</sub>O) volatilization and nitrate (NO<sub>3</sub>) leaching, through the tightening of soil nutrient cycles associated with increased soil organic SOC (Bowles et al 2015).

Of critical significance for the Governor's Healthy Soils Initiative, the use of compost in crop, pasture and rangeland ecosystems offers the most rapid means of directly increasing SOC, through direct additions of stable, beneficial soil organic matter (SOM), enabling the rapid elevation of SOC to levels that would take decades or more to achieve through agroecosystem management alone. Compost further provides necessary plant nutrients in organic form, helping to displace the use of nutrient cycle-forcing synthetic fertilizers, including nitrogen (N), a significant source of the potent GHG, N<sub>2</sub>O. Compost also offers a means of transferring biomass and associated nutrients from areas of excess to areas of deficit, greatly facilitating the recycling and balancing of nutrients at landscape and regional scales.

While it is true, as stated in the Draft, that "There is considerable variation among organic growers in the use of compost for plant nutrient provision," the greatest value of compost is not provision of plant nutrients per se, but to provide solar energy –in the form of photosynthetically fixed organic carbon- to the soil ecosystem. This biologically fixed solar energy drives soil processes that in turn support plant nutrition, soil health and agricultural productivity. It is precisely the distinction between the conventional "limiting factor" approach and the "soil organic matter" approach to soil fertility that, classically, defines the difference between "organic" and "conventional" agricultures (Fukuoka 1985, Rodale 1960, Steiner 1958, Turner 1951, Sykes 1949, Howard 1943, Balfour 1943).

Despite these paradigmatic distinctions, we strongly support incentivizing compost application on conventional and organic croplands equally. Because most of the working land soils of the state have the capacity to hold significantly more organic carbon than they currently do (Kong et al 2005), the promise of the Healthy Soils Initiative can best be realized by optimizing SOC increases wherever possible. We urge CDFA, as it establishes incentives for compost application, to look beyond compost nutrient content to embrace the understanding that it is SOC, as embodied solar energy, that drives soil ecological processes and thus soil-plant-water relations within the agricultural ecosystem (figure 1).

### **Compost Application Frequency**

Compost application frequency is an important compost utilization parameter critically missing from the Draft. Intensive cropping systems, whether organic or conventional, tend to be both more destructive of SOM through tillage and have a higher overall nutrient

demand. The incentivized compost application rate should, therefore, pertain to *each cropping cycle*. For example, a producer growing four vegetable crops per year should be able to access incentives to apply compost up to four times per year, depending upon soil conditions and crop requirements. Alternatively, a winter wheat producer, with a summer cover crop, for example, might need to apply compost only one or two times annually to maintain an increase in SOC over time (Kong et al 2005).

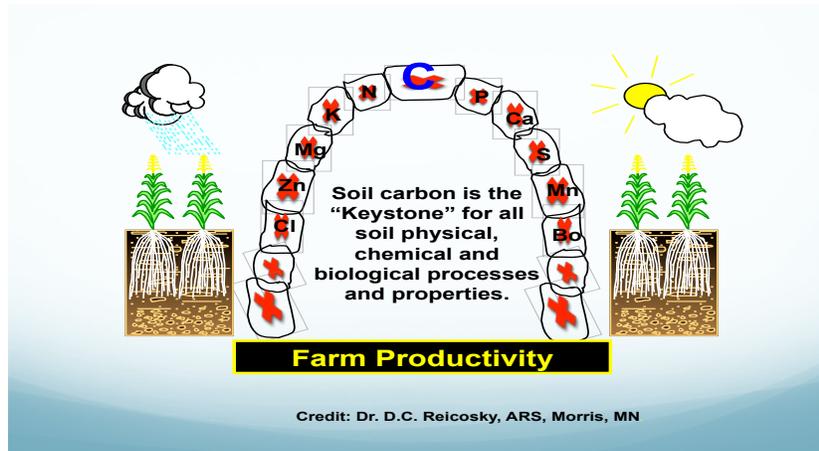


Figure 1: Carbon as the keystone for soil processes

Incentivized application of compost to grazed rangelands, on the other hand (as distinct from pastures), should probably be limited to once every ten years or so, based on evidence of persistence of the effects of compost application in grazed rangeland ecosystems (Ryals and Silver 2013, Ryals et al, 2015). Exceptions to this general rule of thumb may certainly be justified under conditions of rangeland degradation, where compost application may be warranted for several years during the restoration process. CDFA may wish to consider an upper incentive threshold of, for example, 4% SOM on rangelands, based on the lower inherent productivity of these systems relative to croplands or pastures.

### Compost Application Rate

We concur with the Draft that there is too much variability among both cropping systems and soils to define a single compost application rate. However, much of the variability among composts can be addressed by defining compost explicitly as finished compost produced in accordance with Cal Recycle standards. Compost eligible for the program *should conform with CalRecycle standards*, whether or not it is divided into further categories based upon C:N ratio, N content, or other criteria.

We disagree with the distinction made within the Draft that:

“use of compost with C:N greater than 11 should be thought of as a practice that is *in addition to* the nutrient management system on the farm, whereas

compost with C:N less than 11 should be thought of as a practice that is *part of* the nutrient management system on the farm.”

This distinction fails to recognize the importance of carbon as the keystone element in soil fertility and crop production (figure 1), thereby directly undermining the core tenant of the Soil Health Initiative.

If compost is used, it is part of the farm nutrient management system, regardless of its C:N ratio. Ultimately, lower C:N composts may help drive photosynthetic capture of additional soil C more quickly, while higher C:N composts may help retain, and recycle, available N within the system (Bowles et al 2015) while also accelerating the rate of accumulation of more recalcitrant forms of SOC (Ryals et al 2015). Higher soil carbon levels can lead to significantly tighter plant-soil nutrient cycles, reducing N susceptibility to losses via leaching or volatilization and providing adequate plant nutrition despite soil analyses showing insufficient levels of available soil N (Bowles et al 2015). Over time, soils with higher organic carbon content will also tend to have higher organic N content and thus be able to provide an increasing percentage of slowly released N to meet crop demand, as organic N is rendered gradually available through dynamic soil biochemical processes, helping to remove the need for synthetic N inputs.

### **Nitrogen as a Limiting Factor in Compost Application Rates**

The N content of compost, along with its phosphorous and potassium contributions, provides an opportunity to incentivize reductions in the amount of synthetic fertilizer applied to California’s agricultural systems (Bowles et al 2015, ARB 2011). As currently structured, however, the Draft proposes a compost application rate that would supply less than 10% of first year crop demand for N. By way of comparison, NRCS requires at least a 15% reduction in synthetic N to recognize N<sub>2</sub>O emission reductions under Conservation Practice Standard 590 (COMET-Planner).

At a minimum, this suggests CDFA should increase its incentivized cropland application rate by at least 50%, –to at least 12 tons of compost per acre- to at least match the minimum 15% reduction required under NRCS 590 synthetic N reduction use standard. However, the broader question remains: why is CDFA basing its cropland compost incentive on meeting only 5.55% (C/N > 11/1) or 9.7 % (C/N < 11/1) of crop demand for N (Draft)? With such low rates, CDFA misses both the opportunity to reduce emissions associated with synthetic fertilizer manufacture and use, and to increase soil C directly with significant compost additions (Kong et al 2005). It seems both reasonable and conservative to incentivize a compost application rate that would provide at least 25% or even 50% of first year crop demand for N (assuming concomitant reductions in synthetic N use where it occurs).

Fears raised in the Draft and by members of the public around surface water pollution caused by runoff from compost should be allayed by the well-documented use of compost in soil erosion prevention and degraded site mitigation (Risse 2012, ARB 2011) at rates of 140 tons/acre and above (CIWMB 2007). Cal Recycle’s significant body of research on this

issue is notably absent from the Draft, and should be reviewed for data relevant to this question (CalRecycle 2009, CIWMB 2007).

Based upon issues discussed above, we encourage CDFA to bring its high C/N compost definition in line with CalRecycle (2009) guidelines for compost (C/N  $\geq$  14/1-20/1), and increase incentivized application rates for such compost to 20 (twenty) dry tons per acre, *per cropping cycle*.

### **Environmental Impact of Greatest Concern**

The potential environmental impact of greatest concern for the Healthy Soils Initiative must be GHG reductions. To this end, incentivized compost use rates should be maximized with respect to both avoided emissions (displacement of synthetic N sources, and CH<sub>4</sub> and N<sub>2</sub>O avoidance) and CO<sub>2</sub> sequestration as SOC. While it is certainly true that organic N can be a source of ground and surface water contamination and N<sub>2</sub>O emissions- excess ecosystem N is driven primarily by the anthropogenic forcing of the N-cycle through the manufacture and use of synthetic N fertilizers (CAN 2015).

While some percentage of the N contained in compost –particularly that derived from conventional livestock manures- may originate from synthetic sources, using compost in place of synthetics enables recycling of already-fixed nutrients, rather than further forcing global nutrient cycles through manufacture and use of additional synthetics. While we recognize the role of synthetic fertilizers in modern conventional cropping systems, the use of organic forms of nutrients, and particularly of biologically stable compost, should be incentivized to relieve, to the maximum degree possible, continued forcing of global nutrient cycles and associated environmental pollution, including GHG emissions and surface and ground water contamination.

Overall, 69% of the N added annually to cropland statewide is derived from synthetic fixation (CAN 2015). It is this ongoing infusion of manufactured N fertilizer into California's working land soils that is forcing the state's N-cycle, providing much of the excess available N driving ground and surface water pollution and increasing atmospheric N<sub>2</sub>O as a potent GHG. Cropland soils and manure management together represent 32% of N<sub>2</sub>O emissions in the state (CAN 2015), and N input is the most reliable proxy for calculating N<sub>2</sub>O emissions (VCS 2013).

Leaching from cropland represents 88% of N input to groundwater, with roughly one third of that from dairy manure (CAN 2015). By reducing use of synthetic N fertilizers, demand for livestock manures, and particularly composted manures, can be increased, leading to greater spatial distribution of manure nutrients, less overall N entering the environment, and less loss of N to the environment.

N in manure and inorganic fertilizers is generally highly labile, while compost N is largely complexed with carbon, leading to slower mineralization rates (Ryals and Silver 2013, Sikora and Szmids 2001, Eghball 2000). This means compost application rates based on *total* N may need to be much higher than equivalent rates of manures or synthetic

fertilizers in order to meet crop demand for *labile* N from compost alone (Delonge et al, 2013). Comparisons of compost with inorganic fertilizers should consider available N, rather than total N, in the compost.

Recent work by Bowles et al (2015) illustrates the complexity of soil-plant-water N dynamics, and confirms that soils with higher organic matter content can provide sufficient crop N while appearing deficient in “available N” when evaluated by common agronomic metrics. Tightening of N cycling, from organic matter to bacteria to plant, or directly from organic matter to plant, means N losses, whether through leaching or volatilization, can be virtually absent from higher carbon soils.

As noted by Rosenstock et al (2013), “Overuse of nitrogen fertilizer threatens the health of the state’s agricultural, human and natural resources.” Excess N can speed decomposition (Parton et al. 2007) and thereby lower (Khan et al. 2007) or maintain (et al. 2009) soil C stocks that might otherwise increase. Because SOC is the primary carbon pool of concern for the Healthy Soils Initiative, the high potential for applications of synthetic N fertilizers to result in a net decline in SOC underscores another important consideration in the evaluation of relative impacts of compost versus synthetic fertilizers.

### **Rangelands**

As noted in the Draft, “Concerns about the impact of compost addition on rangeland plant diversity are grounded in studies that have documented significant changes in plant community composition in response to synthetic N fertilizer addition.” Critically, *these studies do not reflect actual impacts of compost application on rangelands*. To allay fears of those unfamiliar with compost and compost use, we support a site-specific risk assessment approach to CDFA-incentivized compost application to rangelands. To this end, CDFA-supported use of compost on rangelands should occur within a controlled context, such as that prescribed by the ACR grazed grassland compost protocol, or NRCS Conservation Practice 590 (or other NRCS Conservation Practice Standard), providing abundant opportunity to define the precise circumstances under which CDFA-supported compost application on rangeland would actually occur, including the C:N ratio of the compost applied. For example, the American Carbon Registry Protocol for Compost Additions to Grazed Grassland explicitly excludes Serpentine soils, Histosols, and intact native plant communities from consideration for compost application (Haden et al 2014).

Climate change and increasing concentrations of GHG in the atmosphere are already driving undesirable changes on global rangelands, including desertification and the loss of soil carbon (Koteen et al 2011; FAO 2009, Lal 2004;). California rangelands consequently are likely to benefit from stable carbon-based soil amendments such as compost (Koteen et al 2011, Ryals et al 2015). Given the significant role of California’s rangelands as a source of much of the state’s water supply, it is critical that the capacity of these lands to capture and filter water is supported and, where possible, improved, particularly in the face of increasing probability of drought and high intensity rainfall events associated with climate destabilization. Compost has been repeatedly shown to provide both water filtering and water absorption benefits (CIWMB 2007). While we support a cautionary approach to

practices that could impact California rangeland plant diversity, it must be noted that compost, as defined by CalRecycle, is highly unlikely to significantly impact plant community species composition (Ryals et al in press).

### **Croplands and Pastures**

Croplands and pastures are likely to benefit from repeated compost applications over time, and CDFA should support incentives for compost applications up to some level of SOM accumulation, perhaps 5%, which NRCS has suggested as an indicator of “soil health.” Incentives for increasing SOM on croplands and pastures beyond 5% may be difficult to justify in the face of limited compost availability and the vast extent of California soils with SOM levels well below that threshold. CDFA may also wish to prioritize incentives for soils with SOM levels of 3% and below, based on work by Dr. David Johnson at New Mexico State University. Johnson has shown that it is at approximately 3% SOM that fungal/bacterial ratios in soils shift in favor of fungal dominance (KRWG 2014). The significance of this for long-term soil carbon persistence and soil health generally is only beginning to be explored, but could be considered in CDFA’s incentive protocol. Lal (2015) has similarly posited 1.5% SOC (approximately equivalent to 3% SOM) as a minimum desirable SOC content.

### **COMET-Planner: Available Now for Assessment of GHG Impacts of Compost on Cropland and Pasture**

Contrary to statements within the Draft, the USDA-NRCS COMET-Planner on-line tool can, today, be used to quantify GHG reductions from compost applications on croplands and pastures (but not rangelands) under NRCS Conservation Practice Standard 590 (Nutrient Management). CDFA use of this framework would directly address environmental concerns raised within the Draft, as the NRCS 590 practice standard requires a nutrient risk assessment. While a nutrient risk assessment may be cumbersome, and, as noted above, poorly reflects the dynamics of compost as compared with synthetic fertilizers, this approach does provide CDFA with a means to use COMET-Planner to evaluate GHG benefits of compost additions on the state’s croplands and pastures while also addressing N, P or other concerns on a site-by-site basis.

Nevertheless, while COMET-Planner is available now for use as a compost GHG quantification methodology, we support CDFA’s proposed use of ARB’s (2011) compost offset methodology as a preferred approach. ARB’s methodology, as proposed within the Draft, appears to be more comprehensive, is directly responsive to quantity of compost applied, is rooted in California, including CalRecycle, data, and offers what we believe is a more accurate, and attractive, GHG mitigation value of 0.42 MT CO<sub>2</sub>e/ton of compost *feedstock*, rather than 1.0 MT CO<sub>2</sub>e *per acre*, within COMET-Planner.

It should also be noted here that this is an example of the fundamental conservatism of COMET-Planner, as further evidenced by the significant discrepancy between COMET-Planner output and actual GHG data for compost applications on Mediterranean California rangelands (Ryals and Silver 2013) and for the GHG benefits of riparian restoration (Lewis et al 2015). This conservatism should provide both CDFA and ARB with confidence that

offsets or sequestration quantified through use of COMET-Planner is more likely to be under, rather than over, estimated. On the other hand, it does leave producers under-recognized for their management practices' contributions to California's efforts to stop and reverse global warming.

It should also be noted that, as currently structured, NRCS Conservation Practice Standard 590 does not apply to compost used at the time of perennial crop establishment, though it does permit quantification of GHG benefits of compost applied on pasture, established perennial crops and/or annual croplands. ARB's 2011 methodology would, presumably, recognize compost use in establishment of perennial crops. This is important, in so far as use of compost at this critical phase of crop establishment offers a unique opportunity to incorporate compost into the soil matrix (eg, individual planting holes). This can have a significant impact on both immediate SOC enhancement and subsequent crop-soil-water dynamics, and thus crop carbon capture capacity and agricultural productivity following crop establishment.

### **Conclusion**

*The goal of the Healthy Soils Initiative is to engage the enormous potential of California's working lands to reduce anthropogenic forcing of the climate system.* It is imperative, therefore, that all the ways in which forcing of the climate is being aggravated, and all the opportunities for its reversal, are addressed to the fullest extent possible. Compost - particularly mature composts made in accordance with Cal Recycle standards- is produced from existing feedstocks, which are largely waste products that would otherwise require some manner of disposal with associated potential negative environmental impacts. Critically, therefore, its use does not result in additional forcing of global nutrient cycles, as occurs with the manufacture and use of synthetic fertilizers.

CDFA should, therefore, incentivize compost use as a means to both directly increase soil carbon -a highly laudable and essential Healthy Soils Initiative goal- and maximize overall program effectiveness by incentivizing (not regulating) the replacement of synthetic N fertilizers with organic sources -preferably compost (clearly defined) whenever possible. By shifting demand away from synthetics toward compost, composting of waste streams can be encouraged and spatial distribution of composted organic waste streams in California can be increased, helping to relieve pressure on eutrophic soils, ground waters and surface waters, while simultaneously reducing CH<sub>4</sub> and N<sub>2</sub>O emissions, sequestering C in soils, enhancing agricultural resilience to climate change, including drought, increasing soil water holding capacity, and supporting working land productivity.

As the recent Conference of Parties in Paris has shown, the time for timid steps and half measures in response to climate change is far behind us. California cannot meet its GHG reduction goals without an ambitious, soil focused, working lands program. Compost, while appropriately recognized as only one component of such a program, is particularly effective because of its multi-faceted impacts on the climate change equation.

Decomposition within the compost environment maximizes carbon conservation as compost biomass while minimizing emissions of short-lived climate pollutants, including

black carbon associated with burning of biomass that might otherwise have been composted, and CH<sub>4</sub> and N<sub>2</sub>O emissions from anaerobic disposal alternatives. Compost offers the simplest and fastest way to safely increase SOM on working lands. The compost environment effectively pre-processes and stabilizes organic materials for safe and beneficial application to working land soils, eliminating or radically reducing pathogens, weed propagules and soluble nutrients.

In closing, we greatly appreciate the opportunity to comment upon the Draft Report, and reiterate our strong support for a CDFA cost-share incentive program for compost applications to working lands in California.

Thank you,

Sincerely,

Jeffrey Creque, Ph.D.  
Carbon Cycle Institute

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## COMPOST COALITION of SONOMA COUNTY

To: Karen Ross, Secretary, California Department of Food and Agriculture

**RE:** Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program

Dear Secretary Ross:

The Compost Coalition of Sonoma County appreciates the opportunity to comment upon the Draft Report, *Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program (Draft)*. We strongly support CDFA plans to implement a cost-share incentive program for compost applications to working lands in California.

Compost is in the first place a soil amendment, not a fertilizer. Not until recently could compost manufacturers make a claim on the nutrient value in their products. As a soil amendment, compost helps to conserve water, reduce erosion, diversify soil microbial populations and increase carbon in the soil. Increasing soil organic matter is probably the only economically viable way to significantly reduce greenhouse gas emissions to curb global warming. Whereas most efforts to prevent impacts from global warming are aimed at reducing emissions, thereby slowing down the impacts on climate change, increasing soil organic matter can actually reverse global warming through carbon sequestration.

The report puts a lot of emphasis on nitrogen as a guide to determine how much compost can be used. We certainly do not want to solve one problem while creating another. However, given the quantities of compost that are used in agriculture and the slow release of nutrients from mature compost there is no real threat to the environment. In fact, compost has the ability to minimize the impacts of nutrient pollution from conventional fertilizers through immobilization. A focus on nutrients is misplaced and will slow down the benefits that can be gained from carbon sequestration through compost applications.

The report also uses C:N ratios to evaluate the amount of compost to be used. First, C:N ratio is an indicator of maturity of compost, not an accurate indicator of the amount of nitrogen present in the compost. Percent nitrogen would be a much better tool. By introducing C:N ratios as a guide to determine how much compost can be used the discussion has entered a field of confusion.

We would like to make the following recommendations:

- Simplify the approach on how much compost can be used. □
- Omit the use of C:N ratio and abandon the two tier system of high and low nitrogen as □well as the differentiation between organic and conventional agriculture to determine □how much compost can be applied. □
- As a starting point for the incentives, adopt up to 8 wet tons per acre for crop production per crop cycle, 8 wet tons per acre per year for orchards and up to 30 tons per acre for rangeland. As the Marin Carbon Project has demonstrated,

Comments on Report: *Agronomic Rates of Compost  
Application for California Croplands and Rangelands*

- repeat application on rangeland may be necessary only once in 20 years.
- In order to get the maximum impacts from the incentive program, CDFA needs to assess the rate at which the minimum amount of compost applied yields the maximum rate of carbon sequestration.
  - Adjust the incentives program in 2-3 years. □
  - The RCD's will soon start to create Carbon Farming Plans for farmers. We recommend that compost uses, as recommended in the Carbon Farming Plan, will be automatically approved for the incentives □

In closing, we greatly appreciate the opportunity to comment upon the Draft Report, and reiterate our strong support for a CDFA cost-share incentive program for compost applications to working lands in California.

Sincerely,



2/11/2016

Erin Axelrod  
Co-Chair  
Compost Coalition of Sonoma County

TO: Karen Ross, Secretary, California Department of Food and Agriculture RE: Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and rangelands to support a Cdfa Healthy Soils Incentives Program (Version 1.0 – 1/5/2016)

Dear Secretary Ross,

It is with gratitude and joy that I write to you today. I am excited to see the progress on the topic of compost application rates for California and I am thankful that you all have opened this topic up for comments. My name is Joseph Haggard and I am currently a farm apprentice at Raphael Garden at Rudolf Steiner College, a 28 year old organic and biodynamic farm and CSA (Community Supported Agriculture) in Fair Oaks, California. Prior to this apprenticeship I studied Environmental Studies with a focus in Sustainable Agriculture at UC Santa Cruz.

I have read the Draft Proposal “*Agronomic rates of compost application for California croplands and rangelands to support a Cdfa Healthy Soils Incentives Program*” and would like to offer the following comments. First off, I would like to thank all of those who dedicated their time to compose this document as well as to those who provided the research cited in this document. I would like to offer my comments to your proposed compost incentive from the feasible, practical, and ecologically sustainable model I am involved with, as well as offer some considerations.

Raphael Garden is a three acre organic and biodynamic urban farm and garden located in Fair Oaks, California. It was started 28 years ago by Harald Hoven who continues to be the head gardener on site. Raphael Garden is a highly intensive system utilizing compost for nutrient management and attended by relatively high labor. This is reflected in the high yield/acre we are recording. Some of the main elements that make up this farm include a variety of livestock, including: 2 cows, 4 sheep, 40 chickens, and 4 ducks; different plots of land management, including: a little under an acre of orchard grazed by the sheep, an acre of pasture grazed by the cows, and an acre of garden beds planted with over 140 varieties of annual crops; as well as a variety of agricultural programs, including: a seed saving program which accounts for at least 50% of the seed grown on the farm, and a carefully considered composting system. This last year, we harvested over 12 tonnes of vegetables off of the one acre in production, we typically range from 11-15 tonnes harvested per year. These vegetables are sold through a CSA which supports 90 family shares for 50 weeks out of the year.

We use no outside fertilizer products and compost is our sole form of nutrient management, much of which we make onsite from our own biomass. Our composting system is managed by hand and includes two main types of compost: plant compost and manure compost. Both systems are static piles; the plant compost is used mostly for seed starts and is more than a year old, while the manure compost is used for field application after 3-6 months of decomposition. This allows for stable humus formation which helps improve the water and nutrient holding capacity of our soil. The compost piles we construct have a starting ratio of approximately 30:1 C:N. We do not know the C:N ratio of our finished piles. In addition to compost fertility, we graze our animals on the pasture and in the orchard which supplies this land with enough fertility to increase soil organic matter without additional compost application. The manure collected during the night in the pen is used for manure compost.

Our compost application rates vary crop to crop. Each garden bed has compost applied 2-3 times per year on average. We use a crop rotation plan considering heavy and light feeders, plant families and which part of the plant is harvested, such as root, leaf, or fruit. We use a relatively low cover crop (20%-25% per year) in order to maintain economic sustainability for year round harvest. We estimate

that about 35 tonnes of compost is applied to our one acre garden annually. We use more compost to compensate for the low amount of cover crop utilized in winter. Approximately 2/3 of this compost is made from on site organic materials, while the rest is purchased biodynamic compost from local organic/biodynamic farms. This rate is necessary for such an intensive production system which provides the yields we are recording annually.

After 28 years, Mr. Hoven has observed a tremendous improvement in overall soil organic matter as well as general soil and crop health. This improved soil organic matter includes soil organic carbon which is actively sequestered carbon from the atmosphere. Thus, to briefly summarize our garden, we build compost at a 30:1 C:N ratio, apply approximately 35 tonnes of compost/acre/year, and harvest between 11-15 tonnes of vegetables from the one acre per year. All of this is accomplished while simultaneously improving soil quality and building soil organic matter.

I would like to make two recommendations:

1) Providing for an incentive for a higher standard of compost application above 8 wet tonnes of compost/acre/crop for cropland.

2) Adding incentives for on site composting. This would reduce the environmental cost of transportation of resources and would help to localize soil building networks. As the system described above illustrates, our 3 acre production system is capable of producing around 20 tonnes of compost/acre/year on site.

Once again, I would like to extend my gratitude to all of you who have worked so hard to accomplish what has thus far been put forward. I am thankful that this topic is in consideration and I thank you all for opening time and space for comments and questions to be received.

Thank you for your time,  
Joseph Haggard  
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9200 Fair Oaks Blvd, Fair Oaks, California.



CALIFORNIA  
NATIVE PLANT SOCIETY

February 12, 2016

Sent via electronic mail to: [EcoSysServices@cdfa.ca.gov](mailto:EcoSysServices@cdfa.ca.gov)  
[Amrith.Gunasekara@cdfa.ca.gov](mailto:Amrith.Gunasekara@cdfa.ca.gov)

Dear Dr. Gunasekara:

The California Native Plant Society would like to provide the following comments regarding the report, *Agronomic rates of compost application for California croplands and rangelands to support a Cdfa Healthy Soils Incentives Program*.

Regarding the Healthy Soils Incentives Program as it relates to compost application to rangelands, we note the following conclusion from the report:

*“Uncertainties about the drawbacks of rangeland compost application are even greater than the uncertainties about its (statewide) C sequestration benefits.”* [at page 9]

This statement provides a clear reason why we must take a precautionary approach to adopting a statewide practice of compost application to rangelands in the name of greenhouse gas reduction. As we have stated in our public comments, our organization continues to stress the need to determine if application of compost to rangelands will result in an alteration of grassland native species composition and/or structure.

Of particular concern is whether applying compost to rangelands will negatively effect the richness and diversity of native forbs (wildflowers) in treated grasslands. The native forb component of rangelands, i.e., California wildflowers, is essential habitat for California’s pollinator insects. California’s wildflower fields are in decline. Loss of forb biodiversity will lead not only to the loss of one of our state’s iconic landscapes (wildflower vistas), but to the loss of habitat and forage for both native and cultivated pollinator species. Declines in bee populations and their insect allies can lead to declines in pollination and productivity of not only native vegetation communities, but of our cultivated croplands as well.

Recent studies have determined that there is significant value of rangelands as pollinator habitat to California agriculture<sup>1</sup>, that loss of pollinator habitat proximal to croplands can result in declines in crop productivity and revenues because of reduced pollination of crop plants<sup>2</sup>, and that more insects than bees contribute significantly as agricultural pollinators<sup>3</sup> (Radar et al. 2015). It is important that we answer questions of potential negative impacts to rangeland biodiversity - especially related to the native forb component of rangelands - before scaling this proposal statewide.

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<sup>1</sup> Chaplin-Kramer et al., 2011. *Rangelands* 33(3): 33-41.

<sup>2</sup> Garabaldi et al., 2013. *Science* 339: 1608-1611.

<sup>3</sup> Radar et al. 2016. *PNAS* 113(1): 146-151.

Based on our on-going concerns about potential negative effects to biodiversity resulting from the application of compost to rangelands, CNPS recommends the following.

- a. Retired agricultural lands that are being restored or converted to rangelands currently represent areas best suited for rangeland compost applications.
- b. Healthy Soils Incentives should prioritize application onto annual croplands, orchards, and vineyards where fewer negative impacts to native biodiversity would occur, over application onto rangelands where uncertainties of drawbacks outweigh those of benefits.
- c. Uncertain affects to grassland biodiversity need to be assessed before widespread application to rangelands. Our research has demonstrated that assessing grassland biodiversity requires monitoring plots during multiple seasons over multiple years to fully catalog the plant species present. Current ACR method of site assessment and monitoring is not sufficient to assess native biodiversity of potential rangeland application sites. Recent advances in genomic research tools could compliment grassland survey techniques and potentially expedite grassland inventories. CNPS would gladly help identify genomic research opportunities related to grassland biodiversity. Most especially, we would like to help design field studies that can test the following hypothesis: *The application of compost to rangelands will alter the biological composition and structure of treated grasslands.*

CNPS is willing to commit staff resources to help convene a scientific panel to explore how best to test this hypothesis. NRCS field trials that are being developed that can help assess biodiversity questions. However, because NRCS Conservation Field Trial (CFT) policy restricts resources to \$20,000 over 3 years, the scope of monitoring variables and level of effort of the CFTs will be limited. An additional level of effort will be needed to answer the biodiversity questions.

- d. Rare and sensitive grasslands must be deemed ineligible for compost application. Serpentine grasslands, grasslands hosting vernal pool complexes (nitrogen leaching from application to upland vernal pool prairie rangelands will likely end up percolating into vernal pools), alkali grasslands, and desert grasslands fall into this category.
- e. Grasslands currently designated as mitigation and/or conservation lands must be ineligible for compost application. Grassland habitats home to rare, threatened, and endangered species and species of conservation concern are designated for conservation in conservation plans (HCPs, NCCPs) across California. Other grassland habitats have been purchased as mitigation for impacts elsewhere. Because these grasslands have been designated for specific conservation purposes, and because uncertainties related to application of compost to grasslands remain which could negatively impact grasslands, such areas must be ineligible for compost application until biodiversity impacts can be assessed sufficiently to determine negative effects will not occur.
- f. In all cases, we must continue to address and take steps necessary to ensure that compost application to soils will not result in the promotion of invasive non-native species, or the spread of plant pathogens.

Thank you for considering our comments. Please contact me if you have any questions or would like to discuss any of these points further.

Sincerely,

A handwritten signature in black ink that reads "Greg Suba". The letters are cursive and fluid, with the first name "Greg" and last name "Suba" written in a single continuous line.

Greg Suba  
Conservation Program Director, CNPS

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February 11, 2016

### **Public Comment submitted regarding CDFA Healthy Soils Initiative**

I am pleased to have the opportunity to contribute my perspective on the CDFA Healthy Soils Initiative. By way of background, I am a Professor of Biological Sciences at Union College, NY. I am a specialist in the area of the interactions between native and non-native plant species including in California grasslands, and the importance of nitrogen availability to their interactions in particular. I have worked extensively in California grassland ecosystems and am an author or co-author of many peer-reviewed scientific papers related to California grasslands. I am also a co-editor of the book *California Grasslands: Ecology and Management*, published by University of California Press. For these reason, I feel that my perspective may be of use to the CDFA.

I applaud the CDFA for considering all available tools to combat climate change including soil amendments to increase carbon sequestration. When carefully administered and when applied to appropriate ecosystems, they have the potential to increase ecosystem services provided by California landscapes.

*I am concerned, however, that for many California habitats and species-diverse rangelands in particular, compost amendments as proposed by the CDFA have the potential to imperil our unique plant biodiversity. While compost-amendment in an agricultural context can promote plant productivity and ecosystem functioning, I remind readers that not all plants in an uncultivated ecosystem benefit from added nutrients including nitrogen and phosphorus.*

My take-home point is to emphasize the counter-intuitive relationship between California's unique biodiversity and soil nutrient status. *My research, and that of many others, has established that many native California species including ones of particular biodiversity value are at a significant disadvantage under high-nutrient conditions.* Our native forbs and grasses are relatively slow-growing and do not respond when nutrient availability increases to the extent that their more aggressive non-native competitors due.

This is significant in the context of atmospheric nutrient deposition (e.g. downwind of cities) and also invasion by nitrogen-fixing plants. Native biodiversity has been shown to decrease in grasslands where N inputs increase – this is a well-established pattern that has been replicated in grasslands worldwide. The addition of nutrient-rich compost will likely have the same effect. My worry is that, despite its benefits as

far as the State's carbon balance, the Healthy Soils Program will put another nail in the coffin of our biological treasures.

While some rangelands may not have vegetation that is vulnerable to nutrient addition, the fact is that we don't have sufficient surveys available to be able to identify where it would be safe and where it would not be safe. For this reason, I urge caution in implementing the Healthy Soils Program to rangelands other than some very specific types such as irrigated pasture, fallowed fields, or small holding/feeding pastures. Just as the Scientific Advisory Board identified some habitats such as coastal prairie, serpentine, and vernal pools that may not be suitable, I suggest that a significant burden of "no harm" be passed before the Healthy Soils Program is approved for use in other rangeland settings.

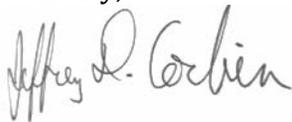
Organic amendments could be an important tool in California's arsenal to control its carbon balance. However, we should be careful not to imperil our biodiversity in the course of implementing such carbon-friendly policies. As is the case with medical care, in conservation we should be sure to *first, do no harm*.

For this reason, I urge you to reconsider the broad approval of compost amendments to rangelands statewide before additional information or criteria can be developed.

I have attached pdf's of relevant writings on the subject. I also list citations of other relevant papers I have published below.

I would be happy to elaborate in any way if it would be useful. I can be reached at (518) 388-6097, or [corbinj@union.edu](mailto:corbinj@union.edu).

Sincerely,

A handwritten signature in cursive script that reads "Jeffrey D. Corbin".

Dr. Jeffrey D. Corbin  
Associate Professor of Biological Sciences  
Union College

### **Relevant Citations:**

- Corbin, J.D.** and B. Oakes. 2014. Species Diversity. In Y.Q. Wang (ed.) Encyclopedia of Natural Resources.
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- D'Antonio, C.M. and **J.D. Corbin**. 2003. Effects of plant invaders on nutrient cycling: Using models to explore the link between invasion and development of species effects. Pp. 363-384 In: C.D.

Canham, J.J. Cole, and W.K. Lauenroth, editors. "Models in Ecosystem Science". Princeton (NJ): Princeton University Press



**Title:**

Cars, Cows, and Checkerspot Butterflies: Nitrogen Deposition and Management of Nutrient-Poor Grasslands for a Threatened Species

**Author:**

[Weiss, Stuart B.](#), Center of Conservation Biology, Stanford University

**Publication Date:**

04-14-1999

**Series:**

[Recent Work](#)

**Permalink:**

<http://escholarship.org/uc/item/81t6n5fb>

**Additional Info:**

Weiss SB. 1999. Cars, cows, and checkerspot butterflies: nitrogen deposition and management of nutrient-poor grasslands for a threatened species. IN: Conservation Biology Volume 13, No. 6, December 1999: pp. 1746-1486.

**Keywords:**

rare species, population decline, limiting factor, Serpentinite, biodiversity, introduced species, deposition nitrogen, air pollution, environmental management

**Abstract:**

Nutrient-poor, serpentinitic soils in the San Francisco Bay area sustain a native grassland that supports many rare species, including the Bay checkerspot butterfly (*Euphydryas editha bayensis*). Nitrogen (N) deposition from air pollution threatens biodiversity in these grasslands because N is the primary limiting nutrient for plant growth on serpentinitic soils. I investigated the role of N deposition through surveys of butterfly and plant populations across different grazing regimes, by literature review, and with estimates of N deposition in the region. Several populations of the butterfly in south San Jose crashed following the cessation of cattle grazing. Nearby populations under continued grazing did not suffer similar declines. The immediate cause of the population crashes was rapid invasion by introduced annual grasses that crowded out the larval host plants of the butterfly. Ungrazed serpentinitic grasslands on the San Francisco Peninsula have largely resisted grass invasions for nearly four decades. Several lines of evidence indicate that dry N deposition from smog is responsible for the observed grass invasion. Fertilization experiments have shown that soil N limits grass invasion in serpentinitic soils. Estimated N deposition rates in south San Jose grasslands are 10–15 kg N/ha/year; Peninsula sites have lower deposition, 4–6 kg N/ha/year. Grazing cattle select grasses over forbs, and grazing leads to a net export of N as cattle are removed for slaughter. Although poorly managed cattle grazing can significantly disrupt native ecosystems, in this case moderate, well-managed grazing is essential for maintaining native biodiversity in the face of invasive species and exogenous inputs of N from nearby urban areas.



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# Cars, Cows, and Checkerspot Butterflies: Nitrogen Deposition and Management of Nutrient-Poor Grasslands for a Threatened Species

STUART B. WEISS

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**Abstract:** *Nutrient-poor, serpentinitic soils in the San Francisco Bay area sustain a native grassland that supports many rare species, including the Bay checkerspot butterfly (*Euphydryas editha bayensis*). Nitrogen (N) deposition from air pollution threatens biodiversity in these grasslands because N is the primary limiting nutrient for plant growth on serpentinitic soils. I investigated the role of N deposition through surveys of butterfly and plant populations across different grazing regimes, by literature review, and with estimates of N deposition in the region. Several populations of the butterfly in south San Jose crashed following the cessation of cattle grazing. Nearby populations under continued grazing did not suffer similar declines. The immediate cause of the population crashes was rapid invasion by introduced annual grasses that crowded out the larval host plants of the butterfly. Ungrazed serpentinitic grasslands on the San Francisco Peninsula have largely resisted grass invasions for nearly four decades. Several lines of evidence indicate that dry N deposition from smog is responsible for the observed grass invasion. Fertilization experiments have shown that soil N limits grass invasion in serpentinitic soils. Estimated N deposition rates in south San Jose grasslands are 10–15 kg N/ha/year; Peninsula sites have lower deposition, 4–6 kg N/ha/year. Grazing cattle select grasses over forbs, and grazing leads to a net export of N as cattle are removed for slaughter. Although poorly managed cattle grazing can significantly disrupt native ecosystems, in this case moderate, well-managed grazing is essential for maintaining native biodiversity in the face of invasive species and exogenous inputs of N from nearby urban areas.*

Carros, Vacas, y Mariposas: Deposición de Nitrógeno y Manejo de Pastisales Pobres en Nitrógeno para una Especie Amenazada

**Resumen:** *Suelos serpentiniticos pobres en nutrientes en el área de la Bahía de San Francisco sostienen pastizales diversos que soportan muchas especies raras, incluyendo a la mariposa checkerspot (*Euphydryas editha bayensis*). La deposición de Nitrógeno (N) por contaminación del aire amenaza la biodiversidad en estos pastizales debido a que N es el principal nutriente limitante para el crecimiento de plantas en suelos serpentiniticos. Investigué el papel de la deposición de N mediante muestreos de mariposas y poblaciones de plantas a lo largo de diferentes regimenes de pastoreo, revisiones de literatura y con estimaciones de deposición de N en región. Varias poblaciones de la mariposa en el sur de San José se precipitaron después de cesar el pastoreo por ganado. Poblaciones cercanas bajo continuo pastoreo no sufrieron disminuciones similares. La causa de las disminuciones poblacionales fue la rápida invasión de pastos anuales introducidos que saturaron a las plantas hospedero de las larvas de la mariposa. Los pastizales serpentiniticos sin ramoneo de la península de San Francisco han resistido las invasiones de pastos por casi cuatro décadas. Diversas líneas de evidencias indican que la deposición de N seco del smog es responsable de la invasión de pastos observada. Experimentos de fertilización han demostrado que el N del suelo limita la invasión de pastos en suelos serpentiniticos. Las tasas de deposición de N en pastizales del sur de San José son de 10-15 kg ha<sup>-1</sup> año<sup>-1</sup>;*

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*Paper submitted September 22, 1998; revised manuscript accepted April 14, 1999.*

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los sitios de la península tienen deposiciones mas bajas ( $4\text{--}6\text{ kg ha}^{-1}\text{ año}^{-1}$ ). El ganado selecciona pastos y el ramoneo conduce a una exportación neta de N pues el ganado es removido al ser sacrificado. Sin embargo, el pastoreo pobremente manejado puede desequilibrar significativamente a ecosistemas nativos. En este caso el buen manejo del pastoreo es elemental para mantener la biodiversidad nativa de cara a invasiones de especies y entradas exógenas de N proveniente de áreas urbanas cercanas.

## Introduction

Humans have greatly increased the flux of reactive nitrogen (N) in the biosphere, which is now recognized as a major component of global change (Vitousek et al. 1997). Extensive areas downwind of air pollution sources receive substantial inputs of N from wet and dry deposition. Many terrestrial ecosystems are presently N-limited and respond strongly to incremental additions of N, exhibiting changes in productivity, species composition, and nutrient retention. At global scales, N deposition may be responsible for a substantial part of the “missing carbon sink” (Townsend et al. 1996; Holland et al. 1997). At regional scales, N deposition has been implicated in a large number of ecosystem changes, including forest decline in Central Europe (Schulze 1989), grass invasions of heathlands in northwestern Europe (Aerts & Berendse 1988), and changes in grassland composition in the midwestern United States (Tilman 1988). Nutrient-poor ecosystems are particularly vulnerable to N deposition (Aerts & Berendse 1988; Bobbink & Roelofs 1995; Power et al. 1995).

At regional and local scales, N deposition on nature reserves may change vegetation, threaten the persistence of target species and communities, and greatly complicate reserve management. Nitrogen deposition presents a major conservation challenge because the source of the problem is outside the boundaries of reserves and can be controlled only at the source by expensive, long-term measures. Deposition is especially high near urban areas, where combustion sources (primarily cars, trucks, and industrial and home heating) produce substantial concentrations of nitrogen oxides ( $\text{NO}_x$ ). Setting aside reserves on the fringe of urban areas is often difficult, contentious, and expensive, and loss of diversity in reserves because of exogenous factors such as N deposition needs careful consideration.

Native grasslands are among the most imperiled ecosystems in the temperate zones (Joern & Keeler 1995; Samson & Knopf 1996; Muller et al. 1998). Conversion to agriculture and urbanization has left only small remnants of many formerly extensive grassland ecosystems. Remnants are threatened by further development, isolation, invasion by introduced species, and increasingly by N deposition. Grasslands can be sensitive to added N (Huenneke et al. 1990; Wedin & Tilman 1996). Nitrogen fertilization of grasslands generally results in loss of plant

species diversity when a few N-loving species become dominant (Silvertown 1980; Tilman 1987; Huenneke et al. 1990). Losses of plant diversity can lead to losses of animal diversity, especially of host-restricted herbivores.

Many grassland remnants need intensive management from grazing, fire, and mowing. Grazing is a traditional land use in many regions, and poorly managed overgrazing has been responsible for massive disruption of ecosystems worldwide. Yet properly managed, moderate grazing can be a useful management tool in specific instances, especially for species that require short, open grassland (Aerts & Berendse 1988; ten Harkel & van der Muelan 1995). For example, some of the most imperiled butterfly species in England require sheep or rabbit grazing to maintain suitably short grass swards (Oates 1995). Both fire and grazing have been used in management of prairie fragments in the midwestern United States (Samson & Knopf 1996) and have differential effects on butterflies (Swengel 1998). Management of N deposition in grasslands requires the removal of N-containing biomass from a site by fire, mowing, or grazing (Hobbs et al. 1991). Mowing for hay removal in Europe is a common management practice in “unimproved” grasslands set aside for conservation (Dolek & Geyer 1997).

I documented near extinctions of the threatened Bay checkerspot butterfly (*Euphydryas editha bayensis*) in grasslands following the removal of grazing. The proximate cause was the rapid invasion of nutrient-poor serpentine grasslands by introduced grasses. Several lines of evidence indicate that N deposition by air pollution—primarily from cars and trucks—is an ultimate cause of the grass invasion. The negative consequences of improper grazing management in these sites are great, and the story provides a striking example of the complexities of managing reserves adjacent to urban areas and of the interdisciplinary nature of effective conservation biology.

## Methods

### Study Organism and System

The Bay checkerspot butterfly is restricted to outcrops of serpentine rock in the San Francisco Bay Area, California. Serpentine weathers to a thin, rocky, nutrient-poor soil with low N, high magnesium, low calcium, local patches of heavy metals, and other unfavorable character-

istics. Introduced annual grasses from the Mediterranean region have invaded the majority of California grasslands on richer soils (Huenneke 1989). Serpentinic soils provide refugia for a diverse native grassland with more than 100 species of forbs and grasses (Murphy & Ehrlich 1989), including dense stands of the host plants (*Plantago erecta*, *Castilleja densiflorus*, and *Castilleja ex-certa*) and nectar sources (*Lasthenia californica*, *Layia platyglossa*, *Allium serratum*, *Muilla maritima*, and *Lomatium* spp.) of the Bay checkerspot butterfly. Several serpentine-endemic plants are listed as endangered or threatened (U.S. Fish and Wildlife Service 1998). Numerous patches of these grasslands have been destroyed by urban development (Murphy & Weiss 1988a), and their protection is a major conservation priority in the San Francisco region (U.S. Fish and Wildlife Service 1998).

The Bay checkerspot butterfly is listed as threatened under the U.S. Endangered Species Act and serves as an "umbrella species" for the serpentinic grassland ecosystem (Murphy & Weiss 1988a; Launer & Murphy 1992). As

of 1998, there was one extant population on the San Francisco Peninsula, at Edgewood County Park (EW; Fig. 1). Many former Peninsula populations are extinct because of urban development, and the small population at Jasper Ridge Biological Preserve (JR) appeared to be extinct as of 1999 (S.B.W., unpublished data). The majority of the habitat surrounds the Coyote Valley in south San Jose and supports a reservoir-satellite metapopulation of the butterfly (Harrison et al. 1988). The major sites I considered were the Silver Creek Hills (SC), Kirby Canyon (KC), and adjacent areas of Coyote Ridge (CR) (Fig. 1).

I estimated densities of postdiapause larvae with the stratified sampling design described by Murphy and Weiss (1988b). During the study period (1991–1998), several areas in the Silver Creek Hills were surveyed, along with sites south along Coyote Ridge (CR) and at Kirby Canyon (KC). For conservation planning purposes, the Silver Creek Hills were divided into subareas (SC1, SC2, and SC3) corresponding to property boundaries. Coyote Ridge was subdivided into four areas (CR1, CR2, CR3, and CRlow).

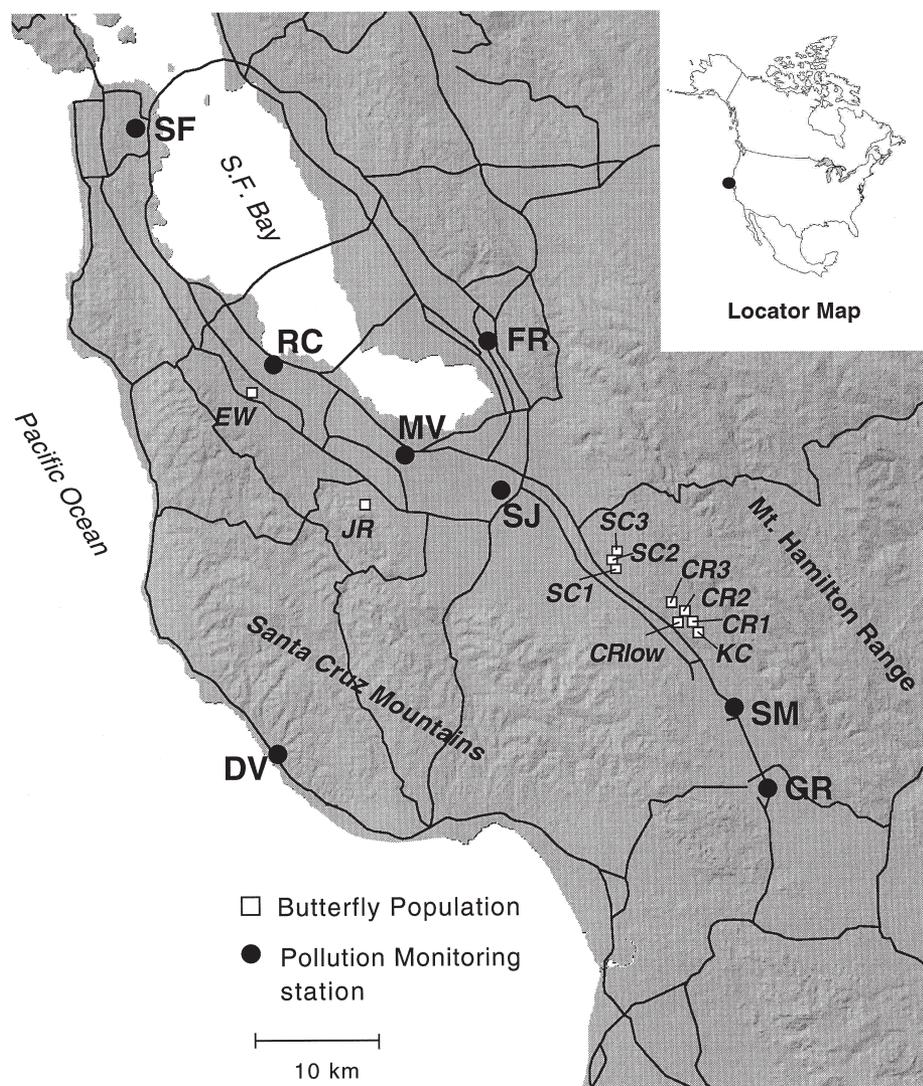


Figure 1. Map of regional air pollution monitoring stations and populations of the Bay checkerspot butterfly in the San Francisco Bay area. Only those air pollution stations and butterfly populations used in this study are identified. Site abbreviations are as follows: KC, Kirby Canyon; CR, Coyote Ridge, numbers 1–3, distinct sites along the ridgetop; CRlow, low-elevation site; SC, Silver Creek Hills, numbers 1–3 are separate properties within the Silver Creek Hills; JR, Jasper Ridge Biological Preserve; EW, Edgewood County Park. Pollution stations are SF, San Francisco; RC, Redwood City; MV, Mountain View; SJ, San Jose; SM, San Martin; GR, Gilroy; and DV, Davenport.

Data on the composition of plant communities across different grazing treatments at KC, CR, and SC were collected in 1996. The treatments were (1) continuously grazed in winter and spring (CR1 and CR2); (2) continuously grazed in summer and fall (CR3); (3) ungrazed since 1990 (SC1); and (4) ungrazed since 1985 (fenced areas adjacent to KC). Transects consisted of five 0.25-m<sup>2</sup> quadrats spaced 5 m apart and were sampled for percent cover of all vascular plant species. Comparisons of vegetation composition across grazing regimes was limited to undisturbed upland transects that made up the primary habitat for the butterfly. From 1995 through 1998, I sampled three such transects at site SC1 to monitor the effects of reintroduced cattle grazing.

A literature review on the effects of N and other nutrient additions to serpentinitic soils provided data on the response of native and introduced species to fertilization (Turitzen 1982; Koide et al. 1988; Huenneke et al. 1990; Hull & Mooney 1990).

Nitrogen deposition estimates were taken from Blanchard et al. (1996). I modified surface composition and pollutant loads to more closely estimate deposition on serpentinitic grassland in south San Jose. Air pollution data were taken from public documents (California Air Resources Board 1990–1996).

To measure relative deposition among sites, ion exchange resin bags were strung between poles 2 m off the ground at JR, SC, and KC. Resin bag construction and analysis followed standard methods (Reynolds et al.

1997). Twenty bags were hung at each site on 1 October 1997. The poles at Jasper Ridge fell during an intense windstorm on 18 November and were discovered on the ground on 24 November. All JR bags were collected then, and 10 bags each from SC and KC were collected the following day. The JR bags were on the ground for the final 6 out of 54 total days (slightly changing the deposition environment), but the bags were still exposed to reactive N in the air and rainfall. Because those last 6 days also had low pollution levels because of windy, rainy weather, the position of the bags should have made only a minimal difference in the total nitrate collected over the 8 weeks. October–November is the regional smog season, so relative deposition among sites during that period is a good indicator of relative rates over the entire year.

## Results

### Population Declines of Butterflies

Population estimates in the Silver Creek Hills (SC) showed a pattern of initial growth followed by rapid declines to extinction (Fig. 2). The number of postdiapause larvae rose substantially from 1991 to 1993. From 1993 to 1994, the numbers in the northern section (SC2) fell by a factor of 10. Numbers in the southern section (SC1) fell by 30%. Larval numbers fell from 14,000 in 1993 to 9000 in 1994 in a smaller property (SC3) to the north (not shown on

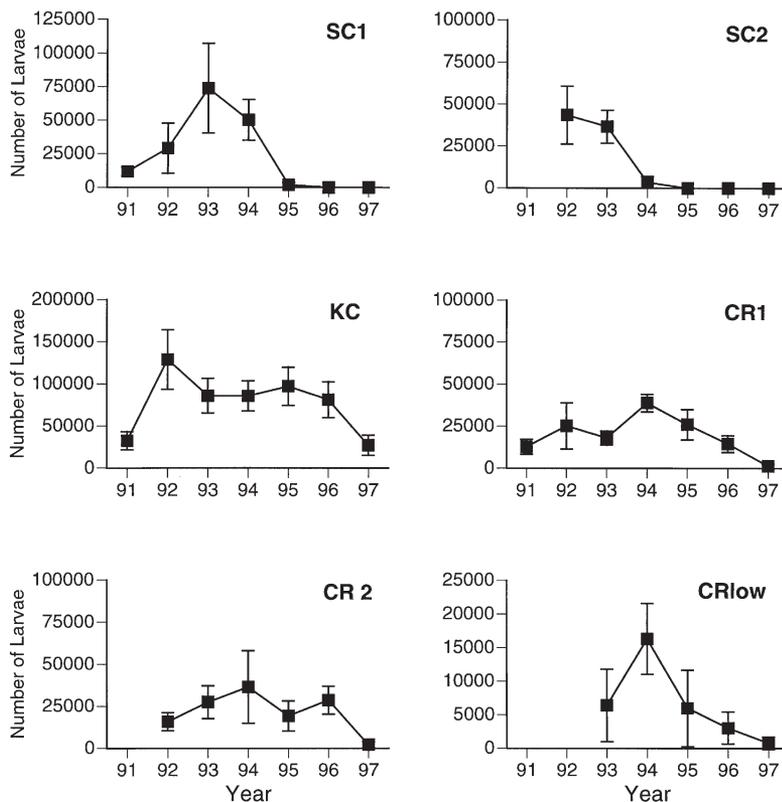


Figure 2. Mean larval densities at sites SC1, SC2, KC, CR1, CR2, and CRlow for the period 1991–1997. Error bars are 95% confidence intervals.

graphs). From 1994 to 1995, larval numbers in SC1 fell by a factor of 20. No larvae have been found at SC1 and SC2 since 1995. Demographic units at KC and CR did not crash over the 1994–1996 time period (Fig. 2).

The cause of population declines was obvious during field surveys (S.B.W. & A. E. Launer, personal observations). Dense stands of *Plantago erecta* and other native forbs were widespread across SC1 and SC2 prior to 1994. By 1995 the grassland was overrun by introduced annual grasses (*Lolium multiflorum*, *Avena fatua*, and *Bromus hordeaceus*) that dominate nearby grasslands on richer soils. *Plantago* was found only on thin soils around rock outcrops and in some areas of gopher disturbance.

The invasion of grasses followed the removal of grazing from sites SC1 and SC2. All three properties in the Silver Creek Hills had been grazed for decades. Grazing was stopped in SC1 in 1989 and in SC2 in 1992, whereas SC3 has been grazed continuously. Although I did not quantitatively sample larval densities in SC3 after 1994 (because of limited access to private property), the grasslands in those areas maintained the forb-rich community typical of serpentinitic grasslands, and a detectable population of larvae persisted in SC3 through 1997 (R. R. White, personal communication). No larvae were found in either SC1 or SC2, and only three adults were observed in SC1 in 1997. No larvae or adults were observed at SC1, SC2, and SC3 in 1998.

### Quantitative Vegetation Data

Qualitative field observations were confirmed by quantitative vegetation data. Vegetation plots from 1996 showed the differences in plant composition between grazed and ungrazed serpentinitic grasslands in south San Jose (Fig. 3). Both winter-spring and summer-fall grazing regimes had significantly higher densities of *Plantago* and significantly lower densities of introduced

grasses than sites where grazing was removed in 1985 and 1990. The amount of grass cover followed the gradient in grazing intensity and time since grazing removal. Grass cover increased and *Plantago* cover decreased along the gradient from winter-spring grazing (CR1 and CR2), to summer-fall grazing (CR3), to grazing removal in 1990 (SC1), and finally to removal in 1985 (adjacent to KC). Of the introduced grasses that invade serpentinitic grassland, *Lolium multiflorum* was most common and was responsible for differences among sites (Fig. 4).

At Kirby Canyon in 1985 and 1986, forb-dominated plots within a grazing enclosure that received no fertilization rapidly responded to the removal of grazing (Koide et al. 1988). Following two growing seasons without grazing, grass numbers and biomass greatly increased: by the spring of 1986 *Avena* appeared in forb plots, *Bromus* numbers and biomass tripled, and *Lolium* increased in biomass by two orders of magnitude. *Plantago* maintained high numbers and biomass over the 2 years. By 1987 the enclosure was dominated by dense stands of *Lolium*. *Plantago* and other small forbs were restricted to shallow soils around rocks. Similar plots outside the enclosure retained high forb biomass and low annual grass biomass (Huenneke et al. 1990; S.B.W., personal observation)

Not all serpentinitic grasslands have been invaded in the absence of grazing. Serpentinitic grasslands at JR on the San Francisco Peninsula have not been grazed since 1960, have maintained high forb diversity and density (especially *Plantago erecta*), and have not been heavily invaded by *Bromus*, *Lolium*, or *Avena* (Hobbs & Mooney 1995). *Lolium* remained a rare occurrence there until 1998, when it increased greatly during record El Niño rains (S.B.W., personal observation). In contrast, within heavily grazed serpentinitic grassland at KC and CR during average to dry years, *Lolium* is often found where soils are deeper (Huenneke et al. 1990) and is a substan-

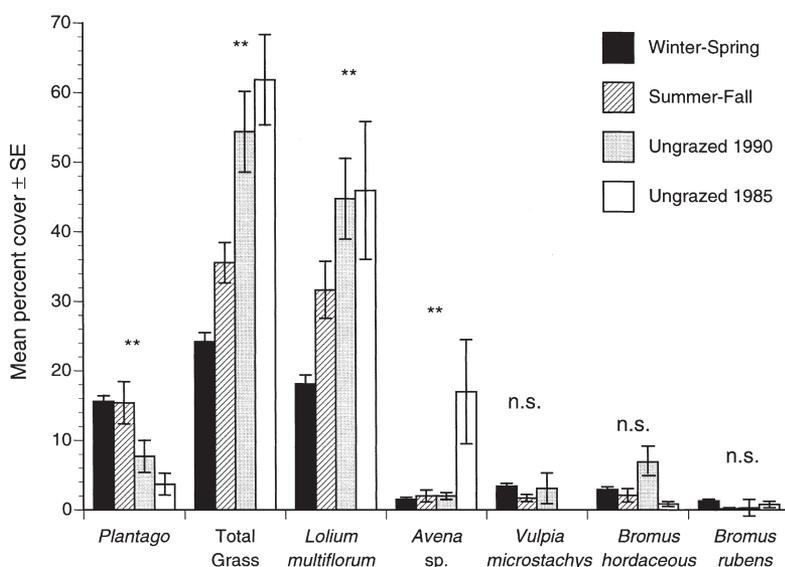


Figure 3. Percent cover of *Plantago* and annual grasses in 1996 at winter-spring grazed, summer-fall grazed, and ungrazed sites (stopped in 1985 near Kirby Canyon, and 1990 in SC1). Grazing was reintroduced at SC1 in 1995. \*\* $p < 0.001$ , Kruskal-Wallis test (nonparametric analysis of variance); n.s., not significant.

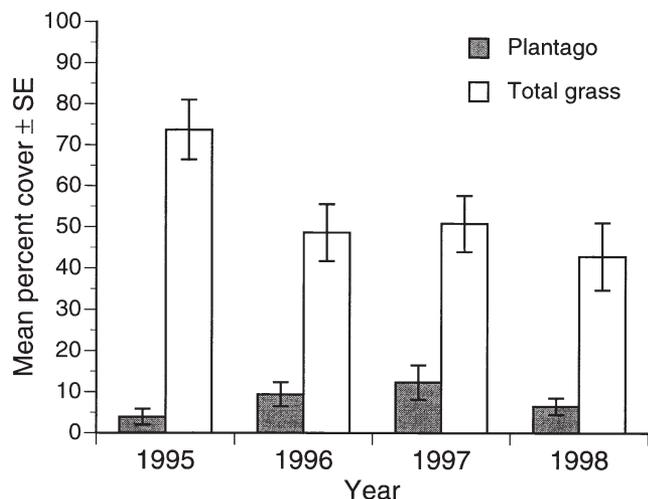


Figure 4. Response of total grass cover and *Plantago* cover to the reintroduction of grazing in site SC1 in summer 1995. Differences in grass cover were significant ( $p = 0.01$ , Kruskal-Wallis nonparametric analysis of variance), but differences in *Plantago* cover were not.

tial component of the grassland community. Koide et al. (1988) describe "a distinct vegetation patterning present at Kirby Canyon which is not present at Jasper Ridge. This consists of alternating patches of annual forb- and grass-dominated vegetation varying in extent from less than  $1 \text{ m}^2$  to greater than  $10 \text{ m}^2$ ." In the 1996 vegetation survey, *Lolium* averaged about 20% cover in the winter-spring grazing site (Fig. 3). *Lolium* cover was highest in swales (often  $>50\%$ ) and in deeper soils, but it was also found at low levels (1–20%) throughout the site.

#### Recovery of Habitat Value

In 1988 the electric fence around the 2-ha enclosure at KC was breached. By 1993, following 5 years of drought, the plot supported high *Plantago* cover, and larval densities ( $0.25 \text{ larvae/m}^2$ ) were indistinguishable from surrounding larval samples (S.B.W., unpublished data).

In the summer of 1995 following the population crash, cattle were reintroduced into site SC1. In 1995, grass cover was around 75%, dominated by *Lolium* (Fig. 4). Following 3 years of grazing from 1996 to 1998, grass cover dropped to about 45%. *Plantago* densities did not increase significantly across the habitat, however, but overall forb densities increased from 10% to about 30%, and the habitat appears to be recovering even if the small remnant butterfly population is not (Fig. 2).

#### Nitrogen Limitation in Serpentinic Grasslands

Experimental work has addressed nutrient limitations in serpentinic grasslands. Nitrogen additions in pots (equivalent of  $200 \text{ kg/ha}$ ) greatly enhanced the growth

of grasses (Turitzin 1982). Fertilization with a nitrogen-phosphorus-potassium slow-release fertilizer ( $313 \text{ kg N/ha/year}$ ) stimulated dense growth of grasses at Jasper Ridge (Hobbs et al. 1988). In field experiments at Kirby Canyon, addition of  $100 \text{ kg N/ha/year}$  into forb-dominated plots stimulated rapid increases of annual grasses (Koide et al. 1988), and factorial design with other nutrients (phosphorus, potassium, calcium) showed that N was the primary limiting factor for grass growth (Huenneke et al. 1990).

*Lolium* is the introduced grass that accounts for most of the community biomass increases observed under N fertilization (Koide et al. 1988; Huenneke et al. 1990). Detailed studies of *Lolium* and other grasses show that *Lolium* has high N assimilation rates and relative growth rates (Hull & Mooney 1990), and it responds rapidly to fertilization in the field and greenhouse.

#### Estimates of Nitrogen Deposition

In many regions, such as Europe and eastern North America, wet and dry deposition may be of equal magnitude (Vitousek et al. 1997). Because of the long, dry summer and the winter rains directly off the Pacific Ocean, N deposition in urban coastal California is dominated by dry deposition (Blanchard et al. 1996; Bytnerowicz & Fenn 1996), often by a factor of 10–30. Dry deposition is difficult to measure, and estimates of total deposition rely on models that combine aerial concentrations of reactive-N species with deposition velocities (Hicks et al. 1985, 1987). Deposition velocities are surface specific and may change with meteorological conditions. Expected uncertainties in dry deposition estimates by this inferential method are on the order of 30–50% (Blanchard et al. 1996).

Although the complexities of smog photochemistry and modeling dry deposition fluxes are beyond the scope of this paper, some background is essential (Seinfeld & Pandis 1998). The major N species responsible for dry deposition are (1) nitrogen dioxide ( $\text{NO}_2$ ), (2) nitric acid vapor ( $\text{HNO}_3$ ), (3) ammonia ( $\text{NH}_3$ ), (4) particulate nitrate ( $\text{pNO}_3^-$ ), and (5) particulate ammonium ( $\text{pNH}_4^+$ ). Nitrogen dioxide is formed by oxidation of nitric oxide from combustion in a series of rapid reactions involving ozone ( $\text{O}_3$ ). Nitrogen dioxide is taken up primarily through plant stomata and has a relatively low deposition velocity on inert surfaces. Nitric acid vapor is formed by reactions of  $\text{NO}_2$  with hydroxyl radicals ( $\text{OH}^-$ ) on a time scale of hours. Nitric acid vapor "sticks" to virtually all surfaces equally and has a high deposition velocity. Ammonia is produced from soils and animals, and like  $\text{NO}_2$  is taken up primarily by plant stomata. Particulate nitrate and ammonium are formed by reactions between soot and dust particles and gases and have relatively uniform deposition velocities among different surfaces.

Blanchard et al. (1996) provide the best current estimates of dry and wet deposition at various sites in California. The only deposition station in the San Francisco Bay area is at Fremont (Fig. 1). Estimated rates of dry N deposition at Fremont were around 6 kg N/ha/year, with 47% from  $\text{NO}_2$  and 21% from  $\text{HNO}_3$  (Fig. 5a). Wet N deposition at a station in San Jose was 0.89 kg N/ha/year (<15% of dry N deposition) and contributed amounts comparable to  $\text{NH}_3$ ,  $\text{pNO}_3^-$ , and  $\text{pNH}_4^+$ .

The estimates for Fremont are not directly applicable to the serpentinic grasslands from Silver Creek to Kirby Canyon. Several adjustments need to be made for (Fig. 5a): (1) surface composition, (2) seasonality of the grassland, and (3) higher pollution levels. The surface composition used for deposition modeling in Fremont was 70% inert surface (asphalt, roofs, etc.), 15% grass, and 15% tree. When 100% grass was used, deposition from  $\text{NO}_2$  increased from 2.2 to 5.2 kg N/ha/year because  $\text{NO}_2$  has a much higher deposition velocity on grass than on inert surfaces. Ammonia deposition increased from 0.91 to 2.9

kg N/ha/year. Nitric acid vapor deposition did not change appreciably because it has similar deposition velocities on all surfaces. The total deposition at Fremont for a 100% grass surface was estimated at 10 kg N/ha/year (C. Blanchard, personal communication.).

Nonirrigated grasslands in California are green in the rainy season (November–April) and largely brown in the dry season (May–October). Deposition varies seasonally because proportions of N species vary over the year (Fig. 5b). Nitrogen dioxide deposition is highest in fall, whereas  $\text{HNO}_3$  deposition is highest in summer. To simulate this seasonality, the figure for 100% grass was used only for the fall and winter months, whereas the figure for the “urban” mix was used for the spring and summer. At Fremont this adjustment reduced deposition to 100% grassland to 9 kg N/ha/year (Fig. 5a)

South San Jose has more air pollution than Fremont. Deposition from a N species is generally proportional to concentration. The ratio in annual  $\text{NO}_2$  concentration between San Jose and Fremont is around 1.3 (Fig. 6a), and the ratio of  $\text{pNO}_3^-$  concentrations is around 1.2 (Fig. 6b, both comparisons  $p < 0.01$ , paired  $t$  test by year). Increasing ozone levels (Fig. 6c) indicate that  $\text{HNO}_3$  increases toward the south because there is generally a tight relationship between ozone and  $\text{HNO}_3$  vapor in polluted areas (Blanchard et al. 1996). The ratio of 1:1.2 in ozone concentration between Fremont (FR) and San Martin (SM, the site closest to Kirby Canyon, paired  $t$  test by year  $p < 0.01$ ) suggests a 20% increase in  $\text{HNO}_3$  deposition. Based on these adjusted pollutant concentrations, the estimate for dry deposition into seasonal grassland in San Jose increased to 11 kg N/ha/year (Fig. 5a).

The average yearly input of wet deposition at San Jose has been 0.89 kg N/ha/year for a total of around 12 kg N/ha/year. Wet deposition within a region usually varies proportionally with rainfall. Because the ridgetop at KC receives about 640 mm rainfall, twice that at the wet deposition station in San Jose (330 mm), wet deposition at Kirby Canyon is estimated to be about 1 kg N/ha/year greater. Given uncertainties in deposition calculations, 10–15 kg N/ha/year is a reasonable approximation for sites such as Silver Creek and Kirby Canyon.

Jasper Ridge lies upwind of most pollution sources and receives much of its air as northwest winds off the Pacific Ocean that pass over the virtually undeveloped Santa Cruz Mountains (Fig. 1). Marine air is low in  $\text{NO}_2$ , as evidenced by low levels at Davenport, directly on the coast (Fig. 6c). That prediction is confirmed by the resin-bag sampling. Aerial bags at JR collected about 40% of the  $\text{NO}_3$ , compared with bags at SC and KC (JR =  $2.3 \pm 1.1$   $\mu\text{g/mL}$  extracted, SC =  $5.3 \pm 1.1$   $\mu\text{g/mL}$ , KC =  $6.8 \pm 0.7$   $\mu\text{g/mL}$ , mean  $\pm$  SE,  $n = 4$  for all sites). The difference between JR and two other sites was highly significant ( $p < 0.001$ ), but the difference between SC and KC was not (Tukey-Kramer HSD test). If that ratio holds for total deposition throughout the year, then JR may receive about 4–6 kg N/ha/year.

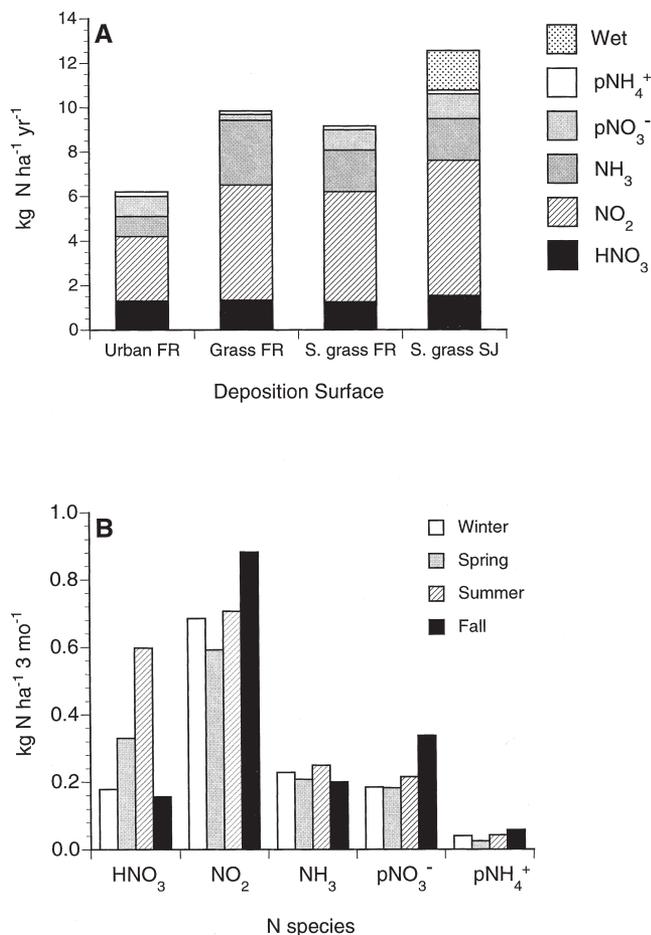


Figure 5. (a) Adjusted estimates of nitrogen deposition for grassland and seasonal (S) grassland at Fremont (FR) and San Jose (SJ). (b) Seasonality of nitrogen deposition by species at Fremont, California. (Adapted from Blanchard et al. 1996.)

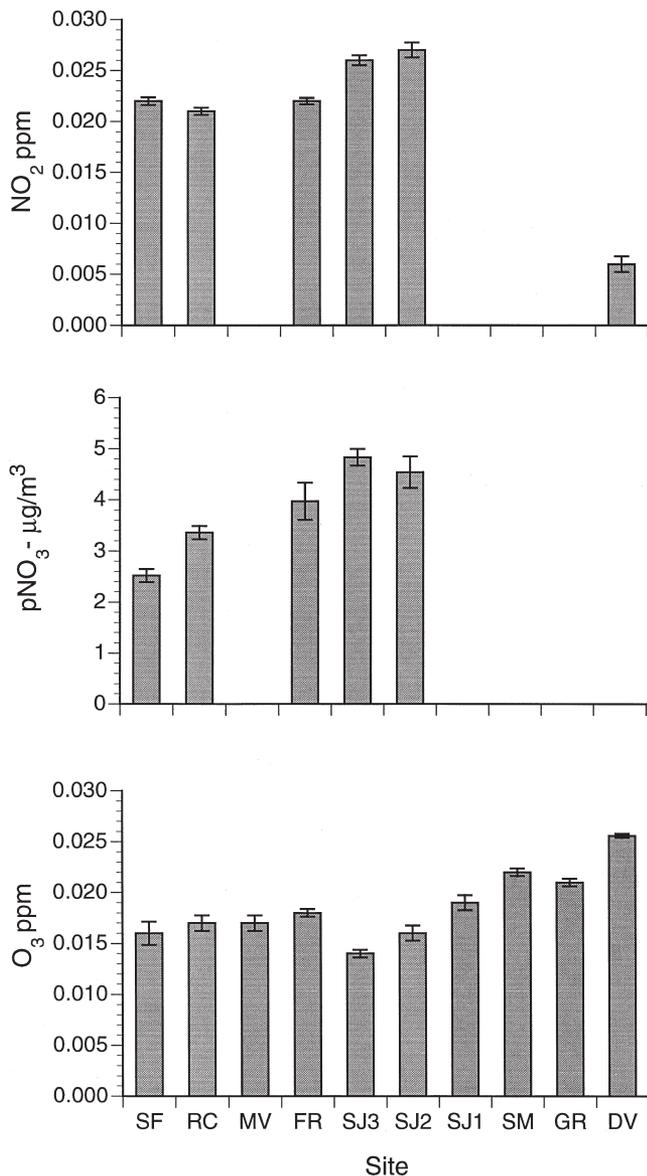


Figure 6. Annual average concentrations (1990–1996) of ozone, NO<sub>2</sub>, and particulate NO<sub>3</sub><sup>-</sup> at pollution monitoring stations in the San Francisco Bay area (data from California Air Resources Board 1990–1996).

## Discussion

The population surveys showed that the Bay checkerspot butterfly experienced severe population crashes in 1994 and 1995 after grazing was removed from two sites in the Silver Creek Hills. Concurrently, other populations in continuously grazed areas did not crash, indicating that widespread climatic factors such as drought were not responsible for the population crashes at SC1 and SC2.

The proximate cause of the crashes was invasion by introduced grasses. The dense sward of grasses, espe-

cially *Lolium multiflorum*, reached heights of 0.75 m and crowded out small native annual forbs, especially *Plantago erecta*. Similar invasions of serpentinitic soils by annual grasses have been observed around the Santa Clara Valley (Ehrlich & Murphy 1987; Murphy & Weiss 1988a; Huenneke et al. 1990). Extensive serpentinite outcrops in Santa Teresa County Park, where grazing was eliminated in the late 1970s, are dominated by grasses (S.B.W. & A. E. Launer, personal observation). Few if any checkerspot butterflies are found in ungrazed areas. Similar grass invasions have not occurred at JR, despite the removal of cattle grazing in 1960.

These observations suggest a fundamental difference between JR and the south San Jose sites. My review of experimental work showed that serpentinitic grasslands are largely N limited and can be rapidly invaded by introduced grasses when N is added. The most responsive species is *Lolium multiflorum*, the most common grass at ungrazed sites in south San Jose. *Lolium* has been rare or absent from Jasper Ridge.

I estimated that N deposition levels in grasslands in south San Jose are 10–15 kg N/ha/year. Total deposition may be even greater. My calculations did not include nitric oxide (NO), which may also dry-deposit into grasslands (Hanson & Lindberg 1991); average NO concentrations are not reported, nor was NO included in the Blanchard et al. (1996) estimates. In addition, areas of bare soil typical of serpentinitic grasslands may have high deposition velocities for NO<sub>2</sub>, greater than that of the plants themselves (Judeikis & Wren 1978; Hanson et al. 1989). Nitrogen deposition levels at Jasper Ridge are much lower, based on local meteorology and preliminary measurement with aerial resin bags.

These levels of N deposition in south San Jose are high enough to act as fertilizer and enhance the growth of annual grasses at the expense of native annual forbs. Over the course of several years, the amounts of N deposition at SC and KC approach the yearly amounts used in fertilization experiments. Most grassland ecosystems respond incrementally to N additions (Tilman 1988). For example, the effects of the addition of 50 kg N/ha/year on annual grass growth were intermediate between 0 and 100 kg N/ha/year in field experiments at JR (Hull & Mooney 1990). Serpentinitic grasslands are highly retentive of N (Reynolds et al. 1997; Hooper & Vitousek 1998), and leaching losses are small (Hooper & Vitousek 1998). Thus, incremental additions of N are incorporated into plants and microbes and may build up over several years.

The magnitude of dry N deposition flux in south San Jose is comparable to the measured yearly N uptake by plants in serpentinitic grasslands. Hungate et al. (1997) measured plant N uptake at 30–50 kg N/ha/year. Hooper (1996, personal communication) estimated similar uptake in high-diversity experimental plots. A substantial portion of dry deposition (especially NO<sub>2</sub>) is directly absorbed by stomata and is ready for immediate assimilation into the plants.

The evidence indicates that N deposition is greatly affecting the habitat suitability of serpentine grasslands for the threatened Bay checkerspot butterfly and other native species. Invasion of grasses following grazing removal is not a subtle phenomenon, can occur rapidly, and takes many years to reverse. Given the repeated responses of these grasslands to the removal of cattle grazing, the key conservation recommendation is that well-managed, moderate grazing must be maintained on sites that are expected to support the butterfly. Any conservation plan that sets aside land must include long-term grazing management.

The mechanisms by which grazing directly and indirectly affects the plant community and the N cycle are diverse. Most directly, cattle selectively graze introduced annual grasses in preference to forbs. The taller grasses can rapidly outgrow shorter forbs; grazing maintains a low, open plant canopy. Grasses also outcompete native forbs through buildup of dense thatch (Huenneke et al. 1990). Cattle break down the thatch by trampling and feeding in the dry season. Cattle also disturb the surface and compact soils, but the effects of this disturbance alone on plant composition are not well understood.

Grazing effects on the N cycle are complex. On an ecosystem-wide basis, cattle remove N as they gain weight and are removed for slaughter. Some N may also be exported via ammonia volatilization from droppings and urine (Holland et al. 1992). Although cattle droppings and urine lead to local deposition of N, in terms of N, cattle "eat globally and deposit locally." Local fertility islands (<1 m<sup>2</sup>) immediately around cattle droppings support lush stands of annual grasses (usually *Lolium*). Cattle grazing can also lead to enhanced N availability in the soil by speeding up the rate of N cycling via allocation patterns of plants (Holland et al. 1992).

Grazing regimes must be well managed and of moderate intensity. The rancher at KC and CR regularly monitors cattle weight gain and grass availability. When biologists asked for removal of cattle from KC during drought years to prevent overgrazing, the rancher had already made the decision to move his cattle to other pastures. This self-regulation of the grazing regime has been a great benefit for the Bay checkerspot butterfly and for many of the native plant species that survive on serpentine soils.

The butterfly also persists in areas where grazing is concentrated in the summer and fall. Although grass cover is higher in these areas than in the winter-spring grazed areas (Fig. 3), *Plantago* and numerous nectar sources are still abundant. Multiple management regimes may help to spread risks associated with particular grazing regimes. For example, grazing during the winter and spring undoubtedly leads to some direct mortality of butterfly larvae, eggs, and pupae by crushing; this source of direct mortality is avoided by summer-fall grazing at the cost of higher grass cover in the habitat.

Some plant species do not do well with grazing. *Dudleya setchellii* (listed as endangered) lives on rock outcrops, is often chewed up by cattle, and tends to be more abundant and vigorous in ungrazed areas (U.S. Fish & Wildlife Service 1998; personal observation). Fencing off selected rock outcrops that provide little habitat value for the butterfly is effective mitigation. Grazing may not be the perfect solution to grass invasions, but, given the current state of our knowledge, existing moderate grazing regimes appear to be a suitable management prescription for most serpentine grasslands in south San Jose.

Ungrazed, grass-dominated sites can recover to acceptable levels of habitat quality for the butterfly following the reintroduction of grazing, but recovery may take years. Introduced grasses perform well during wet years and poorly during droughts, and *Plantago erecta* performs well during droughts (Hobbs & Mooney 1995). The years following reintroduction of grazing in SC1 (1995-1998) all had above-average rainfall, including the record El Niño winter of 1997-1998, so grass growth was favored. Conversely, a severe drought period may slow the grass invasion and speed recovery; recovery of the KC grazing enclosure took place over 5 years of drought.

Fire and mowing may not be appropriate management tools for serpentine grasslands. Fire in these grasslands is poorly studied, and extensive controlled experimental studies would be required to properly assess the effects of fire. Fire may kill Bay checkerspot larvae that are in summer diapause beneath rocks and in the soil. Mowing and hay removal are not feasible because of rock outcrops and steep slopes.

Documenting the effects of grazing removal creates a dilemma for scientists and conservationists attempting to protect biologically rich serpentine grasslands. Removal of grazing is a rapid route to diminished habitat quality and population-level extinctions of the one protected species that can stymie broad development plans on serpentine soils. One landowner (SC2) has already followed that course in hopes of eliminating the habitat value from his parcel. The U.S. Fish and Wildlife Service (1998), however, is aware of the problem and still regards the degraded habitat as important to protect. Whether the Endangered Species Act can be invoked to force landowners to continue grazing is an open question.

Although N deposition on the peninsula is lower than in south San Jose, it still may have chronic long-term effects. *Lolium* may require several years of drier weather to disappear from JR. Edgewood Natural Preserve, which has consistently supported patches of high-density *Lolium* throughout the grassland (S.B.W., personal observation), may be affected by short-range deposition from an eight-lane freeway that bisects the site. Introduction of grazing or fire at EW and JR would be difficult without careful research and planning to address scientific and political concerns.

The ultimate solution is to eliminate sources of excess N, a much larger societal problem that will take decades to solve. Air pollution standards based on direct human health issues may not be stringent enough to avoid negative effects on N-sensitive ecosystems. The San Francisco Bay area (and all of California) presently meet federal and state standards for mean annual NO<sub>2</sub> concentrations (California Air Resources Board 1996). Cars are the major regional source of NO<sub>x</sub> (60% or more), and little progress has been made in reducing car use despite chronic traffic problems. Proposed new air quality rules will only incrementally decrease NO<sub>x</sub> levels and offer no short-term relief.

The threads of this story highlight the interdisciplinary nature of conservation biology. Much of the evidence linking N deposition to the grass invasions is still circumstantial and inferential. More research is needed in a number of areas: (1) refinement of deposition estimates; (2) effects of low-level chronic N additions over several years; (3) effects of grazing regime on serpentine, both in terms of plant composition and N dynamics; (4) time course of restoration following the reintroduction of grazing; (5) recovery of Bay checkerspot populations; and (6) alternatives to grazing, especially fire. Such research would necessarily draw on atmospheric chemistry, plant ecology, ecosystem ecology, range management, population biology, and other disciplines and would be a major, long-term research program.

The number and identity of rare species in California beyond the Bay checkerspot butterfly that are being negatively affected by N deposition is unknown. Dry N deposition in California varies greatly with location (Blanchard et al. 1996): 24–29 kg N/ha/year in the Los Angeles Basin; 10–20 kg N/ha/year in Central Valley cities (Bakersfield and Sacramento); 6 kg N/ha/year in Fremont and Santa Barbara; and around 1 kg N/ha/year in rural locations at Gasquet (far northern California) and at Sequoia and Yosemite National Parks in the Sierra Nevada. Other estimates of N deposition in California range up to 45 kg N/ha/year in forests in the San Bernardino Mountains in the Los Angeles basin, which are showing signs of N saturation (Bytnerowicz & Fenn 1996). Coastal sage-scrub communities are being transformed by the invasion of annual grasses driven by N deposition (Allen et al. 1998).

## Conclusions

The enhancement of the global N cycle is but one aspect of global change. Land-use alterations and invasive species are two widely recognized components of global change (Vitousek 1994), and the plight of the Bay checkerspot butterfly demonstrates how all three factors interact to threaten local biological diversity. Land-use change (urbanization) directly threatens the serpentine grasslands of the San Francisco Bay area (Murphy & Weiss 1988a). The invasive grasses that have dramati-

cally changed California's grasslands are poised to dominate the last refugia for the native grassland flora and fauna, given the chance. That chance is provided by smog-induced fertilization, but only with the additional land-use change of removing grazing. The economics of grazing adjacent to major urban areas may not be viable over the long term. It is ironic that grazing, which has contributed so greatly to the transformation of California's native grasslands, may prove necessary for their maintenance on nutrient-poor soils downwind of major pollution sources.

Nitrogen deposition is a long-term regional and global problem that deserves increased attention from scientists and policymakers (Vitousek et al. 1997). Many other rare ecosystems, communities, and species worldwide may be affected by N deposition from nearby (or even distant) air pollution sources. The effects of additional N may be obvious, as observed in this study, but also may be more subtle, such as changes in soil chemistry and plant tissue that can affect herbivores and overall nutrient cycling. Identification of the problem and establishment of suitable management regimes will undoubtedly take much research and experimentation by conservation biologists and managers.

## Acknowledgments

The following people gave valuable advice and assisted with data collection and analysis: E. Allen, C. Blanchard, C. Boggs, A. Bytnerowicz, J. Dukes, P. Ehrlich, C. Field, L. Hildemann, S. Hobbie, R. Hobbs, D. Hooper, A. Launer, P. Matson, H. Mooney, D. Murphy, J. Neff, T. Peterson, T. Pierce, D. Pritchett, S. Thayer, M. Torn, P. Vitousek, A. Weiss, R. White, and L. Zander. Waste Management, Inc., Castle and Cooke, Inc., United Technologies Corporation, Inc., Shea Homes, and J. Bumb provided site access and financial support. S.B.W. was supported by P. and H. Bing.

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3-1999

# Bush Lupine Mortality, Altered Resource Availability, and Alternative Vegetation States

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## BUSH LUPINE MORTALITY, ALTERED RESOURCE AVAILABILITY, AND ALTERNATIVE VEGETATION STATES

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**Abstract.** Nitrogen-fixing plants, by altering the availability of soil N, potentially facilitate plant invasion. Here we describe how herbivore-driven mortality of a native N-fixing shrub, bush lupine (*Lupinus arboreus*), increases soil N and light availability, which promotes invasion by introduced grasses to the detriment of a native plant community.

Soils under live and dead lupine stands contained large amounts of total N, averaging 3.14 mg N/g dry mass of soil (398 g/m<sup>2</sup>) and 3.45 mg N/g dry mass of soil (438 g/m<sup>2</sup>), respectively, over four years. In contrast, similar lupine-free soil was low in N and averaged only 1.66 mg N/g dry mass of soil (211 g/m<sup>2</sup>) over three years. The addition of N fertilizer to lupine-free soil produced an 81% increase in aboveground plant biomass compared to plots unamended with N. Mean rates of net N mineralization were higher under live lupine and where mass die-off of lupine had occurred compared to soils free of bush lupine. At all sites, only 2.5–4.2% of the total soil N pool was mineralized annually.

Soil enriched by lupine is not available to colonists while lupines are alive. The dense canopy of lupine shades soil under shrubs, reducing average photon-flux density in late spring from 1725  $\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$  (full sunlight) to 13  $\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$  (underneath shrubs). Stand die-off due to insect herbivory exposed this bare, enriched soil. In January, when annual plants are establishing, average photon-flux density under dead lupines killed by insect herbivores was 370  $\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ , compared to the photon-flux density under live lupines of the same age, which averaged 83  $\mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ . The availability of bare, N-rich patches of soil enabled nonnative annuals (primarily *Lolium multiflorum* and *Bromus diandrus*) to colonize sites, grow rapidly, and dominate the plant assemblage until lupines reestablished after several years. The N content of these grasses was significantly greater than the N content of the mostly native plants that occupied adjacent coastal prairie devoid of bush lupine. Between 57 and 70% of the net amount of N mineralized annually was taken up by introduced grasses and subsequently returned to the soil upon the death of these annuals. Even in the absence of further N inputs, we estimate that it would take at least 25 yr to reduce the soil N pool by 50%, indicating that the reestablishment of the native prairie flora is likely to be long term.

**Key words:** bush lupine; insect herbivory; plant assemblages; plant invasion; soil nitrogen mineralization and enrichment.

### INTRODUCTION

Nitrogen-fixing plants have long been known to have important community- and ecosystem-level impacts. These effects have been traditionally studied within primary successional communities, where enrichment of N-poor soil by native N-fixers can facilitate successional change (Olson 1958, Lawrence et al. 1967, Connell and Slayter 1977, Walker and Chapin 1987, Morris and Wood 1989). Recently, however, with the increasing spread of nonnative N-fixing trees and shrubs, concern has focused on the role of exotic N-fixers in native plant communities (Vitousek et al. 1987, Witkowski 1991, D'Antonio and Vitousek 1992). Because soil N enrichment is thought to decrease species richness and to increase the susceptibility of communities to invasion (Hedde and Specht 1975, Heil

and Diemont 1983, Bobbink and Willems 1987, Tilman 1987, Aerts and Berendse 1988, Hobbs et al. 1988, Huenneke et al. 1990, Bobbink 1991, Hobbs and Huenneke 1992, Marrs 1993, Wedin and Tilman 1996), alien N-fixers potentially facilitate the spread of other exotic plants through their effects on soil N availability (Vitousek and Walker 1989, Witkowski 1991, Stock and Allsop 1992, Stock et al. 1995).

Studies of the effects of soil N enrichment on plant assemblages usually involve fertilizing plots and examining subsequent changes in species composition (Willis and Yemm 1961, Tilman 1987, Bobbink et al. 1988, Hobbs et al. 1988, Huenneke et al. 1990, Burke and Grime 1996, Wedin and Tilman 1996). More recently, atmospheric deposition of N has been linked to changes in plant composition (Press et al. 1986, Bobbink and Willems 1988, Morris 1991). But what of the effects of native N-fixing plants on soil N availability and invasion? We know surprisingly little about this

issue, despite the fact that many communities increasingly contain a suite of alien grasses and forbs that might benefit from high N soil created by native N-fixers.

Limited evidence suggests that any facilitating effect of N-fixing plants on invasion might not be realized until after the N-fixer dies. For example, in early successional communities on a recently erupted volcano, Mount St. Helens, *Lupinus lepidus* enriches N-poor soil and, in doing so, appears to facilitate colonization by other plants, principally the exotic, *Hypochaeris radicata*. However, sites enriched by *L. lepidus* only become colonized after *L. lepidus* dies (Del Moral and Bliss 1993). This raises the question of whether herbivores that kill N-fixing plants may modulate the impact of these plants on invasion. Although studies have shown that herbivory on dominant plants can alter community and ecosystem properties (McNaughton 1979, Jefferies 1988, Pastor et al. 1988, 1993, Whicker and Detling 1988, Jefferies et al. 1994, McNaughton et al. 1997), these effects have been little explored in systems where N-fixing plants are involved. Hence, while ecosystem changes initiated by N-fixing plants are well recognized (Vitousek and Walker 1989, Witkowski 1991), feedbacks between these plants, herbivory, altered ecosystem-level characteristics, and invasion are poorly documented. In this study, we examine how herbivore-driven changes in the abundance of a dominant N-fixing shrub alter ecosystem processes to the benefit of exotic plants.

Californian coastal prairie grasslands often are dominated by bush lupine (*Lupinus arboreus*), a large, native, N-fixing shrub that is capable of rapidly enriching soil (Gadgil 1971, Palaniappan et al. 1979, Baker et al. 1986, Bentley and Johnson 1991, 1994). At our study site, bushes are frequently attacked by a variety of insect herbivores, some of which kill plants (Strong et al. 1995, Maron 1998). On a fine spatial scale, mortality of isolated islands of lupines growing within mostly native coastal prairie promotes subsequent invasion into dead lupine patches by weedy introduced plants (Maron and Connors 1996). Invasion on this scale appears to be facilitated by the dual combination of soil enrichment and shrub death. Lupines enrich patches of soil; subsequent herbivore-driven lupine death exposes these enriched sites to light, making them available to weedy colonizers (Maron and Connors 1996).

Here we examine the ecosystem-level impacts of bush lupine in more detail, and we consider whether the same processes that promote invasion on a fine spatial scale, into individual dead lupine patches, might occur on larger spatial scales, after entire lupine stands are killed by herbivores. In particular, we ask how episodic recruitment and the establishment of dense lupine stands and subsequent death from insect herbivory alter the availability of soil N and light, and whether changes in the availability of these resources might be responsible for promoting changes in the prairie plant

assemblage. To address these questions, we determined whether prairie soils devoid of bush lupine are N-limited for plant growth. We then quantified the extent to which lupine occupancy and herbivore-driven mortality alter total soil N pools, rates of net N mineralization, and light availability beneath shrubs, which appears to be important in enabling N-rich patches created by lupine to be colonized. Finally, we measured the fraction of the total annual pool of mineralized N that is utilized by invasive annual plants that colonize enriched sites, and we estimated, in the absence of N inputs, the time required for the soil N pool to be depleted to a value comparable to that of soil free of lupine.

## METHODS

### *Study area*

This study took place on the 147-ha University of California Bodega Marine Reserve (BMR), situated on a coastal headland in Sonoma County, California, USA (Barbour et al. 1973). The site experiences a typical Mediterranean climate, with seasonal rains heaviest from November to March. Bush lupine, a perennial evergreen shrub, is abundant on BMR, where it forms dense stands within a coastal prairie plant community. Bushes can live at least 10 yr if they escape herbivory, but many shrubs die at younger ages (Davidson 1975; J. L. Maron, *personal observation*). The interstitial spaces between bush lupines primarily support the introduced annual grasses *Bromus diandrus* and *Lolium multiflorum*, along with a few native and introduced forbs, most of which are annuals. In the few grassland sites free of lupine, the prairie plant community is composed of several species of native perennial bunchgrasses and many small native annual forbs, including the annual lupine, *Lupinus nanus* (see Maron and Connors 1996 for a full description of this plant community). These remnants of mostly native coastal grassland vegetation are increasingly rare in California; most coastal grasslands in California are composed of introduced grasses and forbs (Heady et al. 1995).

To determine whether lupine occupancy and herbivore-driven die-off alters soil N relations, we compared soil N dynamics and plant community properties within three large (1–1.5 ha) contiguous sites (Fig. 1): a dense lupine stand (hereafter referred to as LL, for live lupine), a dead lupine stand composed of >40 000 dead plants that were killed by subterranean ghost moth (*Heppialus californicus*) caterpillars in the summer of 1992 (hereafter referred to as DL, for dead lupine), and an area mostly free from bush lupine (referred to as NL, for no lupine). The vegetative history of these sites is known from aerial photographs taken at various times over the last 40 years. At both DL and LL, dense stands of lupine have repeatedly died off, but subsequently regenerated from recruitment out of a dense seedbank (Strong et al. 1995, Maron and Simms 1997). In contrast, NL (and several smaller sites nearby) has his-

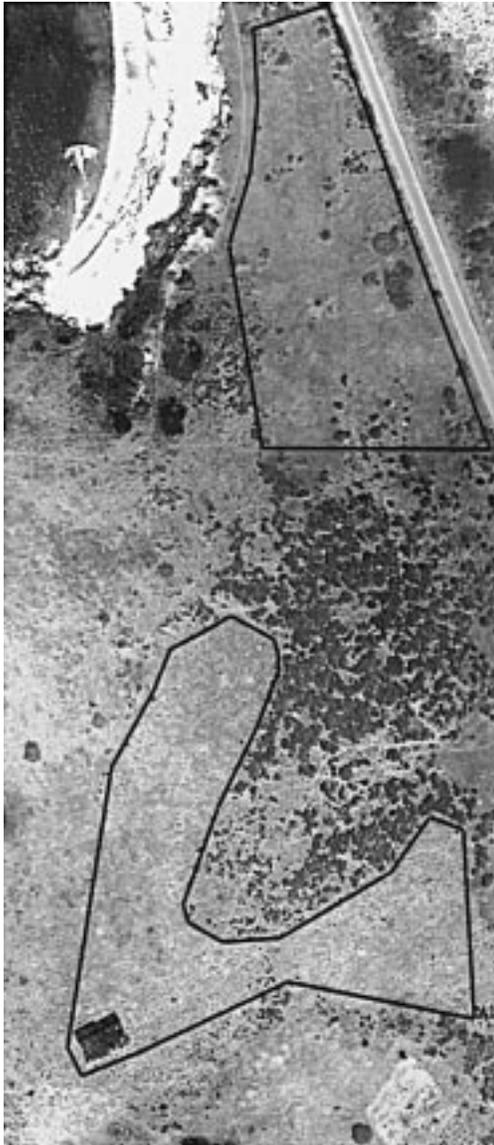


FIG. 1. Aerial photo showing NL (no lupine), LL (live lupine), and DL (dead lupine) sampling sites. NL (closest to the ocean at the top of the photo) and DL are outlined; LL is unmarked and lies between NL and DL. The photo was taken in August 1994.

torically lacked lupine, probably because prior agricultural activities on Bodega Head >35 yr ago kept plants out of these areas. Lupine colonization of these sites has been slow, because heavy lupine seeds do not disperse far from parent plants. As a result, these areas presently contain only isolated islands of bushes rather than dense stands, which dominate most of the grassland habitat at BMR. Although colonization of lupine-free sites has been slow, once established, the lupines flourish (J. L. Maron, *unpublished data*). Apart from differences in lupine density, there are no obvious differences in gross site characteristics between NL, DL,

and LL. However, in order to compare soil characteristics at the three primary sites, we took 50 g of soil at a depth of 5 cm, just below the surface layer of loose litter, at eight randomly chosen locations within each of the three areas. The samples were analyzed for pH; percentage of sand, silt, and clay; and cation-exchange capacity, using standard analytical procedures at the University of California at Davis soil laboratory. We used one-way ANOVAs to compare soil characteristics among sites. These and all other statistical analyses were performed using SYSTAT 5.0 for Windows (SYSTAT 1992).

An advantage of knowing the vegetative history of contiguous sites with similar soil characteristics is that it enabled us to determine how historical differences in lupine occupancy affect soil N relations and plant community characteristics. However, a drawback to this design is a lack of replicated sites with different known histories of lupine occupancy. Since there was only one stand of lupine that died en masse during most of this study (DL), as well as only one site about which we have unequivocal information (from >40-yr-old aerial photographs) of the lack of lupine (NL), it was not possible to replicate these large (>1.5 ha) sites. This lack of site replication hampers our ability to generalize beyond the sites we studied. However, we sampled total soil N and plant biomass at three additional smaller sites that have lacked lupine for at least 15 yr, as well as beneath live lupine bushes in several additional stands (as a replicate of the LL stand). In the case of the dead lupine stand, the only possibility for replication came from sampling the soils and plant assemblages in the interstitial spaces between lupines at three additional sites. Aerial photographs and the remains of dead wood in the soil show that these interstitial spaces represent places where lupines have died in the recent past. Additionally, we measured plant-available soil N under isolated dead lupines within NL, to compare with the majority of sites at NL that lacked lupine.

#### *Nitrogen limitation in soil free from bush lupine*

In order to determine whether N might be limiting in soils historically free from bush lupine, we established experimental blocks in eight randomly selected locations at NL. Each block contained two 1-m<sup>2</sup> plots that received either N or no amendment. Fertilization was randomly assigned to a plot within a block. Wire mesh, 75 cm in height, was erected around each 1-m<sup>2</sup> plot to exclude aboveground mammalian grazers. The plots were fertilized on 18 November 1994 with the addition 14.34 g of reagent grade ammonium nitrate (5.02 g N/m<sup>2</sup>). This amount of N was approximately equivalent to annual difference in amount of mineralized inorganic N found in areas with and without lupines (see *Total soil nitrogen and carbon*). On 17 February 1995, we again added N to plots (5.02 g N/m<sup>2</sup>) in an attempt to maintain high inorganic N concentrations in the upper levels of soil in the presence of extremely heavy rains (152 cm/yr, over twice the

annual average at our site; Bodega Marine Laboratory, unpublished data).

On 6 May 1995 at peak biomass for annual species, we subsampled total plant biomass within all plots by cutting and bagging aboveground vegetation within a  $30 \times 30$  cm quadrat placed in the center of each plot. The vegetation was sorted into three categories—grass, forbs, and *L. nanus*—dried at 60°C for 36 h, and then weighed. We used a one-way ANOVA to test for differences in mean plant biomass between fertilized and nonfertilized plots.

#### *Total soil nitrogen and carbon*

In order to examine the influence of bush lupine on total soil N and carbon, we collected 50 g of soil at the rooting depth of most grasses and forbs in the community, between 5 and 10 cm below the soil surface, immediately below the loose litter layer. We collected samples at randomly selected locations within each of the three different primary study areas. Soil samples were collected at LL and DL sites in April 1993, at the NL site in July 1994, and at all three sites in both May and October 1994, in October 1995, and in May 1996 ( $n = 3-6$  in each sampling period, except on 18 October 1994, where  $n = 2$ ). In May 1996 we collected four soil samples at the three additional sites that lacked lupine, at one additional site under live lupine, and at three additional sites in the interstitial spaces between live lupines (where lupines had died in the recent past). In order to determine soil N levels at soil depths  $>5$  cm, on 19 October 1995 we collected three additional soil samples at each of the primary areas at depths of 10 cm and 20 cm. All soils were dried at 50°C for 1 wk, and total soil N and C were measured with a LECO 600 carbon-hydrogen analyzer (St. Louis, Missouri, USA).

We compared concentrations of total soil N among sites by first averaging samples within sites and within year and then performing one-way ANOVAs on log-transformed data. Least square means post hoc comparisons (with Bonferroni adjustment) were made to examine differences between individual sites. Average C:N ratios of soils were compared among sites using a one-way ANOVA on arcsine-transformed data.

#### *Exchangeable levels of inorganic soil nitrogen and net rates of nitrogen mineralization*

To determine how soil N pools varied on a fine spatial scale between isolated dead lupine bushes (killed in September 1993) scattered throughout NL and surrounding prairie that lacked lupine, we measured exchangeable levels of  $\text{NH}_4^+$  and  $\text{NO}_3^-$  in soil, monthly, from November 1993 to March 1995. We sampled soil under solitary dead lupines at NL and at sites several meters away. For comparison with fine-scale variation in N caused by lupine colonization at NL, we also sampled soil monthly at LL and at DL from April 1993 through April 1995. From May to August 1995 (in-

clusive), measurements were interrupted; further measurements were made at NL (away from dead lupines only), at LL, and at DL, from September 1995 to April 1996. Under isolated dead lupines at NL we took only one sample per month for determination of inorganic N. At all other sites, each month we took two 50-g samples between 5 and 10 cm below the soil surface. In each area we sampled soil at six randomly chosen locations separated by at least 20 m. One of the two soil samples was used for the determination of exchangeable levels of soil  $\text{NH}_4^+$  and  $\text{NO}_3^-$ . The other sample was incubated in the field at 5 to 10 cm beneath the soil surface for one month to determine net rates of N mineralization. Samples to be incubated were placed in a polythene bag (Glad sandwich bag) porous to air exchange; each bag was enclosed in a wire cage to protect it from pocket gophers (*Thomomys bottae*). When soil was collected monthly for measurement of extractable N, we also collected the buried bags and replaced them with bags containing new soil sampled from randomly chosen locations.

Immediately after samples were collected, we measured percent soil moisture in 10–15 g subsamples. These subsamples were weighed, dried at 60°C for 2–7 d, and then reweighed. Moisture values were used to express rates of net N mineralization and levels of exchangeable ammonium and nitrate on a dry mass basis.

Ammonium and nitrate ions were extracted from 10 g of soil placed in a flask containing 50 mL of 1 mol/L KCl, which was kept at room temperature and shaken periodically. Twenty-four hours later, we filtered each sample through Whatman 40 ashless filter paper into vials and froze the filtrate. Some samples were analyzed for  $\text{NH}_4^+$  and  $\text{NO}_3^-$  on a Carlson diffusion-conductivity analyzer (Carlson 1986); others were analyzed colorimetrically using a Technicon auto analyzer (Technicon Industrial Systems, Tarrytown, New York, USA). The phenol-hypochlorite method was used to detect  $\text{NH}_4^+$ , while Marshall's reagent was used to detect  $\text{NO}_3^-$  (as  $\text{NO}_2^-$ , following cadmium reduction). The analytical error associated with the use of this equipment is  $<2\%$ , and detection limits are similar between the two analyzers.

Net N mineralization rates were calculated monthly by determining ammonium and nitrate levels in soil within buried bags after incubation, and subtracting amounts of ammonium and nitrate found in unbagged samples at the beginning of the incubation. The values were summed for each site from May 1993 to April 1994. Mean net mineralization rates from mid-September to mid-April at each site were also calculated by summing monthly values. We used an ANOVA to compare mean annual and 7-mo mineralization rates between sites. We used least square means post hoc comparisons (with Bonferroni adjustment) to examine specific differences between individual sites. We also calculated total soil N and net N mineralization rates on a 1-m<sup>2</sup> basis by estimating the bulk density of a  $1 \times$

1 × 0.1 m block of soil 5–10 cm below the soil surface (127 kg). For comparisons of levels of total inorganic N, we determined the 4 mo when total net exchangeable inorganic N was highest at each site (November–February and October–January for 1993–1994 and 1994–1995, respectively). We then summed these 4-mo values for each replicate within each location and performed a two-way ANOVA (with year as a random factor and location as a fixed factor) on log-transformed data to compare the averages of these summed values between sites and across years. We used least square means post hoc comparisons (with Bonferroni adjustment) to compare means between sites.

#### *Species composition*

In February 1993 we established six 4 × 4 m permanent plots at randomly chosen locations at the DL site in order to characterize the plant community that established where lupine had died in the late summer of 1992. At this time the woody skeletons of lupines were intact, and the soil was bare of vegetation. In May of each year, from 1993 to 1996, we subsampled vegetation in two permanent 0.5 × 0.5 m subplots within each plot. Within the subplots, we recorded shoot frequency of all plant species within a grid of 25 × 10 cm squares and calculated average changes in frequency for each species each year within each of the six plots.

In order to determine the amount of N in the new plant biomass and to express it as a percentage of the soil N pool, we measured total plant biomass and tissue carbon and N. We sampled plant biomass in late April or early May (1993–1996 inclusive), when the standing crop was at or close to its peak. Outside the 4 × 4 m plots, we placed a 30 × 30 cm quadrat in three (1993) or four (1994–1996) randomly chosen sites dominated by *L. multiflorum*, *B. diandrus*, and the forbs *Claytonia perfoliata* and *Stellaria media*. We sampled the biomass of each of these two groups separately in 1993 ( $n = 3$ ), but in subsequent years the vegetation was almost exclusively composed of annual grasses, and hence separate samples of biomass were not collected.

In May 1996, we sampled aboveground plant biomass in the interstitial spaces between live lupines, where individual lupines had died at the primary LL site, and within three additional live lupine stands. We further sampled vegetation under live lupines at the LL site. We took four samples of interstitial plant biomass from each site and under canopies at the LL site, as described above. Grasses and forbs were separated before drying.

Each May from 1993 to 1996, we sampled aboveground plant biomass within four 25 × 25 cm quadrats placed in randomly selected locations within the NL site, and in May 1996 at the three additional sites that were free from bush lupine. The samples collected in 1996 were separated into grasses and forbs and weighed. Vegetation sampled at all sites was dried at

50°C for at least 7 d, weighed, ground, and the C and N contents of tissues were determined with a LEECO 600 analyzer (LECO Corporation, St. Joseph, Missouri, USA).

In May 1996, we measured the root biomass and root : shoot ratio of 10 randomly selected plants of *B. diandrus* and *L. multiflorum*. Plants were excavated, gently washed in water to remove soil, and separated into shoots and roots, which were dried for 55 h at 60°C and weighed. The N and C contents of tissues were analyzed as described above.

#### *Light measurements*

To quantify the extent to which lupines decrease light availability, we used a LI-COR quantum sensor to measure photon-flux density (PFD) above and at ground level under the canopies of six randomly chosen 2-yr-old lupines. Measurements were made in full sunlight at midday solar time on 9 May 1996. To quantify how dead lupine skeletons affect light penetration to the soil surface during winter, when most annual grasses are establishing within areas where lupines have died, on 5 January 1998 we measured PFD at midday under and above the skeletons of eight randomly chosen 2-yr-old dead lupines and under and above eight randomly chosen live lupines of the same age. Dead lupines were within a large lupine stand that had died during fall 1997 (J. L. Maron and S. Harrison, *unpublished data*). We expressed light penetration as the percentage of PFD (measured without obstruction, above lupines) reaching the ground surface, and then compared the mean level of PFD penetrating through live and dead lupines using a *t* test.

#### RESULTS

Soils at each of the three primary sites were similar in mineral composition and pH (Table 1). Soil free from bush lupine at the NL site, however, contained significantly less organic material (ANOVA,  $F_{2,21} = 13.0$ ,  $P < 0.001$ ), which may explain its lower cation-exchange capacity ( $F_{2,21} = 15.6$ ,  $P < 0.001$ ) compared to sites with live and dead lupine.

#### *Nitrogen limitation in soil free from lupine*

Addition of fertilizer to plots at the NL site produced significant effects on total aboveground plant biomass ( $F_{1,14} = 18.4$ ,  $P < 0.001$ ). Biomass harvested from fertilized plots averaged 6563 g/m<sup>2</sup>, an 81% increase over biomass in control plots, which was largely the result of an increase in the growth of grasses relative to that of forbs. On average, 80% of the total biomass sampled in fertilized plots was composed of grasses, whereas grasses only contributed an average of 58% of the biomass in control plots. Average forb biomass was similar in fertilized and control plots ( $F_{1,14} = 0.011$ ,  $P = 0.92$ ).

TABLE 1. Soil characteristics at live lupine (LL), dead lupine (DL), and lupine-free (NL) sites ( $n = 8$  at each site). Data are means  $\pm 1$  SE.

Characteristic	LL	DL	NL
Organic matter (%)	5.1 $\pm$ 0.31	5.6 $\pm$ 0.54	2.9 $\pm$ 0.13
Sand (%)	88.4 $\pm$ 0.32	87.5 $\pm$ 0.42	86.6 $\pm$ 0.46
Silt (%)	6.9 $\pm$ 0.48	6.25 $\pm$ 0.31	7.0 $\pm$ 0.78
Clay (%)	4.75 $\pm$ 0.25	6.25 $\pm$ 0.25	6.4 $\pm$ 0.5
pH	5.6 $\pm$ 0.1	5.5 $\pm$ 0.08	6.0 $\pm$ 0.04
Cation exchange capacity (mmol/100 g)	14.3 $\pm$ 0.73	14.9 $\pm$ 0.68	10.6 $\pm$ 0.26

### Total soil nitrogen and carbon

Concentrations of total soil N were significantly different between sites ( $F_{2,21} = 10.3$  in 1994,  $F_{2,6} = 34.8$  in 1995,  $F_{2,15} = 29.2$  in 1996,  $P < 0.001$  in all years). Soil from the primary live and dead lupine sites contained higher amounts of total N than that in the lupine-free site (least squared difference post hoc multiple comparison with Bonferroni correction,  $P < 0.05$  in all years), although mean total soil N was not significantly different in any year between LL and DL (post hoc comparison  $P > 0.05$ ). Depending on the year, soil from the lupine-free site contained between 36 and 55% less N than lupine-influenced soil (Table 2). Total soil N averaged 342–462 g/m<sup>2</sup> at the LL site, 394–521 g/m<sup>2</sup> at the DL site, and 190–254 g/m<sup>2</sup> at the NL site. At each of the three sites, soil N declined with soil depth (Table 3).

Despite differences among sites in total soil N, average C:N ratios were not statistically different in 1994 or 1995 ( $F_{2,21} = 0.25$  in 1994,  $F_{2,6} = 0.72$  in 1995,  $P > 0.05$  in both years). In 1996, however, the average C:N ratio at the NL site was significantly lower than that at the DL site, although not significantly different from the C:N ratio at the LL site ( $F_{2,15} = 3.86$ ,  $P =$

TABLE 2. Mean ( $\pm 1$  SE) soil N pool sizes, C:N ratios, and cumulative net N mineralization rates, at LL, DL, and NL. (See *Methods* for sampling dates and sample sizes used to calculate annual averages of total soil N and C:N ratios.) Annual (1993–1994) or 8-mo (1994–1995 and 1995–1996) net mineralization rates are sums of monthly averages from each site ( $n = 6$  at each site).

Year	LL	DL	NL
Pool size, total soil N (mg N/g dry mass soil)			
1993	3.7 $\pm$ 0.3	4.1 $\pm$ 0.3	Not sampled
1994	3.1 $\pm$ 0.2	3.3 $\pm$ 0.3	2.0 $\pm$ 0.1
1995	3.1 $\pm$ 0.1	3.3 $\pm$ 0.3	1.5 $\pm$ 0.03
1996	2.7 $\pm$ 0.1	3.1 $\pm$ 0.2	1.5 $\pm$ 0.1
C:N ratio			
1993	12.8 $\pm$ 0.5	13.5 $\pm$ 0.2	
1994	12.4 $\pm$ 0.7	12.4 $\pm$ 0.9	13.1 $\pm$ 0.6
1995	13.7 $\pm$ 0.4	13.1 $\pm$ 0.6	13.1 $\pm$ 0.04
1996	14.2 $\pm$ 0.4	15.0 $\pm$ 0.5	13.3 $\pm$ 0.4
Net mineralization rate ( $\mu$ g/g dry mass soil)			
1993–1994	107.9 $\pm$ 12.6	82.4 $\pm$ 9.4	51.4 $\pm$ 9.2
1994–1995	78.9 $\pm$ 17.1	92.8 $\pm$ 12.8	36.3 $\pm$ 3.9
1995–1996	86.2 $\pm$ 26.4	80.2 $\pm$ 13.9	30.5 $\pm$ 2.7

0.044; post hoc least square means with Bonferroni adjustment = 0.042 between values from NL and DL sites).

Patterns of total soil N at additional replicate sites sampled in 1996 were similar to those at the primary sites. Total soil N was significantly lower at the three additional lupine-free sites (mean value for all three sites = 207 g N/m<sup>2</sup>), compared to total soil N under lupines (mean value = 305 g N/m<sup>2</sup>), or compared to that in interstitial areas between live lupines (mean value = 283 g N/m<sup>2</sup>;  $F_{2,25} = 6.98$ ,  $P < 0.005$ ). The mean value for the lupine-free sites was not significantly different from that at the primary NL site ( $F_{3,14} = 0.619$ ,  $P = 0.614$ ).

### Exchangeable levels of inorganic soil nitrogen and net rates of nitrogen mineralization

Exchangeable NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> concentrations varied seasonally at all sites; peaks in NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> occurred from early to midwinter, with the onset of rain (Figs. 2 and 3). During peak periods in midwinter, total inorganic N was consistently different among sites and years ( $F_{3,40} = 6.39$  and  $F_{1,40} = 5.1$ ,  $P < 0.01$  for site and year, respectively), with year-to-year differences varying by site (as indicated by a significant site-by-year interaction;  $F_{3,40} = 3.4$ ,  $P < 0.03$ ). In each year, soil under live lupine and dead lupine contained significantly higher inorganic N compared to soil from the lupine-free area (LSD post hoc comparisons with Bonferroni correction based on separate ANOVAs for each year,  $P < 0.001$ ; Fig. 4).

Rates of net N mineralization were significantly different among sites (Table 4). In 1993–1994, greater amounts of N were mineralized annually in soils from LL than in NL (post hoc comparison,  $P < 0.02$ ), but there was no significant difference in net mineralization rates between LL and DL or DL and NL (post hoc

TABLE 3. Mean ( $\pm$  SE) values for total soil N at different depths at LL, DL, and NL sites.

Soil depth (cm)	Total soil N (mg N/g dry mass soil)		
	LL	DL	NL
5	3.1 $\pm$ 0.1	3.3 $\pm$ 0.3	1.5 $\pm$ 0.03
10	2.3 $\pm$ 0.5	2.1 $\pm$ 0.2	1.1 $\pm$ 0.1
20	1.5 $\pm$ 0.1	2.1 $\pm$ 0.2	1.1 $\pm$ 0.2

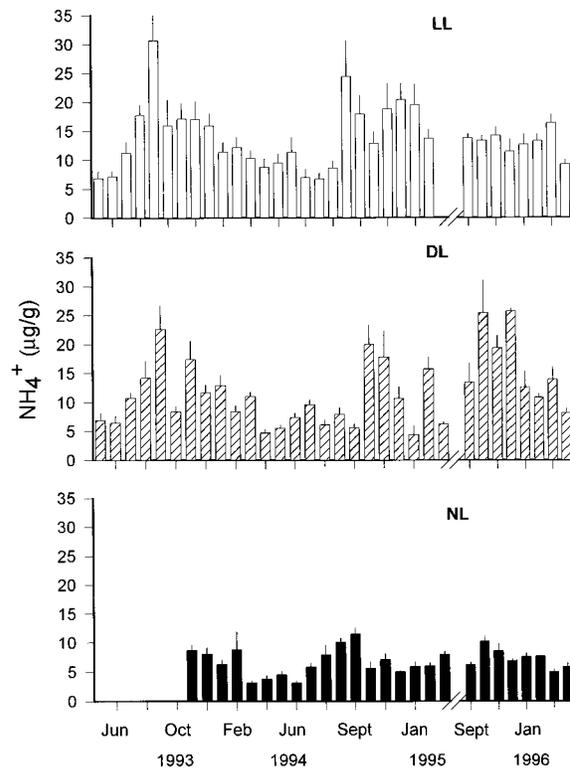


FIG. 2. Ammonium ( $\text{NH}_4^+$ ) concentrations (mean + 1 SE) from within a live lupine stand (LL), a dead lupine stand (DL), and an area mostly free from lupine (NL);  $n = 6$  at each site. Samples were taken every 28–32 d.

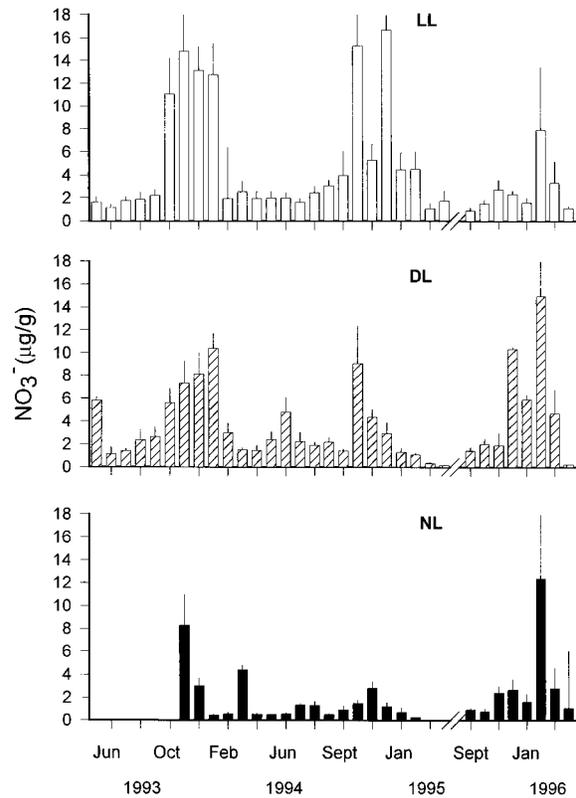


FIG. 3. Nitrate ( $\text{NO}_3^-$ ) concentrations (mean + 1 SE) from within a live lupine stand (LL), a dead lupine stand (DL), and an area mostly free from lupine (NL);  $n = 6$  at each site. Samples were taken every 28–32 d.

comparison,  $P > 0.05$ ). From mid September to mid April 1994–1995 and 1995–1996, mineralization was significantly higher in soils from both the live and dead lupine stands than in soils from the NL site (Table 2; post hoc comparison,  $P < 0.05$  for both LL vs. NL and DL vs. NL). Net N mineralization rates also showed seasonal patterns in both 1993–1994 and 1994–1995, with mineralization rates being highest at all sites in mid winter (Fig. 5).

Only a small fraction of the total pool of soil N in the different sites became available as net mineralized N each year. For both 1993–1994 and 1994–1995, this percentage (2.5–4%) was not substantially different among sites.

*Species composition, plant biomass, and plant nitrogen content*

In the first spring after stand die-off, the plant assemblage that established in the bare soil within the dead lupine stand was composed primarily of the native forb *C. perfoliata*, the introduced forb *S. media*, and the exotic grasses *B. diandrus* and *L. multiflorum* (Fig. 6). Growth of these colonists was rapid, and the concentration of foliar N was high (Table 5). *B. diandrus*/*Lolium multiflorum* contained an average of 1.52% N. The comparable value for the combined forb biomass

was 1.42% N. In 1995 and 1996, the DL plant community was dominated by *L. multiflorum* and *B. diandrus* (Fig. 6). The combined biomass of these species contained an average of 1.3% and 0.96% N in 1995 and 1996, respectively. In 1996, belowground biomass of the grasses averaged  $11.7 \pm 1.26\%$  (mean  $\pm 1$  SD) of the aboveground biomass. Nitrogen concentration was 88% of that of the aboveground biomass. In contrast to the DL site, plant biomass was low in the lupine-free NL site. Mean values for plant biomass were 295, 288.8, and 324 g dry mass/m<sup>2</sup> in 1993, 1995, and 1996, respectively. These values were all significantly lower than corresponding values for the DL site (for each year,  $P < 0.02$ ). At the NL site, forbs contributed an average of 22% of the aboveground biomass, whereas, at the DL site, over 99% of the biomass was composed of introduced grasses in 1996.

In 1995 in the NL site an average of 4.2 g N/m<sup>2</sup> was present in aboveground vegetation, compared to 9.7 g N/m<sup>2</sup> in vegetation at the DL site. A similar pattern was found in 1996, when the vegetation at the NL site contained, on average, 4.7 g N/m<sup>2</sup> (forbs = 1.33 g N/m<sup>2</sup>, grasses = 3.34 g N/m<sup>2</sup>), again a smaller amount of N than that present in aboveground biomass at the DL site, which averaged 7.4 g N/m<sup>2</sup>.

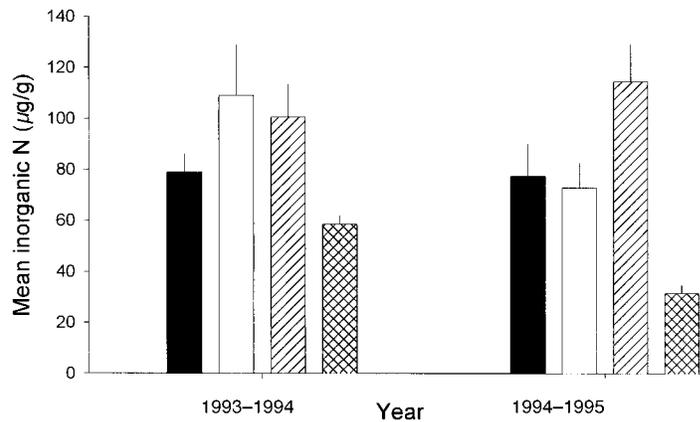


FIG. 4. Total inorganic N (mean + 1 SE) under isolated dead lupine at NL (black bar), within a dead lupine stand at DL (open bar), under a live lupine stand at LL (hatched bar), or at NL away from isolated dead lupines (cross-hatched bar). The figure presents means from six values at each site, with each value being the sum of measurements taken monthly across a 4-mo period when levels of exchangeable N were highest.

Amounts of aboveground biomass at the additional sites sampled in 1996 were similar to those of the primary sites. The three sites sampled (in addition to NL) that were lupine-free supported total aboveground biomass that averaged 289, 166.5, and 241 g dry mass/m<sup>2</sup>, respectively (compared to 324 g/m<sup>2</sup> at NL and 771 g/m<sup>2</sup> at DL in 1996). Forbs contributed an average of  $18.3 \pm 1.89\%$ ,  $29.5 \pm 3.73\%$ , and  $24.9 \pm 1.5\%$  to the biomass at the three additional sites, respectively. In addition, average biomass at these sites was significantly lower than that sampled from the interstitial spaces between lupines at the three additional sites ( $F_{1,18} = 37.36$ ,  $P < 0.0005$ ), where mean values for aboveground biomass were 513.9, 479.5, and 623.8 g dry mass/m<sup>2</sup>, respectively. The interstitial plant community growing between lupines was composed almost entirely of grasses at all sites (99% of the total biomass). Amounts of plant biomass at the three additional sites within each habitat type (LL, DL, NL) were not significantly different from corresponding values for the primary sites ( $P = 0.33$ ; ANOVA with site nested within habitat).

#### Measurements of photon-flux density

The average PFD above the lupine canopies in the full sunlight in May 1996 was  $1725 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ ,

whereas under the lupine canopy, at ground level, the average value fell to  $13 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$  ( $n = 6$ ), <1% of the PFD in full sunlight. In January 1998, when lupine canopies are less dense than in spring, the average PFD under live lupine canopies was  $83 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ , which was 8% of the average PFD in full sunlight ( $1040 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ ). Significantly more sunlight penetrated through dead lupine skeletons (mean PFD =  $370 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ ) than through the can-

TABLE 4. ANOVA testing for differences in mean net N mineralization rates across 12 mo (1993-1994) or 7 mo (1994-1995 and 1995-1996) between LL, DL, and NL sites.

Source	df	MS	F	P
1993-1994				
Site	2	4800	7.3	0.006
Error	15	660		
1994-1995				
Site	2	5202	5.5	0.016
Error	15	945		
1995-1996				
Site	2	3.1	1.5	0.01
Error	15	0.24		

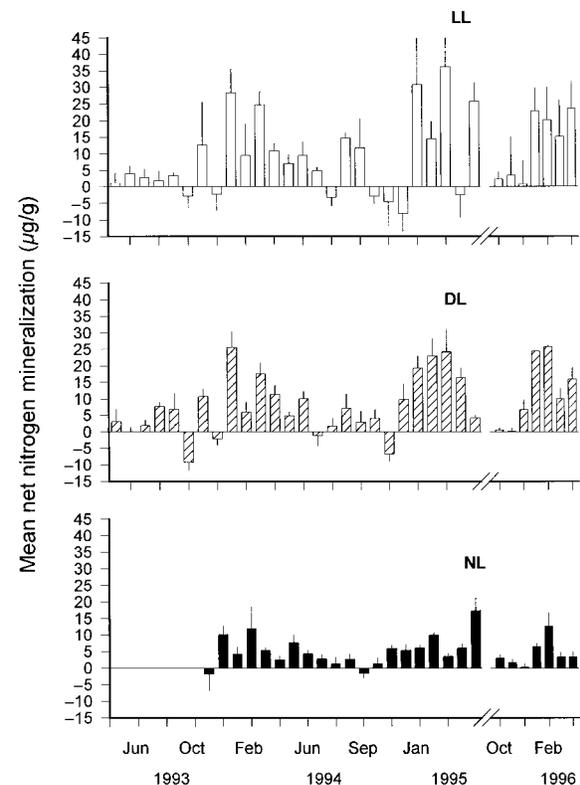


FIG. 5. Monthly rates of net N mineralization (mean + 1 SE) within a live lupine stand (LL), a dead lupine stand (DL), and an area free from lupine (NL);  $n = 6$  at each site. Samples were taken every 28-32 d.

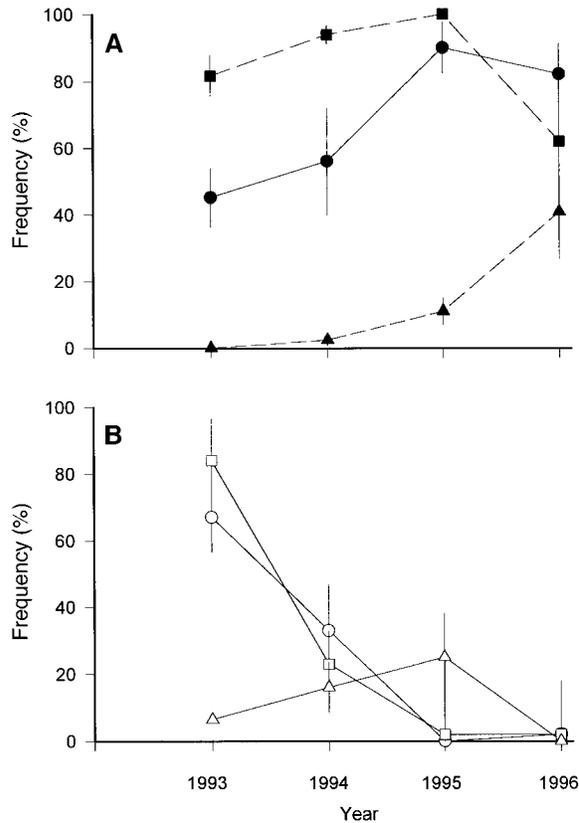


FIG. 6. Mean ( $\pm 1$  SE) frequency (%) of (A) grasses (solid squares, *L. multiflorum*; solid circles, *B. diandrus*) and bush lupine (solid triangles, *L. arboreus*), and (B) forbs (open squares, *S. media*; open circles, *C. perfoliata*; open triangles, *Lasthenia* spp.) in permanent plots at the DL site ( $n = 6$ ). Only plants that occurred at a frequency  $>5\%$  in two of the four years were plotted. The next most common species in plots included the introduced grasses *Aira caryophylla* and *Vulpia bromoides*, the introduced forbs *Cerastium glomeratum*, *Galium aparine*, and *Rumex acetosella*, and the native forbs *Achillea millefolium* and *Trifolium gracilentum*.

opy of live lupine ( $t$  test with separate variance,  $t = 2.88$ ,  $P < 0.025$ ).

DISCUSSION

Continual lupine occupancy and turnover has resulted in N enrichment of sandy soils on BMR, some of which rival highly managed agricultural soil in their fertility (Sprent 1987). In contrast to the lupine-influenced soil, which contained over a three-year period an average of 414 g N/m<sup>2</sup> at 5–10 cm immediately below the loose litter, soil at contiguous lupine-free sites contained over the same period an average of 221 g N/m<sup>2</sup> (Table 2). Organic N content of soils under live lupines was high, and C:N ratios were low. These conditions favor microbial decomposition and high rates of N mineralization. Our results on the effects of lupine on soil N relations are similar to those found for bush lupine at locations where the plant has been introduced

TABLE 5. Mean standing crop and N content of grasses and forbs in DL ( $n = 4$  for both grasses and forbs in all years except 1993, where  $n = 3$ ). The final row shows percentage of annual amount of mineralized N (estimated for 0–10 soil depth; mineralization rates calculated from May to April for both 1993–1994 and 1994–1995) that is sequestered in grasses at the end of the growing season in May. Ellipses indicate absence of forbs from samples in 1994–1996.

Parameters	1993	1994	1995	1996
Standing crop grasses (g dry mass/m <sup>2</sup> )	898	568	767	771
Standing crop forbs (g dry mass/m <sup>2</sup> )	525	...	...	...
N content of standing grass crop (g/m <sup>2</sup> )	13.6	6.0	10.0	7.32
N content of standing forb crop (g/m <sup>2</sup> )	7.2	...	...	...
Percentage of annual pool of net mineralized N that is cycled through plant biomass		56	72	

(Gadgil 1971, Palaniappan et al. 1979, Baker et al. 1986).

Lupine occupancy leads to the continual creation of patches of N-rich soil, which significantly boosts plant biomass, and presumably leads to the increased dominance of exotic grasses, to the detriment of slower growing native species. Yet, while lupines are alive, enriched soil is unavailable for colonists, because light availability beneath the dense lupine canopy is reduced by 98–99%. Insect herbivory and subsequent lupine stand mortality, however, open these enriched sites to light, which then makes them available to invasive fast-growing colonists. Thus, unlike vertebrate herbivores and defoliating insects, which can alter rates of N cycling or change pool sizes of N (Bocock 1963, Chew 1974, McNaughton 1979, Ohmart et al. 1983, Schowalter 1981, Swank et al. 1981, Hollinger 1986, Jefferies 1988, Whicker and Detling 1988, Pastor et al. 1993, Lovett and Ruesink 1995, McNaughton et al. 1997), insect herbivory on lupine acts to increase light availability, which makes N-rich sites available to weedy colonists. A similar result was found for leaf beetles (*Trihabda canadensis*) that fed on goldenrod (*Solidago missouriensis*). Herbivory on goldenrod monocultures led to invasion by other plants; changes in resource (light and N) availability initiated the process that facilitated invasion (Brown 1994).

Neither enrichment nor change in light availability, alone, appears sufficient to promote invasion. For example, fertilization of lupine-free sites does not immediately result in the increased dominance of invasive plants (J. L. Maron and R. L. Jefferies, unpublished data). This may be because fertilized sites do not have a legacy of light limitation, which limits existing species. As well, although plant biomass responds immediately to fertilization, it usually takes several years for plant species composition to change in response to fertilization. By the same token, creation of bare soil alone does not result in invasion. Introduced species

are no more common in disturbed patches in sites free from lupine than they are in undisturbed patches in these same areas (Maron and Connors 1996). Thus, both soil enrichment and lupine death, as dual processes, appear essential in facilitating invasion.

At BMR, sites previously occupied by lupine are invaded primarily by *B. diandrus* and *L. multiflorum* and to a lesser extent by *Vulpia bromoides*. These grasses germinate in early winter, when mineralized soil N is high, and take up between 57 and 70% of the annual mineralized N in the DL site. Plant biomass at this site and in the interstitial spaces between lupines in LL stands is significantly greater than the biomass of the more species-rich (mostly native) plant assemblages growing in lupine-free sites.

Once sites become enriched, we predict that it will take years for the N status of these soils to decline to levels present in soils free from lupine, even in the absence of further inputs of N. At the DL site, where bush lupines were absent for 3–4 yr, N levels remained high. Only a small fraction of the total soil N is mineralized annually, and much of this N appears to be recycled in the litter of annual plants. Calculations based on net N mineralization rates and on N sequestered in annual plants indicate that, in the absence of new N inputs, it would take ~25 yr to reduce the amount of soil N by 50%. Such calculations are based on the rate of turnover of the highly labile fraction of N. Rapid depletion of this fraction within a few years (in the absence of reinvasion of lupine) may result in lower rates of N release in the soil, in spite of the persistent, large soil N pool, as increasingly recalcitrant fractions dominate turnover dynamics. However, the estimate of 25 yr is similar to long-term results from land that has been left fallow, where it has taken >41 yr for the initial pool of soil N to decline by half in the absence of N inputs (Addiscott 1988). Prior N fertilization of sites can affect N mineralization years after fertilization has ceased (Vinton and Burke 1995), and even in unfertilized grasslands, total soil N content declines slowly (Olff et al. 1994).

The amount of N mineralized varied between 2.5 and 4.2% of the total N pool in the three areas, values that are similar to those obtained in other studies of temperate or subarctic soils (Nadelhoffer et al. 1983, Pastor et al. 1984, Hart and Gunther 1989, Wedin and Pastor 1993). Similarities among sites in the percentage of the total soil N pool that is mineralized reflects, in part, the fact that C:N ratios of soil are similar among the three sites. As well, the relative sizes of the most highly labile fraction of the total N pool are comparable in the LL, DL, and NL sites. Values of the annual net amount of N mineralized in the three areas varied between 6.6 g/m<sup>2</sup> at the NL site to 16.6 g/m<sup>2</sup> in soils at the LL site, values that are similar to those from soils of old fields where plots were fertilized with N (Pastor et al. 1987). As found in other studies, seasonal variation in net rates of N mineralization was high (Davy

and Taylor 1974, Taylor et al. 1982, Morecroft et al. 1992), with large pulses of mineralization in the late fall and early winter of 1994, 1995, and 1996.

Although annual grasses dominate sites after lupine mortality, germination from a long-lived seed bank allows lupines ultimately to reestablish in these sites (Maron and Simms 1997). Over periods of 4–10 yr, this produces a cyclical change in the plant community, characterized by stand development, herbivory, stand death, colonization by introduced grasses and forbs, and then reestablishment of dense lupine stands. We have documented one such cycle of change in the DL site, but historical photographs indicate that similar oscillations in the plant assemblages have taken place in the past, both at the DL site and other locations on BMR (Strong et al. 1995). In fact, at the end of this study, we witnessed another large lupine die-off that killed ~95% of the bushes within LL, as well as bushes in nearby areas. Plants died as a result of both heavy defoliation by an unusually dense outbreak of tussock moth (*Orgyia vetusta*) caterpillars and subsequent flooding during an extremely wet winter.

Within Californian coastal prairies, shrub species such as *L. arboreus*, *Ulex europaeus* (another N-fixer), and *Baccharis pilularis* frequently colonize formerly heavily grazed pasture (McBride and Heady 1968, Heady et al. 1995). Although prairie grasses may reestablish at these sites where shrubs do not invade, it is evident from the present study that if N enrichment occurs, it may preclude the establishment of most native species. Active management will be necessary to lower total soil N, especially the size of the highly labile pool. Common practices include cropping of vegetation, removal of surface soil and shrubs, and burning of aboveground vegetation and litter in order to maintain prairie plant assemblages (Marrs 1993). These practices alone, however, may be insufficient to reestablish native flora if poor dispersal limits native plant recruitment into these sites.

Results from this study and those from a companion study (Maron and Connors 1996) suggest that bush lupines and the phytophagous insect herbivores that kill them together influence community and ecosystem processes. The combination of N-fixation and rapid growth, community dominance, and susceptibility to insect herbivory make bush lupine particularly influential at our site in determining coastal prairie plant assemblages.

#### ACKNOWLEDGMENTS

We gratefully thank Barbara Shubin, C. Dean, and D. Ginsburg for field assistance. C. D'Antonio, C. Daehler, A. J. Davy, S. N. Gardner, T. Grosholtz, S. Harrison, P. Kittelson, D. Strong and D. A. Wedin kindly read and improved various versions of this paper. Much thanks goes to the staff of the University of California's Bodega Marine Laboratory, particularly Peter Connors. Financial support was provided to J. L. Maron by grants from the Bodega Field Conference, the California Native Plant Society, Mildred Mathias Fund, the Hardman Foundation, the U.C. Davis Ecology Graduate

Group, Jastro-Shield Fund and Center for Population Biology, and by a post-doctoral fellowship (NSF grant DEB-9508546 to Susan Harrison). R. L. Jefferies was funded by the National Science and Engineering Research Council of Canada.

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# Can Carbon Addition Increase Competitiveness of Native Grasses? A Case Study from California

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## Abstract

There is growing interest in the addition of carbon (C) as sucrose or sawdust to the soil as a tool to reduce plant-available nitrogen (N) and alter competitive interactions among species. The hypothesis that C addition changes N availability and thereby changes competitive dynamics between natives and exotics was tested in a California grassland that had experienced N enrichment. Sawdust (1.2 kg/m) was added to plots containing various combinations of three native perennial bunchgrasses, exotic perennial grasses, and exotic annual grasses. Sawdust addition resulted in higher microbial biomass N, lower rates of net N mineralization and net nitrification, and higher concentrations of extractable soil ammonium in the soil. In the first year sawdust addition decreased the degree to which exotic annuals competitively suppressed the seedlings of *Nassella pulchra* and, to a lesser extent,

*Festuca rubra*, both native grasses. However there was no evidence of reduced growth of exotic grasses in sawdust-amended plots. Sawdust addition did not influence interactions between the natives and exotic perennial grasses. In the second year, however, sawdust addition did not affect the interactions between the natives and either group of exotic grasses. In fact, the native perennial grasses that survived the first year of competition with annual grasses significantly reduced the aboveground productivity of annual grasses even without sawdust addition. These results suggest that the addition of sawdust as a tool in the restoration of native species in our system provided no significant benefit to natives over a 2-year period.

**Key words:** annual grass, coastal prairie grassland, grassland restoration, labile carbon, native perennial grass, nitrogen-enriched soils, sawdust.

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## Introduction

Ecosystem nitrogen (N) enrichment is a common barrier to plant community restoration. Past fertilization (Vitousek et al. 1997), atmospheric N deposition (Bobbink 1991), and invasion by N-fixing shrubs (Vitousek et al. 1987; Maron & Connors 1996) can all increase soil N availability and, in turn, can favor a few fast-growing species at the expense of slower growing species. Although general characteristics of non-native-invading plant species have proven elusive (Mack et al. 2000), N enrichment has been shown to favor invasive non-indigenous species in a variety of habitats (Huenneke et al. 1990; Vinton & Burke 1995; Maron & Connors 1996). Restoration in N-enriched habitats must therefore deal with the question of how to promote slower growing native species in competition with faster growing exotic species under high N conditions (Corbin et al. 2004).

Addition of C in the form of sucrose or sawdust has been suggested as a tool to reduce plant-available N and thereby increase the competitiveness of slower growing

natives (Morgan 1994; Alpert & Maron 2000; Paschke et al. 2000; Blumenthal et al. 2003). For this tool to work the addition of sucrose or sawdust must increase microbial N immobilization and decrease plant-available N. Under lower N conditions, growth of all vegetation would be expected to decrease, but if faster growing species are disproportionately affected by lower soil N concentrations, slower growing native species may benefit indirectly due to reduced competition from fast-growing exotic species.

Previous experimental tests of the benefits of C addition to native species have considered the impact of sucrose or sawdust on mixed plots of native and exotic plant species (Table 1). C addition has been shown to reduce inorganic N levels (Wilson & Gerry 1995; Zink & Allen 1998; Reever Morghan & Seastedt 1999; Paschke et al. 2000; Torok et al. 2000; Cione et al. 2002; Blumenthal et al. 2003) and exotic plant biomass (Reever Morghan & Seastedt 1999; Alpert & Maron 2000; Blumenthal et al. 2003). However C addition has not been shown to consistently benefit native species. Zink & Allen (1998) and Paschke et al. (2000) demonstrated positive effects of mulch and sucrose addition, respectively, on growth rates and relative abundances of native species, while other studies reported no benefit of sawdust addition to native species (Wilson & Gerry 1995; Reever Morghan & Seastedt 1999; Alpert & Maron 2000; Cione et al. 2002). C addition has been most successfully used by Blumenthal et al. (2003), who

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**Table 1.** Amendments of labile carbon (C) to reduce plant-available nitrogen and increase competitiveness of slower growing species.

Study	C Source Quantity and Form	Study Duration (years)
This study	1.2 sawdust	2
Blumenthal et al. (2003)	Various quantities of sucrose and sawdust	2
Cione et al. (2002)	2.5 cm depth leaf and bark mulch	1
Alpert & Maron (2000)	1.5 sawdust	2
Paschke et al. (2000)	1.1 sucrose	3
Torok et al. (2000)	0.1 sucrose + 0.03 kg/m <sup>2</sup> sawdust	1
Reever Morghan & Seastedt (1999)	1.0 sucrose + 0.7 kg/m <sup>2</sup> sawdust	1
Hopkins (1998)	24 g C/m <sup>2</sup> as sucrose and sawdust	3
Zink & Allen (1998)	3 cm depth pine bark, oat straw	3
Wilson & Gerry (1995)	0.4 sawdust	1
McLendon & Redente (1992)	1.1 sucrose	3

Quantity is kilogram of sawdust or sucrose per square meter added through the entire experiment. Frequency and method of application varied among studies.

reported that exotic biomass decreased and native biomass increased when a mixture of sucrose and sawdust was added to a tallgrass prairie restoration site. Sawdust remains an enticing tool for native restoration because of its ease of application and its cost effectiveness relative to sucrose. But if it decreases the production of all vegetation (natives and exotics), as has been reported in a number of studies, then sawdust would be less useful as a restoration tool than it would be if native biomass or competitiveness increases.

We tested the ability of sawdust addition to benefit native perennial grass species in competition with two different types of exotic grasses—exotic annual and exotic perennial grasses—in an ecosystem that had undergone N enrichment. We hypothesized that sawdust addition would lessen the competitive impact of exotic species on native species during the establishment phase. We expected that the impact of sawdust addition would be the strongest on native perennial, exotic annual interactions, because exotic annuals are known to be particularly responsive to changes in nutrient availability (Huenneke et al. 1990; Maron & Connors 1996). We chose an experimental design that permitted the examination of the relative impacts of native and exotic grasses on each other with and without sawdust addition. This represents a more complete test of the mechanisms of the benefits to native species of sawdust addition than previous studies. We further hypothesized that the effect of sawdust addition on native species in the absence of exotic competitors would be negative, indicating that the benefits of C addition to natives are mediated by the reduction in competitiveness of exotic species rather than by a direct positive effect of sawdust addition on growth of natives.

## Methods

### Study Area

This study took place at Tom's Point, a private nature preserve adjacent to Tomales Bay in Marin County, CA, U.S.A. (38° 13' N, 122° 57' W). Tomales Bay experiences highly seasonal rainfall patterns typical of Mediterranean

climates. Annual rainfall is 790 mm per year, mainly falling between November and April. Peak growth for annuals is generally in late February through the end of March. For perennial species peak growth extends longer into the spring. Rainfall patterns vary among years, although typically the first rainfall events that stimulate germination occur in late October or November and the last events before the summer drought occur in April. Mean monthly temperature ranges from 13.9 °C in March to 18.1 °C in September. The soil at the study site was a Tomales Series sandy loam (fine, mixed, and mesic Ultic Paleustalf).

The site has not been grazed by livestock for at least 30 years. A portion of the reserve has been periodically dominated by *Lupinus arboreus* (bush lupine), a short-lived native N-fixing shrub that invades coastal grassland systems throughout central and northern California. At nearby Bodega Marine Reserve (BMR) *L. arboreus* has been shown to reduce native plant diversity through N enrichment of formerly native-dominated ecosystems (Maron & Connors 1996; Maron & Jefferies 1999). A portion of Tom's Point Preserve, where we established our experimental treatments, has become depleted of native species during repeated cycles of *L. arboreus* expansion and dieback. The high N content and low C:N ratio of the soils at our site (Table 2) is consistent with the pattern after *L. arboreus* colonization described by Maron and colleagues (Maron & Connors 1996; Maron & Jefferies 1999) at BMR. Few *L. arboreus* individuals were present at the beginning of the experiment due to widespread insect-driven dieback (Strong et al. 1995; Maron 1998),

**Table 2.** Mean (±SE) total nitrogen (N) and total carbon (C), and texture of the soil (top 10 cm) at the study site before the experiment.

Total soil N (mg N/g dry mass soil)*	4.1 (0.1)
Total soil C (mg C/g dry mass soil)*	49.5 (1.6)
C:N ratio	12.1 (0.05)
Sand (%)	83
Silt (%)	13
Clay (%)	5

\*Carlo-Erba CHN autoanalyzer (Fisons Instruments, Milan, Italy).

and the area was dominated by introduced annual grasses including *Avena barbata* (slender wild oat), *Bromus diandrus* (ripgut brome), and *Vulpia myuros* (zorro fescue), introduced perennial grasses including *Festuca arundinacea* (tall fescue), *Holcus lanatus* (velvet grass), and *Phalaris aquatica* (harding grass) and such exotic annual and biennial forbs as *Carduus pycnocephala* (Italian thistle) and *Conium maculatum* (poison hemlock).

### Experimental Design

Seeds of the introduced annual grasses *A. barbata*, *B. diandrus*, and *V. myuros*, the introduced perennial grasses *F. arundinacea*, *H. lanatus*, and *P. aquatica*, and the native perennial grasses *Agrostis idahoensis* (bent grass), *Festuca rubra* (red fescue), and *Nassella pulchra* (purple needlegrass) were collected by hand at Tom's Point Preserve in Spring 1998. Seeds of the introduced and native perennial grasses were planted in individual Conetainers (Stuewe and Sons, Corvallis, OR, U.S.A.) in September 1998 and allowed to germinate under greenhouse conditions.

In summer 1998, standing *C. pycnocephala* and *C. maculatum* individuals at Tom's Point Preserve were cut at ground level and removed. All remaining vegetation was sprayed with 5–10% glyphosate-based herbicide and removed, leaving bare soil. We established 70 1.5 × 1.5-m plots (with a 1–2 m buffer between plots) and assigned each plot to one of five treatments (Table 3). Because of the differences in size and longevity between exotic annual and exotic perennial grasses, it was not possible to equate the competitive environment experienced by native perennial grasses growing with each exotic group. Therefore we chose two different experimental designs to test the interactions between the native perennials and the exotics. We used an additive design in the case of the exotic annuals, whereby the density of individuals in plots containing both native perennials and exotic annuals was the sum of the number of individuals in single-group plots. In contrast, we used a replacement design in the case of the exotic perennials, whereby the density of individuals in each treatment remained constant (Table 3).

Annual seed densities applied in fall 1998 (Table 3) were chosen to fall within the range reported by Heady (1956) for Californian annual grasslands. Annual grass seeds were applied to appropriate plots in fall 1999 (second year) both

by allowing established plants from the previous growing season to set seed and by supplementing this natural seed rain with seed collected outside the experimental plots. Seeds were added at the same level as the previous season, with the exception that the number of *Vulpia* seeds was reduced to 7,750 seeds/plot. Perennial seedlings were transplanted from Conetainers between 5 and 14 January 1999 using a 2.5-mm soil corer to dig holes. Seedlings were planted in a 12 × 12 grid totaling 144 individuals per plot (Corbin & D'Antonio in press). Each plant was separated from its neighbors by 12 cm. The location of the species neighborhoods within each plot's grid was determined by randomly selecting grid location so that each species had the same number of individuals per plot. Plots were weeded of dicots and non-target grasses three times each year to maintain species composition and density.

We randomly selected six of the 14 plots to receive sawdust amendments, leaving eight plots that did not receive sawdust amendments. All plots were raked (top 2–4 cm) between 7 and 12 December 1998. We expected that frequent applications of sawdust would be more effective than a single large dose in influencing soil N cycling (Alpert & Maron 2000). Sawdust additions of 200 g/m<sup>2</sup> were repeated in February (immediately before peak vegetation growth) and April 1999. Additions in the 1999–2000 growing season were performed in November 1999 (1 week after the first significant rains of the season), March 2000, and April 2000 at similar rates as in the previous growing season. All sawdust additions after the December 1998 application were cast onto the soil surface, rather than raked, so as to minimize the disturbance of the vegetation and soil matrix. The sawdust in the first season was a hardwood mixture obtained from a local lumber supply center; the following season the sawdust was obtained at a furniture-making store that used only maple wood (*Acer* sp.). Sawdust from both sources was finely textured and was observed to become well mixed into the top 10 cm of the soil during the experiment, though this effect was not quantified.

### Plant Growth

Aboveground biomass of annual grasses was destructively sampled at peak biomass in Spring 1999 and 2000. All

**Table 3.** Treatments and initial planting densities (seeds or seedlings/2.25 m<sup>2</sup>).

Treatment	<i>Avena</i> Seeds	<i>Bromus</i> Seeds	<i>Vulpia</i> Seeds	Native Perennial Seedlings	Exotic Perennial Seedlings
Annual only	3,400	1,750	9,000	0	0
Native perennial	0	0	0	144	0
Exotic perennial	0	0	0	0	144
Annual and native perennial	3,400	1,750	9,000	144	0
Exotic perennial and native perennial	0	0	0	72	72

Exotic annual seed densities were chosen to fall within the range reported by Heady (1956) for California grasslands.

aboveground vegetation was clipped in three randomly selected  $0.25 \times 0.25$ -m subsamples in each plot and separated by species. After drying ( $60^\circ\text{C}$ ) to constant mass, each sample was weighed and returned to appropriate plots to decompose.

Aboveground biomass of perennial grasses was sampled twice each season using non-destructive methods. In the first season biomass was sampled 1 month after transplanting (March 1999) and in July 1999. Measurements were repeated in the fall (October 1999) and early summer (June 2000) of the second season to coincide with the minimum and maximum plant sizes, respectively. The basal diameter, height, and the number of flowering culms (where present) of 32 perennial grasses in each plot were sampled at each date. Allometric relationships between the three measures of plant size and aboveground biomass (clipped to  $<1$  cm height) were constructed for each species by harvesting 29–45 individuals of each species representing a range of plant sizes (*A. oregonensis*:  $n = 36$ ,  $r^2 = 0.92$ ; *F. arundinacea*:  $n = 32$ ,  $r^2 = 0.92$ ; *F. rubra*:  $n = 45$ ,  $r^2 = 0.95$ ; *H. lanatus*:  $n = 29$ ,  $r^2 = 0.68$ ; *N. pulchra*:  $n = 36$ ,  $r^2 = 0.85$ ; *P. aquatica*:  $n = 26$ ,  $r^2 = 0.85$ ) (Corbin & D'Antonio in press). Each species' relationship between plant size and aboveground biomass was found to vary over time, so separate allometric equations were used in each growing season. Growth each season was calculated as the difference between each plant's biomass in the early summer and its biomass in March or October 1999.

### Soil Analyses

Soil was analyzed for extractable ammonium and nitrate and net N mineralization in March, May, October, and November 1999 and in April 2000 to determine the impact of sawdust addition on soil mineral N. At each sampling period three 10 cm-deep  $\times$  2 cm-wide cores were collected from each plot, bulked, and sieved ( $<2$  mm). One subsample of soil (10 g) was collected from each sample and immediately extracted with 50 mL of 2.0 M KCl for the determination of extractable ammonium and nitrate. Another soil sample was incubated in capped polyethylene vials at field moisture for 7 days at  $25^\circ\text{C}$ . The incubated samples were extracted with KCl as above. An additional subsample was weighed and dried overnight at  $>100^\circ\text{C}$  for gravimetric water content (GWC) correction (calculated as [wet – dry]/dry). At the time of the November 1999 and April 2000 sampling, additional subsamples of soil (10 g) were analyzed for microbial biomass N using the chloroform-fumigation extraction method (Brookes et al. 1985). One subsample was immediately extracted with 40 mg of 0.5 M  $\text{K}_2\text{SO}_4$  for determination of initial extractable N. The other subsample was fumigated with chloroform for 5 days. The fumigated subsamples were then extracted with  $\text{K}_2\text{SO}_4$  as above. Organic N in the  $\text{K}_2\text{SO}_4$  extracts was converted to  $\text{NO}_3\text{-N}$  in a sulfuric-salicylic acid Kjeldahl digestion (Howarth & Paul 1994).

Ammonium and nitrate concentrations in all extracts were measured using a Lachat flow-injection autoanalyzer at UC Berkeley, then converted to microgram of  $\text{NO}_3$  and  $\text{NH}_4$ /g soil using GWC-corrected soil weights. Net mineralization of N was calculated as extractable nitrate + ammonium in the incubated sample minus extractable nitrate + ammonium in the initial extracts. Net nitrification was calculated as extractable nitrate in the incubated sample minus extractable nitrate in the initial extracts.

### Statistical Analyses

The effect of sawdust addition on extractable ammonium and nitrate concentrations, net N mineralization rates, and GWC was tested using repeated measures ANOVA (SAS Institute 2000). The model used included Block, Sawdust, and Time, and the interaction between Sawdust and Time. The effects of Time and Sawdust  $\times$  Time were analyzed with MANOVA using Roy's greatest root (Scheiner 1993). Where the analysis indicated different main effects in different time periods (i.e., significant Sawdust–Time interactions), ANOVA was run separately for each sample period. GWC was log transformed to meet the assumptions of ANOVA. The effects of the competition treatments and sawdust addition on the growth of each species were tested using ANOVA with a model including Block, Competition, Sawdust, and a Competition–Sawdust interaction. Native species responses to exotic annual and exotic perennial species were tested separately. Native biomass in June 2000 was also analyzed using ANOVA. The growth and biomass of each species were log transformed to meet the assumptions of ANOVA.

## Results

### Soil Responses to Sawdust Addition

Repeated measures ANOVA revealed significantly higher extractable ammonium levels in sawdust-amended plots ( $F_{[1,58]} = 6.39$ ;  $p < 0.02$ ) (Fig. 1). However, when the repeated measures ANOVA was performed excluding October 1999 (the only individual time in which extractable ammonium concentrations were significantly different between sawdust-amended and non-sawdust-amended plots (ANOVA: October 1999  $F_{[1,58]} = 14.32$ ,  $p < 0.001$ ; all other sample times,  $p > 0.1$ ), the effect of sawdust on ammonium concentrations was no longer significant. There was no effect of sawdust on extractable nitrate concentrations. Sawdust addition did decrease net N mineralization (repeated measures ANOVA:  $F_{[1,58]} = 6.30$ ,  $p < 0.02$ ) and net nitrification (repeated measures ANOVA:  $F_{[1,58]} = 8.65$ ,  $p < 0.01$ ), as we had predicted, though April 2000 was the only sample period in which either net N mineralization (ANOVA:  $F_{[1,58]} = 44.94$ ,  $p < 0.001$ ) or net nitrification (ANOVA:  $F_{[1,58]} = 74.20$ ,  $p < 0.001$ ) were significantly different among treatments (Fig. 1). As with the ammonium pool sizes, when this single date was excluded

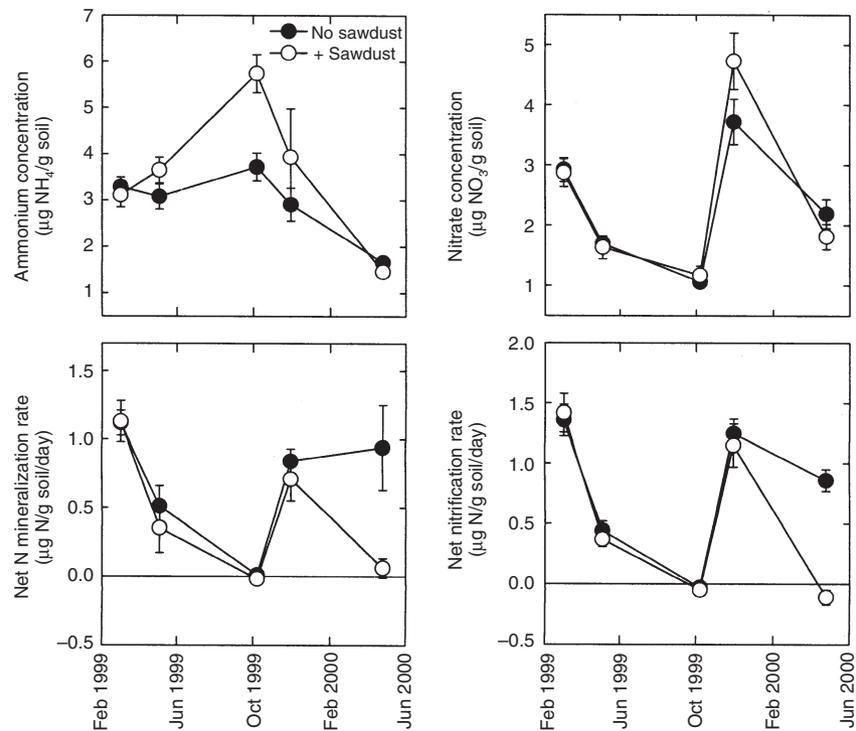


Figure 1. Extractable ammonium and nitrate concentrations, net nitrogen (N) mineralization rate, and net nitrification rate in unamended and sawdust-amended plots. Points represent mean  $\pm$  1 SE.

from the repeated measures ANOVA, the effect of sawdust on net N mineralization and net nitrification was not significant.

Microbial biomass N was significantly higher in sawdust-amended plots than non-sawdust plots in November 1999 ( $0.080 \pm 0.003$  versus  $0.031 \pm 0.003$   $\mu\text{g N/g soil}$  [mean  $\pm$  SE]) and April 2000 ( $0.095 \pm 0.005$  versus  $0.041 \pm 0.004$   $\mu\text{g N/g soil}$ ) (repeated measures ANOVA:  $F_{[1,58]} = 10.09$ ,  $p < 0.01$ ). GWC was not significantly different in sawdust-amended plots as compared with non-sawdust-amended plots (repeated measures ANOVA:  $F_{[1,58]} = 1.86$ ,  $p > 0.15$ ).

#### Plant Responses to Sawdust Addition

The aboveground growth in the first growing season of all the three native species was negatively affected by competition with exotic annual species (Table 4; Fig. 2a). The effect of sawdust on one of the native species, *Nassella pulchra*, in that year was dependent on the presence or absence of annual species. When grown in the absence of annual competitors, sawdust had a negative effect on *N. pulchra*. When annuals were present, *N. pulchra* growth was greater with sawdust addition than without (annuals-sawdust interaction) (Table 4). Growth of one other native species, *Festuca rubra*, showed a trend toward the same pattern (Table 4). Competition with annual species reduced growth of *Agrostis oregonensis* and *F. rubra* in the second growing season (Fig. 2b), but any advantage

that sawdust may have provided for species competing with exotic annual species was no longer detectable: ANOVA revealed no significant interactions between sawdust and competition with annual grasses (Table 4). Furthermore none of the native species' aboveground biomass at the end of the 2-year experiment was significantly different between sawdust-amended plots and plots that did not receive sawdust ( $p > 0.1$ ).

Competition with exotic perennials reduced the growth of all three native species in 1999, but there was no significant effect of exotic perennials on native growth in the second growing season (Table 4; Fig. 2). There was also no significant interaction between exotic perennial competition and sawdust for any of the native species in either season.

Among the exotic annual grasses, only *Vulpia myuros* was slightly affected by competition with native grasses in the first growing season, when its aboveground biomass was significantly lower with competition from native species (Fig. 3a). In contrast native species decreased annual grass production in the second season by over 50%, with all species showing significant declines (Fig. 3b). In neither season did sawdust addition influence the production of the three annual species or the total annual grass production.

The presence of native species did not reduce the growth of exotic perennial grasses in either year. In fact, *Festuca arundinacea* appeared to grow better with native

**Table 4.** ANOVA of effect of competition (with exotic annual and perennial grasses) and sawdust addition on native perennial grass growth in (A) 1998–1999 growing season and (B) 1999–2000 growing season.

Source	<i>Agrostis</i>		<i>Festuca rubra</i>		<i>Nassella</i>	
	$F_{[1,23]}$	$p$	$F_{[1,23]}$	$p$	$F_{[1,23]}$	$p$
(A) 1999						
Annuals	<b>19.24</b>	<b>0.0002</b>	<b>36.15</b>	<b>0.0001</b>	<b>18.19</b>	<b>0.0003</b>
Sawdust	0.40	0.5	0.72	0.4	1.31	0.3
Annuals × sawdust	0.11	0.7	3.10	0.092	<b>6.28</b>	<b>0.02</b>
Exotic perennials	<b>4.69</b>	<b>0.04</b>	<b>15.72</b>	<b>0.0006</b>	<b>3.90</b>	<b>0.06</b>
Sawdust	0.14	0.7	0.20	0.7	<b>3.69</b>	<b>0.07</b>
Exotic perennials × sawdust	0.83	0.4	0.03	0.9	1.87	0.2
(B) 2000						
Annuals	<b>22.64</b>	<b>0.0001</b>	<b>14.87</b>	<b>0.0008</b>	0.91	0.4
Sawdust	0.08	0.8	0.04	0.8	0.99	0.3
Annuals × sawdust	0.92	0.4	0.08	0.8	0.01	0.9
Exotic perennials	0.79	0.4	0.48	0.5	1.68	0.2
Sawdust	0.69	0.4	0.58	0.5	2.64	0.1
Exotic perennials × sawdust	0.01	0.9	0.71	0.4	0.17	0.7

Numbers in bold are significant.

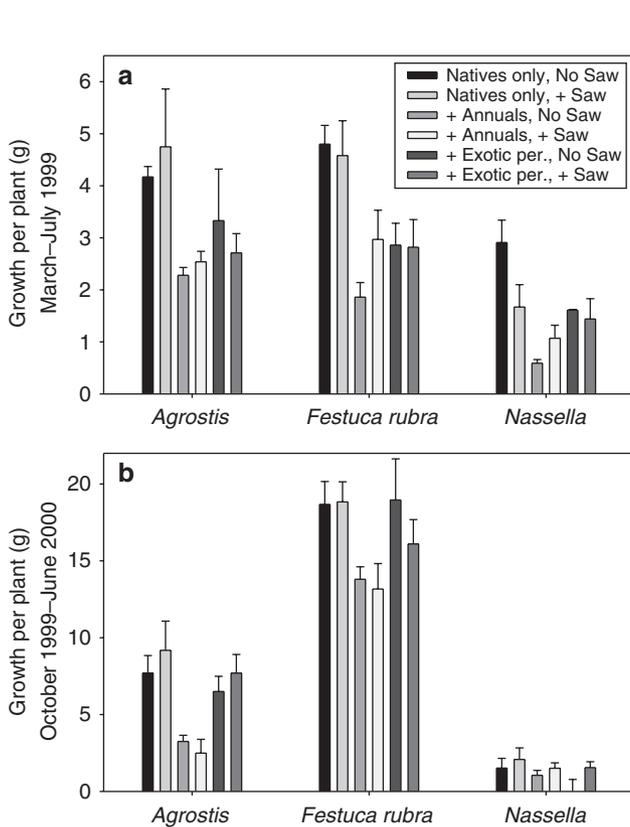


Figure 2. Growth of each native perennial grass in 1999 (a) and 2000 (b). Competition treatments: native perennials only, natives with exotic annual grasses, and natives with exotic perennial grasses. Each competition treatment was crossed with sawdust addition. Bars represent mean  $\pm$  1 SE (See Table 4 for ANOVA).

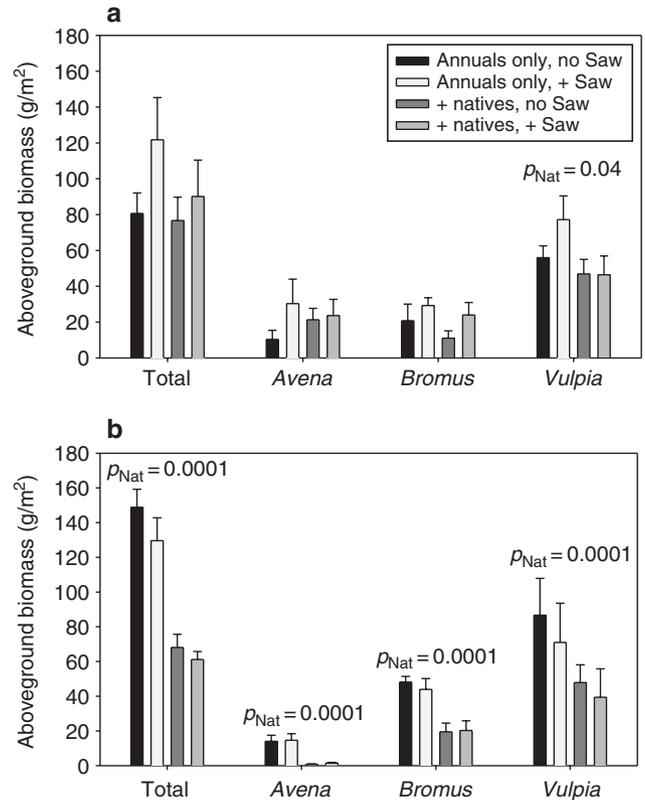


Figure 3. Biomass of each exotic annual species and total annual biomass in 1999 (a) and 2000 (b). Competition treatments: exotic annuals only and annuals with native perennial grasses. Bars represent mean  $\pm$  1 SE. Significant  $p$  values from ANOVA (model: biomass = block + native competition (Nat) + sawdust (Saw) + Nat  $\times$  Saw) for each species are indicated.

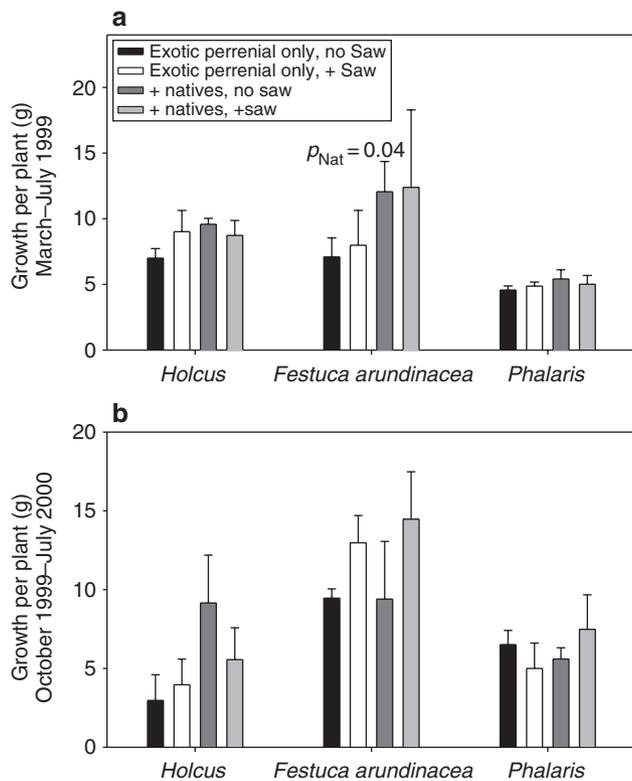


Figure 4. Growth of each exotic perennial grass in 1999 (a) and 2000 (b). Competition treatments: exotic perennials only and exotic perennials with native perennial grasses. Bars represent mean  $\pm$  1 SE. Significant  $p$  values from ANOVA (model: biomass = block + native competition (Nat) + sawdust (Saw) + Nat  $\times$  Saw) for each species are indicated.

perennial grasses than with other exotic perennial grasses in 1999 (Fig. 4). Growth of exotic perennials was not affected by sawdust addition in either growing season.

## Discussion

In the first year, when natives were grown with exotic annual competitors, growth of *N. pulchra* and, to a lesser extent, *F. rubra* was greater in plots that received sawdust addition than in plots that did not receive sawdust addition. These results were consistent with our prediction that sawdust addition would benefit native species, particularly those competing with exotic annual grasses. In the second year, however, we did not find evidence that sawdust addition increased the competitiveness of natives. Instead, native species substantially reduced annual grass production regardless of whether sawdust was added or not. Sawdust addition did not influence the growth of native species competing with exotic perennial grasses in either growing season. At the end of the experiment the above-ground biomass of native species receiving sawdust addition was not greater than the biomass of natives that did not receive sawdust. We conclude therefore that the addition of sawdust as a tool in the restoration of native species

in our system, where target individuals were planted as seedlings and survival was high in all treatments, provided no significant benefit to natives over a 2-year period.

By the second growing season, sawdust additions had stimulated microbial immobilization of N, consistent with the findings of Torok et al. (2000), and had reduced rates of net N mineralization. We did not find evidence that sawdust addition decreased N availability or N cycling rates in the first growing season. Addition of sucrose and/or sawdust has frequently been shown to reduce soil inorganic N levels in other systems (Zink & Allen 1998; Reeve Morghan & Seastedt 1999; Blumenthal et al. 2003), though the efficacy of sawdust and sucrose may be dependent on initial soil fertility, quantity of C added, and the form of C added (e.g., sucrose versus sawdust) (Blumenthal et al. 2003). It is possible that the N enrichment from dying lupines in our coastal prairie site (Table 2) was high enough that the rate of sawdust addition was not sufficient to induce N limitation until the second year.

We did not find support for the expected mechanism by which lower N would impact native vegetation, namely by reducing competitors' production and indirectly benefiting natives' growth. Aboveground production of exotic annual or exotic perennial grasses was not affected by sawdust in either season and hence it is difficult to explain the impact of sawdust on natives competing with annual species in the first growing season. The effect of sawdust on one of the natives, *N. pulchra*, growing without exotic competitors was negative, but otherwise there was no evidence that sawdust negatively affected native growth. The possibility remains that shifting allocation to above- versus below-ground structures may have masked significant differences in the productivity of exotic annuals or exotic perennial species between plots receiving sawdust and plots that did not. It is, however, unlikely that sawdust addition had a direct facilitative effect on native growth, because sawdust addition did not increase the growth of natives in treatments without competitors.

Sucrose and sawdust addition to other systems has produced more lasting benefits to native species than we found. Paschke et al. (2000) and McLendon & Redente (1992) found that sucrose addition to two successional shortgrass steppe ecosystems favored slower growing perennial species over faster growing annual grasses such as *Bromus tectorum* (cheatgrass) and annual forbs. However sucrose is expensive relative to other C sources such as sawdust and therefore is less practical as a restoration tool (Reeve Morghan & Seastedt 1999). Zink & Allen (1998) reported improved growth and survival of hand-planted *Artemisia californica* (California sagebrush) seedlings after 2 years of pine bark and oat straw addition to a California coastal sage scrub habitat. Sawdust addition (with or without sucrose addition) has been shown to negatively affect exotic competitors but has been less successful in benefiting native species (Wilson & Gerry 1995; Reeve Morghan & Seastedt 1999; Alpert & Maron 2000;

Cione et al. 2002). However the ability to increase native growth may be a function of the quantity of sawdust added relative to site fertility. Blumenthal et al. (2003) reported decreasing availability of NO<sub>3</sub> and weedy biomass and increasing biomass of native species as the quantity of sucrose and sawdust input increased. They suggested, based on their data and a review of previous studies, that a threshold level of C likely must be added before decreases in soil N or increased competitiveness of native species is detected (Blumenthal et al. 2003). We cannot discount the possibility that greater quantities of sawdust addition, or a longer duration of sawdust application, may have led to different results in our study. Further investigations should examine whether the quantity of sucrose and/or sawdust that is added influences the outcome in other systems, as well.

### Acknowledgments

J. Kelly and Audubon Canyon Ranch generously provided access to Tom's Point. This study grew out of discussions with K. Haubensak and other members of the D'Antonio laboratory group at UC Berkeley. The authors would also like to acknowledge the field assistance of K. Haubensak, J. Schue, S. Gulamhussein, T. Bouchier, M. Thomsen, and D. Corbin. We thank M. Mazzola, R. Blank, R. Anderson, E. Allen, and two anonymous reviewers for comments on an earlier draft. Funding was provided by grants from the Marin Community Foundation and the National Science Foundation (DEB 9910008).

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# Experimental Approaches to Conservation Biology

*Edited by*

MALCOLM S. GORDON AND SORAYA M. BARTOL

*University of California Press*

BERKELEY LOS ANGELES LONDON

## 11 Tipping the Balance in the Restoration of Native Plants

### *Experimental Approaches to Changing the Exotic:Native Ratio in California Grassland*

Jeffrey D. Corbin, Carla M. D'Antonio,  
and Susan J. Bainbridge

#### SUMMARY

As exotic species increasingly threaten native biodiversity, habitat managers have turned to a variety of tools designed to increase the efficiency of plant-restoration projects. These efforts include eliminating exotic competitors through mechanical removal, herbicide application, or fire, and increasing native species' competitiveness relative to that of exotic species through reduction of soil nitrogen availability, grazing, prescribed burning, or biological control. In this chapter, we evaluate the ability of experimental tests of these techniques to favor native species in California grassland ecosystems. We found no evidence that any of the strategies consistently favored native species relative to exotic species. Outcomes were highly case specific and likely varied with biotic and abiotic conditions in the experimental systems. Several studies suggest that these techniques are more successful in reducing specific invasive plant species in California grasslands rather than in increasing the success of native revegetation.

Limited availability of native propagules in the experimental systems likely limited the extent to which restoration techniques actually promoted native species. The most promising strategy for increasing native components in invaded ecosystems is likely to be the coordination of multiple strategies that address exotic-species abundance, native-seed or -seedling availability, and the postestablishment competitiveness of the native species. Such an application of an integrated "pest" management approach to the restoration of degraded habitats holds greater promise for the successful reestablishment of native biodiversity than simply targeting exotic species for removal.

## INTRODUCTION

Exotic species increasingly threaten native biodiversity in natural habitats worldwide. Habitat managers trying to restore native-species richness and abundance face the daunting challenge that exotic species frequently are superior competitors in sites where the natives and exotics co-occur. For example, invasive *Spartina* spp. (cordgrass) in western North American estuaries are capable of excluding such native salt-marsh species as *Spartina foliosa* (California cordgrass) and *Salicornia virginica* (pickleweed) (Daehler and Strong 1996). Reintroduction of native species into these invaded ecosystems is unlikely to succeed as long as the exotics are competitively superior.

Habitat managers and restoration ecologists must utilize a variety of tools to tip the competitive balance toward native species and away from exotic species while increasing the efficiency of plant-restoration projects. Current tools include eliminating exotic competitors through mechanical removal, herbicide application, or fire, and increasing native species' competitiveness relative to exotics' through reduction of soil nitrogen (N) availability, grazing, or prescribed burning or through the introduction of biological control agents. These tools are often applied repeatedly or in combination, but to provide a lasting increase in the native component of degraded ecosystems they frequently need to be used in conjunction with reintroduction of native-plant species either as seeds or seedlings. The application of these tools may also be constrained by practical considerations such as safety or toxicity concerns (e.g., fire or herbicide application) or limited habitat area (e.g., fire or grazing).

The need to enhance the success of native-plant restoration and to increase native species' competitiveness is particularly urgent in California grassland ecosystems. Grasslands are a major component of the state's natural vegetation, comprising nearly 10 million ha, or 25% of the state's surface area (Heady et al. 1991). The state's grasslands are used extensively for livestock production (Wagner 1989) and recreation and are habitat for many of California's state-listed threatened and endangered plants. During the last two centuries, invasion by European annual grasses and forbs into California grasslands, modifications of land use, and, possibly, changes in the region's climate have resulted in a dramatic, large-scale conversion from dominance by perennial bunchgrasses, forbs, or both to dominance by Eurasian annual species (Burcham 1970; Crampton 1974; Bartolome, Klukkert, and Barry 1986; Baker 1989; Hamilton 1997). Whereas perennial species such as *Nassella pulchra* (purple needlegrass), *Bromus carinatus* (California brome), *Elymus glaucus* (blue wildrye), *Danthonia californica* (California oatgrass),

*Poa secunda* (pine bluegrass), and *Festuca* spp. (fescue) were thought to dominate some of the region's grasslands prior to European settlement, introduced grasses such as *Bromus diandrus* (ripgut brome), *Bromus hordeaceus* (soft chess), *Avena* spp. (wild oat), and *Vulpia* spp. (annual fescue) are dominant today, even in stands where some native bunchgrasses have persisted.

Life-history and growth characteristics of exotic species offer substantial advantages over those of native perennial species in disturbed habitats that are frequently the targets of restoration efforts. Seedbank composition in California grasslands is highly skewed toward exotic annual species (Champness and Morris 1948; Major and Pyott 1966; Dyer, Fossum, and Menke 1996; Holl et al. 2000; Alexander 2001). Seed production by annual species substantially exceeds the number of seeds necessary to replace the population (Young and Evans 1989), whereas the establishment of perennial species has frequently been shown to be limited by seed availability (e.g., Peart 1989a; Kotanen 1996; Hamilton, Holzapfel, and Mahall 1999). Furthermore, annual seeds in growth chambers have been shown to germinate earlier and under a wider range of temperatures than native perennial seeds (Reynolds, Corbin, and D'Antonio 2001). The more abundant and earlier-germinating annual grass species can form dense stands and monopolize resources, thereby restricting the growth and survival of native seedlings (Bartolome and Gemmill 1981; Dyer, Fossum, and Menke 1996; Dyer and Rice 1997; Hamilton, Holzapfel, and Mahall 1999; Brown and Rice 2000). As a result, competitive interactions between native and exotic grasses in California have usually been shown to strongly favor the exotic species, especially in recently established native populations (Dyer, Fossum, and Menke 1996; Dyer and Rice 1997; Hamilton, Holzapfel, and Mahall 1999; Brown and Rice 2000).

The large competitive advantages that some exotic species enjoy over natives suggest that efforts to restore native-plant biodiversity in exotic-dominated grasslands in California must improve the competitiveness of native species relative to that of exotic species. In this chapter, we review the successes and failures of techniques either being proposed or employed over a large scale to improve conditions for native species during grassland restoration. These techniques include the reduction of N in N-enriched habitats through sawdust addition or repeated biomass removal, grazing, prescribed burning, herbicide application, and biological control. We focus on efforts to alter composition in sites that are currently grassland rather than on the *de novo* creation of grassland from recently plowed or otherwise heavily disturbed sites.

## REDUCTION OF PLANT-AVAILABLE NITROGEN

Ecosystem nitrogen enrichment is a common barrier to native-plant restoration. Past fertilization (Vitousek et al. 1997), atmospheric nitrogen deposition (Bobbink 1991; Jefferies and Maron 1997), fire (Wan, Hui, and Luo 2001), habitat disturbance (Hobbs and Mooney 1985), and invasion by nitrogen-fixing shrubs (Vitousek et al. 1987; Maron and Connors 1996) can all increase soil nitrogen availability. While general characteristics of non-native invading plant species have proven elusive (Mack et al. 2000), enhancement of N availability has been shown to favor fast-growing invasive species in a variety of habitats (e.g., Huenneke et al. 1990; Vinton and Burke 1995; Maron and Connors 1996). Restoration in N-enriched habitats must, therefore, deal with the question of how to promote slower-growing native species in competition with faster-growing exotic species.

Invasion of northern coastal prairie grasslands in California by a variety of N-fixing shrubs commonly known as brooms—for example, *Genista monspessulana* (French broom) and *Cytisus scoparius* (Scotch broom)—and *Ulex europaeus* (gorse) has been shown to have significant impacts on soil N availability and plant community composition (Randall, Rejmánek, and Hunter 1998; Haubensak 2001). Haubensak (2001) found that N availability was three times as high in a broom-invaded grassland than in an adjacent uninvaded grassland. The colonization of coastal prairie grasslands by the native shrub *Lupinus arboreus* (bush lupine) has had similar effects on N cycling and community composition as broom invasion (Maron and Jefferies 1999). Individual shrubs grow rapidly, producing a dense canopy that shades out native grassland species. In northern California coastal prairies, repeated cycles of lupine colonization and death lead to a doubling of total soil N, greatly increased N availability, and thus increased vegetative production (Maron and Jefferies 1999). Maron and Connors (1996) documented that these cycles cause a large-scale shift in grassland composition from native perennial to exotic annual species. Increased N levels even after broom removal or lupine dieback may continue to favor exotic species and hamper efforts to reintroduce native species.

Two promising methods to reduce plant-available N and increase the competitiveness of slower-growing natives in such N-enriched habitats are (1) the addition of a labile carbon source such as sucrose or sawdust (e.g., Morgan 1994; Alpert and Maron 2000; Paschke, McLendon, and Redente 2000) and (2) repeated mowing followed by biomass removal (Collins et al. 1998; Maron and Jefferies 2001). The addition of a carbon source is assumed

to increase microbial N immobilization and decrease plant-available N (Morgan 1994; Alpert and Maron 2000; Paschke, McLendon, and Redente 2000). Repeated mowing and biomass removal are assumed to remove N in plant biomass that would otherwise be remineralized as plant litter is produced and decomposes. Under lower N conditions, growth of all vegetation would be expected to decrease, but if faster-growing exotic species are disproportionately affected by lower soil N concentrations, slower-growing native species may benefit indirectly owing to reduced competition.

#### *Carbon Addition*

Carbon addition has successfully reduced the abundance of exotic species in California grassland (Alpert and Maron 2000), shrubland (Zink and Allen 1998), and sagebrush–bunchgrass (Young et al. 1998) communities, as well as in shortgrass steppe ecosystems in Colorado (Reever Morghan and Seastedt 1999; Paschke, McLendon, and Redente 2000). Alpert and Maron (2000) tilled 1.5 kg m<sup>-2</sup> of sawdust into bare N-rich patches left after the death of bush lupine individuals in a coastal prairie site. The patches were not experimentally seeded with natives, and recruitment into them depended on seed rain or the seedbank. Native biomass comprised only 8–12% of the total biomass in all treatments, reflecting the highly invaded nature of these ecosystems. Sawdust addition significantly reduced the aboveground biomass of exotic grasses, although the biomass of exotic forbs was unaffected. Sawdust addition also showed no significant benefit for native-species richness or biomass. Thus, while sawdust addition successfully reduced exotic grass abundance, there was no evidence that it increased the occurrence of native species.

Two studies in our lab employed similar experimental approaches to assess the ability of sawdust addition to benefit native species in ecosystems invaded by two different types of N-fixing shrubs (Corbin and D'Antonio 2004a; Haubensak 2001). Following removal of shrubs and understory vegetation in a broom-invaded coastal scrub ecosystem and another lupine-invaded coastal prairie, seedlings of three species of native perennial grasses were transplanted into experimental plots, half of which were seeded with exotic annual grasses. In the postlupine site, we added a third treatment consisting of three species of exotic perennial grass competitors. Sawdust was added to half the plots (600 g m<sup>-2</sup> yr<sup>-1</sup>) for 2 years.

Native species did not benefit from the addition of sawdust following removal of French and Scotch broom in either growing season (Haubensak 2001). Instead, exotic annual grasses significantly reduced the growth and survival of all three native species, whether sawdust had been added or not.

Apparently, the effect of competition with annual species was so strong that it overwhelmed any potential effect of sawdust addition.

Sawdust addition showed greater promise in reducing the competitive advantage of exotic annual grasses in the lupine-invaded coastal prairie (Corbin and D'Antonio 2004a). In the first growing season, sawdust decreased the competitive suppression of seedlings of two native grass species by exotic annual grasses, but there was no benefit for native species competing with exotic perennial grasses. In the second year, sawdust addition did not affect the competitive interactions between natives and either exotic annual or exotic perennial grasses. In fact, the native perennial grasses that survived the first year of competition with annual grasses significantly reduced the aboveground productivity of annual grasses, even without sawdust addition. We concluded that sawdust addition provided no significant benefit to native plants in this system, where target individuals were planted as seedlings, and survival was high in all treatments. Competition between native species and exotic annual grasses was most asymmetric (in favor of the exotics) in the first growing season, after which native species were capable of significantly reducing the productivity of annual grasses (Corbin and D'Antonio 2004b). The possibility remains that sawdust addition may provide greater benefit to restoration projects in which seedling survival in the first year is less certain or in which native species are introduced as seeds.

#### *Mowing and Biomass Removal*

Maron and Jefferies (2001) examined the effectiveness of mowing and removing aboveground biomass in reducing soil N and favoring native species in a coastal prairie grassland that had experienced lupine invasion and dieback. The mowing and removal of plant biomass for five growing seasons reduced exotic grass biomass and doubled the number of forb species present as compared to unmanipulated control plots. However, mowing had no effect on the number of perennial grass species, most likely because of a lack of native propagules. The 5-year experiment removed approximately 9% of the total soil N as plant biomass but was not sufficient to induce N limitation of vegetation. In fact, unmowed plots experienced a significant reduction in soil N in the form of nitrate leaching losses in the fall and early winter, a reduction that was nearly equivalent to the biomass removal in mowed plots. The authors concluded that while mowing was effective in reducing exotic biomass and increasing the species richness of forbs (although many were exotic), mowing was unable to reduce soil N levels enough to favor the reestablishment of native grasses. The study also

suggested that reduction of soil N content in ecosystems that have become suitably enriched may require long-term treatment, owing to the slow turnover of soil organic N pools.

#### *Livestock Grazing*

Over the past decade there has been increased interest in the use of livestock grazing to reduce the biomass of introduced species and increase the diversity and abundance of native species in California grassland settings (e.g., Menke 1982; Edwards 1995, 1996; Reeves and Morris 2000). The California Cattlemen's Association, for example, suggested that carefully controlling the timing and intensity of livestock grazing can promote native diversity in California grasslands (Reeves and Morris 2000). Grazing may benefit native vegetation by disproportionately targeting exotic biomass, thereby reducing the exotics' competitive advantages; by reducing exotic seed production; or both. By contrast, some conservationists believe that livestock grazing has contributed to the degradation of many California grasslands and that its persistence is inimical to restoration of native-species richness (Fleischner 1994; Painter 1995).

D'Antonio et al. (2001) reviewed livestock-grazing studies from throughout California in an attempt to quantitatively evaluate the use of grazing as a tool to reduce exotic-species cover and promote native biological diversity (see Table 11.1 for a complete listing). Their initial goal was to conduct a meta-analysis of the size and direction of grazing's effects on native and exotic plants using all the available published and unpublished data sets from California. Meta-analysis is a statistical way of synthesizing results from different studies on a common topic (Gurevitch and Hedges 1993). They calculated an "effect size" in each study for each response variable (e.g., native forb cover) based on the ratio of the variable in the treated area (grazed) compared to the control (ungrazed). They assessed the effect of grazing on the measured response variables across studies using the mean of the pooled effect sizes. They found that most studies lacked adequate controls, lacked replication, or had no available measurement of among-plot variability and hence were not useable for meta-analysis. A summary of the six studies that fit the meta-analysis criteria demonstrated that livestock grazing was associated with an increase in the cover of native perennial grasses for those sites (Figure 11.1). Contrary to the claims of others (Thomsen et al. 1993; Kephart 2001), these studies showed a slight negative effect of livestock grazing on native-forb abundance and a positive effect of grazing on the abundance of exotic forbs. However, the results should be interpreted with caution because this small number of studies is inadequate for a true meta-analysis, and the

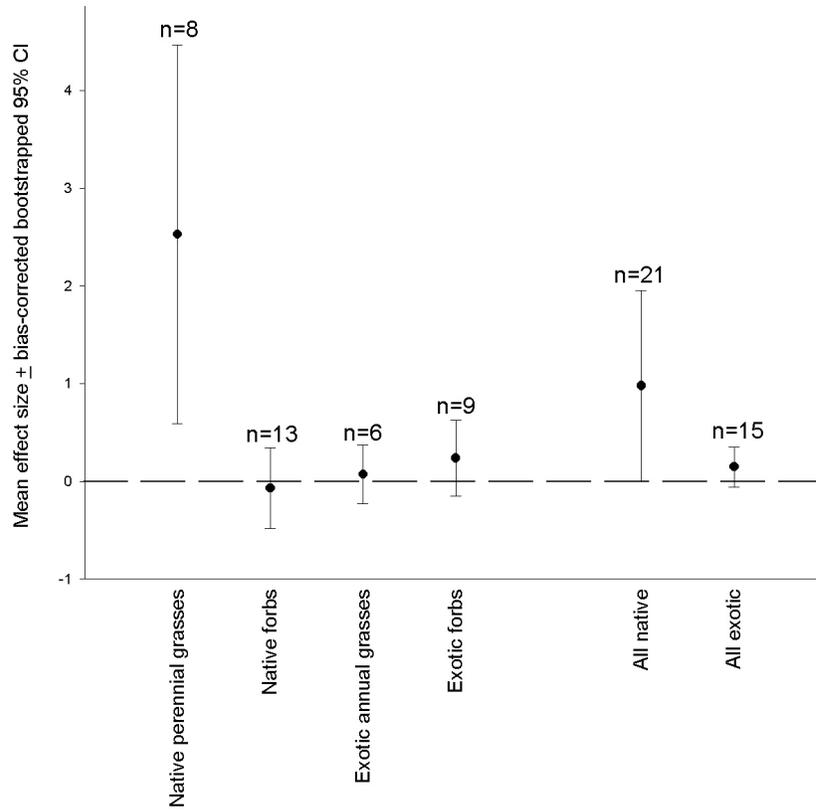


Figure 11.1. Effect of grazing on grassland plant life-form groups (based on studies reviewed by D'Antonio et al. [2001]). Values are the cumulative effect sizes (mean natural log of the response ratio  $[X_{\text{grazed}} / X_{\text{control}}]$  weighted by study variances  $\pm$  95% C.I.; n = number of effect sizes). Number of effect sizes may be greater than the number of published studies, owing to multiple comparisons within the same study.

addition of just a few studies could reverse the overall outcome. In addition, these studies represent a small subset of the California grassland and may not be representative of the state as a whole. Sadly, the often heated debates over the impact of livestock grazing and the role, if any, that grazing should play in grassland restoration are likely to continue until many additional careful quantitative studies are conducted across the full range of grassland habitats in the state.

Many grazing studies in California focus on the responses of particular native species, such as the native perennial bunchgrass *Nassella pulchra*, to

TABLE 11.1 *Studies of the impact of fire, grazing, or both on the species composition of California grasslands*

<i>Reference</i>	<i>Study Type</i>	<i>Grassland Type</i>
Ahmed 1983	Fire	Valley and foothill
Arguello 1994	Fire	Bald hills
Bartolome, Stroud, and Heady 1980; Jackson and Bartolome 2002	Grazing	Annual, valley, and foothill
Bartolome et al. in press	Fire and grazing	Valley and foothill
Bartolome and Gemmill 1981	Grazing	Various
Bett 2003	Fire	Valley and foothill
Cooper 1960	Grazing	Coastal prairie
Cox and Austin 1990	Fire	Vernal pool
Delmas 1999	Fire	Wildflower field
DiTomaso, Kyser, and Hastings 1999	Fire	Valley and foothill
Dyer and Rice 1997	Fire and grazing	Vernal pool
Dyer, Fossum, and Menke 1996; Fossum 1990	Fire	Vernal pool
Eller 1994	Grazing	Annual
Elliot and Wehausen 1974	Grazing	Coastal prairie
Foin and Hektner 1986	Grazing	Coastal prairie
Garcia-Crespo 1983	Fire	Valley and foothill
Graham 1956	Fire	Annual grassland–savannah
Hansen 1986	Fire	Alkali grassland–vernal pool
Harrison 1999	Grazing	Serpentine-forb
Hatch, Bartolome, and Hillyard 1991	Fire and grazing	Valley and foothill

the cessation of grazing or to particular grazing regimes. This species is, arguably, the most commonly used species in grassland restoration projects and the best studied of the native grasses. Yet a review of the relevant literature readily demonstrates the difficulties inherent to generalizing about the effects of livestock grazing. Huntsinger et al. (1996) and Dennis (1989) found substantial variation in the response of *N. pulchra* individuals to simulated grazing (clipping) among different populations. Genetic differences among the populations may explain the differential population response, although this variable was not explicitly examined. Likewise, enclosure

TABLE 11.1 (continued)

Reference	Study Type	Grassland Type
Hatch et al. 1999	Fire and grazing	Coastal prairie
Heady 1956	Grazing	Valley and foothill
Hektner and Foin 1977	Grazing	Coastal prairie
Keeley, Lubin, and Fotheringham 2003	Grazing	Oak woodland
Kephart 2001	Fire and grazing	Valley and foothill
Kneitel 1997	Fire	Valley and foothill
Langstroth 1991	Fire and grazing	Vernal pool
Larson and Duncan 1982	Fire	Annual
Marty 2001, 2002	Fire and grazing	Vernal pool
Merenlender et al. 2001	Grazing	Valley and foothill
Meyer and Schiffman 1999	Fire	Annual
Micallef 1998	Grazing	Annual, valley, and foothill
Parsons and Stollgren 1989	Fire	Annual
Pollack and Kan 1998	Fire	Vernal pool
Porter and Redak 1996	Fire	Valley and foothill
Reeves and Morris 2000	Grazing	Various
Saenz and Sawyer 1986	Grazing	Bald hills, woodland
Stromberg and Griffin 1996	Grazing	Valley and foothill
Thomsen et al. 1993	Grazing	Annual
TNC 2000	Fire and grazing	Wildflower field
White 1967	Grazing	Valley and foothill
York 1997	Fire	Annual
Zavon 1977	Fire and grazing	Annual

studies from several parts of central and northern California have demonstrated increases (Hatch et al. 1991), decreases (Hatch et al. 1999), and no change (White 1967; Stromberg and Griffin 1996) in the abundance of *N. pulchra* in response to protection from grazing. Some studies have observed fluctuations of *Nassella* abundance at the same site, but because of a lack of appropriate controls, the changes could not clearly be attributed to cessation of grazing (Bartolome and Gemmill 1981; Merenlender et al. 2001).

Several investigators have attempted to use livestock grazing to control particular exotic species, such as *Centaurea solstitialis* (yellow starthistle),

in California grasslands. For example, Thomsen et al. (1993) found that the timing of grazing was critical to the outcome of their experimental grazing treatments: late-spring and early-summer grazing greatly reduced yellow starthistle abundance relative to controls, although grazing did not eliminate the starthistle populations. At the same time, though native-plant-species richness was not recorded, the investigators observed an increase in populations of three disturbance-responsive native forbs with this late-spring grazing. Such targeted grazing may be useful in grassland restoration projects if the goal is simply to reduce a noxious weed.

Careful timing of grazing to coincide with the period of exotic seed production has the potential to benefit native species by reducing exotic germination and productivity in subsequent growing seasons. However, few studies have quantified the effect of grazing on the seed production of exotic species. Stromberg and Kephart (1996) argued that mowing or grazing for 2–3 years following native-plant restoration is likely to reduce exotic annual biomass and exotic seed production. Maron and Jefferies (2001) found that mowing reduced annual propagules and induced seed limitation of one of the most abundant exotic grass species, *Bromus diandrus*. We do not know, however, whether livestock grazing is capable of producing the same effect.

Overall, the existing data are insufficient to conclusively discern a relationship between livestock grazing and California's native grassland plants, or to evaluate the potential of grazing to enhance native-species richness and cover. Grazing has been shown to benefit native species in some individual studies, but its effects do not appear to be generalizable among studies or among years. Studies such as those of Stromberg and Griffin (1996) and Safford and Harrison (2001) suggest that grazing does not have as strong an effect on native species as has previously been suspected, but more research is needed to explore the generality of such conclusions. Many managers are now using controlled burning in combination with grazing to reduce exotic species and promote natives, and grazing may have a more predictable impact in combination with other techniques such as fire. Specific grazing regimes, with modest levels of grazing carefully timed to coincide with critical periods of exotic vegetation growth and seed production, have the greatest potential to be useful in a restoration context. Further research that employs both extensive quantitative surveying of properties with different grazing histories over a range of carefully recorded environmental conditions plus properly designed experiments is necessary to clarify the utility of grazing in increasing native competitiveness.

### *Prescribed Fire*

Land managers are increasingly turning to prescribed fire in an attempt to reduce the dominance of exotic plant species such as N-fixing shrubs (e.g., brooms and gorse), herbaceous forbs (e.g., yellow starthistle and *Taeniatherum caput-medusae* [medusahead]), and exotic grasses (Pollack and Kan 1998; DiTomaso, Kyser, and Hastings 1999; Bossard, Randall, and Hoshovsky 2000; Alexander 2001). Fire has the potential to instantaneously reduce exotic vegetation biomass, including standing biomass and residual litter, and can be applied to a relatively large landscape. Fire can also influence the seed crop and germination of native and nonnative species in subsequent growing seasons. Although fire can directly kill seeds on adult plants or fire-intolerant seeds in the soil, transient increases in light availability, soil surface temperatures, and soil nitrogen availability frequently associated with fire can also increase seed germination and seedling survival of fire-tolerant species. Frequent fires, however, may reduce available soil N and grassland productivity because they cause repeated volatilization of N and increased root death (Seastedt, Briggs, and Gibson 1991).

Efforts to generalize as to the impacts of fire on native-plant abundance and species diversity have proven difficult, in part because of the differential responses of various life-form groups to fire. Whereas some studies have shown dramatic increases in native-forb abundance in the first year following fire (Pollack and Kan 1998; Meyer and Schiffman 1999; DiTomaso, Kyser, and Hastings 1999), other studies have demonstrated minimal or negative effects of fire on native perennial grasses (Dyer, Fossum, and Menke 1996; Hatch et al. 1999). The effects of fire on the grassland vegetation also vary with time because the reductions in annual productivity that are frequently observed following fire tend to be temporary (Keeley 1981).

D'Antonio et al. (2001) reviewed the role of fire in structuring California grassland vegetation and the abundances of four life-form groups: native perennial grasses, native forbs, exotic annual grasses, and exotic forbs (see Table 11.1 for a complete listing). They conducted a meta-analysis on nineteen studies of prescribed or natural fires in California grassland. The investigators did not augment native propagules or seed availability, so the effect of seed limitation on the grassland response was not a controlled factor. They found that fire tended to shift grassland composition toward native forb species in the first year, but native perennial grasses were generally negatively affected by fire. The abundance of exotic species was, for the most part, unaffected by fire in the first growing season, apparently because the composition of exotic vegetation shifted from annual grasses, which

decreased, to annual forbs, which increased, after fire. In subsequent seasons following a single burn, total plant biomass increased to an average of 13% more in burned areas than in unburned areas. Where native perennial grasses were studied, *Nassella pulchra* abundance generally rebounded during the second postfire year, whereas *Danthonia californica* was slower to recover. Although germination of native grasses increased following fire, there was no detectable increase in native-grass abundance in subsequent years. Similarly, Dyer, Fossum, and Menke (1996) found that the establishment of native perennial grass seedlings was about the same in burned and unburned areas, whereas seedling mortality in burned areas was high. By the third year, the cover of native species relative to that of exotic species was not significantly different in burned areas and unburned areas, probably because of the rapid recovery of exotic annual grasses.

The observation that repeated burning reduces soil N availability suggests that prescribed burning could be used to tip the balance of competition in favor of native species if nitrogen is an important resource for both native and nonnative grassland species. For example, Seastedt, Briggs, and Gibson (1991) found that productivity of frequently burned tallgrass prairie grasslands was limited by N availability. The influence of fire frequency on soil N levels and N limitation of productivity in California grasslands is not known, though 2 or 3 consecutive years of burning have been shown to either decrease (Parsons and Stohlgren 1989; Delmas 1999) or have no significant effect on (Hansen 1986; DiTomaso, Kyser, and Hastings 1999) aboveground productivity beyond the effect of a single burn. D'Antonio et al. (2001) concluded that annual burning (after two or three burns) in ungrazed grassland resulted in higher native-forb and exotic-forb abundance than a single burn, but exotic annual grasses apparently did not respond further to the additional fires. Unfortunately, data were insufficient to conduct a meta-analysis on the effects of repeated burning on native perennial grasses. A single study of the effects of repeated fire reported a dramatic increase in native perennial grasses, particularly *Nassella pulchra* (DiTomaso, Kyser, and Hastings 1999), but more work is needed at other sites to evaluate the generality of this finding. Further investigation of the importance of fire frequency to soil N levels, N limitation, grassland productivity, and native species' competitiveness in California grasslands would help to determine whether repeated burning provides benefits to native biodiversity.

The meta-analysis by D'Antonio et al. (2001) determined that a combination of a single burn and cattle grazing likely did not improve the magnitude of the benefits of a single burn for native forbs, but grazing did sustain the benefits of a single fire for native forbs into the third postfire growing

season. Likewise, grazing sustained the decrease in exotic annual grasses observed in the first year after a single fire into the third year. Grazing also lessened the negative impact of fire on native perennial grasses in the first growing season. Exotic forb suppression was successful only when a site was burned annually for several consecutive years and also grazed. However, this suppression did not appear to benefit native forbs: there was no increase in the abundance of native forbs by the third year in repeatedly burned and grazed sites. So, as with other techniques described in this chapter, combining fire and grazing reduced exotic species but did not increase the diversity and abundance of native forbs.

The time of year in which controlled burns are performed may have a significant effect on the impact on grassland species composition. D'Antonio et al. (2001) found that the month in which grasslands were burned significantly influenced native perennial grasses, with growing-season burns (e.g., November–June) having significantly more detrimental impacts on native cover than summer or fall burns. Burn season did not have a strong effect on native forbs or exotic annual grasses. Burns can also be specifically timed to limit invasive species' seed dispersal. Fires targeting medusahead and yellow starthistle before mature plants dispersed their seeds effectively suppressed these species and their soil seedbanks (Pollack and Kan 1998; DiTomaso, Kyser, and Hastings 1999).

In addition to trying to manipulate the abundance of native and exotic grasses and forbs using fire, many managers use fire to control woody invaders in California grasslands. Alexander (2001) surveyed species composition and broom seedbank density and aboveground cover in numerous managed grassland sites in northern California to determine whether controlled burning was capable of controlling invasive shrubs (primarily *Genista monspessulana* and *Cytisus scoparius*) and opening suitable habitat for native species. She found that although fire reduced aboveground biomass of adult broom plants, it stimulated germination of broom seeds from the soil seedbank, which resulted in very dense broom stands in the first few years. The germination occurring after the first fire significantly reduced the number of live broom seeds in the soil seedbank, but because of the resulting increase in number of new broom plants, the seedbank has a great potential to become large again if the new crop of seedlings is not controlled. Hence, later fires must occur before the new crop of broom seedlings becomes reproductive (within 3–4 years). Alexander (2001) also found that the postbroom grasslands created by controlled burning were dominated largely by non-indigenous grasses and forbs. The only places where the postbroom grassland had substantial native cover were a few sites where broom

had been pulled repeatedly by hand and no burning had been conducted. In these few sites, native perennial grasses and forbs codominated with exotic annual grasses. Overall, Alexander's study suggests that repeated fire effectively reduces the broom seedbank and the cover of adult plants. However, the study points out that the reduction of broom dominance through prescribed burning alone is unlikely to increase native-species richness and cover in landscapes in which exotic forbs and grasses are so abundant.

#### *Herbicide Application*

Herbicide application has been suggested as a way to reduce established exotic vegetation in heavily invaded ecosystems and to control the flush of exotic annual species from the soil seedbank prior to planting of native species (e.g., Wilson and Gerry 1995; Stromberg and Kephart 1996; Rice and Toney 1998). Herbicides such as glyphosate, picloram, and clopyralid have been shown to substantially reduce exotic biomass and increase native-seedling establishment in a variety of grassland systems (e.g., Wilson and Gerry 1995; Rice et al. 1997; Rice and Toney 1998). Stromberg and Kephart (1996) recommended repeated herbicide application to reduce the exotic annual seedbanks prior to native-plant establishment in coastal California old fields. Our own experience in a northern coastal prairie grassland dominated by a mixture of exotic annual and perennial grasses and biennial forbs supports the utility of herbicide application in favoring native-grass establishment (Corbin and D'Antonio 2004a). Though we are not aware of controlled experiments comparing the efficacy of herbicides in restoration of California grasslands, herbicide application is likely an effective tool to facilitate site preparation and reduce exotic reestablishment (Anderson and Anderson 1996; Stromberg and Kephart 1996).

#### *Biological Control*

The introduction of biological control agents holds great promise in reducing the competitiveness of invasive plants in cases where insects, pathogens, or vertebrates specifically target undesirable species (Hoddle, this volume). However, in spite of the advantages of biological control—which include relatively low costs and long-term, self-sustaining management of invasive species—cases of successful control of nonnative grassland species in California are rare. DeLoach (1991) found that of twenty-three native and exotic weed species in western rangelands (including the northwestern United States and western Canada) where biological control had been attempted, only seven were successfully controlled. In California several insects have been introduced to control seed production of the pernicious yellow star-

thistle, and although the insects have been established successfully in many areas, starthistle remains a widespread, abundant weed (Turner, Johnson, and McCaffrey 1995; Villegas 1998; Roché et al. 2001). The introduction of three biological control agents in coastal Oregon grasslands to control *Senecio jacobaea* (tansy ragwort) has been far more successful, reducing the exotic to 1% of its former abundance (McEvoy, Cox, and Coomes 1991; McEvoy and Coomes 1999). One of the few investigations of biological control of exotic grasses (Carsten et al. 2000) found evidence that augmentation of a natural crown rust of *Avena* spp. (wild oat) on San Clemente Island reduced seed production of these annual grasses. In the cases of both yellow starthistle and wild oats, the biological control agents have been more effective at reducing seed production than at reducing exotic population sizes, but the agents could reduce exotic competitiveness in combination with other control techniques (DiTomaso 2000).

#### DISCUSSION

A review of attempts to increase the efficiency of California grassland restoration did not yield a strategy that consistently favored native species over exotic species. Although some individual techniques showed promise for increasing native-plant growth or seedling survival, no technique consistently increased native-grass or native-forb diversity or biomass. Rather, the outcomes were highly case specific and likely varied with such factors as initial vegetation composition, nutrient availability, past land-use history, and climatic conditions. Further exploration of the restoration tools under a wider range of habitat conditions is required before habitat managers can predictably apply them to revegetate degraded ecosystems.

Some techniques showed promise in reducing the exotic components of degraded ecosystems even though they were unable to increase the native component. Reduction of plant-available N (Alpert and Maron 2000; Maron and Jefferies 2001), mowing or grazing (Thomsen et al. 1993; Stromberg and Kephart 1996), prescribed fire (Pollack and Kan 1998; DiTomaso et al. 1999; Alexander 2001), and herbicide application (Stromberg and Kephart 1996) were able to reduce specific invasive plant species in California grasslands. While these techniques would be of even greater use in a restoration context if they were capable of consistently increasing the competitiveness or abundance of native species, the control of exotic species is frequently a primary goal of habitat management (Ehrenfeld 2000).

The absence of native plant species, either as individuals or as seeds from nearby populations, frequently limits the success of efforts to restore de-

graded habitats. Many natural areas that are chosen for restoration are chosen precisely because their native component has been lost. For example, invasion by exotic species can be a major factor in the elimination of native-plant populations in natural habitats (Baker 1989; Bossard, Randall, and Hoshovsky 2000). Agricultural activities, especially plowing and other intense farming activities, are also capable of directly removing native individuals and likely exhausting the seedbank of the former dominants (Milberg 1992). The effects of agricultural activities on native-grass abundances have been shown to persist for decades after cessation of the agricultural activity (Stromberg and Griffin 1996). In cases where exotic species invasion or past land-use history have degraded native abundance, native species are unable to take advantage of even the most diligent efforts to remove exotic species or modify the competitive environment.

Habitat restoration strategies must, therefore, take into account not only the vulnerability of native individuals to competition with exotic species but also the limited source of native propagules in degraded ecosystems. There is strong evidence that the colonization of appropriate habitats by native-plant populations is often limited by seed availability. The soil seedbanks of native grasses and some forb species have been shown to be negligible in both disturbed (Kotanen 1996) and undisturbed (Peart 1989a) habitats in California. Seed rain of native grasses and forbs is generally substantially lower than that of exotic species when all groups are growing in the same environment (Hobbs and Mooney 1985; Peart 1989a; Kotanen 1996). Native species also have limited ability to repopulate degraded sites from nearby remnant populations (if such populations even exist), owing to low seed production (Hobbs and Mooney 1985; Peart 1989a; Kotanen 1996) and limited seed dispersal relative to exotic species (Hobbs and Mooney 1985; Peart 1989a–c; Kotanen 1996, 1997). We believe that restoration projects that augment the pool of native propagules via seed addition or seedling out-planting are much more likely to succeed than projects that rely on natural seed dispersal and recolonization. Some efforts to increase the competitiveness of native species, including sawdust addition (Alpert and Maron 2000), mowing and biomass removal (Maron and Jefferies 2001), grazing (Hatch et al. 1999) and prescribed burning (e.g., Alexander 2001 and others), may have had more success if more native propagules had been available to take advantage of the modified competitive environment.

Seedling establishment and persistence exert a major influence on plant population dynamics (Harper 1977), particularly in a restoration context in which native-plant species must revegetate habitats from which they have been extirpated. A variety of studies in California grasslands have demon-

strated that exotic grasses restrict the establishment of native perennial grass (e.g., Peart 1989a; Dyer and Rice 1997; Hamilton et al. 1999; Brown and Rice 2000). However, there are indications that mature native perennial grasses are capable of competing with exotic species and reducing future species invasion (Peart 1989b; Corbin and D'Antonio 2004b; but see Hamilton, Holzapfel, and Mahall 1999, in *N. pulchra*-dominated grassland). We have found that mixed communities of native perennial bunchgrasses are able to reduce the aboveground productivity of exotic annual grasses and resist invasion by exotic forb species within 2 years after native-seedling establishment (Corbin and D'Antonio 2004b). These results indicate that the period of seedling establishment in the first year after seed or seedling planting is a critical phase for native-grass restoration. Habitat managers should, therefore, concentrate on increasing native competitiveness during this window of establishment, after which mature native individuals may be better able to compete with exotic species.

We believe that a strategy that uses multiple tools to both reduce the competitiveness of exotic plant species and increase the establishment of native species holds promise for more successful restoration of native biomass. The coordination of multiple approaches to deal with undesirable species, a hallmark of integrated pest management (IPM) theory (Buhler, Liebman, and Obrycki 2000; Hoddle, this volume), has been well-developed in agricultural systems and to a lesser extent in rangeland management (reviewed in DiTomaso 2000). The most common application of IPM in the control of exotic species in natural systems is the introduction of biological control agents, but IPM can also include such strategies as prescribed burning and mechanical removal. Stromberg and Kephart (1996) argued that the establishment of native grasses in abandoned agricultural fields in central California is facilitated by a three-step program that includes site preparation by either plowing or applying herbicides to exotic grasses, seeding or planting seedlings of the desired native species, and instituting postestablishment management practices designed to increase the competitiveness of the native species. Such a program, though likely more expensive and labor intensive than other restoration techniques, should be considered if it is more likely to accomplish the goal of increasing population sizes of native-plant species and creating communities with greater resistance to further invasion.

#### ACKNOWLEDGMENTS

The authors would like to acknowledge Coleman Kennedy, Meredith Thomsen, Karen Haubensak, Sally Reynolds, and other members of the

D'Antonio lab group and James Bartolome for valuable discussions that contributed to this chapter. Cynthia Brown, John Maron, and an anonymous reviewer made valuable comments to an earlier version. Jeffrey Corbin would also like to thank Malcolm Gordon and Soraya Bartol for organizing the IoE Conference under difficult circumstances. The National Science Foundation (DEB 9910008) and the David and Lucille Packard Foundation supported the preparation of this chapter.

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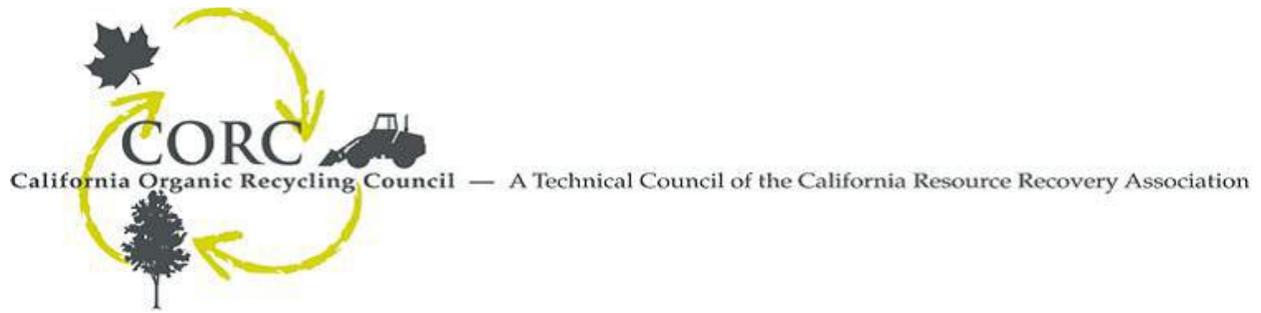
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February 12, 2016

TO: Karen Ross, Secretary, California Department of Food and Agriculture

RE: Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program (Version 1.0 – 1/5/2016)

Dear Secretary Ross,

We appreciate the opportunity to comment on CDFA's *Draft Agronomic Rates of Compost Application for California Croplands and Rangelands*, which was presented to the Environmental Farming Act Science Advisory Panel to support a CDFA Healthy Soils Incentives Program. We are following up on comments provided during the workshop.

Firstly, we are expressing our strong support for CDFA to put incentives in place to build carbon in our agricultural soils through compost applications. We commend CDFA for their leadership with regards to the Healthy Soil Initiative and we would like to express our support for the agency's effort to create a cost share incentive program for the use of compost in our agricultural crop and range land systems to build soil organic matter.

We do, however, have several concerns about some of the information provided in the report and workshop discussion. In particular, we feel the report provides mixed messages regarding the use of compost, primarily in its questionable portrayal as a significant source of N migration into surface or ground water. While there may be overall concerns regarding total N applied to enhance the fertility of the soils, the addition of compost discussed is a very small contributor (5-10% of total N), with little discussion of the benefits of increased organic matter and microbes that will aid in the stabilization of all nutrients, significantly increasing plant availability and minimizing their migration in the environment. With 90-95% of the nutrient load on agricultural lands coming from other (often synthetic) sources, nutrient migration should not be identified as a limiting factor to the expanded use of finished compost. Conversely, with the increased tilth of the soil provided by the use of compost, fertilizer application can be curtailed, making it a part of the solution to efficient nutrient management.

Fundamentally, we understand a desire to take a conservative approach in moving this concept forward, and looking at potential environmental impacts is a part of such assessment. We believe that the draft study has failed to recognize some of the essential benefits of compost application on the basis of a lack of qualified study work during the literature review. It would be helpful to the stakeholder community to better understand where CDFA believes that information gaps exist – what additional, perhaps California-specific study work needs to be conducted to best support maximizing the benefits of compost application to our native soils. Even more helpful would be a prioritized list of the outstanding issues so that efforts may be undertaken to facilitate immediate action to resolve these voids and fully realize the climate change goals of the Healthy Soils Initiative through a more robust program.

While we agree that there is “too much variation in the scientific data within both “croplands” and “compost” to define a single application rate”, we believe the current methodology of solely using the C/N ratio is overly simplistic and does not adequately explain the expected nutrient availability or release from the applied materials. Furthermore, the study unreasonably limits the proposed application rates to well below what is considered typical or recommended usage for actual field applications in current practice.

We have fully review comments provided on this matter by Dr. Jeff Creque of the Carbon Cycle Institute and wholly support his analysis and recommendations, in addition to those provided in this letter.

We look forward to continued discussion in the development of the Healthy Soils Initiative and will continue efforts to help secure Greenhouse Gas Reduction Fund allocations proposed in the Governor’s Budget to support this worthy program.

Will Bakx



CORC Chair

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11 February 2016

re: "Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program."

Via email: EcoSysServices@cdfa.ca.gov

To Whom It May Concern,

I am providing comments on the CDFA's Healthy Soils Incentives Program and accompanying report, "Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program."

My expertise is in coastal grasslands (a.k.a. "coastal prairie") in California. I have published many peer-reviewed scientific papers about the ecology of these grasslands as well as co-authoring the chapter on this ecosystem in UC Press' text *Terrestrial Vegetation of California* (Ford & Hayes 2007). My specialty is specifically in the conservation of plant species, and this is the specific concern I bring to the proposal being considered by CDFA.

Nutrient addition, such as that being proposed with compost addition to rangelands, has a strong potential to negatively impact the most imperiled plant species of California's grasslands and could negatively affect the long-term productivity of our state's rangelands. In the world's most species rich grasslands, either increased nitrogen or phosphorus reduce plant species richness (Ceulemans et al. 2013). This is partly due to nutrient availability favoring competitive species, which are normally grasses and especially weedy grasses – species that negatively impact native species in California's grasslands (Hobbs et al. 1988, Huenneke et al. 1990, Brooks 2003). Nutrient enhanced grasslands with increased grass growth produce higher light competition, reducing the diversity and abundance of lower-statured species (Bobbink et al. 1987, Borer et al. 2014) as well as reducing the establishment of less competitive native species (Brown & Rice 2000). We know little in California about threshold limits to nutrient inputs, beyond which species diversity declines, but such thresholds have been established in other arid U.S. grasslands (Tipping et al. 2013). In other species-rich Mediterranean grasslands, crossing these thresholds, reducing species diversity has lasted more than 100 years even with attempts to improve management (Forey & Dutoit 2012).

Preliminary analyses on compost addition impacts on native plant abundance/diversity in native grasslands support the aforementioned concerns. Although these experiments took place on especially species-poor sites, even these sites experienced a reduction of cover of the few native species present with a corollary increase in the most invasive, non-native and competitive species, especially non-native grasses (Ryals et al. 2016).

California's coastal prairies, in particular, and California's native grasslands, in general, are of extremely high conservation value. These grasslands are one of the top ten most endangered ecosystems in the United States (Noss et al. 1995). Coastal prairies have been destroyed by urbanization more than any other major ecosystems in the nation (Loveland & Hutcheson 1995). At the same time, coastal prairies are the most diverse grasslands in North America (Stromberg et al. 2002). Many of the rarest native species of California's coastal prairies are short-statured annual wildflower species which are already threatened by the invasion of tall-statured weeds (Ford & Hayes 2007); further nutrient inputs would increasingly endanger those species.

In conversations with advocates of compost addition to rangelands practices, I have heard several suggestions to mitigate the concerns I raise above. The first is that professionals could easily determine areas of low conservation concern where compost addition would have the least potential to impact native species diversity. This is problematic because of California's grasslands high interannual variability, where "non-native" grasslands appear more like "native" grasslands in some years, without apparent correlation to rainfall or other factors (Hobbs & Mooney 1995). The second suggestion has been to avoid areas with known rare, threatened, and endangered species. This is also problematic for the reason previously stated, and because few locations of such species have been well documented. The third suggestion has been that careful grazing management would ameliorate any potential negative effects of compost addition, and that qualified rangeland managers would carefully construct (complex) grazing plans and monitor the results, adjusting management as necessary. This last suggestion is particularly perplexing as science has yet to satisfactorily answer the most basic questions about how changes of grazing management can better sustain our grassland plant species diversity: the addition of considerable acreage (under known threat by nutrient addition) to the burden of rangeland science would not be a solution, but a further and onerous burden.

In conclusion, I suggest that compost addition is best applied in purely intensive agricultural systems, avoiding the natural or semi-natural grasslands where the risks are too high and the precautionary principle requires that California's citizens be better informed by much more research than is currently available.

Many thanks for your consideration,



Grey Hayes

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February 11, 2016

**Comments re: “Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program”**

Submitted via email: [EcoSysServices@cdfa.ca.gov](mailto:EcoSysServices@cdfa.ca.gov)

The Nature Conservancy (TNC) and Dr. Claire Kremen appreciate the opportunity to provide comments on the California Department of Food and Agriculture’s (CDFA) Healthy Soils Incentives Program and white paper, “Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program.”

TNC is an international non-profit organization dedicated to conserving the lands and waters on which all life depends. Our on-the-ground work is carried out in all 50 states and in 35 countries and is supported by one million members. The foundation of TNC’s work is our commitment to using the most up-to-date conservation science information and methodologies to guide decision-making.

Dr. Claire Kremen is an American biologist, and professor of conservation biology at University of California, Berkeley. She collaborates with The Nature Conservancy on various projects related to agriculture, biodiversity and ecosystem services. Her work focuses on understanding and characterizing the relationship between biodiversity and ecosystem services, and utilizing this information to develop conservation and sustainable management plans, considering both protected areas and the working lands matrix around them. She has won numerous honors, including the prized MacArthur Foundation Fellowship for her contributions to ecology, agriculture and biodiversity (2007) and the Honorable JC Pritzlaff Conservation Award (2014). She also co-directs the Berkeley Food Institute and the Center for Diversified Farming Systems.

TNC strongly supports using natural and working lands as part of a climate change strategy, and our climate team has been actively working on these issues for two decades. We also support ongoing efforts to reduce greenhouse gas (GHG) emissions and the related goal of diverting all organic waste from landfills by 2025 as a key emission reduction strategy in California.

The effort to improve soil health in a way that also sequesters carbon is laudable, and Healthy Soils represents an exciting payment for ecosystem services program. Here we identify several issues in the white paper that would benefit from further assessment or clarification, make recommendations for the calculation method and agronomic application rates for rangelands, and suggest a path forward that would ensure the program can commence quickly while also including appropriate environmental safeguards and natural resource protection.

Our concerns with creating an incentive for the application of compost under the Healthy Soils Incentive Program and the approach proposed in the white paper fall into three primary categories: 1) determining standardized, appropriate rates of compost application to crop and rangelands; 2) impacts of potential nutrient runoff and loading (particularly nitrogen); and, 3) potential impacts to native biodiversity in California's highly biologically diverse grasslands and other rangelands. We offer the following comments and recommendations that address these concerns:

- **Applied rates of compost should be based on measured nitrogen content.** The agronomic compost application rates proposed were developed based on *average* pounds of nitrogen per ton of dry compost from CalRecycle's 1364 compost samples (Table 1 in Gravuer white paper). However, the amount of nitrogen in compost varies enormously and can be 3 times the average. Therefore, using averages instead of measured content will result in unpredictable outcomes for landowners/managers (e.g., forage production) and the environment. We therefore recommend including information on the ranges of nitrogen levels in compost in Table 1 (i.e. standard deviations), and that application rates be based on the measured content of the compost. CDFA could provide a simple spreadsheet calculator for producers to use instead of recommended rates.
  
- **Safeguards should be included in the program to prevent nutrient runoff and loading in adjacent streams and waterbodies.** Nitrogen (N) and phosphorus (P) are essential nutrients in terrestrial and aquatic ecosystems but excessive levels cause significant negative impacts to waterways and natural communities. Over 300 waterbodies in California are listed under section 303d of the Clean Water Act for nutrient pollution due to agricultural activities (per State Water Resources Control Board data: [http://www.waterboards.ca.gov/water\\_issues/programs/tmdl/integrated2010.shtml](http://www.waterboards.ca.gov/water_issues/programs/tmdl/integrated2010.shtml)). We would like to see the Healthy Soils Program address the following issues to reduce potential for nutrient runoff:
  1. Avoiding impacts of phosphorus additions is currently explicitly addressed in "Other Considerations" and specifically as part of the proposed "Rangelands site assessments" (page 17 of white paper). In addition, we recommend that more specific safeguards should be developed prior to implementation of the program for both crop and rangeland sites that include, but are not limited to, downward adjustment of application rates or avoidance of application in areas with higher slopes and potential for runoff, in watersheds with already-impacted waterbodies, and in areas where little is known about transport processes. We recommend specific guidelines be developed through consultation with the UC Davis Rangeland Watershed Laboratory or other science advisors with topical expertise.

2. Nitrogen runoff is not currently assessed in the white paper; it should be added to the white paper in the crop section as well as “Rangeland site assessments” and reflected in the guidelines for adjusting to application rates, similar to those proposed for phosphorous. Specific guidelines should be developed concurrently with those for P.
- **The rangeland compost application program should be implemented in phases, with near-term applications focusing on (1) scenarios that available science suggests are low risk and/or (2) carefully planned demonstration projects over a wider range of conditions to refine implementation guidelines and resolve uncertainties about ecosystem impacts.** Rangelands in the Western US are diverse physically and ecologically, and do not respond to perturbations predictably (i.e., they exhibit non-equilibrium dynamics) ([Booker et al. 2013](#)). As a result, soils and biological communities cannot be expected to respond uniformly to compost additions. California rangelands, especially grasslands and oak savannahs, are largely dominated by non-native species, notably annual grasses. Nevertheless, California’s native floral diversity remains among the highest on earth, with ~6500 native and over 2100 endemic species (Jepson Flora Project (eds.). *Jepson eFlora*, <http://ucjeps.berkeley.edu/IJM.html>, accessed on January 20, 2016). Many of these California native plants depend on our rangeland ecosystems, which are home to over 50% of California’s rare and endangered species. It is important to understand, however, that most of these native species found in rangelands currently persist in low relative abundances. **A critical consideration for the potential use of compost additions in rangelands is that the native plant species richness of rangelands is often determined—paradoxically—by relatively nutrient-poor soil conditions** (Harpole et al. 2007). Furthermore, the diversity and abundance of native forb species is critical for sustaining our wild native pollinators, which in turn supply valuable pollination services both in rangelands and to neighboring crops ([Chaplin-Kramer et al. 2011](#)). Conservation of pollinators is now considered a national policy objective under the National Strategy to Promote the Health of Honeybees and Other Pollinators (<https://www.whitehouse.gov/sites/default/files/microsites/ostp/Pollinator%20Health%20Strategy%202015.pdf>). Although there have been very few compost-addition studies, *per se*, completed in California’s extensive rangelands that confirm long-term GHG benefits, it is well documented that higher nutrient levels (specifically, N and P) in many rangelands, particularly lower-productivity sites, results in shifts in plant community composition that favor invasive annual grass species and reduce native plant cover. In order to meet both the goal of increased carbon storage and avoid impacts to natural communities, we propose a phased approach to implementing the rangelands compost program, with the following elements:

- 1. Appropriate determination of agronomic application rates.** Critical nutrient level thresholds for most rangelands have not been identified. However, as the white paper acknowledges, even low rates of N addition (~6 kg N/ha/year) can encourage growth of invasive annual species, leading to declines in native species in areas with low natural productivity. Sites with higher levels of soil P, typically found in association with higher N in composts, have also been documented to correlate with the absence of native species in extensive rangelands of California (Gea-Izquierdo et al. 2007). On the other hand, the addition of carbon (without N or P) can have neutral or even beneficial effects on native plant species composition in some California grasslands ([Alpert 2010](#)). *We therefore recommend that only higher C:N ratio composts be used in areas other than converted rangelands (defined below) until further information is available to ensure there are no adverse impacts to native biodiversity. We also recommend that application rates be limited to 3-5 tons/acre in areas other than converted rangelands or where ecologically-relevant studies (i.e., with similar climatic, topographic, species composition, and soil conditions) have documented no or low impacts to native species diversity and abundance.* By supporting a post-doctoral researcher (see below), TNC is committed to helping rapidly develop studies that would identify critical thresholds for a wider range of rangeland types, thereby helping expand the program responsibly over time and increasing opportunity for GHG mitigation.
- 2. Implement Program in Phases.** We recommend that a first phase of the program be limited to rangelands that have been converted (e.g., plowed, irrigated, heavily seeded, or otherwise disturbed such that the natural communities and soil conditions are no longer present). In addition the first phase should include only the use of lower C:N ratio composts and include appropriate restrictions that will limit N and P runoff or impacts to sensitive areas. Concurrently, several demonstration sites across California along a gradient of soil, climate, species composition, and management conditions should be established and outcomes tracked. Following this first phase, we recommend that a second phase of the Program include unconverted rangelands that are naturally more mesic (i.e. wet) and with naturally rich soils and high productivity, provided that research currently underway and demonstration sites established through Healthy Soils conclusively demonstrate that compost amendments do not cause reduction of native species abundance or richness. A scientific panel should be assembled to assess these outcomes and make recommendations. Finally, future phases of the Program could include drier lower-productivity sites if outcomes from demonstration sites show that compost addition can lead to carbon storage

without resulting in loss of biodiversity (as measured by native species richness and abundance/cover, or per the recommendations of the panel).

- 3. Exclude application of compost in all sensitive ecological areas.** Areas that are considered sensitive to addition of nutrients and therefore not eligible for the Program are discussed on page 14 of the white paper. A more complete list should include, but not necessarily be limited to:
- Serpentine and serpentine-influenced soil types;
  - Sites containing vernal pools;
  - Sites containing federal, state, and/or CNPS listed native plants; and/or animals that require low-stature rangelands for their life history, including but not limited to San Joaquin Kit Fox, Giant Kangaroo Rat, Tiger Salamander, and/or Burrowing Owl;
  - Wet meadows or other seasonally inundated rangelands, regardless of slope (e.g., floodplains);
  - Desert grasslands;
  - Coastal prairie;
  - Chaparral, coastal sage scrub, and other systems dominated by native shrubs;
  - Sites that have recently burned;
  - Sites in watersheds already impacted by N or P, unless appropriate mitigating practices are included (as described above).

We commend the work of CDFA staff as they developed this program and appreciate this opportunity to offer our recommendations. As previously stated, we strongly support investments in enhancing the carbon sequestration of natural and working lands and reducing emissions in those sectors. As mentioned above, we have committed to hiring a post-doctoral researcher for the next two years through the NatureNet Fellowship Program to study the climate benefits and environmental outcomes of organic amendments to California's agricultural soils, and look forward to coordinating with CDFA and other key agencies and land managers on those studies. We are also available to work with CDFA to further identify how this exciting program can forward with the safeguards as suggested in our comments. Our science team can provide additional mapping resources, and/or relevant bibliographic and ecological information, and we would be able to serve as advisors to the program to develop demonstration projects.

Please feel free to contact us with any questions or comments.

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Dr. Claire Kremen, Professor, [ckremen@berkeley.edu](mailto:ckremen@berkeley.edu)

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## Appendix: Data Sources for Sensitive Habitats

- Serpentine and serpentine-influenced soil types
  - [http://www.conservation.ca.gov/cgs/information/publications/pub\\_index/pages/gis\\_data.aspx](http://www.conservation.ca.gov/cgs/information/publications/pub_index/pages/gis_data.aspx)
- Sites containing vernal pools
  - USFWS Vernal pool critical habitat [http://www.fws.gov/sacramento/es/Critical-Habitat/Vernal-Pool/es\\_critical-habitat-maps\\_vernal-pool.htm](http://www.fws.gov/sacramento/es/Critical-Habitat/Vernal-Pool/es_critical-habitat-maps_vernal-pool.htm)
  - CADFV vernal pools (under “other biogeographic data”) <https://www.dfg.ca.gov/biogeodata/gis/clearinghouse.asp>
- Sites containing federal, state, and/or California Native Plant Society listed native plants; and/or animals that require low-stature rangelands for their life history, including but not limited to San Joaquin Kit Fox, Giant Kangaroo Rat, Tiger Salamander, and/or Burrowing Owl;
  - Mostly native plants: from NRCS National Resources Inventory Rangeland Resource Assessment, data descriptions here: <http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/nri/results/?cid=stelprdb1253602>
  - CNDDDB covers specific species - <http://www.dfg.ca.gov/biogeodata/cnddb/>
  - Also critical habitat (Kit fox, tiger salamander, etc) <http://ecos.fws.gov/ecp/report/table/critical-habitat.html>
- Wet meadows or other seasonally inundated rangelands, regardless of slope (e.g., floodplains)
  - USFWS National Wetlands Inventory <http://www.fws.gov/wetlands/Data/Mapper.html>
  - CalFire FVeg 2015, includes wetland category (WHR10NAME) which covers wet meadows and emergent wetlands <http://frap.fire.ca.gov/data/frapgisdata-subset>
  - FEMA floodplains and flood hazards data <https://hazards.fema.gov/femaportal/wps/portal/NFHLWMS>  
<https://www.fema.gov/national-flood-hazard-layer-nfhl>
- Desert grassland
  - CalFire FVeg 2015, includes desert vegetation categories <http://frap.fire.ca.gov/data/frapgisdata-subset>
- Coastal prairie
  - CalFire FVeg 2015, includes coastal scrub and chaparral vegetation categories <http://frap.fire.ca.gov/data/frapgisdata-subset>
- Sites that have recently burned

- CalFire records of fire perimeters (2014)  
[http://frap.fire.ca.gov/projects/fire\\_data/fire\\_perimeters\\_index](http://frap.fire.ca.gov/projects/fire_data/fire_perimeters_index)
- Sites in watersheds already impacted by N or P, unless appropriate mitigating practices are included (as described above). Data available from State Water Resources Control Board- [http://www.waterboards.ca.gov/water\\_issues/programs/tmdl/integrated2010.shtml](http://www.waterboards.ca.gov/water_issues/programs/tmdl/integrated2010.shtml)

TO: Karen Ross, Secretary, California Department of Food and Agriculture  
RE: Draft Report for the Environmental Farming Act Science Advisory Panel,  
Agronomic rates of compost application for California croplands and rangelands to support  
a CDFA Healthy Soils Incentives Program (Version 2.0 – 02/13/2016)

Dear Secretary Ross,

Thank you for the opportunity to comment on CDFA's *Draft Agronomic Rates of Compost Application for California Croplands and Rangelands*.

We commend CDFA for their leadership with the Healthy Soil Initiative and we would like to express our support for the agency's effort to create a cost share incentive program for the use of compost in our agricultural and range land systems.

Compost application in crop, pasture, and rangeland ecosystems is the most direct and rapid way to build soil organic matter. It is a catalyst, "turning on" a vast photosynthetic carbon sink offered by these working lands<sup>1</sup>. In addition, increasing carbon in soils via compost application has beneficial effects on almost every soil property: chemical, physical, and biological, including soil tilth, nutrient availability, water holding capacity, cation exchange capacity, bulk density, and aggregation<sup>2</sup>. Unlike nitrogen fertilizers, which feed plants, compost feeds the entire soil food web, which in turn builds the healthy soils central to your effort.

While we are greatly supportive of the effort overall, we ask that you reconsider the choice of C:N ratio as the defining metric of characterization for compost types that will be supported under the new proposed program. We are concerned that this metric demonstrates a lack of understanding of how compost functions as a soil amendment (rather than as a fertilizer) and believe that there are more meaningful and accurate metrics that could be used for this incentive.

Our comments:

1) C:N ratio is used by composters, both at the beginning of the process as an overall indication of feedstock balance and at the end of the process as a rough indicator of compost maturity. **We recommend percent of inorganic nitrogen coupled with a maturity indicator, as indicated by the CCQC Maturity Index<sup>3</sup> (see attachment 1), be used as alternative metrics to manage for undesirable nitrogen instead of C:N.**

- a. The C:N ratio, by itself, does not give an accurate indication of the type or amount of nitrogen present in the finished product. As CDFA states on pages 5 and 6 of the Draft Report, there are different forms of nitrogen: nitrate [and nitrite which converts to nitrate rapidly], ammonium, and organically-bound nitrogen. Controlling for nitrate (the water soluble form of nitrogen

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<sup>1</sup> *Effects of Organic Matter Amendments on Net Primary Productivity and Greenhouse Gas Emissions in Annual Grasslands*. Ryals & Silver, Ecological Applications 23(1) 46-59.

<sup>2</sup> Brown, BioCycle March 2009, Vol. 50, No. 3, p. 44

<sup>3</sup> *CCQC Maturity Index*, Test Methods for the Examination of Compost and Composting, April 2002, 05-02.13

which is of concern for water quality) is more a matter of the maturity rather than the C:N ratio of the product.

- b. C:N ratio does not give an indication of how much nitrogen is available in the compost.
- 2) The use of C:N in this proposal incorrectly categorizes compost as a fertilizer and not a soil amendment. Fertilizers feed plants, whereas compost feeds the entire soil system. **We suggest that revisions be made to clarify the effects of compost on soil structure, cation exchange capacity, and nutrient availability, and that CDFA remove the comparison to fertilizers with respect to nitrogen effects in the soil system. At a minimum, comparisons of compost with inorganic fertilizers should consider available N, rather than total N, in the compost. The use of an upper bound of % N coupled with a requirement on maturity prior to application should sufficiently address nitrogen concerns with regard to application of the finished compost.**
- a. We are strongly supportive of protections for our state's threatened water resources. Contamination of water sources from nitrate leaching is a significant issue presented by the use of synthetic fertilizers and direct application of uncomposted manure. We recognize that contamination of water can be a concern during the process of composting certain organic materials, especially manures, and are supportive of the current CA State Water Resources Board proceedings in this regard. However, there is far less of a concern of contamination of water with the application of mature, finished compost to soils, the definition of which is given by CalRecycle as, *"the end product resulting from the controlled biological decomposition of organic material from a feedstock into a stable, humus-like product that has many environmental benefits...fully mature compost...may be safely applied directly to existing landscapes."*
  - b. Compost is a powerful tool to help manage for excess nitrates both from a perspective of substitution of synthetic fertilizers and from that of soil stabilization. It increases the cation exchange capacity of the soil, causing the soil to hold onto nutrients (including water soluble forms of N) like a sponge or magnet<sup>4</sup>. Additionally, increasing carbon in the soil increases water holding capacity, thereby decreasing the opportunity for leaching and runoff. Over time, soils with higher organic carbon content tend to have higher organic N content, and are thus able to provide an increasing percentage of slowly released N to meet crop demand. As organic N is rendered gradually available through dynamic soil biochemical processes, the need for synthetic N inputs is reduced.
  - c. CDFA is inconsistent in the way it presents the discussion of nitrogen. We are concerned that this could provide inaccurate or unsuitable precedent for future compost incentives. CDFA acknowledges that compost has an

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<sup>4</sup> Brown, BioCycle March 2009, Vol. 50, No. 3, p. 44

overall positive effect on soil structure that mitigates issues of water-soluble nitrate in a footnoted comment on page 6. However, the focus on nitrogen in C:N ratio as the metric of choice inappropriately implies that nitrogen in compost functions in the soil in the same way as nitrogen from synthetic fertilizers. The Draft Report states, *“The comparison of nitrogen in compost to synthetic fertilizers was made simply to mimic soil physio-chemical behaviors of the nitrogen and not compare the amendment to synthetic fertilizers.”* This comparison is confusing and inaccurate.

- 3) The proposed CDFA incentive offers an opportunity to support the development of a sector and product that has a cascade of positive benefits associated with it. However, the narrow focus on nutrients could limit the achievement of the full suite of benefits available to rebalance both the carbon and nitrogen cycles, and restore the resilience of our working lands. **We recommend CDFA increases its incentivized cropland application rate by 50%, to at least 12 tons of compost per acre/per crop cycle, and we support incentivizing compost application on conventional and organic croplands equally.**
- a. If CDFA is seeking to address concerns associated with compost nitrogen, then **we suggest a broader view of the N cycle be considered** based upon the understanding that providing a strong incentive for the creation and use of compost will have positive consequences with regard to mitigating the impact of undesirable forms of nitrogen on water quality and greenhouse gas emissions. Positive consequences include: 1) avoidance of the production of methane and nitrous oxide associated with changes in waste management; 2) displacement of synthetic sources of N and avoidance of their negative water quality and greenhouse gas effects; 3) increased photosynthetic capacity for carbon sequestration and plant productivity; and 4) minimization of agroecosystem nutrient losses, including nitrous oxide production and nitrate leaching.
  - b. CDFA’s proposed rates for compost application on croplands are very low and may not offer significant incentive to farmers currently using synthetic fertilizers. With these low proposed rates, CDFA may miss the opportunity to reduce emissions associated with synthetic fertilizer manufacture, potential ground water contamination, and nitrous oxide emissions.

In closing we wish to express our support and gratitude to CDFA for leading the Healthy Soils Initiative and for this new ground “making” initiative.

Thank you,

John Wick  
Co-Founder, The Marin Carbon Project

Calla Rose Ostrander

January 13, 2016

Submitted to [EcoSysServices@cdfa.ca.gov](mailto:EcoSysServices@cdfa.ca.gov) on January 25, 2016

Don Cameron  
Chair, Environmental Farming Act Science Advisory Panel  
California Department of Food and Agriculture  
1220 N Street  
Sacramento, CA 95814

RE: Joint State Agency Public Meeting of the Environmental Farming Act Science Advisory Panel (EFA SAP), CalRecycle, and the California Department of Food and Agriculture

Dear Chairman Cameron:

EDF applauds the research that CDFA has undertaken to develop the science behind maximizing the environmental benefits of California's working lands. We support the rigorous research that has gone into determining the optimal rates of compost application to crops, orchards, vineyards, and grasslands.

Given the expansiveness of California's grasslands and their immense potential to support wildlife, sequester carbon, and provide recreational benefits to our families, we are grateful for your continued efforts to protect them. The studies conducted so far to determine the effect of carbon sequestration resulting from compost addition have been limited to a small subset of California's grassland ecosystems that are not representative of the variety that exists across the state. Scientific literature indicates that grazing management, fire, soil type and moisture, are some of numerous factors that can influence carbon storage in grasslands<sup>1</sup>. Without considering this diversity, it is not possible to accurately calculate the additional carbon sequestered or understand any potential side-effects of these practices. For these reasons, we strongly support CDFA's plans to coordinate with NRCS to expand demonstration projects to evaluate the effects of compost additions and assumptions across a range of geographies, soil types, and climate zones.

Over the past five years, EDF and our partners have conducted research on various aspects of the application of compost to grazed grasslands and we would be happy to provide those materials and our experience to CDFA or its partners. Furthermore, we are currently developing a series of tools that can help landowners and policymakers understand carbon sequestration benefits of preserving grasslands. We have developed a mapping tool that estimates the carbon sequestration potential of preserving grasslands which accounts for a number of significant factors, such as land use history, climate, geography, and soil type. This

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<sup>1</sup> Schuman, G. E., Janzen, H. H., & Herrick, J. E. (2002). Soil carbon dynamics and potential carbon sequestration by rangelands. *Environmental pollution*, 116(3), 391-396.

analysis could be beneficial in determining the location of pilot. All of this research paves the way for a successful Healthy Soils Incentives Program.

CDFA has done a tremendous job supporting the conservation of working lands in California over the years. EDF thanks CDFA for this opportunity to offer comments. We look forward to continued collaboration to implement policies and strategies for the preservation of California's working lands.

Sincerely,



Robert Parkhurst  
Director of Agriculture Greenhouse Gas Markets  
Environmental Defense Fund



**TO:** Karen Ross, Secretary,  
California Department of Food and Agriculture

**RE:** Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program

Dear Secretary Ross:

Thank you for the opportunity to respond to the Draft Report for the Environmental Farming Act Science Advisory Panel, Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program. I am following up on the comments I made during the workshop. I am expressing my strong support for CDFA to put incentives in place to build carbon in our agricultural soils through compost applications.

Compost is in the first place a soil amendment, not a fertilizer. Not until recently could compost manufacturers make a claim on the nutrient value in their products. As a soil amendment compost helps to conserve water, reduce erosion, diversify soil microbial populations and increase carbon in the soil. For that matter, increasing soil organic matter is probably the only economically viable way to significantly reduce greenhouse gas emissions to curb global warming. Whereas most efforts to prevent impacts from global warming are aimed at reducing emissions, thereby slowing down the impacts on climate change, increasing soil organic matter can actually reverse global warming through carbon sequestration.

The report puts a lot of emphasis on nitrogen as a guide to determine how much compost can be used. We certainly do not want to solve one problem while creating another. However, given the quantities of compost that are used in agriculture and the slow release of nutrients from mature compost there is no real threat to the environment. In fact, compost has the ability to minimize the impacts of nutrient pollution from conventional fertilizers through immobilization. A focus on nutrients is misplaced and will slow down the benefits that can be gained from carbon sequestration through compost applications.

The report also uses C:N ratios to evaluate the amount of compost to be used. First, C:N ratio is an indicator of maturity of compost, not an accurate indicator of the amount of nitrogen

present in the compost. Percent nitrogen would be a much better tool. By introducing C:N ratios as a guide to determine how much compost can be used the discussion has entered a field of confusion.

We would like to make the following recommendations:

- Simplify the approach on how much compost can be used
- Omit the use of C:N ratio and abandon the two tier system of high and low nitrogen as well as the differentiation between organic and conventional agriculture to determine how much compost can be applied
- As a starting point for the incentives, adopt up to 8 wet tons per acre for crop production per crop cycle, 8 wet tons per acre per year for orchards and up to 30 tons per acre for rangeland. As the Marin Carbon Project has demonstrated, repeat application on rangeland may be necessary only once in 20 years
- In order to get the maximum impacts from the incentive program, CDFA needs to assess the rate at which the minimum amount of compost applied yields the maximum rate of carbon sequestration (sweet spot)
- Adjust the incentives program in 2-3 years to reflect the sweet spot
- The RCD's will soon start to create Carbon Farming Plans for farmers. We recommend that compost uses as recommended in the Carbon Farming Plan will be automatically approved for the incentives

In summary, we applaud you for moving forward to incentivize the use of compost to promote soil health and maximize soil carbon sequestration. However, the draft report use of C:N ratio brings in confusion and should be abandoned. There should be a focus shift from nitrogen to rate of carbon sequestration. The draft can be greatly simplified to achieve this lofty goal toward improving soil health.

Thank you,

Will Bakx



[willbakx@sonomacompost.com](mailto:willbakx@sonomacompost.com)

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West Marin Compost  
6290 Nicasio Valley Rd.  
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From: Ron Alexander <alexassoc@earthlink.net>  
Sent: Tuesday, February 23, 2016 12:05 PM  
To: Gunasekara, Amrith@CDFA  
Subject: RE: HSI

Amrith,

Thank you for forwarding me the compost application rate data. I am excited about your new program and look forward to it beginning....and then data being generated. We all hope that the efforts to add stable organic matter to crop and rangeland acreage will reduce the impact of climate change, while improving plant growth and conserving water.

My main comments are few.....

1. C:N Ratio – pretty much all commercial composts have a C:N between 12-30:1, and most are between 15-25:1, so all of the composts are going to be considered as high C:N products. Perhaps we should discuss this, as maybe the ‘higher’ and ‘lower’ differentiation is not necessary.
2. Application rates – of annual crops-8t/a, tree crops-6-8t/a, and rangeland-15-30t/a are fine. However, I would suggest the higher end of the application rates for rangeland, as they better relate to the Marin Carbon Project research.
3. Nitrogen concerns – I do know how experienced the expert team is with compost, but excess nitrogen should not be considered a great concern. In my experience (30 years plus) good composts contain nitrogen that is 90-95% in organic form and less than 1% in combined ammonia and nitrate forms. Further, most data suggests that only 10-20% of the organic-N is available the first year, then rates reduce from there in subsequent years. Therefore nitrogen drift, especially in agricultural applications, seems unlikely.
4. Test methods - FYI, good compost test methods exist, so no need to ‘re-invent the wheel’ here. The US Composting Council manual is readily available (TMECC – Test Methods for the Examination of Composting and Compost) and contains them. Testing using TMECC test methods is required in the national testing program (Seal of Testing Assurance Program), and all composts sold to Caltrans are in the program.

Thanks again, and please feel to call on me if you would like my opinion on any of this. It’s very exciting.  
Ron

From: Tracy V Hruska <t.hruska@berkeley.edu>  
Sent: Friday, February 12, 2016 1:48 PM  
To: CDFA Environmental Stewardship@CDFA  
Subject: Comments on "Agronomic rates of compost application for California croplands and rangelands..."

To Whom it May Concern,

I am writing to express concern about the manner by which compost is proposed to be added to rangelands in California, specifically the lack of a systematic approach to researching the effects of this practice. As was pointed out in the document, there are a number of potential risks associated with fertilizing rangelands, particularly in California where the vast majority of grasslands are dominated by non-native species. I am concerned that the proposal to apply compost does not include a budget for ongoing research/monitoring, nor does there appear to be a systematic approach to starting the program, where small, specific areas might be selected for treatment and subsequently carefully monitored.

It is important to note that most of the literature on fertilizing rangelands generally - not just in California - is ambiguous at best on potential benefits. This is especially true in more arid regions. The fact that the only two studies conducted thus far have been in comparatively wet, coastal grasslands means that we do not yet know how composting would affect the drier parts of the state (i.e. the majority of it). I recommend reading up on the subject in the document "Conservation Benefits of Rangeland Practices" put out by the Natural Resource Conservation Service. It is available for free here: <http://www.nrcs.usda.gov/wps/portal/nrcs/detail/in/home/?cid=stelprdb1045811>.

While adding compost to crop lands has been standard practice for millennia, there may be good reason that it is rarely done in arid or semi-arid rangelands (beyond what is deposited by grazing livestock, of course). I highly recommend that those responsible for this project modify the current plan to include a slower, more systematic delivery of the program on rangelands, and that adequate funding be dedicated to monitoring of results for at least 3 years before scaling up the program.

Regards,  
Tracy Hruska

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UC - Berkeley

From: Thayer Tomlinson <info@biochar-international.org>  
Sent: Friday, February 12, 2016 6:44 AM  
To: CDFA Environmental Stewardship@CDFA  
Subject: Comment on Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program  
Attachments: Chapter 25 Composting and Growing Media.pdf; Compost\_biochar\_IBI\_final.pdf

Dear CDFA Members,

Thank you for the opportunity to provide comments to the compost application document. We (the International Biochar Initiative) have been compiling information on the co-beneficial effects of adding biochar to composting operations. I attach here for your review (and possible consideration to add biochar to the document) a literature review on biochar and compost as well as Chapter 25 from the recent book "Biochar for Environmental Management" that highlights research on biochar and composting/growing media. As noted in the literature review, recent research has found that "the benefits of adding biochar to the composting process may include shorter compost times; reduced rates of GHG emissions (methane, CH<sub>4</sub> and nitrous oxide, N<sub>2</sub>O); reduced ammonia (NH<sub>3</sub>) losses; the ability to serve as a bulking agent for compost; and reduced odor. For the biochar material itself, undergoing composting helps to charge the biochar with nutrients without breaking down the biochar substance in the process."

Please feel free to follow up with any additional questions.

Best regards and thank you for your consideration,  
Thayer

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# Biochar as an additive to compost and growing media

*Christoph Steiner, Miguel A. Sánchez-Monedero and Claudia Kammann*

## Introduction

Recycling of organic wastes through composting has been practiced for ages (Gajalakshmi and Abbasi, 2008). Composting is a biological treatment of biodegradable solid waste and an option to convert the enormous quantities of organic waste that are generated in the world into a valuable soil amendment. Composting can be done at various scales, ranging from single household compost bins to large industrial facilities. Piled organic matter with an appropriate moisture and nutrient content decomposes quickly and a large proportion of its original carbon (C) content is lost until decomposition slows down. Once decomposition rates are reduced to relatively low levels (stabilization) the product is a valued soil conditioner, providing nutrients and soil organic carbon (SOC).

Carbonization is another option to stabilize organic wastes. The product is biochar, if the carbonized plant material is used as a soil amendment. The two products, compost and biochar, do not necessarily compete for the

same resources and their production and utilization can be synergistic.

## Composting

During composting the organic material is consumed by several bacteria, actinomycetes and fungi. This biological degradation liberates carbon dioxide (CO<sub>2</sub>), ammonia (NH<sub>3</sub>), water (H<sub>2</sub>O) and heat (Bernal et al, 2009). In poorly managed composts under anaerobic conditions, also nitrous oxide (N<sub>2</sub>O) and methane (CH<sub>4</sub>) may be produced. Complex organic compounds are partially degraded and transformed into more stable substances. During the thermophilic phase a compost pile can exceed 60°C and the degradation takes several weeks. During this phase the high temperature destroys weed seeds, pathogenic microorganisms, undesired insects and other unwanted organisms; the effect of this hygienization can be tested via standardized seed germination tests in mature composts (Kehres, 2003). This peak in biological

activity is followed by slower degradation (maturation phase). Once the product has a low mineralization rate it is called compost. Applied to soil its further mineralization liberates plant nutrients and CO<sub>2</sub>.

The application of immature compost can inhibit seed germination and even damage roots, if rotting material is applied to the soil (Wong, 1985). Negative responses of plants could also be caused by the presence of phytotoxic substances such as phenolic and organic acids, nitrogen (N) immobilization due to a high C/N ratio, an excess of ammonium (NH<sub>4</sub><sup>+</sup>), or osmotic stress due to high salinity (Gajalakshmi and Abbasi, 2008). To avoid these negative effects, only mature composts with well-stabilized organic matter should be applied. The ideal C/N ratio of mature compost is below 20 (Bertoldi et al, 1983) but other maturity indices also exist, based on different physical, chemical and biological properties of composts. The benefits of compost addition are mostly associated with an increase of SOC and the nutrients released during its mineralization (Stevenson, 1994). Well-made composts are known to improve soil structure, which facilitates air exchange, water infiltration and retention (Bertoldi et al, 1983).

## Biochar

The benefits of compost addition are manifold and are similar to those reported for biochar additions. For example, the suppression of plant pathogens by the addition of compost to container media was reported by Gajalakshmi and Abbasi (2008), while a systemic induced disease resistance due to biochar addition to growing media was reported by Elad et al (2011). The combined use of biochar and compost as a soil amendment presents benefits to crops and nutrient cycling, especially with respect to N use efficiency (Steiner et al, 2007, 2008; Asai et al, 2009; Gathorne-Hardy et al, 2009). However, investigations of the interactions between biochar and organic matter during the composting process have only begun in recent years. Fischer and Glaser (2012) suggested that the co-composting of biochar with fresh organic matter and nutrients would lead to an accelerated composting process and the production of a substrate with enhanced fertility and C-sequestration potential. Grob et al (2011) suggested that the combination of the durability of biochar and its environmental benefits may drive the development of new products and market strategies.

## A complementary approach

Mixing biochar with compost provides several advantages for both materials. As feedstocks for biochar production are often nutrient-poor materials, biochar would be enriched with plant nutrients derived from the nutrient-rich compost feedstocks. For biochar to be an effective soil conditioner, a source of N is of particular importance. Several studies report the synergistic effects of N fertilization and biochar as soil amendment (Chan et al, 2007; Steiner et al, 2007; Asai et al, 2009; Gathorne-Hardy et al, 2009).

Biochar improved the N use efficiency in these studies while the less stable C fraction of biochar can cause N immobilization.

## Feedstock for compost and biochar

In principle, all organic residues can be converted to compost, if suitable conditions for biodegradation are provided. At first glance, this may suggest that composting and carbonization compete for the same resources.

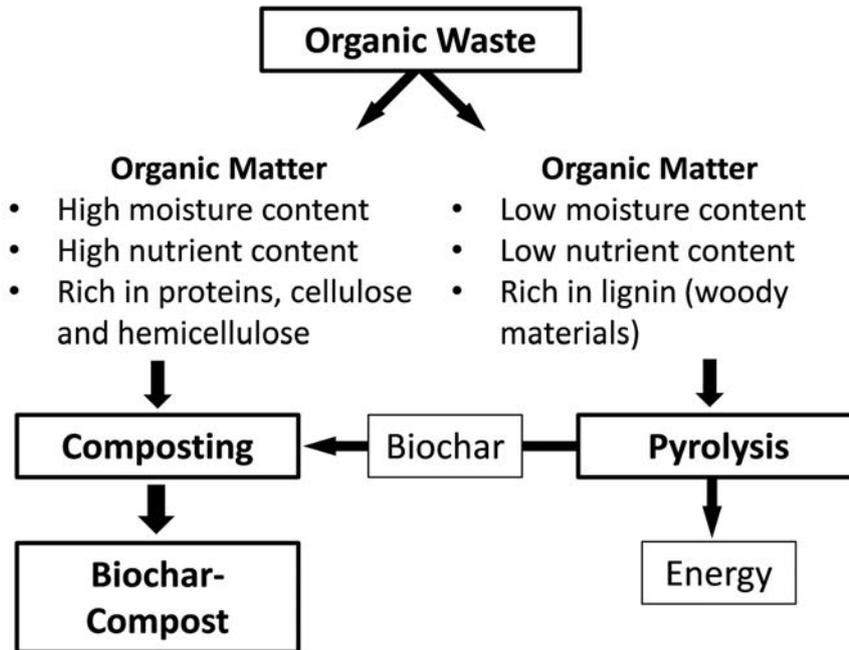
However, in practice, many of the materials that make good composts are rather difficult to convert into biochar, whereas some materials are ideal for biochar production but are less suitable for composting. For example, woody biomass has high lignin content, making it relatively resistant to microbial decomposition. The density of wood is relatively high, facilitating better heat transfer during pyrolysis. Materials such as straw, with a low bulk density, are insulating and thus hindering heat transfer. However, these materials are frequently used as bulking agent to improve aeration and adjust the C/N ratio. In contrast to ligneous materials, those high in protein, cellulose and hemicellulose decompose quickly, making them well suited for composting (e.g. green waste, vegetable and manures). Nitrogen is also a major factor as over 50 per cent of the dry bacterial cell mass is protein. An initial C/N ratio of 30 is desirable in composting mixes and the moisture content should be between 60 and 70 per cent (Gajalakshmi and Abbasi, 2008).

In contrast, the feedstock for pyrolysis should be as dry as possible. A high moisture content of the feedstock reduces the pyrolysis efficiency, as a significant proportion of the heat is consumed to dry the feedstock. The N content of the feedstock is not an obstacle for pyrolysis, but a considerable amount of N is lost at elevated temperatures (Gaskin et al, 2008) and the remaining N that mostly resides in the aromatic biochar structure is not plant available (Knicker and Skjemstad, 2000). Therefore, pyrolysis of N-rich materials will not recycle available N for plant nutrition, but rather sequester and volatilize most of the N. In addition, elements such as potassium (K) and chlorine (Cl) can cause severe damage to the pyrolysis equipment. Therefore, compost is ideally made from moist and nutrient-rich materials, whereas biochar is ideally made from dry porous materials with low nutrient contents (Figure 25.1).

## Compost and biochar

Biochar has been successfully used as a composting additive over a wide range of application rates, from small additions of 6 to 10 per cent of biochar (Jindo et al, 2012b; Theeba et al, 2012) up to 50 per cent of the composting mass (Dias et al, 2010). In all cases, biochar has proved to be effective in enhancing the process and the quality of the final composts (Table 25.1). Ishizaki and Okazaki (2004) observed reduced organic matter degradation when biochar was added at 30 per cent in the starting mixture. Dias et al (2010) also observed a sharp decrease in the levels of soluble organic compounds in a composting mixture prepared with a high dose of biochar (50 per cent), but these authors did not find any negative impact on the performance of the composting process. This is surprising as the biochar-C is rather persistent and an upper limit of application rates may be expected where composting performance declines.

The distinct properties of biochar, such as cation exchange capacity, porous structure, large surface area, water holding capacity, etc., are known to lead to complex interactions with the components of soil (Joseph et al, 2010). These interactions are also expected to occur within a composting matrix, characterized by a high nutrient and organic matter content and an active microbial biomass. Similar to the positive effects of biochar on soil biota, reviewed in Chapter 13, one of the main impacts of the addition of biochar is the stimulation of the microbial activity in the composting mix. Yoshizawa et al (2005, 2007) observed a proliferation of microorganisms and a colonization of the biochar's surfaces in the composting pile, which was attributed to the increased surface available for the microorganisms, the favorable moisture levels resulting from the increased water-holding capacity and the



**Figure 25.1** Organic waste separation into materials appropriate for pyrolysis and composting. The products are energy and a nutrient enriched biochar-compost

sorption of available C compounds which can be readily used by the microorganisms. Biochar addition can also reduce bulk density and facilitate aeration in the composting mix, not only as a bulking agent but also as a consequence of the micropores within the biochar structure which can facilitate micro-aeration. Hence, the addition of biochar may provide a habitat with favorable environmental conditions that stimulate microbial growth. Theeba et al (2012) measured an increase of total count of microorganisms in a poultry manure pile enriched with rice husk biochar. The increase in microbial activity can be reflected as an increase in pile temperature and CO<sub>2</sub> respiration rate during the process, as reported by Steiner et al (2011) during composting of poultry litter

with relatively high doses of pine wood biochar.

Table 25.1 exhibits an overview of the impacts of biochar on the abundance and activity of microorganisms during composting. Several authors have recently reported an enhanced microbial population of bacteria, fungi and actinomycetes and changes in the microbial community structure during composting of different organic wastes amended with biochar (Hua et al, 2011; Jindo et al, 2012a, 2012b; Theeba et al, 2012). The changes may influence important functions, such as organic matter degradation, mineralization and immobilization of nutrients (nutrient cycling), GHG emissions, the interaction with pollutants and the suppression of pathogens (Fischer and Glaser, 2012).

**Table 25.1** Overview of the main impacts of biochar as an additive to organic waste composting

Observed impacts	Dose	Feedstock	Biochar characteristics	Composting mix	Reference
<b>Impact on the microbiology of composting</b>					
				T of pyrolysis	
• Biochar served as matrix for the proliferation of composting microorganisms	10% (dw)	Bamboo		650°C	Yoshizawa et al (2005)
• Enhanced microbial population (fungi)	2% (vol)	Broad-leaved tree ( <i>Quercus serrata</i> )		400–600°C	Jindo et al (2012b)
• Enzymatic activities and pile temperature					
• Prolonged thermophilic phase	3, 6, 9% (fw)	Bamboo		600°C	Chen et al (2010)
• Accelerated composting process	4, 6% (fw)	Rice husk		550–600°C	Theeba et al (2012)
• Enhanced microbial population (fungi)					
• Moisture and nutrient retention					
• Higher temperature and CO <sub>2</sub> emissions	5, 20% (fw)	Pine wood		400°C	Steiner et al (2011)
• Change in microbial community structure	10% (dw)	Broad-leaved tree ( <i>Quercus serrata</i> )		400–600°C	Jindo et al (2012a)
• Enhanced microbial population (bacteria, fungi and actinomycetes) and community diversity	7% (fw)	Bamboo		600°C	Hua et al (2011)
• Change in the structure of microbial community	46% (fw)	Bamboo		650°C	Yoshizawa et al (2007)
<b>Impact on biochar properties</b>					
• Changes in biochar surface chemistry	0.1% (fw) <sup>#</sup>	Beech wood Beech and oak wood		550°C 700°C	Prost et al (2013)
• Reduction of Cu and Zn mobility	1, 3, 5, 7, 9% (fw)	Bamboo		600°C	Hua et al (2009)
• Reduction of Cu and Zn extractability	3, 6, 9% (fw)	Bamboo		600°C	Chen et al (2010)

**Table 25.1** continued

Observed impacts	Biochar characteristics		Composting mix	Reference
	Dose	Feedstock		
<b>Impact on C and N dynamics</b>				
• Reduced N losses	1, 3, 5, 7, 9% (fw)	Bamboo	Sewage sludge + rapeseed bagasse	Hua et al (2009)
• Reduced N losses	7% (fw)	Bamboo	Sewage sludge + rapeseed bagasse	Hua et al (2011)
• Reduced N losses	3, 6, 9% (fw)	Bamboo	Poultry manure + sawdust	Chen et al (2010)
• Reduced N losses and H <sub>2</sub> S emissions	5, 20% (fw)	Pine wood	Poultry litter	Steiner et al (2010)
• Enhanced organic matter processing	50% (fw)	Eucalyptus ( <i>Eucalyptus grandis</i> )	Poultry manure	Dias et al (2010)
• Enhanced organic matter processing	2% (vol)	Broad-leaved tree ( <i>Quercus serrata</i> )	Poultry manure + apple pomace + rice straw + oak bark	Jindo et al (2012b)
• Enhanced organic matter processing	3% (fw)	n.a.	Swine manure + wood chips + sawdust	Tu et al (2013)
• Reduced organic matter degradation at highest dose (30%). No significant effect on N losses	5, 10, 30% (fw)	Cattle manure	Compost from different animal manures + sawdust	Ishizaki and Okazaki (2004)
• Questioned the suitability of conventional maturation indices	5, 10% (fw)	Macadamia nut shell Hardwood shavings Poultry litter	Poultry manure + sawdust	Khan et al (2014)
<b>Impact on GHG emissions</b>				
• Reduction of CH <sub>4</sub> emissions	10% (dw)	Broad-leaved tree ( <i>Quercus serrata</i> )	Poultry manure + apple pomace + rice straw + rice bran	Sonoki et al (2013)
• Reduction of CH <sub>4</sub> emissions	10% (fw)	Oak	Organic fraction of municipal soil wastes + green waste	Vandecasteele et al (2013)

#: estimated dose, biochar in litter bags; fw: fresh weight; dw: dry weight; n.a.: not available.

## The effect of co-composting on biochar properties

It has recently been shown that co-composting biochar changes its properties and effects in soil considerably. Due to its chemical properties (Chapter 6), biochar's mineralization rate in the soil is significantly lower than that of the biomass it was produced from (Chapter 10). Biochars with low H/C ratios and low contents of volatile matter are not expected to undergo relevant degradation during the composting process despite the favorable conditions for biological degradation. Prost et al (2013) studied the changes in biochar surface chemistry before and after composting with farmyard manure and found that the degradation of the pyrogenic C (PyC) forms of biochar (Chapter 6) during composting was negligible. However, biochar prepared at relatively low temperatures may contain a labile fraction of organic matter that can be degraded during the process.

Despite their basic molecular structure that persists in composts and soils, biochar particles can undergo surface oxidation that alters their physicochemical properties (Chapter 9). This process, known as biochar aging or weathering, is often an abiotic process consisting of oxidation of C rings with a high density of  $\pi$ -electrons and free radicals (Joseph et al, 2010). These oxidative processes may be enhanced when biochar is added to a composting pile. The high temperatures reached during the process enhance abiotic oxidation and also stimulate biological oxidation (Chapter 13). These oxidative processes affect the surface chemistry of biochar and its physicochemical and sorptive properties (Hua et al, 2009, 2011). In a laboratory study in boxes, Hua et al (2009) co-composted 1, 3, 5, 7 or 9 per cent (w/w) of a bamboo biochar produced at 600°C with a surface of roughly 320m<sup>2</sup> g<sup>-1</sup> and a pH of 7.3 in a mixture of sewage sludge (90 per cent) and

rapeseed marc (10 per cent; likely fresh weights) with (initially) forced ventilation. The authors found an increase in the acidic groups on the bamboo biochar surface 1.7-fold after 42 days of composting; most of the change happened before day 28. In particular, the carboxylic groups increased 2.4-fold over 42 days of composting while the phenolic and lactonic groups increased 1.5-fold. Thus, the relative contribution of carboxylic groups to the total acidic groups was larger at the end of the composting (35 per cent) than it was initially (25 per cent). Such a change is likely beneficial since carboxylic groups have an important function for the biochar's ability to retain nutrients (Glaser et al, 2001). However, bamboo biochar might be different in several regards compared to biochar from other feedstock. Usually it has a much larger surface and adsorption capability (Hua et al, 2009; Chen et al, 2010). Prost et al (2013) observed a considerable increase in co-composted biochar's potential cation exchange capacity (CEC) and suggest that this increase was due to interception and sorption of organic leachates during the composting process. The surface area of biochar declined during composting due to the sorption of compost-derived organic materials, causing the clogging of micropores (Prost et al, 2013). Similar observations about pore clogging have been made during exposure in soil (Chapter 9).

In addition, Borchard et al (2012) found that the relationship between Cu(II) sorption and the biochar surface properties (CEC, specific surface area and aromaticity) suggested that sorption was largely determined by complexation with organic matter, rather than surface oxidation (Chapter 20). Chen et al (2010) also found a decreased extraction efficiency of Cu and Zn with increasing amounts of biochar in the composting mixture and it was attributed to the sorption capacity of biochar for dissolved organic mat-

ter during composting. Therefore, adsorption of organic matter onto biochar surfaces during composting may decrease or increase the adsorption of metals, depending on their affinity and adsorption behavior.

### The effect of co-composting on N-losses

The use of biochar as a compost additive would be especially interesting for organic materials with high nutrient loads. In particular, composting of N-rich materials such as animal excrements requires the addition of appropriate bulking agents (Bernal et al, 2009). Bulking agents are used to increase the porosity of the substrate, facilitate gas exchange and adjust the C/N ratio. These factors and the biodegradability of the C source (bulking agent) influences N-immobilization (Bernal et al, 2009), which is an important factor influencing the typically high N losses due to  $\text{NH}_3$  volatilization. Substantial loss of  $\text{NH}_3$  reduces the nutrient value of the compost product and may lead to environmental problems (Kithome et al, 1999) and a severe odor problem in the composting facility. Losses of up to 88 per cent of the original N content have been reported (Ogunwande et al, 2008). Although biochar typically mineralizes slowly and has a relatively high pH (another factor favoring  $\text{NH}_3$  losses), the use of biochar to reduce N losses was already recommended by Allen in 1846. Used as bulking agent, biochar could significantly reduce N losses by 64 per cent with only 9 per cent bamboo biochar addition to sewage sludge (Hua et al, 2009) or by 65 per cent in the case of poultry manure composting (Chen et al, 2010). Furthermore, it seems that N loss decreases more with increasing amounts of biochar. A 20 per cent biochar addition to poultry litter reduced the  $\text{NH}_3$  concentration in the emissions by up to 64 per cent and N losses by up to 52 per cent without negatively influencing the composting process

(Steiner et al, 2010, 2011). Biochar is capable of adsorbing  $\text{NH}_3$  (Iyobe et al, 2004). It may also reduce  $\text{NH}_3$  emissions by adsorbing precursors such as  $\text{NH}_4$ , urea and uric acid.

Prost et al (2013) described increased soluble N contents in the co-composted biochar, in the form of nitrate and total soluble N. The following mechanisms are thought to reduce N losses: (1) improved aeration (Chen et al, 2010; Steiner et al, 2010) and (2)  $\text{NH}_3$  adsorption (Taghizadeh-Toosi et al, 2011) and organic-N sorption (Prost et al, 2013), possibly facilitated by the adsorption of dissolved organic C during composting and/or development of acidic functional groups (Hua et al, 2009), either by surface oxidation (Cheng et al, 2006) or by adsorption of material carrying these groups.

Biochar additions to poultry litter also reduced  $\text{H}_2\text{S}$  emissions during composting (Steiner et al, 2010). Both  $\text{H}_2\text{S}$  and  $\text{NH}_3$  are easily noticed by their malodor. In general, minimal odor production is a sign of aerobic composting conditions (Brown et al, 2008). Biochar's capacity to adsorb excess moisture might be one of its key advantages as a bulking agent. Other frequently used bulking agents like straw and wood chips may have a negative influence on compost maturation (Adhikari et al, 2009) and N availability (Wang et al, 2004). Odor problems caused by  $\text{NH}_3$  loss can severely reduce the acceptance of a composting facility and this waste-management strategy as a whole by neighbors, a problem that should not be underestimated. Hence, improvement of this problem, e.g. by biochar addition during composting, may have merits beyond pure N retention in the end product.

### The effect of co-composting on compost mineralization

The mineralization rate of compost (stability, maturity) is another important quality crite-

tion (Jiménez and Garcia, 1989; Bernal et al, 2009). Thermo-chemical degradation confers a low mineralization rate to biochar (Chapter 10), which therefore does not need to undergo further biological degradation in order to become stable in the context of composting. Furthermore, biochar may influence the stabilization of the more labile C in composting mixes as discussed for soil organic C and plant litter (Chapter 16). Biochar additions to poultry manure could affect C dynamics, leading to a more rapid composting process (Dias et al, 2010). These authors reported a decrease in soluble organic matter due to the sorption of soluble compounds on biochar surfaces, especially during the maturation phase.

Jindo et al (2012b) studied the impact of a small addition of biochar (2% v/v) on composting by means of  $^{13}\text{C}$ -NMR; they observed the incorporation of aromatic groups into the non-biochar composting matrix, probably as a consequence of the presence of biochar particles intimately mixed with the composting matrix. The presence of biochar may therefore represent a limitation for the assessment of compost maturity by conventional indices. Biochar incorporation can interfere with C dynamics during composting, affecting parameters traditionally used as maturity indices such as the C/N ratio, the decline in water soluble compounds and the increase of so-called humification indices (Dias et al, 2010; Tu et al, 2013). Khan et al (2014) recently questioned the suitability of the C/N ratio as a maturity index in biochar amended composts since the incorporation of recalcitrant C would increase the levels of C/N ratio above the conventional values accepted for mature composts, even in materials that have achieved an adequate degree of compost stability. However, the amount of biochar-C can be accounted for by analytical methods distinguishing between biochar-C and non-biochar-C. Calculating the relative abundance of

biochar-C would require knowledge about the amount of biochar-C added to the compost and an estimation of labile C lost during the composting process. Thies and Rillig (2009) also highlighted the interference of biochar in the determination of microbial parameters in biochar amended composts and proposed the use of spiking assays as internal standards for assessing the interference of biochar in the methodology. For this reason, the interpretation and suitability of conventional maturity indices need to be re-evaluated in biochar amended composts to avoid the interference of biochar.

### **The effect of co-composting on greenhouse gas emissions**

Management options avoiding anoxic conditions and emissions of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  involve composting, combustion, gasification and pyrolysis of biomass. During composting, the microbes use C as an energy source and consequently the oxidation reactions involve the release of C as  $\text{CO}_2$ . When animal manures are composted, the C losses are between 50 and 70 per cent of the original C contained in the manure (Bernal et al, 2009). These  $\text{CO}_2$  emissions are not considered as additional greenhouse gas (GHG) emissions as the feedstocks are considered to be part of the short-term C cycle. Therefore, GHG emissions reductions are mainly generated through  $\text{CH}_4$  and  $\text{N}_2\text{O}$  avoidance, when the organic wastes would otherwise be deposited in landfills or lagoons (Brown et al, 2008). However, Boldrin et al (2009) assessed the GHG emissions and savings from compost production and utilization and also counted C stored as relatively stable SOM. Compost mixtures consist of readily degradable, slowly degradable and relatively low degradable organic matter. Only the least degradable organic matter had a turnover time of 100 to 1000 years (Boldrin et al, 2009). The C remaining

in soils for 100 years has been estimated to be 2–10 per cent of the original C in the compost mix by the same authors.

Despite the question of whether the least degradable C compounds are considered as sequestered C or not, accurate GHG accounting is relatively complicated for composting and compost utilization. In addition to fossil fuel requirements for transport, chipping and turning, emissions of CH<sub>4</sub> and N<sub>2</sub>O need to be monitored (Brown et al, 2008). Therefore the emission reductions due to composting can vary between significant savings and a net load of CO<sub>2</sub>-equivalents (Boldrin et al, 2009).

Biochar C sequestration is much easier quantified in comparison to GHG emissions associated with composting and compost utilization. A further potential of biochar in composting systems is the reduction of CH<sub>4</sub> and N<sub>2</sub>O emissions during composting. In soils, biochar addition has been associated with a significant reduction in N<sub>2</sub>O formation (Chapter 17). Adding biochar (3% w/w) to pig manure, wood chips and sawdust compost mixture could reduce N<sub>2</sub>O emissions by 26 per cent (Wang et al, 2013).

Biochar addition has successfully reduced the emission of CH<sub>4</sub> from poultry manure composting piles (Sonoki et al, 2013). The

authors measured a decrease of methanogenic bacteria and an increase of methanotrophs in the composting pile containing biochar. The addition of biochar as bulking agent would improve the physical properties of the composting matrix facilitating gas diffusion in the pile and improving aeration, avoiding the formation of anaerobic spots (Sonoki et al, 2013). Similar effects were observed in soils amended with biochar, where a decline in soil methanogenic activity and an increase in CH<sub>4</sub> oxidation was reported (Spokas, 2013). Vandecasteele et al (2013) also reported a reduction of CH<sub>4</sub> emissions in a full scale composting plant adding 10 per cent biochar to a mixture of the organic fraction of municipal solid wastes and green wastes.

In anaerobic digesters, the retention time of biodegradable organic matter is based on economic optimization and therefore often less than required for complete CH<sub>4</sub> recovery (Brown et al, 2008). Consequently, the CH<sub>4</sub>-slip is a considerable problem. As biochar has the ability to improve aeration if used as bulking agent, it might also be capable of reducing the CH<sub>4</sub>-slip when added to previously anaerobically digested material.

### Box 25.1 Biochar-compost production

Biochar is produced from paper sludge, grain husks and yard waste. The pyrolysis unit (Figure 25.2) carbonizes approximately 3t feedstock (70 per cent dry matter) per day in continuous flow, producing 1Mg of biochar per day. The excess heat is used to pre-dry the feedstock. Currently, the biochar is mainly purchased by farmers and used to improve liquid and solid manures. Biochar additions to manures should reduce N losses and malodor of the manures. Some farmers use the biochar for their compost production and have had good experience with an addition of 5 per cent. A proportion of the biochar is used to produce a biochar-compost. Thirty per cent (by volume) biochar is added to the initial compost mix, in order to produce this biochar-compost. The company also develops a slow release N fertilizer based on biochar. Biochar is sold at €600 (including tax) per t<sup>1</sup> or packed in Big Bags for €200 per 200kg.



**Figure 25.2** *Slow pyrolysis unit at a compost production facility in Austria. The company is producing biochar and various soil products with compost, biochar and inorganic amendments*

## Biochar as *sphagnum* peat substitute in growing media

### Properties and importance of horticultural peat

Modern horticulture requires dependable, quality-assured growing media. *Sphagnum* peat has been the most important constituent of growing media for several decades because of its properties that make it ideal for this purpose. Indeed *Sphagnum* peat, after it has been fertilized, is the sole constituent of many growing media. The favorable characteristics of horticultural peat are its large water holding capacity (WHC), its high air capacity at 100 per cent WHC (container capacity), the homogeneity and availability of the product, the absence of weed seeds and pathogens, its low bulk density, low pH, low microbiological activity, low nutrient contents and its low salt content (Reinhofer et al, 2004; Schmilewski, 2008; Michel, 2010). The low pH and nutrient contents enable adjustments of pH (liming) and nutrients (fertilization) as desired to meet the plant specific requirements.

Germany and Canada account for over half of the global horticultural peat extrac-

tion. Worldwide, Germany is the largest manufacturer of growing media for the professional and hobby markets (Schmilewski, 2008). According to the United States Geological Survey Mineral Resource Program, the world's annual horticultural peat consumption was on average 12.1Mt in the years 2006 to 2010 (Indexmundi, 2013).

### Environmental concerns

Peat bogs are valuable habitats and important C stocks. They also provide crucial functions in their local environment, e.g. regulation of the local water quality and water regime or flood protection (Alexander et al, 2008). Although peat is thousands of years old, it decomposes quickly once removed from its protective anoxic and acidic environment. Drained, aerated, limed and fertilized, extracted peat can decay to CO<sub>2</sub> within years and is thus a source of GHG. Although the area of peatland under extraction is small compared to the total peatlands in Canada, the extraction causes GHG emissions and it would take approxi-

mately 2000 years to restore the original SOC pool after successful restoration. A life cycle analysis estimated that the peat extraction in Canada emitted 0.54 10<sup>6</sup>t of GHG in 1990, increasing to 0.89 10<sup>6</sup>t in 2000 (Cleary et al, 2005). Thus, the conservation of peatlands is important for both adapting to and mitigating climate change.

The ecological value of peat bogs is increasingly recognized and they are important for species diversity and conservation. In many countries peat lands belong to the most threatened habitats with unique species diversity. Therefore, the UK government recently introduced ambitious targets for peat replacement (Alexander et al, 2008). Peat is not considered to be a renewable resource and is excluded from the Ecolabel for growing media and soil improvers by the European Commission. Consumer and corporate awareness of environmental, social and sustainability issues has grown markedly in recent years and is likely to continue to exert a strong influence on retail markets for peat in the future. It is likely that peat use in growing media will be further restricted, thereby increasing the importance of suitable substitutes (Rivière and Caron, 2001).

### **Peat substitutes other than biochar**

Peat containing products are used in gardens and for the production of potted plants. Most of the alternative materials (such as compost) are used in gardens as soil improvers (Alexander et al, 2008). Many alternative media are still inconsistent in their composition; N immobilization properties potential and the origin of the materials is frequently unknown and there are problems with structural stability and water holding capacity (Reinhofer et al, 2004). So far no single material has been identified to substitute peat

entirely (Reinhofer et al, 2004; Schmilewski, 2008; Michel, 2010).

Wettability is a particularly important property of horticultural growing media because it determines its ability to re-wet itself once it has dried out (Michel, 2010). The poor rewettability is one of the negative characteristics of peat (Alexander et al, 2008). Adding composts, wood fibre products, bark and composted bark, or coir to peat-based growing media can improve the rewettability and air capacity or change other properties if desired. The disadvantage of compost is its variability, relatively high pH, high nutrient (potassium, K) content and bulk density which increases transportation and handling costs. Wood fibres have a low shrinkage value and good rewettability and are free of seeds and pathogens, but can cause N immobilization and have a low water holding capacity. Composted bark can increase the air capacity, drainage, CEC and pH buffering. However, pH and salt content of composted bark can be too high. Coir (i.e. coconut fibre) is expensive due to long routes of transport. Many of its physical and chemical characteristics can vary; overall, it has good rewettability, extremely high air capacity, but a low WHC (Schmilewski, 2008).

The production of some of these alternative materials requires additional chemical and physical treatments (Reinhofer et al, 2004). Elevated temperature and pressure is used to split woody biomass into fibres. Chemical treatments to reduce mineralizability and additions of N should prevent N immobilization by suppressing microbial degradation or supplying additional N, respectively. Inorganic products such as mineral wool, perlite, vermiculite, sand, clay and clay products may also be used. Environmental constraints also influence the manufacturing, transport and, after use, elimination of these substrates.

### The possible role of biochar in growing media

Physical properties such as aeration and water retention are essential for growing media (Table 25.2). As outlined above, materials that have favorable attributes in both of these respects are rare (Michel, 2010). Depending

mainly on the degree of decomposition, peat properties can vary significantly. According to Michel (2010) only certain white *Sphagnum* peat possesses entirely suitable aeration and water retention when used as sole media bulk component; therefore many commercially available growing media contain a blend of different materials.

**Table 25.2** *The most important properties of peat and wood biochar as growing media*

Properties	Peat	Biochar
Homogeneous quality	Available	Available
Nutrient content/adjustability	Low/adjustable	Low/adjustable
pH	Low	Mostly neutral to high
Water holding capacity <sup>1</sup>	High	Medium
Air capacity <sup>1</sup>	Medium	High
Balanced water and air capacity <sup>1</sup>	Good	Good
Weeds and pathogens	Mainly free	Free
Structural stability	Medium	Exceptionally high
Bulk density <sup>1</sup>	Low	Low
Texture <sup>1</sup>	Uniform	Uniform
Rewettability	Poor <sup>2</sup>	Good
Disease suppressive properties	Neutral	Evidence <sup>3</sup>
Availability (technical)	High	Currently low
GHG emissions	High	Depending on feedstock and production
Environmental damages	High	Depending on feedstock and production

<sup>1</sup> depending on particle size distribution (level of decomposition in peat)

<sup>2</sup> Alexander et al (2008)

<sup>3</sup> at low doses of 1 to 5 per cent (Elad et al, 2010)

Depending on production conditions, feedstock and particle size, the properties of biochar can also vary significantly (Gaskin et al, 2008; Novak et al, 2009). Biochars can likely be produced that possess the properties needed to replace or complement peat (Chapter 19). In addition, biochar has an exceptional structural stability (Tian et al, 2012) and mineralizes very slowly (Kuzyakov et al, 2009) compared to peat. Biochar products may also increase the CEC and buffering capacity of a growing medium (Doydora et al, 2011) and indications exist that biochars can be produced that have a good rewettability.

Due to the high temperatures during pyrolysis, the biochar is free of pathogens and weeds. In addition, recent research has shown that adding biochar (even in low doses, 1–5 per cent w/w) to growing media can influence the root-associated bacterial community structure (Kolton et al, 2011) and promote systemic resistance of plants to several prominent foliar pathogens (Elad et al, 2010). This remarkable benefit may have important horticultural consequences, in particular for organic growers. The low mineralizability of biochar does in most cases not require any further treatment to minimize microbial N immobilization, in comparison to wood fibres that are often chemically treated to reduce mineralizability (Reinhofer et al, 2004).

Not all biochars may be suitable as peat substitutes. Some feedstocks (e.g. poultry litter) are rich in minerals and produce biochars with high pH values and salt content, i.e. they would cause osmotic stress in plants when used in larger amounts. However, biochar produced from pure wood has a very low salt and nutrient content (Gaskin et al, 2008). Biochar produced from wood chips was used as growing media by Steiner and Harttung

(2014). The EC was similar to that of unfertilized peat ( $612\mu\text{S cm}^{-1}$  and  $633\mu\text{S cm}^{-1}$  respectively). When mixed with peat, the blend could contain up to 80 per cent biochar without raising the pH above 7. The growth of a miniature sunflower was similar in perlite, clay granules (Seramis), peat and peat-biochar mixtures (Steiner and Harttung, 2014). Plant-available nutrients may be removed by rinsing with water, while the relatively high pH of biochar can be reduced by acid treatments (Doydora et al, 2011). Compost is frequently acidified with sulphur. Peat-based growing media are acidic and limed to adjust the pH. If biochar is used as an additive to peat it could replace lime. Elevated K contents may be displaced from ion exchange sites by saturating with calcium (Ca) and magnesium (Mg) as done to produce buffered coir pith (Schmilewski, 2008).

In contrast to mined peat, biochar may be obtained as by-product from energy generation using renewable resources (Chapters 26 and 29). Substituting peat with biochar would in most cases not only avoid GHG emissions associated with peat decay, but would also sequester the C contained in the biochar. The potential GHG emissions reductions depend largely on the feedstock and technology used for biochar production (Box 25.2). Not including potential emissions reductions from fossil fuel substitution (e.g. heating greenhouses with pyrolysis gases) the replacement of peat with biochar could avoid  $4.5\text{t CO}_2\text{e t}^{-1}$  of peat substituted (Steiner and Harttung, 2014). Assessing the emission from the off-site use (decay) of horticultural peat is only done by the United Kingdom, but applies for all European countries that produce horticultural peat (Barthelmes et al, 2009).

### Box 25.2 BlackCarbon case study: GHG emissions reductions by peat substitution with biochar

Pyrolysis of biomass is the key technology of BlackCarbon A/S in Denmark. The BC-300 unit converts wooden crates to energy (heat and power, CHP) and biochar. The annual production of each BC-300 unit is 715MW of heat, 227.5MW of electrical power and 162.5t of biochar. BlackCarbon is developing biochar utilization pathways to maximize GHG emission reductions. One such application is the substitution of peat in growing media. The estimated annual GHG reductions per BC-300 unit range from 694 (without fossil fuel substitution) to 1032t CO<sub>2</sub>e, with an estimated average of 915t CO<sub>2</sub>e.

## Recommendations for future research

Several studies report a reduction in gaseous N losses (NH<sub>3</sub> and N<sub>2</sub>O) if manures are co-composted with biochar. Reduced N<sub>2</sub>O emissions were explained by an increased abundance of N<sub>2</sub>O-consuming bacteria (Wang et al, 2013). More research is needed to discern other mechanisms of N conservation and the biochar properties responsible for an alteration of the microbial community. The microbial composition and activity in biochar-compost and the potential alteration of physical and chemical properties of biochar during the composting process are interesting research topics. However, due to difficulties cleaning and separating biochar particles from the compost, the biochar alterations due to co-composting are not easy to study.

The use of biochar as a bulking agent could be optimized by identifying the ideal biochar particle size, feedstock and produc-

tion conditions for a particular product. Similarly, these two properties (particle size and feedstock) are most likely to influence the suitability of biochar as a peat substitute or additive, but only preliminary research has been conducted. Biochar has mainly been studied in soil mixtures and rarely in very high concentrations (> 50 per cent). However, biochar might be used in concentrations > 50 per cent, if used in growing media formulations. The physical properties of pure biochar need to be assessed in order to evaluate its utility as a growing medium. If biochar is used as a growing medium, the potential after-use would be worth investigation. The growing media might be pyrolysed (sterilized) again, used in compost or as a soil amendment. Potential adverse effects, such as the leaching of biochar dust, should also be taken into consideration.

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# The Use of Biochar in Composting

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February 2015; for more information, please see [www.biochar-international.org](http://www.biochar-international.org).

Reviewed by Mariluz Cayuela and M.A. Sánchez-Monedero

Both compost and biochar production are methods to utilize and recycle organic wastes. This paper provides information on the use of biochar in composting and highlights the potential benefits, which include the ability to accelerate the process of composting and reduce the loss of nutrients, among others.

- **Composting** is a technology for the treatment and disposal of biodegradable waste. Almost any food waste, industrial food waste, and sewage sludge can be composted. The main advantages of successful composting are a decrease in waste volume; the elimination of most organic toxic compounds, pathogens and pests (potentially present in the original waste); the transformation of organic matter; and associated nutrients into a product that acts as a slow release fertilizer (referred to as ‘stabilization’ in the compost literature).
- **Biochar** is a solid material obtained from the thermochemical conversion of biomass in an oxygen-limited environment. It has a greater persistence than the uncharred precursor biomass. Biochar can be used as a product itself or as an ingredient within a blended product, with a range of potential applications as an agent for soil improvement. When the right biochar is added to the right soil, biochar can, among other benefits, improve resource use efficiency, remediate and/or protect soils against particular environmental pollution, and become an avenue for greenhouse gas (GHG) mitigation<sup>1</sup>.



## Compost and Biochar: In Competition for Feedstocks?

Although both biochar and compost use organic wastes as feedstocks, the two operations do not have to be an either/or option; instead, they can be combined for synergistic production and utilization. For example, many materials that make good compost, such as food waste and wet manures, are not easily used for biochar production since a large amount of heat would be needed to dry the materials prior to producing biochar. Ideal feedstocks for composting have from 60 – 70% moisture, high nutrient levels, and low lignin content<sup>2</sup>. Ideal feedstocks for biochar have 10 – 20% moisture and high lignin content, such as field residues or woody biomass.

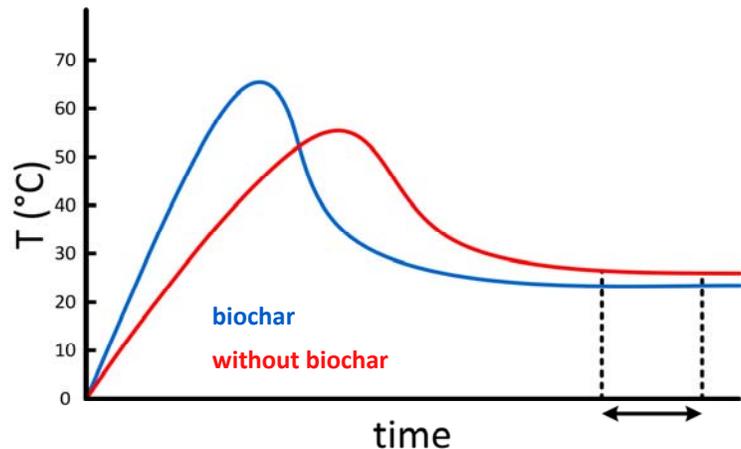
## Biochar Benefits to the Composting Processes

Based on current findings, the benefits of adding biochar to the composting process may include shorter compost times; reduced rates of GHG emissions (methane, CH<sub>4</sub> and nitrous oxide, N<sub>2</sub>O); reduced ammonia (NH<sub>3</sub>) losses; the ability to serve as a bulking agent for compost; and reduced odor. For the biochar material itself, undergoing composting helps to charge the biochar with nutrients without breaking down the biochar substance in the process.

### Adding biochar can enhance the composting process through:

- Accelerating the composting process
- Reducing GHG emissions
- Reducing ammonia loss
- Serving as a bulking agent for the compost
- Reducing odor

A wide range of biochar application rates to compost have been tested, from 5 – 10% to 50% (mass basis) or higher<sup>3,4</sup>. A biochar dose higher than 20 – 30% (mass basis) is not recommended as an excessive amount relative to the composting material could interfere with biodegradation. At adequate doses, biochar has been found to accelerate the composting process—mainly through improving the homogeneity and structure of the mixture and stimulating microbial activity in the composting mix. This increased activity translates to increased temperatures and a shorter overall time requirement for compost development<sup>5</sup>. This may have important economic implications since accelerated composting is a desirable effect.



Biochar increases the temperature in a compost process, accelerating the time needed for material decomposition<sup>4,6,7</sup>

One challenge to compost operations is the loss of nutrients and the emission of GHGs during the composting process—specifically CH<sub>4</sub> and N<sub>2</sub>O. Adding biochar at 3% (mass basis) to a pig manure, wood chips, and sawdust compost mixture was found to reduce N<sub>2</sub>O emissions by 26%<sup>8</sup>. For methane, a recent study found that the addition of biochar reduced CH<sub>4</sub> emissions from poultry manure composting piles<sup>9</sup>. However, other studies have shown that biochar has no impact on the overall GHG emissions since these emissions were offset by the enhanced microbial activity on the composting mix containing biochar<sup>10</sup>. In those cases there may not be a net impact on GHG emissions.

The porous nature of biochar can reduce the bulk density of compost and facilitate aeration in the composting mix. For compost feedstocks that are high in nitrogen (N), such as animal manures, biochar offers the opportunity to reduce the overall N loss over the process, especially that of NH<sub>3</sub>. The odor problems caused by NH<sub>3</sub> loss during composting are not only unpleasant, but can reduce the acceptance of a composting facility by a community. A 20% (mass basis) biochar addition to poultry litter reduced the NH<sub>3</sub> concentration in the emissions by up to 64% and N losses by up to 52% without negatively influencing the composting process<sup>11, 12</sup>.

A common problem during the composting of manure is the formation of big lumps upon drying that stops the process. The addition of 3% (mass basis) of wood biochar to poultry manure co-composted with straw was able to significantly reduce the formation of big lumps in the pile, improving the composting process and the overall structure of the final compost<sup>13</sup>.

## Biochar and Compost: Looking Ahead for Wider Commercial Use

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Although initial publications show measurable benefits on the impact of biochar on composting, the number of studies is still very limited. Many of the traditional indices used for evaluating the quality ('stability') of compost (e.g., the carbon/nitrogen, C/N, ratio) are not valid when biochar is included in the mixture (6), since biochars have very different properties than the rest of the composting material (e.g. high C/N ratio, which will not decrease during the composting process in contrast to the remaining organic material) and may even influence compost quality assays if controlled for biochar (e.g., water-soluble C may adsorb to biochar). Establishing compost quality indices that take the benefits of biochar into account could help increase the commercial use of this activity.

At this time there is not a significant industry for compost amended with biochar, even though many current biochar producers sell a biochar/compost blend. The 2013 *IBI State of the Industry Report* found that compost was the most common additive to biochar when biochar was sold as part of a blend<sup>14</sup>. There are a few companies that are actively taking a leading role in commercializing biochar-amended compost blends by producing biochar onsite and utilizing those feedstocks for biochar production which would be less ideal for composting.

Increasing the use of biochar in compost operations requires education on the benefits of biochar to producers, not only on emissions and odor reductions, but also on the potential economic benefits of accelerated composting time to offset the additional price of producing/purchasing biochar.

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From: Judith Redmond <judith@fullbellyfarm.com>  
Sent: Monday, February 08, 2016 9:33 AM  
To: CDFA Environmental Stewardship@CDFA  
Subject: Agronomic Rates of Compost Application...

Dear Authors of “Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program”

I think there is a typo in the table in Box 1, pg 8.

I’m looking at Example 2, but I think the same mistake exists in Example 1.

In the final bullet, you have calculated that 15.6 tons of N are applied per acre. This calculation results from multiplying tons by lb/ton. Shouldn’t it read 15.6 lbs N applied per acre?

Please advise.

Judith Redmond  
Co-Owner Full Belly Farm  
530-796-2214

From: Peter Alpert <palpert@bio.umass.edu>  
Sent: Wednesday, February 03, 2016 11:23 AM  
To: CDFA Environmental Stewardship@CDFA  
Subject: Comment on proposed application of compost to California croplands and rangelands  
Attachments: alpert10rangelands.pdf; alpert10rangelandsonlinematerials.pdf

Dear Sirs,

I would like to offer a public comment on the draft document, "Agronomic rates of compost application for California croplands and rangelands to support a CDFA Healthy Soils Incentives Program". I am a professor in the Biology Department at the University of Massachusetts -- Amherst specializing in plant ecology and have worked for over a decade in coastal dune and grassland systems in central coastal California. I also served on the federal Invasive Species Advisory Committee from 2006-2012.

I feel that the document clearly states the key ecological issues involved in application of compost to rangelands with the possible exceptions of introduction of seeds and microbes and effects of burial of plants or seeds by applications. However, the basis for setting thresholds for application to rangelands seems weak, both because of the very limited amount of relevant study and because the use of plant diversity as a criterion is flawed if diversity includes introduced species and minimally sensitive if diversity is measured by number of species instead of the relative abundances of species, which I could not tell from the document.

In central coastal, semi-natural and natural Californian grasslands where annual grasses are the most abundant introduced plants, there is probably a negative correlation between total plant productivity and relative abundance of native species; if the goal of composting is to increase productivity, then the goals of composting and maintaining native biodiversity may inherently counter each other. Application of organic carbon with almost no nitrogen, such as application of sawdust, is likely to have no effect or a positive effect on native plants, as documented in the attached, peer-reviewed review paper, but that does not seem to be within the proposed policy. My recommendation on ecological grounds would be to either leave rangelands out of the new program or to restrict applications to rangelands to compost with almost no nitrogen.

Sincerely,  
Peter Alpert

# Amending Invasion With Carbon: After Fifteen Years, a Partial Success

By Peter Alpert

One of the factors most strongly associated with high abundances of human-introduced plant species in natural and seminatural systems in North America and Europe is high availability of soil nutrients, particularly nitrogen (N).<sup>1,2</sup> For example, areas within grasslands or shrublands that have soils with higher levels of inorganic N tend to be more invaded than areas with lower levels; adding nutrients to grasslands experimentally often increases the absolute and relative abundance of introduced plants whose propagules are present; and the common introduced species in partly invaded grasslands often increase their growth more in response to high N than do the common native species when plants are grown separately.

This suggests that reducing the availability of N in the soil might slow or reverse the spread of introduced plants in grasslands.<sup>3,4</sup> One relatively quick and specific, if probably temporary, way to do this is to promote the uptake of N by soil microbes by amending the soil with a metabolic substrate for bacteria or fungi that is very low in N, such as sucrose, cellulose, lignin, or most types of sawdust or other fine residue from the cutting or milling of wood. Since these substances are all high in carbon (C), the metabolic energy from them being derived largely from splitting C-C bonds, this method has been commonly called “carbon addition,” not to be confused with the use of activated, pure C to adsorb and reduce availability of organic compounds such as allelochemicals.

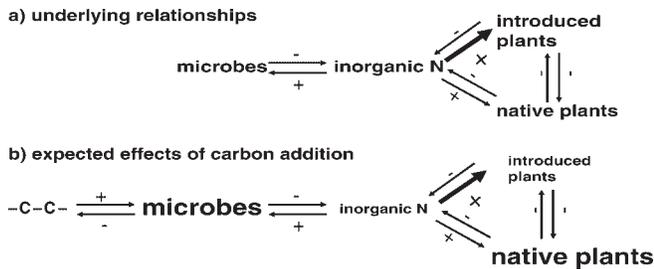
The generally accepted mechanism by which C addition to soil might reduce the abundance of introduced plants and promote the abundance of native ones involves not only competition for inorganic N between plants and microbes, but also competition between introduced and native plants as well as contrasting responses of introduced and native plants to availability of N (Fig. 1a). Reduction in supply of inorganic N due to uptake by microbes is expected to have a direct negative effect on natives. However, reducing N is also expected to have an indirect positive effect on natives, by negatively affecting introduced plants and reducing their negative effects on natives. If the response of introduced

plants to N availability is greater than that of natives, the indirect positive effect on natives of reducing N could be greater than the direct negative effect. Adding C and fueling higher metabolism of microbes should then decrease inorganic N, decrease growth of introduced plants, and increase growth of natives (Fig. 1b).<sup>5</sup>

Perhaps the least certain assumption of this model for the mechanism of C addition is that availability of N will have a stronger effect on introduced than on native species. This assumption is consistent with the association between high N and invasion. However, the assumption has not always been found to be true, especially when species are grown separately. The model also fails to account for possible direct effects of microbes on plants, such as through mycorrhizae. For example, if a mycorrhizal fungus translocates N to its plant host, this could alleviate decrease in availability of N in the soil.

Testing the efficacy of C addition as a countermeasure to the spread of introduced species of plants is of high societal interest because of the widespread elevation of N availability due to agriculture and pollution and of the potential advantages of C addition over other methods of control. In some regions N deposition following the burning of fossil fuels has caused major changes in vegetation that might be reversed, if desired, by reducing N availability back to levels before human-caused deposition. Much of the area available for restoration of some types of natural grasslands that have been made rare is on former agricultural fields, where N availability is typically elevated. These and other widespread, human-caused increases in N availability increase the likelihood that many introduced species have become widespread in part because they respond positively and strongly to high N.

If effective, C addition could have important practical advantages over other methods for the control of invasive species. Because C addition is expected to control whole sets of introduced species, it could replace multiple and sometimes conflicting efforts to control individual introduced species. It could also replace the use of herbicides. If specific



**Figure 1. a,** Relationships underlying the hypothesized mechanism by which carbon addition might counter the spread of introduced species of plants in a terrestrial habitat. The thicker arrow is intended to show that effects of high availability of nitrogen (N) must be greater on introduced than on native species. **b,** Expected effects of amending the soil with a low-N metabolic substrate for soil bacteria or fungi. Changes in type size between the two panels show the expected increases or decreases in abundances of microbes, N, and plants.

techniques such as twice-yearly applications of sawdust are effective, this could greatly reduce the cost and controversy over control.

Experiments with C addition are also of theoretical interest. A leading hypothesis to explain the spread of introduced species, or its lack, is biological resistance, that competition from native species on the same trophic level and predation by native generalists on the next trophic level reduce the growth and survival of introduced species. A main reason to expect that native species will outcompete introduced ones is that the natives are locally adapted. Recent, marked changes in environmental conditions are likely to change selection pressures and reduce the difference in fitness between native and introduced competitors. This could shift the balance of competition in favor of the latter, especially if conditions have changed to be more like those in their native range. Human-caused rise in N availability may be an example of such change, and, if reversing the rise reverses the spread of introduced species, this would support the hypothesis.

Published research on C addition to control introduced plants dates at least to 1994,<sup>6</sup> although the technique was used earlier as a way to investigate nutrient relations<sup>7</sup> or increase rates of succession.<sup>8</sup> The purpose of this review is to catalog the results of these last two decades or so of research on C addition and use them to address three questions: 1) Does C addition reduce N availability to plants? 2) Does it reduce the abundance of introduced plants? 3) Does it increase the abundance of natives?

## Findings

### Range of Studies

Most of the 55 relevant studies found (see online materials [www.srmjournals.org] for details of search methods and tabulations of studies) focused on a few major vegetation types and regions. A plurality of studies had been conducted on prairies of central North America (15 studies exclusively on prairies). Relatively large numbers of studies had also

been conducted on mixed shrublands and grasslands of the intermontane United States (9), grasslands and heathlands in Europe that seem likely to be human-caused (8), and Mediterranean-type grasslands and shrublands in coastal California and southern Australia (7).

Many but not most studies added C exclusively as sucrose (21). Nine studies added exclusively sawdust, three added only a blend of sucrose and sawdust, and five compared separate additions of sucrose and sawdust. Only a few studies identified which types of trees sawdust was from. Other forms of C added were dextrose, glucose, cellulose, straw, bark, wood chips or fragments, urban wood waste, and leaf litter. Amounts of C added varied by two orders of magnitude, from 0.02 to 6.5 pounds C/yard<sup>2</sup>/year (0.01–2.5 kg C/m<sup>2</sup>/year), with from 1 to 10 applications per year. Most studies (31) added C only once or over only 1 year, but at least nine continued applications for 3 years or more, and one for 10 years. Several studies continued measurements of effects for 1 year after applications ended, and one for 2 years after.

### Effects of C Addition on N Availability and Other Soil Characteristics

Nearly all studies found that C addition reduced likely availability of N to plants (online materials, Table 1). The most common measurements of plant-available N were concentrations of NO<sub>3</sub> and NH<sub>4</sub> in extracts of soils in KCl. Standing concentration of NO<sub>3</sub> was often at least 50% less in soils amended with C (online materials, Table 2). Concentration of NH<sub>4</sub> tended to show less effect of C addition and, unlike NO<sub>3</sub>, was sometimes found to be higher in plots with added C than in those without (e.g., Corbin and D'Antonio 2004; Iannone and Galatowitsch 2008; see online materials for references cited under Findings). However, no study reported higher total concentrations of inorganic N in amended than in control soils. The six studies that reported no significant effect of C addition on N availability each measured either only standing concentration of inorganic N or only net N mineralization; they shared no obvious similarities in experimental system or type or amount of addition. Other observed effects of C addition on N availability included decreases in amount of inorganic N absorbed on resin and in N mineralization, and increases in microbial C and N, in density of microbes, and in production of CO<sub>2</sub> in soil. All these effects are consistent with an increase in immobilization of N by soil microbes, the expected effect of C addition (Fig. 1).

These effects of C addition on soil N sometimes increased with repeated applications and generally decreased within months after applications stopped. For example, Young et al. (1996) saw little effect of sucrose on standing concentration of inorganic N until the second year of applications, and Michelson et al. (1999) found a positive effect on microbial C after 5 but not 2 years of applications. At least six studies documented decrease in effects of C on N within

1 year after applications stopped (online materials, Table 2), although at least one (Blumenthal et al. 2003) reported no decrease within 2 years. There was little evidence that effects decreased as long as applications continued. Only one field study appears to have found that C addition reduced availability of phosphorus (Michelson et al. 1999).

Additions of C were also observed to affect the composition of soil biota. Biederman et al. (2008) reported increases in the abundance of bacterivorous and several other functional groups of nematodes following application of wood waste, while Nieminen (2009) found a decrease in abundance of bacterivorous nematodes in soil amended with sucrose, both opposite to expected changes in the relative abundance of bacteria and fungi with availability of lignin versus labile C. A detailed laboratory study by van der Wal et al. (2006) suggested that microbial decomposition of sawdust began with digestion of cellulose and hemicellulose on the surface of wood particles, and that additions of cellulose and sawdust favored similar soil biota, and glucose a different biota. Larger effects of C addition on nutrient cycling in the soil are further suggested by at least one finding that addition decreased decomposition of litter (Hunt et al. 1988).

A number of studies reported that C addition increased soil moisture (not noted in online materials, Table 2). This could be an effect of decrease in plant mass and hence uptake of water, of increase in organic matter in the soil, or of decrease in evaporation from the soil surface in the case of surface applications of sawdust or other wood products. Tilling in such applications can produce different results than applying the same applications to the surface (e.g., Biederman and Whisenant 2009). There does not yet seem to be enough information to indicate whether any of these differences are consistent, but they could suggest that effects of additions of sawdust may be due at least in part to changes in availabilities of resources other than N.

#### *Effects of C Addition on Introduced Plants*

Effects of C addition on introduced species have been less consistent than those on soil N, but still mainly negative. Of the 30 studies that separately reported effects of C addition on species unambiguously identified as introduced in the context of the experimental system, 20 found generally negative effects, nine found no effect, and one found a positive effect (online materials, Table 1). The most commonly reported negative effects were on aboveground mass and cover, which were often reduced by 30% or more (online materials, Table 2). Other negative effects included decreases in mass per plant, density, survival of planted seedlings, emergence of added seeds, tillering, height, seed production, and leaf N concentration. Negative effects increased with subsequent years of applications in at least one case (Rowe et al. 2009), and diminished the year after treatments ended in at least one case (Averett et al. 2004). In most cases, the introduced species present or measured were mostly annuals or biennials, although negative effects

were noted on perennial introduced grasses in at least three studies (Perry et al. 2004; Gendron and Wilson 2007; Iannone and Galatowitsch 2008).

The studies that found no effect of C addition on introduced species consisted of all five studies in which addition was combined with burning, grazing, or mowing, one study that measured effects only on an introduced shrub (Cassidy et al. 2004), one study that measured only effects on germination (Monaco et al. 2003), one study that used a unique application of bark mulch and eucalyptus leaves (Cione et al. 2002), and one study that did not differ in obvious ways from those that reported negative effects (Corbin and D'Antonio 2004). This suggests that C addition may have little additional effect on introduced species in systems that are also subjected to periodic removals of biomass, but also points out that few studies have tested effects on woody introduced species.

The study that found a positive effect of C addition on introduced species was the only one on mine spoils. Smith et al. (1986) also found positive effects of N addition on native species in this study and concluded that the positive effects of adding sawmill residue of mixed bark, chips, and sawdust were probably due to increased infiltration of water.

#### *Effects of C Addition on Native Plants*

Effects of C addition of native plants have been both positive and negative, with somewhat more studies finding negative than positive effects. The 41 studies that separated out and reported effects on native species variously found the following (online materials, Table 1): generally positive effects—six studies; mixed positive and null effects—three; mixed positive and negative effects—four; no effect—13; transient negative effects—two; mixed null and negative effects—two; weak generally negative effects—one; or generally negative effects—10.

Three of the six studies with generally positive effects began with largely unvegetated land and seeded or planted natives (Smith et al. 1986; Young et al. 1996; Kardol et al. 2008) but were otherwise very different from one another (online materials, Table 2). The other three studies were all on the same system (Prober et al. 2005; Smallbone et al. 2007; Prober and Lunt 2009), in which natives were perennial grasses or forbs.

Of the seven studies with mixed positive and null or negative effects, two showed positive effects only when natives were subject to competition from introduced species (Corbin and D'Antonio 2004; Perry et al. 2004), as expected from the model of underlying relationships (Fig. 1). One study each showed positive effects on native perennials but not on native annuals (Rowe et al. 2009), on perennials but not on annual forbs (McLenden and Redente 1992), on a tree but not on two perennial grasses (Nieminen 2009), of adding bark but not straw (Zink and Allen 1998), and only in more recently abandoned fields (Paschke et al. 2000). Effects were not statistically significant in this last study; each of the others began with bare soil.

The 13 studies with mixed null and negative effects, weak generally negative effects, or generally negative effects included all three studies in largely natural systems not subjected to major disturbance that reported effects on natives, none of the studies in former crop fields, one study of a restored gravel pit with 80% cover of planted native grasses (Seastedt and Suding 2007), seven studies in which only natives were present, and two studies in which it was not clear if only natives were present (Miller et al. 1991; Bleier and Jackson 2007). In both of the studies with mixed null and negative effects (Shaver and Chapin 1980; Blumenthal 1999), effects were negative only on monocots. In at least two cases, sucrose had stronger effects than sawdust (Yarie and Van Cleve 1996; Bleier and Jackson 2007).

### Conclusions

Research to date suggests that additions to soil of either sugar or sawdust in amounts of 0.3 pounds C/yard<sup>2</sup>/year or more (0.1 kg/m<sup>2</sup>/year or more) are very likely to reduce availability of N to plants in many temperate or boreal systems; effects in tropical systems appear untested. Effects of C addition on soil N may build up over at least 2 years. They are likely to persist for at least 5 years as long as additions continue, but also to disappear within a few months to 2 years after additions stop.

Adding C as sugar or sawdust is also likely to reduce the abundance of introduced species in many systems. C addition seems more likely to negatively affect annuals than herbaceous perennials, and graminoids than forbs; effects on woody species are largely untested. C addition may have little additional effect on introduced species in systems that are also managed by grazing, burning, or mowing.

C addition is less certain to increase the abundance of native species. It may be more likely to do so when competition from introduced species is more intense, and in systems that have previously been cleared and enriched in N by human activity than in less disturbed systems. Whether C addition has stronger negative effects on the introduced than on the native plant species in a system, as required for addition to benefit natives, may largely depend on whether the introduced species tend to be annuals and the natives perennials.

Based on current knowledge, managers should consider using applications of sawdust as a way to control introduced

species in grasslands and shrublands where burning and mowing are not being used. However, individual trials in each system will be needed to test whether applications will have positive effects on native plants.

### Acknowledgments

I thank Ed Vasquez and Tom Monaco for organizing this special feature on invasive plants. Research was supported by USDA grant CREES 2006-34439-17024, in collaboration with the California State Parks and the US National Park Service. This is a contribution from the University of California Bodega Marine Laboratory.

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**On-line materials for Alpert, P. 2010. Amending invasion with carbon: after 15 years, a partial success. *Rangelands*.**

*Abstract*

One of the factors most strongly associated with high abundances of human-introduced plant species in natural and semi-natural systems in North America and Europe is high availability of soil nutrients, particularly nitrogen. This suggests that reducing the availability of N in the soil might slow or reverse the spread of introduced plants in grasslands. One way to temporarily reduce N availability is to promote microbial immobilization of N by adding a carbon source such as sucrose or sawdust to the soil. C addition has now been tested for at least 15 years, and a review of the published research suggests that C addition does consistently reduce availability of N to plants. Additions of 0.3 or more lb C/yard<sup>2</sup>/y (0.1 or more kg C/m<sup>2</sup>/y) of either sugar or sawdust also often reduce the abundance of introduced, herbaceous species in shrublands and grasslands, especially introduced annuals. However, adding C to soils has only sometimes increased the abundance of native species. It may be most likely to do so when competition from introduced species is intense, in systems that have previously been cleared and enriched in N by human activity, and when introduced species are mainly annuals and natives mainly perennials.

*Keywords:* carbon addition, invasive plants, nitrogen availability, sawdust, sucrose

*Methods for literature search*

A search for refereed publications on the use of carbon to control invasion or reduce soil nutrient availability, conducted first with word searches using the Web of Science and then by following backwards and forward citations, yielded 58 relevant papers, of which neither abstract nor text

were accessible for 3 (Morgan 1984; Seastedt et al. 1996; Hopkins 1998). The full text of the other 55 papers was read for details on each experimental system and design, especially the nature of carbon additions, and on experimental effects on soil characteristics, introduced plants, and native plants. Additions were converted to units of kg C/m<sup>2</sup>/y where possible, assuming a content of 40% C in wood where content was not specified

Studies were categorized according to how directly and generally their experimental system addressed the utility of carbon addition to control the spread of introduced species: 1) largely natural systems with introduced species present and in which addition was not combined with major disturbance such as burning or tilling that cleared nearly all vegetation, 2) largely natural systems in which addition was combined with disturbance, 3) systems in which the vegetation was probably largely anthropogenic but which had not been used as agricultural crop fields, 4) former crop fields, 5) other areas such as landfill or gravel pits in which both vegetation and soil had been nearly completely disturbed, 6) largely natural systems without introduced species, and 7) greenhouse or laboratory studies.

To help compare results, descriptions of systems were highly abbreviated, descriptions of additions condensed to annual amount, carbon type, and number of years; and effects on soil nutrient availability to plants and on introduced or native species classed as generally negative, positive, or none (Table 1). To make some additional information about individual studies readily available, descriptions of experimental systems were given in slightly more detail, and some key effects of carbon addition on N availability and other soil characteristics, on introduced plants, and on natives were tabulated (Table 2). Full citations for all studies cited in the tables are given following the tables.

**Table 1.** Summary of literature on carbon addition (see Table 2 for more details of studies). Under treatment, the initial number gives kg C/m<sup>2</sup>/y of addition, where possible to calculate; C in sawdust and other wood was assumed to be 40% if unspecified. To convert to English units (lb C/yard<sup>2</sup>/y), multiply by 2.619.

Reference	System	Carbon addition	Effects of added carbon on components of system		
			soil N availability	introduced plants	native plants
<b>Studies in largely natural systems, carbon addition not combined with major natural or experimental disturbance</b>					
Beckstead and Augspurger 2004	shadscale-bunchgrass, in patches with > 85% <i>Bromus tectorum</i> , Utah	0.02, sucrose, 1 y	not reported	-	not reported
Blumenthal 2009	mixed-grass prairie, grazed until study, seeded with 6 introduced spp., Wyoming	0.3-0.9, dextrose, 2 y	-	-	- (monocots), 0 (dicots)
Cassidy et al. 2004	deciduous forest, Massachusetts	0.9, 2.5 sucrose: 1 sawdust, 1 y	not reported	0 (growth of shrub)	not reported
LeJeune et al. 2006	mixed-grass prairie, grazed until 3 y prior, abundant <i>Centaurea diffusa</i> , Colorado	0.2, sucrose, 2 y	-	-	-
Suding et al. 2004	mixed-grass prairie, 55% relative cover of introduced spp., planted with introduced and native spp., Colorado	0.1, sucrose, 2 y	-	-	-
Young et al. 1998	former sagebrush-bunchgrass, now introduced annual grassland, e. California	0.02, sucrose, 5 y	not reported	-	none present
<b>Studies in largely natural systems, carbon addition combined with major natural or experimental disturbance</b>					
Alpert and Maron 2000	coastal Mediterranean-type grassland,, in bare patches left by dead N-fixing shrubs, n. California	0.6, sawdust, 1 y	0	-	0

Cione et al. 2002	coastal sage scrub largely occupied by introduced annual grasses, burned 7 months earlier, s. California	bark and leaves, 2.5 cm deep	- (y 1 only)	0	0
Corbin and D'Antonio 2004	coastal Mediterranean-type grassland, cleared and planted with introduced and native perennial grasses and seeded with introduced annual grasses, n. California	0.4, sawdust, 2y	- (weak)	0	+ (weak, only if with annuals)
Huddleston and Young 2005	highly invaded semi-arid grassland, burned or mowed, and seeded with native perennial grasses, e. Oregon	0.3, sawdust, 1y	- (brief)	0	0
Iannone and Galatowitsch 2008	depressional wetland, cleared, graded, stripped of top soil if to be given sawdust, and sown with native spp. and an introduced perennial grass, Minnesota	sawdust, 7 cm deep, 18 weeks	mixed (brief)	- (sown sp.)	0
Mata-Gonzales et al. 2008	highly invaded semi-arid sagebrush and grassland, former military training center, burned, seeded with native and introduced spp., e. Washington	0.16, sucrose, 3 y	not reported	0	0
Mazzola et al. 2008	former sagebrush scrub, now introduced annual grassland, herbicided, packed, raked, and seeded with introduced perennial grasses, Nevada	0.15, sucrose, 1 y	- (brief)	- (brief)	none present
Rowe et al. 2009	ponderosa pine woodland, seeded with native spp. and raked, Colorado	0.13, sucrose, 2 y	-	-	- (annuals), + (perennials)
Woodis and Jackson 2009	subhumid grassland, former cattle pasture, with mainly introduced spp, grazed or burned, Wisconsin	0.3, sawdust, 2 y	not reported	0	not reported
Young et al. 1996	desert shrubland, ungrazed since 1955, disked and planted with native shrub, Nevada	0.6, sucrose, 3 y	-	-	+ (weak, planted seedlings)
<b>Studies in systems highly transformed by humans but not cropped</b>					

Prober et al. 2005	<i>Eucalyptus</i> woodland, cleared for pasture, C addition crossed with seeding 2 native perennial grasses, New South Wales, Australia	0.8, sucrose, 3.25 y	-	-	+ (perennial grasses)
Prober and Lunt 2009	as above, 9-24 months after end of treatment	as above	mixed	-	+ (perennial grasses)
Smallbone et al. 2007	as above, in plots seeded with 7 and planted with 5 native perennial forbs during last 4 months of treatment	as above	not reported	not reported	+ (perennial forbs)
Storm and Süß 2008	sandy grassland, used by military 1960-1999 then grazed, >90% cover of bryophytes, < 40% cover of vascular plants, Germany	0.055, sucrose; or 0.014, sawdust; ?4 y	not reported	not present?	0
<b>Studies in former crop fields</b>					
Averett et al. 2004	crop field until 11 y prior, mowed, sprayed with glyphosate; seeded with 10 native spp. and planted with 1; in tall-grass prairie, Ohio	2.4, sawdust, 1 application	-	-	- (0 in y 2)
Baer et al. 2003	former crop field in tall grass prairie; disked and seeded with 42 native species, Kansas	2.2, sawdust, 1 application	-	not present?	0
Blumenthal et al. 2003	cropped until study, tilled and seeded with 11 native and 10 introduced spp., Minnesota	0.08-3.3, 6% sucrose and 94% sawdust, once	-	- (annuals)	0
Eschen et al. 2007	former crop fields in chalk grassland, Switzerland and UK, plowed and seeded in Switzerland, fallowed 6 y prior in UK	1.1, sawdust + sucrose or woodchips, 1 y	-	none present	- (y 1 only)
Gendron and Wilson 2007	former crop field in mixed-grass prairie, herbicided and seeded with natives; C addition crossed with 50% and 100% tilling each y; Saskatchewan, Canada	0.16, sawdust, 10 y	0	-	+ (if 50% tilled), - (otherwise)

Horn and Redente 1997	shortgrass prairie, cultivated 60 y earlier, Colorado	0.24, sucrose, 2 y	- (weak)	-?	0?
Kardol et al. 2008	fallowed maize field on sandy glacial deposits, fallowed; addition disked, giving 2 mg C/ g dry soil, crossed with seeding 8 perennials, Netherlands	wheat straw or wood fragments	not reported	- (straw, y 1)	+ (straw, y 2 and 3)
Paschke et al. 2000	shortgrass steppe, uncultivated or cultivated until 1955, 1981, or 1989, Colorado	0.16, sucrose, 4 y	-	-	+(weak, delayed, in newer fields only)
Szili-Kovács et al. 2007	sandy, calcareous farm fields abandoned 1991-1995, 28-42% cover of vascular plants, Hungary	0.09-0.25, sucrose; or 0.03-0.09, sawdust;4y	-	not reported	not reported
Török et al. 2000	farm fields in sandy grassland, abandoned in 1991-1995	0.07, 4 sucrose: 1 sawdust, 1 y	- (weak)	not reported	not reported
Vinton and Goergen 2006	crop field restored to tallgrass prairie in 1970, burned every 3 y, mowed periodically, Nebraska	0.2-0.3, sucrose, 2 y	-	0	0
Wilson and Gerry 1995	former farm field in mixed-grass prairie, mowed, seeded with 41 native spp., Saskatchewan, Canada	0.16, sawdust, 2 y	-	0	0
<b>Studies in other highly human-disturbed areas</b>					
Biederman et al. 2008	mixed-grass prairie, landfill, disked and seeded with native and introduced spp., Texas	urban wood waste, 0.8 or 1.5 g/m <sup>2</sup> , once	not reported	not reported	not reported
Biederman and Whisenant 2009	as above	as above	not reported	not reported separately	not reported separately
Reever Morgan and Seastedt 1999	mixed grass prairie, area dug up to lay utility cable and seeded with native grasses 3-5 y prior, Colorado	0.18, sucrose+ sawdust; 1 y	- (brief)	-	0?

Seastedt and Suding 2007	restored gravel pit in tallgrass prairie, Colorado	0.3, sucrose, 3 y	-	-	-
Smith et al. 1986	spoils of bentonite mine in mixed prairie and <i>Pinus ponderosa</i> woodland, seeded with natives or introduced spp., Wyoming	0.18-0.54, bark, wood chips, and sawdust, once	0	+	+
Zink and Allen 1998	area disturbed to install pipelines on reserve., planted with native shrub or grass, s. California	pine bark or oat straw, 3 cm deep	-	not reported	+(bark)
<b>Studies in largely natural systems without introduced species</b>					
Hunt et al. 1988	shortgrass prairie, montane meadow, and <i>Pinus contorta</i> forest, Wyoming and Colorado	0.06, sucrose, 1 y	- (litter composition)	not reported	0
McLenden and Redente 1992	semi-arid sagebrush-grass, cleared, stripped of some topsoil, and seeded, Colorado	0.16, sucrose, 3 y	not measured	not present?	mixed (weak)
Michelson et al. 1999	alpine heath, n. Lapland, Sweden	0.21, sucrose, 5 y	-	none present	-
Miller et al. 1991	sagebrush-grassland, se. Oregon, ungrazed for 40 y prior	0.019, sucrose, 1 y	0	not reported	-(weak)
Morecroft et al. 1994	acid and calcareous grasslands, grazed, UK	0.3, glucose, 3 y	0	not present?	0
Shaver and Chapin 1980	alpine tundra, central Alaska	0.04, sucrose, 1 application	not reported	none present	-(graminoids), 0 (shrubs)
Yarie and Van Cleve 1996	<i>Picea</i> forest, 3 upland and 4 floodplain sites at different successional stages, Alaska	0.04-2.0, sucrose; or 0.009-2.0, sawdust, once	not reported	not present?	-(leaf N)
<b>Greenhouse and laboratory studies</b>					

Bleier and Jackson 2007	soils from 3 agricultural research stations, seeded with fast-growing or slow-growing grass, Wisconsin	0.4 or 2.1, sucrose or sawdust	- (brief)	-?	-
Bowman et al. 2004	pots with litter from alpine tundra in Colorado, planted with tundra grass; greenhouse experiment	litter from fast- or slow-growing plant	- (litter of fast-growing sp.)	not present	- (slow-growing plant litter)
Eschen et al. 2006	greenhouse experiment, soil from fallow agricultural field, 22 ug inorganic N/g soil, Switzerland	0.25, 0.5, or 1.0, sucrose	not measured	none present?	-
Kardol et al. 2008	greenhouse experiment, pots with top and mineral soil from maize field, fallowed 1 y, Netherlands	wheat straw or wood fragments, 1 cm deep	0	none present	-
Monaco et al. 2003	greenhouse experiment, soil from cold desert proving ground, Utah; containers with 2 introduced annual grasses and 6 perennial native grasses	ground barley straw, 1 mg/kg soil	-	0	0
Nieminen 2009	pots planted with a seedling of the native tree <i>Picea abies</i> and seeds of 2 perennial grasses of boreal forest	sucrose, 8.8 g/pot	not reported	not present	- (grass), + ( <i>P. abies</i> )
Perry et al 2004	greenhouse experiment with sterilized, partly nutrient-depleted soil from wetland in Minnesota, seeded with <i>Carex hystericina</i> and <i>Phalaris arundinacea</i>	sawdust, 1 g : 9 g soil	- if unseeded, + if seeded	-	+ if <i>P. arundinacea</i> present, - if not;
Schmid et al. 1997	pots of <i>Festuca vivipara</i> in intact humus from alpine tundra in n. Sweden mixed 1:4 with expanded clay	glucose, 0.23 or 0.45 µg C/g soil, weekly	-	none present	-
Tilson et al. 2009	soils from 2 sites in Szili-Kovács et al. 2007, laboratory experiment	sucrose or sawdust, 0.6, 1.2, or 2.4 mg C/g dry soil	-	none present	none present

van der Wal et al. 2006	soil samples in lab, from 4 farm fields, abandoned 0, 2, 2, and 21 y prior, and heathland site	glucose, cellulose, or sawdust, 2 mg C/g dry soil	- (brief, field soils), 0 (heathland soil)	not present	not present
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**Table 2.** Detailed summary of literature on carbon addition. Under treatment, the initial number gives kg C/m<sup>2</sup>/y of addition, where possible to calculate; C in sawdust and other wood was assumed to be 40% if unspecified. To convert to English units (lb C/yard<sup>2</sup>/y), multiply by 2.619. Mass is aboveground biomass unless otherwise specified. Characteristics, in context of study, of individual species named: *Acomastylus rossii* – native perennial forb; *Agropyron fragile* – introduced perennial grass; *Andropogon gerardii* – native perennial grass; *Atriplex nuttallii* – native perennial forb; *Bromus inermis* – introduced perennial grass; *Bromus tectorum* – introduced annual grass; *Carex hystericina* – native perennial graminoid; *Centaurea diffusa* – introduced biennial forb; *Conyza canadensis* – introduced biennial forb; *Deschampsia caespitosa* – native perennial grass; *Festuca vivipara* – native perennial grass; *Phalaris arundinacea* – introduced perennial grass; *Purshia tridentata* – native shrub; *Ratibida pinnata* – native perennial forb; *Solidago riddellii* – native perennial forb; *Taeniantherum caput-medusae* – introduced annual grass.

Reference	System	Treatment	Effects of added carbon on components of system		
			soil	introduced plants	native plants
Studies in largely natural systems, C addition not combined with major natural or experimental disturbance					
Beckstead and Augspurger 2004	shadscale-bunchgrass, grazed, in patches with > 85% <i>Bromus tectorum</i> , w. Utah	0.02, sucrose, 1 application, after fencing, at different life stages in different plots	not reported	40% < density of <i>B. tectorum</i> and < mass if applied during establishment or active growth	not reported
Blumenthal 2009	semi-arid mixed-grass prairie, grazed until study, Wyoming	0.3-0.9, dextrose, 5 applications of 0.4 kg dextrose/m <sup>2</sup> in y 1 and 2 in y 2; crossed with addition of water and with addition of NH <sub>4</sub> NO <sub>3</sub> ; plots seeded with 6 introduced spp., of which 3 established	80% < adsorbed inorganic N in y 1 and 2, less effect with added water	over 90% < mass of each of 3 spp. in y 2, < density of 2 spp., 90% < mass/plant in 1 sp. measured	50% < mass of monocots in y 2, no effect on dicots

Cassidy et al. 2004	mixed deciduous forest, Massachusetts	0.9, sucrose, 1.6 kg/m <sup>2</sup> , split between 8 applications over 9 months; plus sawdust, 0.65 kg/m <sup>2</sup> , in 2 applications	not reported	no effect on growth of shrub	not reported
LeJeune et al. 2006	mixed-grass prairie, grazed until 3 y prior, abundant <i>Centaurea diffusa</i> , Colorado	0.17-0.25, sucrose, 2-3 applications/y, 2 y; crossed with addition of gypsum and with removal of <i>C. diffusa</i>	70% < standing NO <sub>3</sub> , 40% < NH <sub>4</sub> , < N mineralization	< cover and relative mass of <i>C. diffusa</i>	30% < cover of grasses and of total, 30% < cover of forbs, > relative cover of grasses
Suding et al. 2004	mixed-grass prairie, grazed, 25% relative cover of <i>Centaurea diffusa</i> , 30% of other introduced spp., Colorado	0.12, sucrose, 2 applications/y, 2 y; crossed with removal of <i>C. diffusa</i> or all species; planted seedlings of 2 introduced and 2 native spp. after 1 y, harvested 1.2 y later	70% < standing inorganic N, < N mineralization	< survival and mass of planted seedlings, < relative growth of <i>C. diffusa</i>	< survival and mass of seedlings; > intensity of competition but no change in relative competitive rankings of introduced or native species
Young et al. 1998	sagebrush-bunchgrass, replaced by <i>Taeniantherum caput-medusae</i> , e. California	0.02, sucrose, 2 applications/y, 5 y; crossed with addition of nitrotyrin	not reported	95-99% < density, 60-75% < mass, 90-95% < seed heads, > seed bank, > dormant seeds; effect of nitrotyrin +/- additive	none present
Studies in mostly natural systems, C addition combined with major natural or experimental disturbance					
Alpert and Maron 2000	moderately to highly invaded coastal Mediterranean-type grassland, ungrazed since 1963, in bare patches left by dead N-fixing shrubs, n. California	0.6, sawdust, 1.5 kg/m <sup>2</sup> , 1 application, tilled in; measured after 1 and 2 y	no effect on standing inorganic N after 1 or 2 y	40% < mass of grasses and total mass. after 2 y	> frequency of forbs after 2 y

Cione et al. 2002	coastal sage scrub, city park largely occupied by introduced annual grasses, burned 7 months earlier, s. California	bark mulch (39:1 C:N) and eucalyptus leaves, 2.5 cm deep; measured 2 y	80% < NO <sub>3</sub> adsorption after 0.2 y, 40% < after 0.3 y; no effect on NH <sub>4</sub> or on N in y 2	no effect on cover of annual grass, < density of annual grass, though non-significant	no effect on density of shrub seedlings
Corbin and D'Antonio 2004	coastal Mediterranean-type grassland, ungrazed for 30 y prior, C:N 12, n. California	0.4, hardwood sawdust, 200 g/m <sup>2</sup> x 5 additions over 2 growing seasons, after removing vegetation and then planting 3 native perennial grasses, seeding 3 introduced annual grasses, both of the above, planting 3 introduced perennial grasses, or planting all 6 perennials	no effect on standing NO <sub>3</sub> , > standing NH <sub>4</sub> at 1 of 4 times, < N mineralization at 1 time, > microbial N after 9 months	no significant effects on perennials or annuals	in y 1, 50% < growth of <i>Nassella pulchra</i> if without annuals or introduced perennials, 2 times > growth if with annuals; 60% > growth of <i>Festuca rubra</i> if with annuals
Huddleston and Young 2005	highly invaded semi-arid grassland, ungrazed since 1988, burned or mowed, e. Oregon,	0.3, sawdust, 2 applications/y, 1 y, plus seeding of 3 native perennial grasses	30% < standing inorganic N after 3 months if unburned, no effect after 6 months	no effect on cover	no effect on cover or seedling density
Iannone and Galatowitsch 2008	depressional wetland, cleared of vegetation, graded, and stripped of the top 7 cm of soil, Minnesota	sawdust from <i>Thuja</i> , 7 cm deep, 50% C, 1.6% N, tilled in to 20 cm; inoculated with wetland microbes, watered to mimic wetland hydrology, weeded for 10 weeks; crossed with addition of 10 native spp. and of <i>Phalaris arundinacea</i> ; measured after 18 weeks	< NO <sub>3</sub> and > NH <sub>4</sub> after 9 but not 18 weeks; note greater and longer decrease in NO <sub>3</sub> when sawdust was lower in C:N in Iannone (2007 [Ph.D. thesis])	30-60% < cover, 40% < height, > light; delayed emergence and 61% < establishment of <i>Phalaris</i> if no other spp. added; growth of <i>Phalaris</i> most decreased if both sawdust and other spp. added	little effect

Mata-Gonzales et al. 2008	highly invaded semi-arid sagebrush and grassland, former military training center, e. Washington	0.16, sucrose, 3 applications/y, 3 y, plus burning; crossed with seeding 21 native and introduced spp.	not reported	little effect	little
Mazzola et al. 2008	former sagebrush scrub, now mostly <i>Bromus tectorum</i> , grazed by cattle until 1 y prior, Nevada	0.15, sucrose, 2 applications, 1 y, after glyphosate, packing, and raking, plus rolling and covering with mesh; crossed with seeding with <i>B. tectorum</i> or with <i>Agropyron fragile</i>	NO <sub>3</sub> on resin decreased 70% during first 6 month but not thereafter, no effect on PO <sub>4</sub>	35% < density of <i>B. tectorum</i> by and mass and seed production by 2/3 after 1 y but no effect after 2 y, decreased tillering in <i>A. fragile</i> and height and density in y 1.	none present
Rowe et al. 2009	ponderosa pine woodland with patches of <i>Bromus tectorum</i> , Colorado	0.13, sucrose (5,900-6,500 kg/ha/y), 9 applications/y, 2 y, after seeding with 9 native spp and raking to 4 cm; crossed with inoculation with soil from native vegetation	40% < adsorbed inorganic N in y 1 and y 2	10% < cover of <i>B. tectorum</i> in y 1 and 2; 40% < cover of other introduced annuals in y 1, 70% < in y 2; also 10% < cover of <i>B. tectorum</i> if inoculated	little effect on establishment; < cover of annuals; 10-30% > cover of perennials in y2, after 20% > in y 1, so significant effect of N x time but not of N, or of N at any one time; < relative cover of annuals, 40-60% > relative cover of perennials in y 2 only
Woodis and Jackson 2009	subhumid grassland, former cattle pasture, with mainly introduced spp, Wisconsin	0.3, sawdust, C:N 174, 2 applications/y, 2 y; crossed with drill-seeding 3 native perennial grasses; combined with grazing or burning; measured after 1 and 2 y	not reported	little effect	not reported

Young et al. 1996	desert shrubland, ungrazed since 1955, with understory dominated by <i>Bromus tectorum</i> and annual forbs, w. Nevada	0.6, sucrose, 580 kg C/ha x 3 applications/y x 3 y, after disking; also nitrapyrin treatment; planted seedlings of <i>Purshia tridentata</i>	in y 1, little effect on inorganic N; in y 2 < NO <sub>3</sub> but not NH <sub>4</sub>	in y 1, < total mass and cover of herbs; in y 2, still < cover	in y 1, > seedling emergence and height; in y 2, still > height
Studies in systems highly transformed by humans but not cropped					
Prober et al. 2005	woodland of <i>Eucalyptus</i> and grass, cleared for pasture, New South Wales, Australia, one site dominated by native perennial grasses and one by introduced annuals	0.8, sucrose, 4 applications/y, 3.25 y; after fencing for grazing; crossed with seeding with 2 native perennial grasses	50% < standing NO <sub>3</sub> at peak season, generally < NH <sub>4</sub>	< mass and cover but not density of introduced annuals and all introduced species groups	40 to 45% > cover of perennial grass at one site, and 5 to 15% > cover at the other
Prober and Lunt 2009	as above	as above, but 9-24 months after end of treatments	on seeded plots, < standing NO <sub>3</sub> , >NH <sub>4</sub>	20% < cover of annual grasses, 50% < total cover, if seeded with natives	30% > cover of grasses or 20% > if seeded, > cover of existing perennial grasses and establishment of one perennial grass
Smallbone et al. 2007	as above, in more invaded site	as above, during last 4 months of treatments, sow 7 and plant 5 native perennial forbs	not reported	not reported	> germination in 3 of 4 sown spp., > survival in 2; > survival of transplants but not > growth
Storm and Süß 2008	sandy grassland, used by military 1960-1999 then grazed, >90% cover of bryophytes, < 40% cover of vascular plants, Germany	0.055, sucrose, 10 applications/y; or 0.014 sawdust, kg C/m <sup>2</sup> /y, 1 application/y; ?4 y	not reported	not present?	little effect on total cover, mass, or height; < cover of graminoids but not significantly so

Studies in former crop fields					
Averett et al. 2004	crop field until 11 y prior, in tall-grass prairie, Ohio	2.4, sawdust, 1 application, tilled in; then mowed, sprayed with glyphosate; seeded with 6 native forbs and 4 native grasses, raked, covered with straw, and planted with <i>Solidago riddellii</i> ; measured after 1 and 2 y	after 6 months, 70% < standing NO <sub>3</sub> , 40% > NH <sub>4</sub> , 30% < inorganic N, > net N immobilization, 90% < N mineralization, > soil moisture, < N in litter	after 1 y, 70% < mass, < leaf N in one measured sp.; after 2 y, 40% < mass	after 1 y, 70% < mass of grasses, 30% < mass of forbs, < leaf N in <i>S. riddellii</i> , across introduced and native species, > effect on grasses than forbs; after 2 y, no effect on mass
Baer et al. 2003	former crop field in tall grass prairie, Kansas	2.2, sawdust, 1 application; tilled in, after excavating, replacing upper 25 cm of soil, ,disking, and fencing for deer; seeded with 42 native species, and covered with native hay; measured 3 y	< standing and adsorbed NO <sub>3</sub> in y 1 and 2 but not 3, < N mineralization, > microbial C	not present?	no effect on productivity, > diversity
Blumenthal et al. 2003	cropped until study, C:N 10.4, Minnesota	0.08-3.3, mix of 6% sucrose and 94% sawdust (39% C, 0.2% N), 84, 133, 210, 33, 529, 666, 839, 1057, 1330, 1675, 2110, 2657, or 3346 g C/m <sup>2</sup> , tilled in to 20 cm; crossed with addition of N; plots seeded with 11 native and 10 introduced spp.; measured after 1 and 2 growing seasons	< standing NO <sub>3</sub> and > water content after 1 and 2 seasons, more effect with larger addition; no effect on NH <sub>4</sub>	mass < with > addition, for total and most annuals but no perennials	if add > 1000 g C/m <sup>2</sup> , > total mass after 1 and 2 seasons; > mass of each of 7 common spp.if > 2000 g c/m <sup>2</sup>

Eschen et al. 2007	former crop fields in chalk grassland, Switzerland and UK, fallowed 6 y prior in UK	1.1, 1 sucrose:1 sawdust or 2 sawdust:1 woodchips, 5 applications/y, 1 y; in Switzerland, plowed, harrowed, sowed with 10 spp. each y; in UK, grazed	60% < NO <sub>3</sub> , sugar faster and more effective; no effect on NH <sub>4</sub> ; > mass of bacteria	none present	30-60% < biomass, effect gone 1 y later, sucrose > effective than sawdust; in Switzerland only, with sucrose, < relative cover of grasses, effect persisting 1 y
Gendron and Wilson 2007	mixed-grass prairie, crop field until 20 y prior, hayed until experiment, dominated by 2 introduced perennial grasses, Saskatchewan, Canada	0.16, sawdust, 2 applications/2, 10 y, after fencing for large herbivores and spraying with gyphosate; seeded with 41 native spp.; crossed with 50% and 100% tilling each y	no effect on standing inorganic N	50% < cover of 2 dominant spp. if 50% tilled	4 times > mass if 50% tilled, 60% < mass if 100% tilled, no effect on diversity
Horn and Redente 1997	shortgrass prairie, cultivated 60 y earlier, e Colorado	0.24, sucrose, 6 applications/y, 2 y; measured 3 y	< adsorbed NO <sub>3</sub> (consistent but not significant), no consistent effect on NH <sub>4</sub> or N mineralization	not considered separately	in y1 but not later, < cover and mass of early- and mid-successional spp., > cover of legumes in y 1 and 2, no effect on late-successional grasses

Kardol et al. 2008	sandy glacial deposits, maize field, fallowed 1 y, Netherlands	wheat straw or wood fragments (< 2 cm <sup>3</sup> ) of <i>Betula pendula</i> , disked in to 10 cm, for final concentration of 2 mg C/ g dry soil, ; crossed with seeding 4 perennial mid-successional grasses and 4 perennial forbs; measured after 1, 2, and 3 y	> mass of fungi, no effect on density of bacteria or community composition of nematodes	< cover of <i>Conyza canadensis</i> in y1 with straw, zero cover in all treatments in y 3	with straw, sometimes < shoot mass in y 1 when early-successional spp. dominated, > shoot and root mass in y 2 and 3 when mid-successional spp. dominated; with wood, > root mass in y 3; little effect on community composition
Paschke et al. 2000	shortgrass steppe, uncultivated or cultivated until 1955, 1981, or 1989, Colorado	0.16, sucrose, 8 applications/y, 4 y	after 2-3 y, 60% < adsorbed NH <sub>4</sub> , 70% < NO <sub>3</sub> , no effect on N mineralization, < litter decomposition	after 2-4 y, < mass	after 3-4 y, > mass but not significant, in newer fields only; across both introduced and native plants, < relative mass of annuals, sometimes < plant N
Szili-Kovács et al. 2007	sandy, calcareous farm fields abandoned 1991-1995, 28-42% cover of vascular plants, Hungary	0.09-0.25, sucrose, 7-10 applications/y; or 0.03-0.09, sawdust, 2-3 applications/y; 4 y	< standing NO <sub>3</sub> and N adsorption on resin, and generally < standing NH <sub>4</sub> and > microbial C and N, > soil moisture	not reported	not reported
Török et al. 2000	farm fields in sandy grassland, abandoned in 1991-1995	0.07, 4 sucrose: 1 sawdust, 4 applications for sucrose and 1 for sawdust; 1 y	> microbial C only at site with less nutrients; at last measurement only, < NO <sub>3</sub> but not < NH <sub>4</sub>	not reported	not reported

Vinton and Goergen 2006	crop field restored to tallgrass prairie in 1970, burned every 3 y, mowed periodically, C:N = 12-60, 80% relative cover of native grasses, Nebraska	0.2-0.3, sucrose, 2-3 applications/y, 2 y	in y 2 but not y 1, 80% < inorganic N, < NO <sub>3</sub> , < NH <sub>4</sub> , < N mineralization, > field respiration	little effect	little effect
Wilson and Gerry 1995	former farm field in mixed-grass prairie, dominated by <i>Bromus inermis</i> and <i>Agropyron cristatum</i> for at least 10 y, Saskatchewan, Canada	0.16, sawdust, 2 applications/y, 2 y mowed, drill-seeded with 41 native spp., mowed and sprayed with glyphosate in y 2; crossed with 50% or 100% tilling;	< standing inorganic N	no effect on cover of perennial grasses	no effect on density of seedlings or cover; > bare ground
Studies in other highly human-disturbed areas					
Biederman et al. 2008	mixed-grass prairie, landfill, disked, seeded with native and introduced spp., selectively weeded, Texas	urban wood waste, C:N 98, 43.5% C, 0.45% N, 56% > 5 cm <sup>2</sup> , 0.8 or 1.5 g/m <sup>2</sup> , on surface or tilled in to 6 cm; measured for 2 y	in surface treatments, > bacterivorous, plant-parasitic, omnivorous, and predatory nematodes	not reported	not reported
Biederman and Whisenant 2009	as above	as above	not reported	not reported separately	not reported separately; in y 2, > total basal area of perennial grasses in low, surface and high, tilled applications
Reever Morgan and Seastedt 1999	mixed grass prairie, area dug up to lay utility cable and seeded with native grasses 3-5 y prior, Colorado	0.18, 1.5 sucrose: 1 sawdust, 5 applications of sucrose, 2 of sawdust, 1 y	80% < inorganic N during treatment, 25% < 2 months after	30% < mass of <i>Centaurea diffusa</i> but not < density	no effect on grass; 40% < total mass of introduced and native plants together

Seastedt and Suding 2007	tallgrass prairie, former gravel pit, refilled with original soil, planted with 6 native grasses now 80% cover, but introduced annuals present, Colorado	0.3, sucrose, 3-4 applications/y, 3 y; crossed with addition of P <sub>2</sub> O <sub>5</sub> or CaSO <sub>4</sub> , with sowing or <i>Centaurea diffusa</i> or native forbs of which only <i>Ratibida pinnata</i> established in large numbers, and with herbicide for grass (no treatment with added sucrose alone)	after 3 y, 40% < standing inorganic N with sucrose + gypsum, no effect on P	< mass, establishment, and density of <i>C. diffusa</i>	after 4 y, 40% < total mass [total probably > 90% native], < establishment of <i>R. pinnata</i>
Smith et al. 1986	mixed prairie and <i>Pinus ponderosa</i> woodland, spoils of bentonite mine, ne. Wyoming	0.18-0.54, sawmill residue (bark, chips, and sawdust), 45, 90, or 135 Mg/ha; crossed with addition of N; seeded with native perennial grasses and <i>Atriplex nuttalli</i> or with introduced perennial grasses and <i>A. nuttalli</i> ; measured after 2 growing seasons	little effect on N mineralization	> mass, > mass than treatment seeded with natives at highest level	> mass except at highest level, > mass than treatment seeded with introduced spp. at 90 Mg/ha
Zink and Allen 1998	area disturbed to install pipelines on reserve ungrazed since 1950, area now occupied by introduced species, s. California	pine bark or oat straw, 3 cm deep on surface; crossed with planting native shrub or native perennial grass; crossed with shallow soil depth of 25 cm; measured after 1 and 2 y	with bark or straw, 50% < standing NO <sub>3</sub> after 0.5-1.6 y, occasionally < NH <sub>4</sub> ; with straw, occasionally > bacterial density; with bark, > fungal hyphae length after 1 y	not reported	with bark, 150% > survival and growth of transplants; with straw, only slight effects
Studies in largely natural systems without introduced species					

Hunt et al. 1988	shortgrass prairie, montane meadow, Pinus contorta forest, se Wyoming and ne. Colorado	0.06, sucrose, 1 application; measured after 1 y	< litter decomposition	not considered separately	no effect on mass
McLenden and Redente 1992	semi-arid sagebrush-grass, nw. Colorado	0.16, sucrose, 8 applications/y, 3 y, after removing vegetation and top 5 cm of soil, mixing the next 35 cm of soil, and seeding with early- and late-successional spp.	not measured	not present?	> cover of shrubs and perennial grasses but not significant, < N concentration in shoots of early-successional forbs; compared to treatment with added NO <sub>3</sub> , > species richness, > relative cover of perennial forbs and annual grasses and < of annual forbs
Michelson et al. 1999	alpine heath, n. Lapland, Sweden	0.21, sucrose, 2 applications/y, 5 y; crossed with addition of NPK and of benomyl	after 2 y, 20% < standing NH <sub>4</sub> (very little NO <sub>3</sub> , no change in P or microbial C or N; after 5 y, 60% > microbial C, < P	none present	after 2 y, < cover of a perennial grass, < mass/stem but not < cover of a shrub
Miller et al. 1991	sagebrush-grassland, se. Oregon, ungrazed for 40 y prior	0.019, sucrose, 1 application; measured over one growing season	no effect on NO <sub>3</sub> or NH <sub>4</sub>	not reported	no significant effect on growth or leaf N of a shrub or a perennial grass, though < mean shoot mass
Morecroft et al. 1994	acid and calcareous grasslands, grazed, UK	0.3, glucose, 2 applications/y, 3 y	no effect on N mineralization	not present?	little effect

Shaver and Chapin 1980	alpine tundra, central Alaska	0.04, sucrose, 1 application	not reported	none present	30-90% < shoot mass of 2 graminoids; in some spp. < inorganic N, P, or K, > total non-structural C
Yarie and Van Cleve 1996	<i>Picea</i> forest, 3 upland and 4 floodplain sites at different successional stages, Alaska	0.04-2.0, sucrose; or 0.009-2.0, sawdust, 45% C, 0.12% N, mixed in surface organic layer; 1 application	not reported	not present?	sucrose sometimes < N or P or > C:N in leaves of trees; with sawdust, sometimes < N in y 1
<b>Greenhouse and laboratory studies (partial list)</b>					
Bleier and Jackson 2007	soils from 3 agricultural research stations, silty loam or loamy sand, starting standing inorganic N 6-60 ug/g, Wisconsin	0.4 or 2.1, sucrose, sawdust, or both, on surface or mixed in; seeded with <i>Bromus inermis</i> (fast-growing C3 grass) and <i>Andropogon gerardii</i> (slow-growing C4 grass) after 6 weeks, later thinned to equal density; grown 18 weeks more	after 2 and 6 but not 24 weeks, < N mineralization; after 6 but not 2 or 24 weeks, < standing inorganic N; sawdust most effective addition, mixing more effective than surface	unclear if <i>B. inermis</i> introduced	5 g/m <sup>2</sup> < mass of <i>A. gerardii</i> , 120 g/m <sup>2</sup> < mass of <i>B. inermis</i> ; sugar > effective than sawdust, 5 kg/m <sup>2</sup> > effective than 1
Bowman et al. 2004	plants and litter from alpine tundra, Colorado; greenhouse experiment	litter from <i>Deschampsia caespitosa</i> (fast-growing) or <i>Acomastylus rossii</i> (slow-growing) mixed into sand inoculated with soils from under both spp. in pots planted with 2-month-old <i>D. caespitosa</i> and given 1/4 strength Hoagland's; measured after 47 days	> microbial C and N and < soil respiration under litter from <i>D. caespitosa</i>	not present	in litter from <i>D. caespitosa</i> compared to that of <i>A. rossii</i> , > growth, < root:shoot mass, < C:N

Eschen et al. 2006	greenhouse experiment, soil from fallow agricultural field, 22 ug inorganic N/g soil, Switzerland	0.25, 0.5, or 1.0, sucrose, 3 applications, one every 3 weeks; pots planted with 1 of 29 spp.; harvested after 2 months	not measured	none present?	< mass of each sp., > effect of > addition, legumes less affected than other spp., > decrease in shoot:root mass in grasses and in annuals than in others
Kardol et al. 2008	greenhouse experiment, pots with top and mineral soil from maize field, fallowed 1 y, Netherlands	wheat straw or wood fragments < 0.5 cm <sup>3</sup> , 1 cm deep; plant 1 seedling of each sp. above; harvest after 2, 4, 6, and 8 months	no effect on inorganic N	none present	with straw, 80% < shoot mass after 2 months but not longer, > root mass; with wood, 65% < shoot mass; change in community composition
Monaco et al. 2003	greenhouse experiment, soil from cold desert proving ground, Utah	ground barley straw, C:N 98, 1 mg/kg soil; containers with 2 introduced annual grasses and 6 perennial native grasses	over 90% < standing NO <sub>3</sub> and NH <sub>4</sub>	no effect on germination	no effect on germination
Nieminen 2009	greenhouse experiment, to simulate boreal forest	sucrose, 8.8 g in pots 10 cm high and 17.3 cm in diameter, planted with a seedling of the native tree <i>Picea abies</i> and seeds of 2 perennial grasses; crossed with addition of 1.175 g wood ash	> enchytraeids, < bacteria-feeding nematodes, > fungus-feeding nematodes, no effect on P	not present	90% < mass of grass, 200% > mass of <i>P. abies</i>

Perry et al 2004	greenhouse experiment with sterilized, partly nutrient-depleted soil from wetland in Minnesota	sawdust, 39% C and 0.2% N, 1 g: 9 g of soil; crossed with seeding with <i>Carex hystericina</i> and with <i>Phalaris arundinacea</i> and with addition of NH <sub>4</sub> ; measured after 25 weeks	< NH <sub>4</sub> but not NO <sub>3</sub> if unseeded, > NH <sub>4</sub> if seeded	< total mass, < competitive effect, and > competitive response of <i>P. arundinacea</i>	> total mass and < competitive response of <i>C. hystericina</i> if <i>P. arundinacea</i> present; < total mass if not; across species, < total mass, > root:shoot mass, > density of plants
Schmid et al. 1997	pots of the native perennial grass <i>Festuca vivipara</i> in intact humus from alpine tundra in n. Sweden mixed 1:4 with expanded clay	glucose, 10 ml of 582 or 1189 µg/ml (0.23 or 0.45 µg C/g soil), weekly; crossed with soil sterilization and with nutrient additions, 0.25 y	>microbial N and P	none present	< plant N and P
Tilson et al. 2009	soils from 2 sites in Szili-Kovács et al. 2007, laboratory experiment	sucrose or sawdust, at 0.6, 1.2, or 2.4 mg C/g dry soil; incubate 19 d (sucrose) or 116 d (sawdust)	with sucrose, < inorganic N, > N mineralization, > microbial C and N, and > CO <sub>2</sub> production; compared to sucrose, sawdust increased inorganic N and microbial N less but CO <sub>2</sub> more; no effect of sawdust on microbial N	none present	none present

van der Wal et al. 2006	soil samples in lab, from 4 farm fields, abandoned 0, 2, 2, and 21 y prior, and heathland site	glucose or cellulose or birch sawdust, 2 mg C/g dry soil, adjusted to C:N 20 with $\text{NH}_4\text{SO}_3$ ; crossed with inoculum from oldest field; 16 weeks	in old field soils with all C additions, > ergosterol (cellulose most), > short-term immobilization of N (glucose most), later > N mineralization (glucose most); with sawdust and to lesser extent cellulose, > hemicellulase and at first cellulase; DGGE bands similar in control and glucose, and in cellulose and sawdust; little effect in heathland soil	not applicable	not applicable
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From: Erich Knight <erichjknight@gmail.com>  
Sent: Monday, January 25, 2016 11:05 AM  
To: CDFA Environmental Stewardship@CDFA  
Subject: Biochar Plus Compost

Dear CDFA,

Lacking in your analysis is the cumulus effects, nitrogen & phosphorous retention and soil-C persistence of Biochar amendments to compost. Numerous protocols have been demonstrated. The most exciting are the cascading effects of feeding Biochar to farm animals and then composting their manure with additional biochar in the EU by the Ithaka Institute. On several hundred farms in Germany & Switzerland.

<http://www.ithaka-institut.org/en/home>

Cheers,  
Erich

Erich J. Knight  
Shenandoah Gardens  
1047 Dave Berry Rd. McGaheysville, VA. 22840  
540-289-9750

Policy & Community Chairman  
2013 North American Biochar Symposium  
Harvesting Hope: The Science & Synergies of Biochar  
October 13-16, 2013 at UMASS Amherst  
<http://pvbiochar.org/2013-symposium/>

From: Smyth, Brenda@CalRecycle <Brenda.Smyth@CalRecycle.ca.gov>  
Sent: Monday, January 25, 2016 10:12 AM  
To: Gunasekara, Amrith@CDFA; Gravuer, Kelly@CDFA  
Cc: Pogue, Kyle@CalRecycle; Larimore, Brian@CalRecycle; Horowitz, Robert@CalRecycle; Levenson, Howard@CalRecycle  
Subject: Healthy Soils - Follow up to SAP meeting on 1/15/16

Hi Ami and Kelly – We are sending you some comments and information as follow up to the Science Advisory Panel meeting on 1/15/16. There were some questions by the audience and the following items provide responses. Let us know if you have any questions.

Thanks for all of the work you are doing on the Healthy Soils Initiative.  
Brenda

1. Title: We suggest changing the title of any presentations or white papers so as to avoid any confusion regarding agronomic rates or standards versus application rates that will be used for the incentive program. As Ami made clear, these rates are just for compost purchases incentivized by the healthy soils program and growers can certainly use more or less than these amounts according to their needs but the incentive program is only going to fund these application rates. Suggested title, “Proposed Compost Application Rates for California croplands and rangelands for a CDFAs Health Soils Incentives Program.”
2. Compost definition. We suggest inserting a definition for compost in the white paper so that there is no confusion over what types of products are eligible for the program. Here are a few options. A definition in PRC 40116: "Compost" means the product resulting from the controlled biological decomposition of organic wastes that are source separated from the municipal solid waste stream, or which are separated at a centralized facility. SWRCB definition: Composting - A controlled microbial degradation of organic wastes yielding a safe and nuisance-free product.
3. Calrecycle database: The data on compost nutrient values originated from Soil Control Labs of Watsonville, CA (<http://compostlab.com/>) We greatly appreciate their sharing this information. Because the data was not originally intended for publication, information about who composted the materials is removed. We request that in the final draft of the report, and any other publications resulting from the use of this data, that Soil Control Labs be credited as the source of this information. A range of different feedstocks were included in the data set and they are reflected by the wide range of associated C:N ratios in the compost products. Composts produced from biosolids and manures generally have higher N while those produced from yard waste and yard waste usually have lower N.
4. Carbon-to-nitrogen ratio vs. % nitrogen. Composters are required by regulation (Title 14 CCR, Article 7, Sections 17868 et seq.) to send finished product samples to a lab prior to sale to ascertain levels for two indicator pathogens and nine heavy metals. Typical lab analyses contain much more information than that, including C:N ratio. Prior to AB 856, it was common practice for a composter to provide a customer with a copy of the lab analysis. Subsequent to CDFAs implementation of AB 856, composters were instructed not to provide the lab analysis, as that would be a competing nutrient claim. Current recommended practice for composters selling CDFAs registered product is to provide the label only, which does not give C:N ratio. Composters who make no nutrient claims nor claims of suitability for organic production may provide an invoice only. However, composters almost always know the C:N ratio of their final product and perhaps for the purposes of the incentives program, it will be ok with CDFAs if composters can share the C:N ratio with growers without having to apply for or include it in their label.

5. Compost quality: We recommend that all compost used in the program be produced by permitted composting facilities which follow state and local regulations for, among other things, pathogen reduction, maximum metal concentrations, and finished product testing.

6. Sudden Oak Death (*P. ramorum*): Calrecycle (then the Integrated Waste Management Board) sponsored researchers at UC Berkeley to determine whether the composting process would be an effective control treatment for SOD quarantined materials. The results were unequivocal: “No *P. ramorum* was recovered at the end of the composting process, regardless of the isolation technique used. By using a PCR assay designed to detect the DNA of *P. ramorum*, we were able to conclude the pathogen was absent from mature compost and not merely suppressed or dormant... Composting is an effective treatment option for sanitization of *P. ramorum* infected plant material.” <https://nature.berkeley.edu/garbelotto/downloads/Swainetal.pdf>

a. The recent article in California Agriculture looked at whether live *P. ramorum* re-introduced to fully matured compost could grow. This phenomena has been observed with other pathogens just because of the natural environment of the compost medium. Best management practices, such as keeping finished product away from areas where raw materials are processed, are designed to eliminate re-introduction of pathogens. The finished product testing regime outlined in Calrecycle regulations is designed to identify and rectify situations where operators are inadvertently contaminating pathogen-reduced product.