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Determining Compost Carryover for Optimal Use in an Organic Corn Squash Rotation

Davey J.R. Olsen

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DETERMINING COMPOST CARRYOVER FOR OPTIMAL USE IN AN ORGANIC CORN SQUASH ROTATION

by

Davey J.R. Olsen

A thesis submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

in

Soil Science

Approved:

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UTAH STATE UNIVERSITY
Logan, Utah
2012
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ABSTRACT

Determining Compost Carryover for Optimal Use in an Organic Corn Squash Rotation

by

Davey J.R. Olsen, Master of Science

Utah State University, 2012

Major Professor: Dr. Jennifer Reeve
Department: Plants, Soils and Climate

Organically certified farms using compost to improve or maintain fertility rarely consider compost carryover and its impact on the determination of economically optimal application rates. Compost carryover is comprised of nutrient and non-nutrient elements. Both affect crop growth, yet carryover is typically described primarily in terms of nitrogen (N)-carryover only. This study tested a new method for estimating compost carryover on organically certified land and expressed carryover in units that capture both the nutrient and non-nutrient components. Compost carryover for five treatment rates was estimated over four years in an organically certified field trial in a corn and squash rotation. Nitrate (NO$_3^-$), phosphorus (P), soil organic matter (SOM) were investigated to determine the residual effect of a one-time compost application. Implications for fertility management and farm profitability were considered. The new method successfully modeled carryover, determining that compost had a persistent and positive effect on crop yields, evident even three years after an initial one-time application. No NO$_3^-$ carryover was observed in any year, suggesting that yield responses were due primarily to non-N carryover. Compositional changes in SOM corresponding to compost input three years
earlier suggested that compost was able to influence non-nutritive soil properties many years after incorporation. High value cash-crops are necessary in organic rotations to offset the high input cost of compost use. In organic fertility management, compost is an important and economical source of non-N fertility, which benefits crop yield many years after incorporation. When used with a dedicated N-fixing cover crop in a rotation that includes a high value cash-crop, complete fertility goals could be met in a sustainable manner.

(153 pages)
In 2008–2011 a graduate project was undertaken by Davey Olsen under the supervision of Utah State University (USU) Plant, Soils and Climate professors, Drs. Jennifer Reeve, Dan Drost, and Astrid Jacobson. The project investigated a new way of measuring the benefits of applying compost to organically certified horticultural crops. In particular, the carryover of these benefits in the three years following a one-time application was studied. A field trial at the USU Organic Farm modeled the carryover of nitrogen and phosphorus in a corn and squash rotation, while laboratory analysis investigated aspects of nitrogen availability and compost decomposition over time. Results indicated that compost continued to provide benefits to crops four years after it was initially applied. It was concluded that the majority of these benefits were due to factors other than nitrogen nutrition. The ongoing benefits to crops from a single compost application are seldom accounted for by growers. The study showed that when these benefits are considered, compost use can become more economically viable. The study also considered the role that compost plays in a crop rotation, and highlighted the need for an additional nitrogen source, such as a legume cover crop, within organic crop rotations.

The results of this study have implications for compost use on organically certified farms by changing the way compost is viewed as a source of plant nutrients, as well as the time frames over which compost is regarded as being beneficial to crops. When compost carryover is considered, growers can more accurately determine the role of compost in their fertility program, and manage its use to optimize economic returns.
ACKNOWLEDGMENTS

I would like to thank my wife, Summer, and daughter Sky, for their sacrifice, patience, love, and support throughout my academic career. For inspiring me to make a career change and igniting my passion for agriculture, I thank Parque Nueva Juventud, San Andres, Guatemala. For their help and support in the lab and field, I thank Alicia Campbell, Jeff Slade, and the students of the USU Organic Farm. I wish to thank Xystus Amakor for his friendship and his example of what it means to be a graduate student. Many aspects of this project would not have been possible without the hard work and assistance of Dr. Jeffrey Endelman, and I thank him. I have greatly appreciated the knowledge and assistance of my committee members, Drs. Dan Drost and Astrid Jacobson. Finally I wish to acknowledge and thank my major professor, Dr. Jennifer Reeve, for her friendship, academic support, and passion for organic agriculture.

Davey J.R. Olsen
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CHAPTER I
GENERAL INTRODUCTION

Organic farming poses unique challenges for fertility management. Although crop nutritional requirements are the same for organic and conventional farms, organic producers often utilize a more diverse range of farm inputs to meet fertility goals, and must operate within a paradigm which mandates responsible management of soil and water resources. They apply natural materials and emphasize practices that retain and recycle nutrients within the soil (Wander et al. 2011). From an organic viewpoint it is more helpful to view soil fertility as an ecosystem concept integrating diverse soil functions, including nutrient supply, which promote crop production (Swift and Palm 2000).

Organic farmers rely on intuition and observation, advice from vendors, conventional soil tests, and their own experience to make decisions about the quantity and types of soil amendments to apply. As a result, there is tremendous variability in both the quantities of nutrients applied and the resulting soil fertility status on organically managed farms (Wander et al. 2011).

Composts and manures are commonly used on organic farms to maintain or improve fertility. Used primarily for their within-season fertilizing contribution, these amendments also play an important role in soil organic matter (SOM) accumulation and long term improvements to soil quality, as well as providing a residual nutritive benefit which is frequently overlooked in fertility planning. Understanding the persistent, residual nature of compost is important if organic farmers are to “manage plant and animal materials to maintain or improve soil organic matter content in a manner that does not contribute to contamination of crops, soil, or water by plant nutrients, pathogenic organisms, heavy metals, or residues of prohibited substances” (National Organic Program 2012).
Where composts are used farmers naturally seek to economically optimize its use. Determining optimal application rates requires an understanding of, and method of accounting for, the fertility carryover of compost which occurs over many years where carryover is understood to represent both nutritive and non-nutritive elements. Because compost has such a beneficial long term effect on SOM, it need not be applied every year. Even a one-time application can provide benefits which span many years.

Research investigating compost carryover is sparse, and is most often conducted in a conventional farming setting, where compost is applied annually and where often the focus of study centers on nitrogen (N) contribution only. Organic agriculture is unique in its challenges and fundamentally different from conventional agricultural systems. Research which investigates compost carryover on organically certified land, and accounts for more than simply N dynamics, is necessary to enable growers to better manage their compost use.

**Nutritive benefits of compost use**

Under organic certification guidelines, growers are able to use cover crops, green manures, compost and a range of organic amendments to meet their fertility goals. Nitrogen, though abundant in the environment, remains the most frequently limiting nutrient for crop production (Wander et al. 2011). In areas with long growing seasons growers are able to grow legume (N-fixing) cover crops to replenish soil-N between cash cropping seasons. The portion of green-manure nitrogen available to a following crop is usually about 40% to 60% of the total amount contained in the legume (Sullivan 2003).

Legume cover crops such as field pea, vetch, and clovers are often grown to provide N for a following cash crop. These crops are able to fix between 112-280 kg N ha\(^{-1}\) in 45-55 days under ideal conditions (Peoples et al. 1995). Given favorable rhizobium inoculation and soil N
concentrations, nitrogen fixing potential of legume crops in irrigated systems is overwhelmingly regulated by plant growth (Peoples et al. 2009). The more biomass, the greater will be the mass of N\textsubscript{2} fixed.

However, for farmers with short growing seasons, such as in the Inter-Mountain West, reliance upon legume cover crops as a sole source of N becomes problematic. The short growing season, averaging 4-5 months, results in insufficient biomass growth post-season. The cover crop struggles to reach maturity, and its full N-fixing potential, before the cash-crop growing season begins again (Gaskell and Smith 2007). Thus, legume cover crops are able to provide only a portion of total crop-N demand. Unless a field is taken out of production and devoted to legume cover cropping for a season, supplemental sources of N are often necessary. Often the costs incurred from loss of productivity due to cover cropping rather than cash cropping may not outweigh the benefits of N fixation (Barker 2010). It is no surprise then that growers in these regions often rely more heavily on compost and manure as a source of N, particularly when high value crops are grown and farm size is small.

Compost-N is predominantly in organic forms requiring mineralization by soil biota before it becomes plant-available. Depending on initial N content, C/N ratio and compost maturity, typically only 10% to 30% of total compost-N is plant-available within the season of application (Amlinger et al. 2003; Eghball et al. 2002; Hadas et al. 1996; Hartz et al. 2000) and often periods of immobilization occur initially upon incorporation. The gradual decomposition and chemical transformation of compost over many years results in a steadily decreasing rate of N mineralization, often described by a decay series in the literature (e.g. Cusick et al. 2006; Klausner et al. 1994).
One complication with compost use, however, is that unlike inorganic fertilizers, many of which supply only one or two mineral nutrients, composts contain a wide range of plant nutrients such as N, P, K and S and micronutrients, which are slowly released into the soil (Smith and Collins 2007). These nutrients vary in their availability with nutrients in the inorganic form being much more readily plant available than those in organic form which must first undergo microbial-mediated mineralization. Total phosphorus availability is > 70% (Eghball et al. 2002) while potassium is highly available with 80-100% of it available in inorganic form (Eghball et al. 2002; Mikkelsen 2007).

Typically the P/N ratio of manures is greater than that of plants, so growers who base their application rates to achieve an N target often apply P in excess of crop needs (Eghball and Power 1999). Excess P can become an environmental pollutant if it is carried into surface or ground waters where it can cause eutrophication and contamination (Daniel et al. 1994). In addition to accumulation of excess P, concentrations of Cu and Zn may also accumulate in soils (Wander et al. 2011). For this reason, growers are often cautioned to avoid over-reliance on animal manures and composts.

*Non-nutritive benefits of compost use*

Compost promotes SOM accumulation, which in turn affects crop growth and yield through a range of non-nutritive soil properties and processes in addition to its nutrient contributions. It is clear that despite comprising less than 5% of a typical soil, SOM exerts a disproportionately large influence on soil properties (Wagner and Wolf 1999). Compost has the potential to increase not only the quantity, but also the quality of SOM by contributing reactive humus-like substances which are important for nutrient supply and storage, and by improving soil physical properties (Rivero et al. 2004).
While nutritive components such as N, P, and K are a function of microbially-mediated mineralization dynamics, non-nutritive components are more directly attributable to soil physical and chemical properties, affected largely by SOM accumulation and residence in the soil. Aside from being a source of nutrient mineralization, SOM contributes to soil fertility through improvement in soil quality indicators that facilitate nutrient uptake and availability (Roe 1998). Non-nutritive benefits include improved chelation of micronutrients (Chen et al. 1998), improved plant root growth due to greater soil porosity (Seiter and Horwath 2004), improved nutrient uptake due to improved water availability (Pinamonti 1998), as well as improvements to soil aggregation, soil water holding capacity, nutrient storage and release, cation exchange capacity, soil pH buffering, anion sorption, and metal mobility (Weil and Magdoff 2004), all of which promote crop growth and ultimately, yield.

The residual effect of compost is predominantly a result of the physical and chemical composition of the compost itself. During composting, easily degradable plant compounds such as carbohydrates and proteins become decomposed, and more recalcitrant plant compounds, such as lignin, together with microbial products and non-identifiable humic substances are relatively enriched (Leifeld et al. 2002). Most of the easily mineralizable N and C is lost, leaving only more stable and recalcitrant forms of N/C (Eghball et al. 1997). The finished product has a higher degree of humification and chemical stabilization than the original raw materials and exhibits a higher aliphatic character and polysaccharide content than native soil (Soler Rovira et al. 2003).

Unlike inorganic fertilizers, which are rapidly solubilized, compost persists in the soil, where it continues to influence nutrient and non-nutritive properties for many years. Soil fertility in a given year is therefore a function of the total compost applied in that year, plus a
proportion of previous years’ applications that are carried over into the current year.

Understanding compost carryover then is the first step in redefining fertility management in organic systems where compost forms an important part of total farm inputs.

*Expressing carryover as a decay series*

The concept of expressing manure carryover as a decay series was first developed by Pratt et al. (1973) with more recent contributions by Cusick et al. (2006). Their work expressed N carryover as a proportion of the total manure N available during successive years after application. Fertility \( F \) in a given year, expressed in terms of N, can thus be expressed by Eq. [1.0] where \( C_t \) terms represent compost applied at time ‘t’, and \( b_x \) terms represent carryover.

\[
F = C_t + (b_1 \times C_{(t-1)}) + (b_2 \times C_{(t-2)}) + \ldots
\] [1.0]

With this concept, decay terms are calculated using reference applications of inorganic N-fertilizer. Yields following an inorganic fertilizer application are used as benchmarks from which to compare the yield from a compost amended plot (Cusick et al. 2006). Carryover is then expressed as a percentage representing an inorganic fertilizer equivalent. Unfortunately these studies often have little relevance to organic farmers, where inorganic fertilizer use is prohibited and where organic inputs follow a more complex mineralization dynamic. Additionally, by focusing solely on N-mineralization, these studies neglect the many non-nutritive contributions of compost. Organic farming systems are fundamentally different from their conventional counterparts and a different approach is required which accounts for total carryover, nutritive and non-nutritive. One which does not rely on N-fertilizer as a reference and which can be conducted on organically certified land.
This study seeks to test a new approach to measuring compost carryover on organic certified land, developed by Endelman et al. (2010). Rather than expressing carryover relative to inorganic fertilizer rates, Endelman et al. (2010) determined carryover by measuring yield relative to new applications of compost. This methodology expresses soil fertility in units of compost equivalents, a term which encompasses both the nutrient and non-nutrient components of compost carryover. We hypothesize that by accounting for total carryover, an economically optimal rate of application can more accurately be determined on organically certified land.

**Determining compost application rates**

While conventional farmers often determine fertilizer rates based on soil test recommendations, organic farmers find these results less useful because recommendations are often given in terms of inorganic fertilizer rates. Not only are these fertilizers prohibited in organic systems, but growers find it difficult to match exact nutrient requirements based on recommendations for individual elements. Also, soil test recommendations are often based on maximum yields, and as such are more suited to conventional agricultural systems that can make use of chemical pesticides and intensive cropping techniques. Additionally, recommendations based on conventional fertilizer usage fail to account for the non-nutritive contributions of organic amendments, which are evident years after initial incorporation.

Compost is a relatively high cost input. It is needed in larger quantities than conventional fertilizers due to its low nutrient content, and therefore often incurs higher costs of purchase, transport and incorporation per unit N. Where farm size is large or where low value crops are grown growers often hesitate to apply compost because they perceive these costs outweigh the benefits to their crops in any given year. Conversely, problems of over application
are more common on small farms growing high-value crops such as vegetables. Additionally, because compost mineralizes gradually over multiple years, its contribution to soil fertility in any given year is difficult to assess. For these reasons, growers often base compost rates on what is affordable in a given year.

The central question for growers becomes; “At what rate is the $ return for each additional $ spent on compost maximized?” This ideal rate should consider the carryover contribution of compost applied in previous years. When this is accounted for, the potential for over or under-application is minimized and input costs are more effectively controlled. Compost over-application is not only wasteful in terms of an economic cost, but it also increases the potential for P-leaching and micronutrient accumulation, which can have adverse environmental consequences. Compost under-application fails to meet fertility goals and results in nutrient mining of soils, which is unsustainable in the long term. Both cases are less than ideal.

When the long-term carryover of compost is considered, even a one-time application may become economically viable, even for growers who have hesitated to use it in the past. Studies have shown that a one-time compost treatment can improve soil quality and nutrient capacity many years after incorporation. Ippolito et al. (2010) found one-time additions of bio-solid compost to rangeland soil resulted in increases in OM that were still evident 13 and 14 years after the original application when the compost was not incorporated. Hartl et al. (2003) reported increases in soil organic carbon, total N, hot water extractable N, NO₃⁻N, in compost treatments three years after application. Butler and Muir (2006) found a curvilinear increase in SOM pH, P, K, and water infiltration evident three years after a one-time compost application at various rates. Marcote et al. (2001) found that in the second year after a single compost application, plots showed higher protease hydrolyzing casein, β-glucosidase and dehydrogenase
activities than control or mineral fertilized soils, indicating that compost had a residual catalyzing effect in soil.

Although many studies have shown one-time compost applications to have a residual nutritive and non-nutritive benefits lasting many years, most of these studies are concerned with rangeland soils, dry-land crops, or forage production. Very little research has been published describing residual contributions of compost in organic farming systems.

Compost composition and behavior in soil

Following compost application, SOM displays enrichment corresponding to the original composition of the compost amendment. Composition here refers to the nature, content and spatial arrangement of functional groups (e.g. carboxyl or hydroxyl groups) within the organic material (Ellerbrock et al. 1999b). Functional groups have a significant effect on both nutrient mineralization and non-nutritive soil properties in the long term. Leifeld et al. (2002) noted that changes in the SOM composition after the addition of compost are characterized mainly by an ongoing decay of the easily degradable OM from the compost, which is mostly polysaccharide and, to a considerable degree, composed of microbial biomass. Polysaccharides, both cellulosic and non-cellulosic, present in composts are primarily of plant and microbial origin and can account for up to 20% of total organic C in compost amended soils (Leifeld et al. 2002). Because polysaccharides are more readily degraded than lignin and other more stable humic materials, they can be used to signal compost degradation. Rivero et al. (2004) noted that higher compost application rates resulted in greater humification and aromaticity of SOM as well as an increase in amount of functional groups. Leifeld et al. (2002) noted that even after 18 months of incubation, lignin contents of soil were fairly stable, indicating that lignin was somewhat
resistant to decomposition during this time. In general, SOM displays functional group enrichment corresponding to the original composition of the compost.

Assessing the persistence of compost in soil

Many infrared wavelengths are known to induce bonding vibrations in a wide range of functional groups, and can therefore be used to characterize organic and inorganic molecules (Stevenson 1994). We hypothesize that compost persistence in soil can be evaluated using infrared spectroscopy to identify the unique functional group composition of compost and track the residence of these groups in soil over time. Fourier Transform Infrared Spectroscopy (FT-IR) is one of the most sensitive infrared techniques and has been used to identify distinct compositional features of compost and SOM (Ellerbrock et al. 1999a, 2001). Many infrared absorption bands display a higher relative intensity for compost, and are almost absent in spectra for control soils (Soler Rovira et al. 2003). These bands can therefore be used as identifying markers to determine the presence or absence of compost in soil as well as provide insight into the degree of chemical transformation as compost is decomposed.

Fourier transform infrared spectroscopy (FT-IR) has been widely used to study aspects of compost and soil organic matter. Compost stability and maturity have been well described and many studies report changes in SOM resulting from agricultural activity or amendment (Sohi et al. 2005; Ellerbrock et al. 1999ab). FT-IR has been used to describe the composting process and determine compost maturity (Chen 2003) as well as for investigating functional group changes during humification and composting (Niemeyer et al. 1992: Inbar et al. 1989). Ellerbrock et al. (1999b) found that FT-IR analysis of soils under different manural practices was useful in detecting changes in functional groups, in particular carboxylic groups, in response to manure treatments. Simon (2007) found soils amended with farm yard manure compost displayed
increased intensity of FT-IR spectra of both aliphatic and aromatic bands compared with control treatment. FT-IR was used by Gerzabek et al. (1997) to show that peat characteristics could be detected in peat treated soils in their long term field trial. Ellerbrock et al. (1999b) used FT-IR to conclude the composition of SOM is influenced by different fertilization practices. Ellerbrock et al. (1999a) found that compositional changes in SOM due to fertilization treatment were evident in IR spectra, while Celi et al. (1997) showed that the content of carboxyl groups of SOM can be determined by FT-IR spectroscopy.

Although FT-IR has been used successfully to study various compositional aspects of compost and SOM, to our knowledge, no studies have investigated compost persistence in soil following a one-time application, on certified organic land. We hypothesize that FT-IR can be used to identify functional groups displaying residual carryover arising from a one-time application of compost one, two, and three years after incorporation. In particular, we propose that spectra representing the polysaccharide band (1030 – 1150 cm$^{-1}$) will show decreased absorbance over time as these polysaccharides are preferentially decomposed.

**FT-IR Spectra**

Absorption bands represent functional groups with varying degrees of reactivity or recalcitrance. Functional groups containing organic-O are equated with SOM reactivity while C, H and/or N containing groups may be considered recalcitrant (Wander and Traina 1996). Additionally, some groups have been correlated with sorption characteristics of the soil, such as carboxylic groups and their correlation with soil CEC (Celi et al. 1997).

Carballo et al. (2008) described the main absorbance bands for composted bovine manure as 2960 – 2850cm$^{-1}$ (C-H stretch of aliphatic structures), 1620 – 1660cm$^{-1}$ (C=O vibrations of ketones, quinnoone’s, carboxylic acids and esters, as well as C=C vibrations of
aromatic components), 1430 – 1455cm\(^{-1}\) (O-H in-plane bend of carboxylic acids, CO2 stretch of carboxylates and aliphatic CH2 alkanes, and also C-O stretch vibration of carbonates), 1030 – 1150cm\(^{-1}\) (polysaccharides), 1504cm\(^{-1}\) (weak peak) and 1595cm\(^{-1}\) (vibration of the aromatic skeleton of lignin’s). These typical IR-absorption bands display a higher relative intensity for compost, and are almost absent in spectra for control soils (Soler Rovira et al. 2003). They can therefore be used as identifying markers to determine the presence or absence of compost in soil.

This thesis investigates: compost carryover resulting from a one-time application, expressed as a decay series where coefficients capture both nutritive and non-nutritive carryover and profitability of a one-time compost application at 5 rates in a corn and squash field trial over a 4-year period (Chapter II); nitrogen carryover following a one-time compost application and expressed as a decay series as well as compost persistence and decay in soil as revealed by infrared spectroscopy following a one-time application (Chapter III); and a summary of these findings (Chapter IV). This thesis is formatted in journal paper format according to Utah State University guidelines. Chapter II is formatted for submission to the Journal of Organic Agriculture and Chapter III is formatted for submission to Environmental Science and Pollution Research.

**References**


CHAPTER II

DETERMINING COMPOST CARRYOVER AND EVALUATING ECONOMIC ASPECTS OF COMPOST USE ON ORGANICALLY CERTIFIED LAND

Abstract

Organically certified farms using compost to improve or maintain fertility rarely consider compost carryover and its impact on the determination of economically optimal application rates. This study was conducted to test a new method for estimating compost carryover on organically certified land. Compost carryover for five treatment rates was estimated over four years using new additions of the amendment as a benchmark in a corn and squash rotation. Carryover was expressed in units of compost equivalents, a term encompassing both the nutritive and non-nutritive components of carryover. The effect of carryover on profitability was then considered. While year to year variability was great, the new method successfully modeled carryover and determined that compost had a persistent and positive effect on crop yields, evident even three years after an initial one-time application. Carryover terms were higher than those typically reported in the literature, indicating that our compost equivalents unit captured more than simply N-carryover. Compost use for corn silage production was not profitable, though the high-value squash crop achieved greater-than-control profitability at all treatment rates. The economically optimal compost rate for squash production was estimated to be 30 Mg DM ha\(^{-1}\). Greater N-fertility was required to optimize yields in the corn. Use of a legume green manure following squash to provide additional N, in conjunction with a 30 Mg DM ha\(^{-1}\) compost amendment to build soil fertility, may provide better fertility conditions for corn in this, or similar rotations.

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Introduction

One of the major problems facing organic farmers is determining how to best manage soil fertility for optimal crop growth. Nitrogen fertility in particular requires careful management. Organic farmers who enjoy long growing seasons rely on leguminous N-fixing cover crops to replenish soil-N between cash cropping seasons. These crops are able to fix between 112-280 kg N ha\(^{-1}\) in 45-55 days under ideal conditions (Peoples et al. 1995). Given favorable rhizobium inoculation and soil N concentrations, nitrogen fixing potential of legume crops in irrigated systems is overwhelmingly regulated by plant growth (Peoples et al. 2009). The more biomass, the greater will be the mass of N\(_2\) fixed.

However in regions with short growing seasons, such as the Inter-Mountain West, reliance upon legume cover crops as a sole source of N becomes problematic. In many locations, short growing seasons, averaging 4-5 months, limit cover crop growth between crop cycles. The cover crop therefore struggles to reach its maximum biomass potential, and its full N-fixing potential, before the cash-crop growing season begins again (Gaskell and Smith 2007).

Growers in these regions therefore cannot rely fully on N-fixing cover crops to meet cash crop N needs unless they take a whole growing season out of production. The costs from loss of productivity due to cover cropping rather than cash cropping may not outweigh the benefits of N fixation (Barker 2010). Few farmers can economically justify taking this approach. In these regions, compost is commonly used as a supplemental source of N, particularly when high value crops are grown and farm size is small.

Compost is fundamentally different from synthetic fertilizers in that it exhibits a carryover effect. Whereas inorganic fertilizers often mineralize completely within one year, compost decomposes gradually, mineralizing nutrients over many years at ever decreasing
rates. Compost also contains a wide range of plant nutrients in addition to N which are slowly released into the soil. In addition, compost influences a range of soil physical and chemical properties which provide many other crop benefits that are non-nutritive in nature. Following an initial application, these nutritive and non-nutritive benefits carryover from year-to-year, a dynamic that is seldom considered by growers. Soil fertility in a given year therefore becomes a function of the total compost applied, plus a proportion of previous years’ applications that are carried over into the current year (Endelman et al. 2010). Understanding compost carryover then is the first step in redefining fertility management in organic systems where compost forms an important part of total farm inputs.

Because compost is a relatively high cost input, both in terms of cost per ton as well as cost of application, farmers naturally seek to economically optimize compost use. Determining optimal application rates requires an understanding of, and method of accounting for compost carryover. Without an understanding of carryover, growers may base their application rates on what is affordable in any given year giving rise to potential for over and under-application. Compost over-application is not only wasteful in terms of an economic cost, but it also increases the potential for P-leaching and trace metal accumulation, which can have adverse environmental consequences (Daniel et al. 1994). Compost under-application not only fails to meet fertility goals but can result in nutrient mining of soils which is unsustainable in the long term.

The carryover behavior of compost is influenced by the chemical and physical composition of the compost itself. During the composting process, much of the readily mineralizable N and C is lost, leaving only more stable and recalcitrant N/C forms (Eghball et al. 1997). These forms are more resistant to decay and thus mineralize more gradually over
multiple years. Depending on initial N content, C/N ratio and compost maturity, typically only 10% to 30% of the total compost-N is available for plant uptake within the season of application (Amlinger et al. 2003; Eghball et al. 2002; Hadas et al. 1996; Hartz et al. 2000) and often periods of immobilization occur initially upon incorporation.

Research investigating compost carryover is sparse, and is most often associated with conventional farming systems. Because nitrogen is the most frequently limiting nutrient for crop production (Wander et al. 2011), carryover studies commonly express carryover as a decay series describing N carryover only. In the majority of these studies, yields following an inorganic fertilizer application are used as benchmarks from which to compare the yield from a compost amended plot. Carryover is then expressed in the form of an inorganic fertilizer equivalent. These studies hold little relevance for organic farmers, where inorganic fertilizer use is prohibited and where organic inputs follow a more complex mineralization dynamic. They also cannot be conducted on organically certified land.

This study seeks to test a new approach to measuring compost carryover on organic certified land (Endelman et al. 2010). Rather than expressing carryover relative to inorganic fertilizer rates, we seek to determine carryover by measuring yield relative to new applications of compost. We hypothesize that soil fertility can be expressed in terms of compost equivalents, a term which encompasses both the nutrient and non-nutrient aspects of compost carryover. We further hypothesize that by considering carryover, an economically optimal rate of application can more be accurately determined.

Methods and Materials

In the spring of 2008, two identically designed experiments were established at the Greenville research farm of Utah State University in North Logan, UT. The soil was a Millville silt
loam (coarse-silty, carbonatic, mesic Typic Haploxeroll). Values for typical soil properties are shown in Table 1. Total C and N were determined by dry combustion (LECO TruSpec C/N). Nitrate-N and ammonium-N were measured in 5:1 extracts (1M KCl) by automated colorimetry cadmium reduction method (Lachat QuickChem AE). Electrical Conductivity (EC) and pH were measured in 1:1 soil/water extracts while P and K were determined from NaHCO$_3$ extracts according to the Olsen method. The site is organically certified and had been planted with various cover crops prior to the trial period.

Four levels of compost (10, 20, 30, 40 Mg DM ha$^{-1}$) were applied in each of the three years (2008, 2009, and 2010). Each year three replicates were assigned, in a completely randomized fashion, to each treatment for a total of 12 plots each year in which compost was applied. A further four plots each year served as control plots and received no compost. Plots received compost for the first time each year and plots received compost only once over the course of the experiment. Each experiment therefore consisted of 40 plots, of which 36 (3 years × 4 treatment levels × 3 replicates) received a one-time compost application and four no compost.

Although originally designed as a three year experiment, a further year was added (2011) because of a poor treatment response observed in the critical third year (2010). In 2011 the experimental design did not allow for the addition of compost to the plots but could still be used to assess compost carryover.

Compost

Compost was purchased in bulk from Miller’s in Hyrum, UT. Miller’s ‘Premium Organic Compost’ brand was used in 2008, while ‘Millers Steer Compost’ was used in both 2009 and 2010. Select chemical analyses of the various composts are shown in Table 2. Total C and N were
determined by dry combustion (LECO TruSpec C/N). Nitrate- N and ammonium-N were measured in 5:1 extracts (1M KCl) by automated colorimetry cadmium reduction method (Lachat QuickChem AE). Electrical Conductivity (EC) and pH were measured in 1:1 compost/water extracts while P and K were determined by HNO₃/H₂O₂ digestion.

Plots were treated with one of four rates of compost (10, 20, 30, 40 Mg DM ha⁻¹). Treatment rates were calculated on a volume basis using compost bulk density. Treatments were spread evenly over the plots with a rake and incorporated with a rototiller. No other additional amendments or fertilizers were used and Table 3 provides the compost incorporation dates for each year.

Crops

Two crops were grown in rotation, summer squash and silage corn. Experiment 1 followed a squash-corn-squash-corn rotation while Experiment 2 followed a corn-squash-corn-squash rotation. Certified organic summer squash (Cucurbita pepo L.) hybrid (‘Golden Zucchini’) was used in 2008, 2009 and 2010. Conventional seed, of the same variety, was used in 2011 due to a lack of availability of the organic version. In 2008 and 2009 plants were started in the field in 50-cell flats on May 27. In 2010 and 2011 plants were started in 50-cell flats at the USU Research Greenhouses on May 13 and 23 respectively. In 2008 the potting mix contained Miller Premium Organic Compost, vermiculite, perlite, and blood meal. In all other years the potting mix was comprised of peat moss (0.22m³), perlite (0.11m³) and vermiculite (0.11m³) with the addition of 5.7L of fertilizer which was comprised of 15 parts bone meal, 10 parts blood meal, 10 parts kelp meal, and 5 parts dolomite. No synthetic fertilizers or amendments were used.

Each year in early June, three sheets of black plastic mulch (1.2m wide) were laid down the full length of the plot area (91.4m) at a spacing of 1.5m. Seedlings were transplanted into
the mulch at a spacing of 0.61m between plants within a row with 2 rows spaced 0.61m apart. Thus each sheet of mulch contained two rows of plants in a staggered pattern. Plant density was 21,500 plants ha⁻¹. See Table 3 for the transplant dates of each year. A severe windstorm occurred on June 13, 2010 and the majority of the crop was lost. On June 15, 2010 a replacement crop was direct seeded into the plastic mulch and the damaged plants removed. Weeds were controlled between mulch rows with a walk behind stirrup hoe and 18in rototiller. Overhead sprinklers on 1.82m (6ft) risers were used in all years. Plots were irrigated once per week for four hours duration at a rate of 2.48cm hr⁻¹ for a total 9.94cm week⁻¹. No signs of water stress were observed.

Squash fruit were picked twice per week for four weeks. See Table 3 for the first harvest date of each year. All fruit larger than 15cm in length were harvested and fresh weights were recorded from 6 plants within the center row of each plot to minimize potential boundary effects. The average cumulative harvest weight per plant was calculated and then this number was scaled to a 1 hectare basis using the density of 21,500 plants ha⁻¹.

The second crop was a certified organic field corn (Zea mays L.) hybrid (Dahlco 2146). Seeds were drilled 91.4m (300 ft.) long, with rows spaced 0.76m (30 in.) apart for a total of 6 rows per plot. See Table 3 for seeding dates for each year. The crop was not systematically thinned and emerged with an average density of 100,000 plants ha⁻¹. Weeds were controlled within row by hand, and between row by a combination of rototiller and walk-behind wheel hoe.

Corn plants were harvested by hand at approximately 30% dry matter. Data was collected from plants within a 1.5m × 1.5m (5 ft. × 5 ft.) area within the center of each plot. Corn ears were removed and fresh weights recorded for both corn stalks and ears. See Table 3 for
harvest dates for each year. Ears and stalks were dried at 50°C for a minimum of 14 days and ears shelled. Dry weights were recorded for both stalks and grain. The average grain and silage yield per plot was calculated and then scaled to a 1 hectare basis using the density of 100,000 plants ha⁻¹.

Cover Crop

A hard red winter wheat (*Triticum aestivium*) cover crop was seeded at a rate of 28 kg ha⁻¹ on all plots following corn harvest in the fall of 2008 and 2009. The corn crop was allowed to over-winter before being tilled under in the following spring. In 2010 hairy vetch (*Vicia villosa*) was planted in combination with winter wheat and seeded at the rate of 30 kg ha⁻¹ for vetch and 28 kg ha⁻¹ for the wheat. Cover crops were first mowed and then incorporated using a rototiller on May 20 in 2009, May 27 in 2010, and May 27 in 2011.

Biomass samples were collected from the winter wheat cover crop just prior to incorporation in the spring of 2009 and 2010. Two plots were selected for each compost rate (10, 20, 30, and 40 Mg ha⁻¹) and three plots for the zero-rate. Three sub-samples were collected from 30 × 60cm areas within the center of each plot and pooled to make one representative sample per plot. Fresh weights were recorded before the samples were oven-dried at 50°C for 7 days and dry weight determined.

Determining Compost Carryover

Methodology for determining compost carryover and quantifying decay series terms is based on that described by Endelman et al. (2010). Central to this method is an experimental design whereby some plots receive compost for the first time each year and any given plot receives compost only once over the course of the experiment. By design, in years two (2009)
and three (2010), some plots received compost for the first time while others received compost in previous years. Current-year amended plots have no carryover and therefore serve as benchmarks from which to determine carryover for plots amended previously.

This project was extended by one year because of the poor treatment response seen in year 3 (2010). In 2011 our experimental design did not allow for the addition of compost to any new plots. Yields for this year were in response to compost applications in the one, two, and three years previously.

In the original model yield (Y) can be related to soil fertility (F) by Eq. [2.4], where (m) is the slope of the relationship and (k) the intercept. Fertility (F) is described in units of compost equivalents, a term representing both nutritive and non-nutritive factors.

\[ Y = mF + k \]  

Equation [2.4]

Soil fertility in a given year is dependent not only upon the current year’s compost rate but also on carryover from previous years’ applications. Soil fertility (F) is therefore a function of compost carryover as expressed in Eq. [2.5] where \( \beta_x \) terms represent carryover coefficients, and \( C_t \) terms represent compost application rates at time ‘t’. Note that current-year applied compost serves as a benchmark and is therefore given a coefficient of 1 (\( \beta_0 = 1 \)). The relationship between yield (Y) and fertility (F) can then be expressed by Eq. [2.6].

\[ F = \beta_0 (C_t) + \beta_1 (C_{t-1}) + \beta_2 (C_{t-2}) \]  

Equation [2.5]

\[ Y = k + m [ (C_t) + \beta_1 (C_{t-1}) + \beta_2 (C_{t-2}) + ... ] \]  

Equation [2.6]

In years where compost was applied (2008-2010), coefficient (\( \beta_x \)) terms were determined using PROC NLIN in SAS (version 9.3). Data was transformed where necessary, so
that model assumptions were met. Initial parameter estimates were generated using PROC REG in SAS (version 9.3).

In 2011, where no compost was applied, the coefficient $\beta_1$ could not be determined. In this case, the carryover coefficients ($\beta_x$) were evaluated in terms of their ratios to one another. Firstly a new variable ($\alpha$) was defined, where $\alpha = m \times \beta_x$ (model slope $\times$ carryover at year $x$). The yield/fertility relationship Eq. [2.6] was then expressed as Eq. [2.7]

\[
Y = k + \alpha_1(C_t) + \alpha_2(C_{t-1}) + \alpha_3(C_{t-2})
\] [2.7]

Conclusions about the change in carryover coefficients can now be made by evaluating the ratio of the $\alpha_x$ terms to each other (ie. comparing the ratio $\alpha_3 / \alpha_2$ to $\alpha_2 / \alpha_1$). Because $\alpha_3 / \alpha_2$ and $\alpha_2 / \alpha_1$ are numerically equal to $\beta_3 / \beta_2$ and $\beta_2 / \beta_1$ respectively, these ratios tell us something of the magnitude of the change in carryover each year. These ratios can be further simplified by Eq. [2.8] and the yield/fertility model expressed as Eq. [2.9]. So that $d_2$ represents second year carryover as a proportion of $\beta_1$. Likewise, $d_3 \times d_2$ represents third year carryover expressed as a proportion of $\beta_1$.

\[
d_x = \frac{\beta_x}{\beta_{x-1}}
\] [2.8]

\[
Y = k + \alpha_1 [(C_{t-1}) + (d_2 \times C_{t-2}) + (d_3 \times d_2 \times C_{t-3})]
\] [2.9]

Terms in the yield model Eq. [2.9] were then determined using PROC NLIN in SAS (version 9.3), with initial parameter estimates generated using PROC REG in SAS (version 9.3).

Statistical Analyses

Cover Crop Biomass. Statistical analysis of winter cover crop biomass was made by first expressing biomass (B) as a linear function of fertility (F) in Eq. [2.1]. Fertility is dependent on
the decay of compost over time as expressed in Eq. [2.2]. Biomass is then expressed as a function of carryover by Eq. [2.3] and PROC NLIN (SAS version 9.3) was used to solve for coefficient terms. Analysis of residuals identified plot 218 as an outlier, and it was removed from the model. Initial parameters for PROC NLIN were generated by PROC REG (SAS version 9.3).

\[
B = mF + k \quad [2.1]
\]

\[
F = \beta_0(C_t) + \beta_1(C_{t-1}) + \beta_2(C_{t-2}) \quad [2.2]
\]

\[
B = k + m [ (C_t) + \beta_1(C_{t-1}) + \beta_2(C_{t-2}) + \ldots ] \quad [2.3]
\]

**Electrical Conductivity and Soil pH.** Electrical conductivity and soil pH was measured in soils (0-30cm) sampled in July 2011. All plots which had received compost treatments the previous year (2010), as well as all control plots were sampled and measured. Analysis of variance of electrical conductivity and soil pH was conducted using PROC GLIMMIX (SAS version 3.1) in a single factor design where compost rate was the only factor. All model assumptions were met.

**Profitability Study**

A profitability study was conducted for both squash and corn silage using yield data from all experimental years and estimates of costs and revenues based on literature review. Yield data was averaged by rate for within-year, first-year, second-year and third-year carryover response. For example, to arrive at the within-year totals, yields were averaged by rate across years for all plots which received compost in that year. Likewise, first-year carryover totals are average yields from plots in 2009, 2010, and 2011 which received compost one year previously. Second-year carryover totals are average yields from 2010 and 2011 plots which received compost two years previously, while third-year carryover totals are average yields from 2011 plots which received compost three years previously. Yields were multiplied by market price to
determine gross profit. Within-year net profits reflect sales revenue minus operating expenses including compost expenses which increase with rate. Average net profits for the years following a compost application, do not incur a compost cost and therefore are adjusted by a yearly operating expense without the additional compost cost. Compost cost was calculated as $104 per Mg DM using average purchase price ($34 m$^{-3}$) and assuming an average compost bulk density of 0.56 Mg m$^{-3}$.

In order to assess profitability based solely on compost rate decisions, a number of assumptions were made concerning costs and market conditions. Costs were assumed to remain constant over the four years of the study. No adjustments were made for inflation or depreciation. The market price received for squash and corn silage was also fixed over the four years. It was further assumed that 100% of the yield was marketable.

Squash establishment and harvest costs were estimated based on a budget prepared by Drost and Ward (2011) for squash production in high tunnels, in conjunction with observed actual costs recorded each year of the study. Expenses associated with high tunnel production were removed and costs scaled to a per hectare basis (Table 4). Squash revenue was determined based on a sales price of $2 kg which was the typical wholesale price of organic squash received from sales to local restaurants and at a local market in 2011.

Corn silage establishment and harvest costs were estimated based on corn silage budgets prepared by Massey (2011), Benson and Green (2008), and University of Tennessee Extension (2007). Costs associated with conventional farming practices, such as fertilizer and herbicide costs, were removed from these budgets and additional costs of weed control by cultivation were included instead (Table 5). Costs were scaled to a per-hectare basis. Yield data, which was recorded as Mg DM ha$^{-1}$, was converted to Mg ha$^{-1}$ at 65% moisture content to reflect
actual market conditions. Commodity prices for corn silage are highly variable by year. Revenues were determined for silage based on a price of $40 ton ($44 Mg) which was a typical market price in 2011.

**Results**

*2011 Season*

No compost was applied during the 2011 growing season, so determination of a within-season yield response was not possible. Compost carryover was instead evaluated as a proportion of first year carryover as described above.

Squash yields in 2011 displayed a linear response to treatment although there was a great deal of variability between plots (Figure 1). Average yields were higher in 2011 than in previous years (Table 6). Ratios of carryover coefficients for, two and three year carryover ($d$, terms) were estimated to be: $d_2=0.86$ (±0.38) and $d_3=0.70$ (±0.43) with a slope of $a_1=0.22$ (±0.09) and an intercept $k=26.49$ Mg DM Ha$^{-1}$ (±1.92) (Table 7). The overall 2011 yield model is shown by Eq. [2.10].

$$Y = 26.49 + 0.22 \left[ (C_{t-1}) + (0.86 \times C_{t-2}) + (0.81 \times 0.86 \times C_{t-3}) \right]$$  [2.10]

These estimates indicate that in 2011, 2009 carryover was 86% of 2010 carryover, while 2008 carryover was 70% of 2010 carryover (0.81 × 0.86). In this case 2010 is the benchmark year and has a fertility rating of 100%. In terms of fertility, the 2009 ratio equates to a 14% decrease (decay) in fertility compared with 2010-amended plots (100% benchmark minus 86%) while the 2008 ratio equates to a 30% decrease (decay) in fertility compared with 2010-amended plots.

Corn yield (Mg DM ha$^{-1}$) was measured as both silage and grain. Yields for the 2011 season, for both silage and grain, were greater than those of the 2009/2010 seasons, and were
similar to the high yields seen in 2008 (Table 6). Silage yield displayed a curvilinear response to treatment for all treatment years (Figure 2), which was best described by a polynomial model of form \( y = Ax^2 + x + k \).

Ratios of carryover coefficients (\( d_k \) terms) for corn silage were estimated to be: \( d_2 = 0.55 (\pm 0.57) \) and \( d_3 = 2.08 (\pm 2.14) \) with a slope of \( a_1 = 0.07 (\pm 0.05) \) and intercept \( k = 11.77 \text{ Mg DM Ha}^{-1} (\pm 1.01) \) (Table 7). The overall 2011 yield model for corn silage is shown by Eq. [2.11].

\[
Y = 11.77 + 0.07 \left[ (C_{t-1}) + (0.55 \times C_{t-2}) + (2.08 \times 0.55 \times C_{t-3}) \right] \quad [2.11]
\]

These estimates indicate that in 2011, 2009 carryover was 55% of 2010 carryover, while 2008 carryover was 114% of 2010 carryover (2.08 \times 0.55), though the standard error was large for both \( d_2 \) and \( d_3 \) terms. In terms of fertility, the 2009 ratio equates to a 45% decrease (decay) in fertility compared with 2010 amended plots. The 2008 ratio equates to a 14% increase in fertility compared with 2010-amended plots.

Corn grain yield for the 2011 season displayed a curvilinear response to treatment in all years (Figure 3). A large amount of variability was seen between plots. Yield data was best described by polynomial model of form \( y = Ax^2 + x + k \). Ratios of carryover coefficients (\( d_k \) terms) for corn grain were estimated to be: \( d_2 = 0.87 (\pm 0.69) \) and \( d_3 = 1.32 (\pm 0.98) \) with a slope \( a_1 = 0.02 (\pm 0.17) \) and an intercept of \( k = 2.64 \text{ Mg DM Ha}^{-1} (\pm 0.35) \) (Table 7). The overall 2011 yield model for corn grain is shown by Eq. [2.12].

\[
Y = 2.64 + 0.02 \left[ (C_{t-1}) + (0.87 \times C_{t-2}) + (1.32 \times 0.87 \times C_{t-3}) \right] \quad [2.12]
\]

These estimates indicate that in 2011, 2009 carryover was 87% of 2010 carryover, while 2008 carryover was equivalent to (114%) of 2010 carryover (1.32 \times 0.87), though the standard
error was large for this $d_3$ term. In terms of fertility, the 2009 ratio equates to a 13% decrease (decay) in fertility compared with 2010 amended plots. The 2008 ratio equates to fertility levels 14% greater than those of 2010-amended plots.

2010 Season

The 2010 season was a poor one for both squash and corn. April received at least 25cm more snowfall than in other experimental years, while average May temperatures were colder and total precipitation was greater than all but 2011(Table 8). Additionally, a severe windstorm on June 13 caused extensive damage to recently transplanted squash seedlings, requiring a complete re-seeding of the squash one week later. As a consequence yields for both squash and corn were lower than in 2011 and no linear treatment response was seen (Figure 4 a, b, c). Variability amongst the plots was too great and due to the lack of treatment response, carryover estimates could not be determined for the 2010 season.

2009 Season

Yields for corn and squash were similar to those seen in the 2010 season, however 2009 plots showed a yield response to the compost treatments and carryover was determined using the method developed by Endelman et al. (2010), and described above. A within-year response to treatment was also reliably measured in 2009.

Squash yields in 2009 displayed a linear response to treatment, though there was a great deal of variability between plots particularly at the higher rates of 30 and 40 Mg DM ha$^{-1}$ (Figure 5). Yields were greater than control for current-year amended plots; however 2008 amended plots showed no difference in yield from control plots.
Estimates of yield model terms revealed a within-year response to be slope \(a_1=0.10\) (±0.03), while the estimated one-year carryover term was \(b_1=-0.08\) (±0.32) and model intercept \((k)\) was 9.58 Mg DM ha\(^{-1}\) (±0.51) (Table 9). The overall 2009 yield model for squash is shown by Eq. [2.13].

\[
Y = 9.58 + 0.10 \left[ \left( C_t \right) - 0.08 \left( C_{t-1} \right) \right]
\]  \hspace{1cm} [2.13]

In terms of fertility, the \(b_1\) term indicates that in 2009, there was no residual fertility carryover in 2008-amended plots. In terms of the overall model, and the effect on yield, a one unit change in 2009 compost rate results in a 0.10 unit increase in yield, while compost applied in 2008, did not affect 2009 yields.

Corn Silage yields in 2009 displayed a linear response to treatment (Figure 6). Yields were greater than control for current-year amended plots, particularly at highest rate 40 Mg DM ha\(^{-1}\), however 2008-amended plots showed no difference in yield from control.

Estimates of yield model terms revealed within-year response to be slope \(a_1=0.15\) (±0.01), while the estimated one-year carryover term was \(b_1=0.34\) (±0.09) and model intercept \((k)\) was 4.47 Mg DM ha\(^{-1}\) (±0.26) (Table 9). The overall 2009 yield model for corn silage is shown by Eq. [2.14].

\[
Y = 4.47 + 0.15 \left[ \left( C_t \right) + 0.34 \left( C_{t-1} \right) \right]
\]  \hspace{1cm} [2.14]

In terms of fertility, the \(b_1\) term indicates that 2008-amended plots had a fertility carryover equivalent to 34% of the rates applied in 2009. In terms of the overall model, and the effect on yield, a one unit change in 2009 compost rate results in a 0.15 unit increase in yield.
plus an additional 0.05 unit increase in yield (0.15 × 0.34) from every unit of compost applied in 2008.

In 2009 corn grain yields displayed a similar trend to silage yield with a linear response to treatment (Figure 7). Yields were greater than control for current-year amended plots, particularly at rates 20, 30, and 40 Mg DM ha\(^{-1}\); however 2008 amended plots showed no difference in yield from control.

Estimates of yield model terms revealed a within-year response to be slope \(a_1 = 0.05\) (±0.01), while the estimated one-year carryover term was \(b_1 = 0.28\) (±0.09) and model intercept (\(k\)) was 0.78 Mg DM ha\(^{-1}\) (±0.26) (Table 9). The overall 2009 yield model for corn silage is shown by Eq. [2.15].

\[
Y = 0.78 + 0.05 \left(C_t \right) + 0.28 \left(C_{t-1}\right)
\]  

[2.15]

In terms of fertility, the \(b_1\) term indicates that 2008-amended plots had a fertility carryover equivalent to 28% of the rates applied in 2009. In terms of the overall model, and the effect on yield, a one unit change in 2009 compost rate results in a 0.05 unit increase in yield plus an additional 0.01 unit increase in yield (0.05 × 0.28) from every unit of compost applied in 2008.

**Electrical Conductivity and Soil pH**

Electrical conductivity was measured to be 296µS/cm in 2007 baseline soils (Table 1), and by 2011 was in the range 281-325µS/cm despite a one-time compost addition during that 4-year period. Electrical conductivity was not statistically different between treatment rates (\(\alpha = 0.05\)) (Figure 8). Soil pH was measured at 8.04 in 2007 (Table 1), and was in the range 8.00-8.03 in 2011 and was not statistically different between treatment rates (\(\alpha = 0.05\)) (Figure 9).
**Winter Wheat Cover Crop**

Winter cover crop biomass was measured in 2010 in response to compost applied 12 months earlier in 2009 and represented the within-year response. Compost applied in 2008 and carried over into the 2009/2010 winter represented the one-year carryover. Biomass yields from 2009-amended plots were no different from yields recorded in 2008-amended plots (Figure 10). A large standard error was apparent at all rates. Nevertheless, biomass yield could be modeled using the methodology described above. Estimates of yield model terms revealed within-year response to be slope $a_1 = 0.01 \ (\pm 0.004)$, while the estimated one-year carryover term was $b_1 = 1.28 \ (\pm 0.45)$ and model intercept ($k$) was $1.87 \text{ Mg DM ha}^{-1} \ (\pm 0.08)$. The overall 2010 winter wheat yield biomass model is shown by Eq. [2.16].

\[ Y = 1.87 + 0.01 \ [ ( C_t ) + 1.28 \ ( C_{t-1} ) ] \]  \hspace{1cm} [2.16]

In terms of fertility, the ($b_1$) term indicates that 2008-amended plots had a fertility carryover equivalent to 28% of the rates applied in 2009, though the high degree of yield variability make this term less reliable. In terms of the overall model, and the effect on yield, a one unit change in 2009 compost rate results in a 0.01 unit increase in yield plus an additional 0.01 unit increase in yield ($0.01 \times 1.28$) from every unit of compost applied in 2008.

**Profitability Study**

Corn Silage yields did not always show a significant response to treatment and revenue was not great enough to overcome the significant cost of compost inputs. Corn operating expenses, without compost cost, were $587 \text{ ha}^{-1}$ (Table 5). Total compost expense was dependent on rate and ranged from $1627 for the low rate (10Mg DM ha$^{-1}$) to $4747 for the high rate (40Mg DM ha$^{-1}$). When this expense was included, operating costs eclipsed silage
revenue in the year of application (Table 10). After four years, total combined profits were
greatest for the control treatment (Table 10). Overall, as rate increased, profitability decreased
until at the higher rates of 30 and 40 Mg DM ha\(^{-1}\), losses were incurred (Table 10).

Squash treatment response was great enough that profits were improved with compost
application, even though operating costs were much higher than for corn. The 30 Mg DM ha\(^{-1}\)
rate returned the greatest 4-year combined profit (Table 11). Operating expenses, minus
compost cost, were $11,749 for one hectare production (Table 4). When compost expense was
included, only the 30Mg DM ha\(^{-1}\) rate resulted in a profit greater than un-amended treatments
in the year of application (Table 11). Overall, after four years total combined profits were
greater than control treatments at all application rates and profitability peaked at $128,988
during the 4 years for the 30Mg DM ha\(^{-1}\) treatment (Table 11). A decrease was seen in total four
year combined profit when rate increased from 30 to 40 Mg DM ha\(^{-1}\).

Discussion

Composts are commonly used on organic farms to maintain or improve fertility. Unlike
inorganic fertilizers, which mineralize rapidly, compost persists in the soil to provide benefits,
both nutritive and non-nutritive, which carry-over for many years. Soil fertility in a given year is
therefore a function of the total compost applied in that year, plus a proportion of previous
years’ applications that are carried over into the current year. Understanding compost
carryover then is the first step in redefining fertility management in organic systems where
compost forms an important part of total farm inputs.

Due to their relatively high cost, farmers naturally seek to economically optimize their
compost use by estimating an optimal application rate. This requires an understanding of, and
method of accounting for, compost carryover. This study investigated compost carryover in a
corn/squash rotation over a four year period, and expressed compost carryover in units of compost equivalents, a term incorporating both the nutrient and non-nutritive components of carryover.

In traditional studies of compost carryover, yields following an inorganic fertilizer-N application are used as benchmarks from which to compare the yield from a compost amended plot (Cusick et al. 2006). Carryover is then expressed as a percentage representing an inorganic fertilizer-N equivalent. Unfortunately these studies often have little relevance to organic farmers, where inorganic fertilizer use is prohibited and where organic inputs follow a more complex mineralization dynamic. Additionally, by focusing solely on N-mineralization, these studies neglect the many non-nutritive contributions of compost.

This study further evaluated the approach to measuring compost carryover on organic certified land developed by Endelman et al. (2010). Rather than expressing carryover relative to inorganic fertilizer rates, or nutrient content, Endelman et al. (2010) proposed a method for determining carryover by measuring yield relative to new applications of compost. This methodology expresses carryover in units of compost equivalents, a term which captures both the nutrient and non-nutrient carryover of compost.

Our findings demonstrated that the methodology developed by Endelman et al. (2010) can be employed to determine compost carryover on organically certified land. Results from 2009 showed that a within-year response and a one-year carryover could be reliably estimated. Problems arose, however, when variability in growing conditions and compost composition, as well as low N-fertility, produced a non-responsive yield result in 2010. This necessitated the introduction of additional terms into the model in order to describe the carryover effect. In the fall of 2010 it was decided to extend the experiment an additional year, and to plant a hairy
vetch cover crop (*Vicia villosa*) to overwinter and boost fertility in the spring. Undersander et al. (1990) noted that a well nodulated hairy vetch crop can enrich the soil with 60-120lb/acre of N (67-134 kg ha\(^{-1}\)). Vetch biomass was not recorded in 2011 so it is unclear how much N was added to the system upon incorporation. However, cover crop biomass collected in 2009 and 2010 showed a linear response to compost rate, so it is likely that the vetch responded to treatment similarly and therefore fixed more N in plots which had previously received higher rates of compost. The 2011 season showed a marked improvement in yields with squash yields the highest of all years, and corn yields similar to the high yields seen in 2008. Given there was no compost treatment in 2011, yields in this year most likely benefited from the improved N-fertility which resulted from incorporation of the vetch cover crop.

As 2010 was designed as the final year of the experiment, there were no plots remaining to apply compost to for the first time therefore the 2011 model was unable to express carryover as a proportion of within-year response. Even so, the model was still able to measure changes in compost carryover, instead expressing them as proportions of first year carryover. Although absolute carryover terms were not estimated, the magnitude of change in carryover over a three year period was determined and provided valuable insight into the persistence of compost in soil.

Yields of amended plots were generally higher than non-fertilized plots for both squash and corn, particularly at the higher treatment rates. This was true even in plots which had been amended three years previously, implying that a one-time compost application provided long term or persistent residual benefits to productivity. Year-to-year variability was great, possibly due to temperature and moisture variations across years, and treatment response depended in large part on which year yields were recorded. Even at the high treatment rate (40 Mg DM ha\(^{-1}\))
maximum yield was not reached for corn, based on county averages. Cache County yield 
averages for corn silage are in the range 18-21 Mg DM ha\(^{-1}\) (Griggs et al. 2005, 2006; Griggs and 
Israelsen 2007). Typical yields of summer squash (zucchini) range from 20.4 Mg ha\(^{-1}\) to 37.7 Mg 
ha\(^{-1}\) (Goldy and Wendzel 2009; Molinar et al. 2005). By comparison, the maximum average 
across all compost levels, of corn silage yield in this trial was 13.65 Mg ha\(^{-1}\) recorded in 2008, 
and the maximum average across all compost levels, for squash was 30.83 Mg ha\(^{-1}\) achieved in 
2011. Regardless, since no yield plateau was observed for either crop, it can only be concluded 
that soil fertility was deficient in some area and compost applied at these rates was insufficient 
to maximize yields.

*Estimates of Compost Carryover*

Corn and squash yields in 2011 were modeled, and the Endelman et al. (2010) 
methodology applied to estimate compost carryover terms. Because no within-year compost 
response was available, as no compost was applied in 2011, the model ratio coefficients 
expressed compost carryover as a proportion of first-year carryover. Model error was large, 
particularly as time from treatment increased, indicating that there was little difference 
between the terms. Squash and corn followed a similar trend whereby the carryover coefficients 
were similar regardless of whether compost had been applied one, two or three years 
previously. Squash terms were \([d_2=0.86] and [d_3 \times d_2=0.70]\), while corn silage terms were 
\([d_2=0.55] and [d_3 \times d_2=1.14]\) and corn grain terms were \([d_2=0.87] and [d_3 \times d_2=1.14]\) (Table 7). 
These estimates represent relatively large proportions of first year carryover, illustrating that 
following the year of amendment compost displays a steady and residual carryover similar in 
magnitude after three years. The 2011 results indicate that beyond the year of incorporation, 
yield response is not dependent on when the compost was applied. Given the degree of error
noted, yield response was generally the same for plots regardless of whether they were fertilized one, two, or three years previously. Significantly, this response was greater than that seen in control plots, indicating that the compost was providing a persistent carryover benefit up to three years after initial application.

The 2011 yield model enabled us to estimate second and third-carryover and express it as a proportion of first-year carryover. However, the model was unable to estimate a value for this first-year carryover term, as we had no within-year response to compare it to. For an estimate of this one-year carryover term, we can look to 2009 yield data. Corn and squash yields for this year were modeled and the Endelman et al. (2010) methodology applied to determine the carryover resulting from compost applied one year earlier in 2008.

In 2009, a strong yield response was seen in corn and one-year carryover terms were estimated to be $b_1=0.34$ and $b_1=0.28$ for silage and grain respectively. If we assume these estimates are representative, we can then use them in conjunction with our 2011 model results to describe compost carryover over three years. For corn silage, using $b_1=0.34$ as our first-year term, our carryover decay series can be expressed 0.34, 0.18 and 0.21. Applying a similar methodology to corn grain, the carryover decay series becomes 0.28, 0.24, and 0.28. The unit of carryover is compost equivalents, where each term in the decay series expresses carryover as being equivalent to some proportion of compost applied in the current year. Importantly, by expressing compost carryover in units of compost equivalents, rather than fertilizer-N equivalents, we are accounting for all of the nutritive and non-nutritive benefits of compost which affect fertility.

In terms of published results, and given the degree of error seen in this study, these estimates are similar to those reported by Endelman et al. (2010). In a similarly designed
experiment, Endelman et al. (2010) determined a decay series of manure slurry to be the equivalent of 0.21, 0.16 and 0.13 units of the new slurry after 1, 2, and 3 years respectively. Decay terms in this study, and the Endelman et al. (2010) experiment describe carryover in units of compost equivalents, which differs from how carryover is typically reported in the literature. Typically compost carryover is described in terms of N-fertilizer equivalents.

Using N-fertilizer as a benchmark, Klausner et al. (1994) estimated decay series values of 0.10, 0.03, 0.03 and 0.02 for the second to fifth years after application. Cusick et al. (2006) investigated dairy manure N-mineralization and reported second and third year carryover of 9-12% and 3-5% of the original manure application respectively, while Amlinger et al. (2003) reported carryover of 2-8% of remaining compost-N in the second and subsequent years. Our carryover estimates, described above, are considerably higher than these, and this difference may be attributed to how the carryover is measured. Both Klausner et al. (1994) and Cusick et al. (2006), measured N carryover only, while our carryover estimates describe fertility in terms of total nutrient and non-nutrient contributions, resulting in higher carryover estimates. It is unclear how much of the carryover seen in this study can be attributed to non-N sources, though yield increases ranging from 5 and 25% have been attributed to these effects (Magdoff and Amadon 1980; Schroder and Dilz 1987).

It is clear though that expressing fertility in terms of compost equivalents gives organic growers a more accurate estimate of total carryover, than the commonly used N-fertilizer equivalent method, which tends to underestimate carryover because it does not consider non-nutritive carryover. This alone, has implications for on-farm budgets by allowing growers to more accurately determine compost application rates and enabling them to reduce instances of over-application.
**Profitability Study**

Yield responses, over four years, to an initial one-time compost application were used in conjunction with typical crop operating budgets and revenue projections to determine the effect of compost carryover on profitability. Lack of a yield plateau hindered our ability to formally model the economically optimal rate however over our treatment range we were able to identify carryover effects and apply these to estimate simple budgets.

Corn did not always show a strong yield response to compost treatment in any year and as such it is hard to justify compost use in this crop. The revenue received for silage was low, compared with squash, and the additional cost of compost use represented a significant proportion of the overall establishment cost. This cost was not outweighed by the returns. Corn silage prices have increased dramatically over the past few years, with the 2011 price ($44/Mg) being relatively high compared with the average of $15-25 seen prior to 2009. Considering this, the case for compost use may be even less appealing.

Where high value crops are grown, compost use makes more sense economically. Squash showed a linear response to treatment at all rates (over control), and profitability was improved with compost use. Even applied one year in four, yields were higher in all years over non-amended plots, justifying compost use. In terms of an economically optimal rate, the 30 Mg DM ha$^{-1}$ rate was identified as the level at which profit was maximized over the 4 year period. Applications at rates higher than this did not result in improved profits due to diminishing response from unit input as well as increased costs of purchase and application. Over the four-year trial, total profitability of treatment plots was twice that of non-amended plots at rate 30 Mg DM ha$^{-1}$.
Future studies should include higher application rates in order to identify the rate at which yield begins to plateau. This would enable marginal profits to be determined and the economically optimal rate to be formally modeled.

**Crop Rotation**

This study investigated compost carryover in a corn/squash rotation, two crops which are both commonly grown in Utah. Given this rotation, we found it difficult to maintain adequate soil fertility for corn with only a one-time compost application and a yearly winter wheat cover crop over the 4-year period. Clearly more N-fertility was needed in this rotation. Squash responded well to spring compost amendments, unlike the corn which has a higher N demand and would have benefited from additional N-fertilization prior to seeding. Compost carryover was simply not great enough to sustain a corn crop following squash. Even within-year rates were not high enough.

Nitrogen was certainly a limiting factor in our rotation, and changes should be made to increase N-fertility in the system. Ideally, a legume cover crop could be grown each year to fix N and supplement the compost. Given that squash requires a shorter growing season than corn, there is opportunity for the grower to gain a real benefit from a post-squash winter cover crop, such as hairy vetch, that fixes N. By late spring the vetch would approach maximum biomass, potentially fixing close to 134kg N ha\(^{-1}\) (Undersander et al. 1990) in readiness for the corn rotation. A vetch cover crop following corn should also be considered. We saw in this study that although the vetch did not achieve maximum biomass following corn, it was still sufficient to adequately fertilize the following squash crop. Squash yields in 2011, following corn and a vetch cover crop and where no compost was applied, were the highest of the study. It is unclear, however, what proportion of this was due to the favorable seasonal growing conditions of 2011.
Compost nevertheless represents an important part of fertility management in this rotation. Compost applications should be considered bi-annually or even every few years, and incorporated in the late spring before the squash rotation. This would provide necessary P, K, S and micronutrients to the system, building non-N fertility which could also carryover to the following year. In addition, compost builds SOM, and regular use can lead to greater non-nutritive benefits over time. This study determined the most cost-effective rate of compost application to be 30 Mg DM ha\(^{-1}\) for squash production. To make this organic system viable, a high value crop such as squash must form part of the rotation. This crop enables the grower to incorporate compost in their fertility plan and still realize a profit. To rely on green manures and cover crops solely, is to neglect the nutritive as well as the many non-nutritive benefits arising from improvements to SOM arising from compost use. Compost is the most economically viable way to incorporate a wide range of macro and micro nutrients in to the soil, and without it growers run the risk of mining their soils of these nutrients.

**Conclusion**

The methodology developed by Endelman et al. (2010) was successfully used to estimate compost carryover on organically certified land. Importantly, this method differs from established methods of expressing carryover because it describes carryover as being equivalent to some proportion of current year compost applications. This captures all of the carryover, both nutritive and non-nutritive, unlike more common N-fertilizer equivalent methods which are concerned with N only. The resulting carryover terms are higher than those typically described in the literature for N-fertilizer equivalents, indicating the method accounts for more than simple N-carryover.
Although a one-time compost amendment can improve crop yields many years after the initial application, it is clear that N-status of the soil needs careful management in organic crop rotations. At our treatment rates, compost was unable to supply adequate N for maximum crop growth and yield. Organic crop rotations should rely on an alternate source of N, such as an N-fixing cover crop to provide the bulk of N-fertility. However, the persistent residual carryover attributable to compost indicates that the long term benefit of compost may be as a source of non-N carryover. In organic farming systems where input costs are high, compost may be the most economical means for improving or maintaining non-N fertility.

In order to profitably incorporate compost into fertility management, growers must be able to cover the high initial costs associated with compost use. Choice of cash-crop needs careful consideration. In the Intermountain West, high value crops with a relatively short growing season, such as squash, can be used in rotation with traditional field crops, like silage corn, to enable growers to apply compost profitably.

In organic fertility management, compost is an important and economical, source of non-N fertility, which benefits crop yield many years after incorporation. When used with a dedicated N-fixing cover crop in a rotation that includes a high value cash-crop, complete fertility goals can be met in a sustainable manner.

References


Table 1 Baseline soil properties. Measured (0-30cm) prior to the commencement of the study in 2007

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Soil test value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic Carbon %</td>
<td>1.3</td>
</tr>
<tr>
<td>Total Nitrogen %</td>
<td>0.15</td>
</tr>
<tr>
<td>Phosphorus mg/kg</td>
<td>5.9</td>
</tr>
<tr>
<td>Potassium mg/kg</td>
<td>143</td>
</tr>
<tr>
<td>pH</td>
<td>8.04</td>
</tr>
<tr>
<td>EC µS/cm</td>
<td>296.7</td>
</tr>
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</table>
Table 2 Compost properties. Concentrations reported on a dry weight basis

<table>
<thead>
<tr>
<th>Property</th>
<th>2008 compost</th>
<th>2009 compost</th>
<th>2010 compost</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dry matter %</td>
<td>58.0</td>
<td>56.6</td>
<td>58.0</td>
</tr>
<tr>
<td>Total N %</td>
<td>1.9</td>
<td>2.3</td>
<td>1.6</td>
</tr>
<tr>
<td>C/N</td>
<td>11.0</td>
<td>15.6</td>
<td>14.7</td>
</tr>
<tr>
<td>Nitrate N (mg/kg)</td>
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<td>256</td>
<td>11</td>
</tr>
<tr>
<td>Ammonium N (mg/kg)</td>
<td>1000</td>
<td>363</td>
<td>1740</td>
</tr>
<tr>
<td>Olsen P (mg/kg)</td>
<td>1000</td>
<td>1900</td>
<td>4300</td>
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<tr>
<td>Olsen K (mg/kg)</td>
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<td>8600</td>
<td>13,500</td>
</tr>
<tr>
<td>pH</td>
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<td>7.7</td>
<td>9.3</td>
</tr>
<tr>
<td>EC (mS/m)</td>
<td>6.0</td>
<td>6.3</td>
<td>8.7</td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>2009</td>
<td>2010</td>
</tr>
<tr>
<td>--------------------------------</td>
<td>-------</td>
<td>-------</td>
<td>-------</td>
</tr>
<tr>
<td>Compost Applied</td>
<td>May 15</td>
<td>May 28</td>
<td>May 27</td>
</tr>
<tr>
<td>Squash Transplant</td>
<td>June 15</td>
<td>June 15</td>
<td>June 8*</td>
</tr>
<tr>
<td>Squash First Harvest</td>
<td>July 3</td>
<td>July 20</td>
<td>Aug 5</td>
</tr>
<tr>
<td>Corn Planting</td>
<td>May 28</td>
<td>June 13</td>
<td>May 27</td>
</tr>
<tr>
<td>Corn Harvest</td>
<td>Sept 1</td>
<td>Sept 22</td>
<td>Sept 3</td>
</tr>
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</table>

*Squash was replanted (direct seeded) on June 15.
Table 4 Squash operating expenses. Establishment and harvest costs for one hectare of production

<table>
<thead>
<tr>
<th>Cost or Expense</th>
<th>Unit</th>
<th>Number of units</th>
<th>Price or Cost/unit ($)</th>
<th>Total ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Supplies</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>soil test</td>
<td>each</td>
<td>1</td>
<td>14.00</td>
<td>14.00</td>
</tr>
<tr>
<td>fuel</td>
<td>gal</td>
<td>50</td>
<td>3.50</td>
<td>175.00</td>
</tr>
<tr>
<td>plastic mulch 30.5cm (ft)</td>
<td>21,000</td>
<td>0.05</td>
<td>1050.00</td>
<td></td>
</tr>
<tr>
<td>organic transplants</td>
<td>each</td>
<td>22,000</td>
<td>0.25</td>
<td>5500.00</td>
</tr>
<tr>
<td>market boxes</td>
<td>each</td>
<td>100</td>
<td>2.40</td>
<td>240.00</td>
</tr>
<tr>
<td><strong>Total supplies</strong></td>
<td></td>
<td></td>
<td></td>
<td>$6739.00</td>
</tr>
<tr>
<td><strong>Labor</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>soil test</td>
<td>hours</td>
<td>1.0</td>
<td>10.00</td>
<td>10.00</td>
</tr>
<tr>
<td>tillage + field prep</td>
<td>hours</td>
<td>6</td>
<td>10.00</td>
<td>60.00</td>
</tr>
<tr>
<td>weeding</td>
<td>hours</td>
<td>30*</td>
<td>10.00</td>
<td>300.00</td>
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<tr>
<td>plastic mulch install</td>
<td>hours</td>
<td>2</td>
<td>10.00</td>
<td>20.00</td>
</tr>
<tr>
<td>planting</td>
<td>hours</td>
<td>150</td>
<td>10.00</td>
<td>1500.00</td>
</tr>
<tr>
<td>irrigation management</td>
<td>hours</td>
<td>12*</td>
<td>10.00</td>
<td>120.00</td>
</tr>
<tr>
<td>harvest and grading</td>
<td>hours</td>
<td>250**</td>
<td>10.00</td>
<td>2500.00</td>
</tr>
<tr>
<td>post-harvest clean-up</td>
<td>hours</td>
<td>50</td>
<td>10.00</td>
<td>500.00</td>
</tr>
<tr>
<td><strong>Total Labor</strong></td>
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<td></td>
<td></td>
<td>$5010.00</td>
</tr>
<tr>
<td><strong>Total Operating Expenses</strong></td>
<td></td>
<td></td>
<td></td>
<td>$11,749.00</td>
</tr>
</tbody>
</table>

*Based on a 12 week season
**Based on a 5 week harvest period
Table 5 Corn silage operating expenses. Establishment and harvest costs for one hectare of production

<table>
<thead>
<tr>
<th>Cost or Expense</th>
<th>Unit</th>
<th>Number of units</th>
<th>Price or cost/unit ($)</th>
<th>Total ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Supplies</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>soil test</td>
<td>each</td>
<td>1</td>
<td>14.00</td>
<td>14.00</td>
</tr>
<tr>
<td>fuel</td>
<td>gal</td>
<td>50</td>
<td>3.50</td>
<td>175.00</td>
</tr>
<tr>
<td>organic seed</td>
<td>50lb bag</td>
<td>0.3</td>
<td>142.00</td>
<td>48.00</td>
</tr>
<tr>
<td><strong>Total supplies</strong></td>
<td></td>
<td></td>
<td></td>
<td>$237.00</td>
</tr>
<tr>
<td><strong>Labor</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>soil test</td>
<td>hours</td>
<td>1.0</td>
<td>10.00</td>
<td>10.00</td>
</tr>
<tr>
<td>tillage + field prep + cultivation</td>
<td>hours</td>
<td>8</td>
<td>10.00</td>
<td>80.00</td>
</tr>
<tr>
<td>Weeding + thinning</td>
<td>hours</td>
<td>8</td>
<td>10.00</td>
<td>80.00</td>
</tr>
<tr>
<td>planting</td>
<td>hours</td>
<td>2</td>
<td>10.00</td>
<td>20.00</td>
</tr>
<tr>
<td>irrigation management</td>
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<td>10.00</td>
<td>120.00</td>
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<td>harvest</td>
<td>hours</td>
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<td>40.00</td>
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<tr>
<td><strong>Total Labor</strong></td>
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<td></td>
<td></td>
<td>$350.00</td>
</tr>
<tr>
<td><strong>Total Operating Expenses</strong></td>
<td></td>
<td></td>
<td></td>
<td>$587.00</td>
</tr>
</tbody>
</table>
Table 6  Average yields for corn and squash by year. Dry weights are reported for corn and fresh weights are reported for squash. Yields were averaged across rate. Standard errors (±) are reported in parentheses

<table>
<thead>
<tr>
<th>Crop</th>
<th>Yield (Mg ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2008</td>
</tr>
<tr>
<td>Corn</td>
<td></td>
</tr>
<tr>
<td>Silage</td>
<td>13.65 (±0.36)</td>
</tr>
<tr>
<td>Grain</td>
<td>3.34 (±0.09)</td>
</tr>
<tr>
<td>Squash</td>
<td>17.17 (±0.87)</td>
</tr>
</tbody>
</table>
Table 7 2011 Compost carryover estimates. Where yield is described by the fertility function:

\[ Y = k + a_1 [(u_{2010}) + (d_2 \times u_{2009}) + (d_3 \times d_2 \times u_{2008})] \]

The \((d_x)\) terms represent decay as a ratio of first-year decay \((b_x/b_1)\). Terms are reported by crop with accompanying standard errors (±).

Two-year ratio \((d_2)\), three-year ratio \((d_3)\), slope \((a_1)\) and intercept \((k)\) in \((\text{Mg DM ha}^{-1})\), are reported.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept (k)</th>
<th>Slope (a_1)</th>
<th>((b_2/b_1))</th>
<th>((b_3/b_1))</th>
<th>((b_2/b_1)) (d_2 \times d_3)</th>
</tr>
</thead>
<tbody>
<tr>
<td>squash</td>
<td>26.49 (±1.92)</td>
<td>0.22 (±0.09)</td>
<td>0.86 (±0.37)</td>
<td>0.81 (±0.43)</td>
<td>0.70</td>
</tr>
<tr>
<td>corn silage</td>
<td>11.77 (±1.01)</td>
<td>0.07 (±0.05)</td>
<td>0.55 (±0.57)</td>
<td>2.08 (±2.14)</td>
<td>1.14</td>
</tr>
<tr>
<td>corn grain</td>
<td>2.64 (±0.35)</td>
<td>0.02 (±0.01)</td>
<td>0.87 (±0.69)</td>
<td>1.32 (±0.98)</td>
<td>1.14</td>
</tr>
</tbody>
</table>
Table 8 Summary of climate data by month and year. Data sourced from the Utah Climate Center, USU Weather Station. Standard errors (±) for averaged data are reported.

<table>
<thead>
<tr>
<th>Month and Year</th>
<th>Maximum Temp. Average (°C)</th>
<th>Minimum Temp. Average (°C)</th>
<th>Precipitation Total (cm)</th>
<th>Snowfall Total (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>April</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>11.0 (±1.1)</td>
<td>-1.7 (±0.7)</td>
<td>2.3</td>
<td>6.6</td>
</tr>
<tr>
<td>2009</td>
<td>12.5 (±1.1)</td>
<td>1.6 (±0.6)</td>
<td>7.7</td>
<td>20.3</td>
</tr>
<tr>
<td>2010</td>
<td>12.8 (±1.2)</td>
<td>1.5 (±0.8)</td>
<td>6.2</td>
<td>52.3</td>
</tr>
<tr>
<td>2011</td>
<td>10.2 (±0.8)</td>
<td>0.3 (±0.5)</td>
<td>11.2</td>
<td>22.8</td>
</tr>
<tr>
<td>May</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>17.3 (±1.1)</td>
<td>5.0 (±0.7)</td>
<td>6.2</td>
<td>3.8</td>
</tr>
<tr>
<td>2009</td>
<td>20.1 (±0.9)</td>
<td>7.1 (±0.7)</td>
<td>4.5</td>
<td>0</td>
</tr>
<tr>
<td>2010</td>
<td>14.3 (±0.9)</td>
<td>3.6 (±0.7)</td>
<td>8.2</td>
<td>5.8</td>
</tr>
<tr>
<td>2011</td>
<td>14.7 (±0.9)</td>
<td>4.4 (±0.5)</td>
<td>13.8</td>
<td>1.3</td>
</tr>
<tr>
<td>June</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>24.5 (±1.3)</td>
<td>10.2 (±0.8)</td>
<td>2.4</td>
<td>0</td>
</tr>
<tr>
<td>2009</td>
<td>22.6 (±0.8)</td>
<td>10.9 (±0.4)</td>
<td>8.6</td>
<td>0</td>
</tr>
<tr>
<td>2010</td>
<td>23.5 (±1.0)</td>
<td>10.7 (±0.6)</td>
<td>3.5</td>
<td>0</td>
</tr>
<tr>
<td>2011</td>
<td>23.5 (±0.9)</td>
<td>9.8 (±0.7)</td>
<td>2.5</td>
<td>0</td>
</tr>
</tbody>
</table>

Utah State University - Utah Climate Center: [http://climate.usurf.usu.edu/#](http://climate.usurf.usu.edu/#)
Table 9 2009 Compost carryover estimates. Where yield is described by the fertility function: $Y = k + (a_1 \times u_{2009}) + (a_1 \times b_1 \times u_{2008})$. Terms are reported by crop with accompanying standard errors (S.E.). One-year decay term ($b_1$), slope ($a_1$) and intercept ($k$) are reported.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept (Mg ha$^{-1}$)</th>
<th>Slope</th>
<th>1 year carryover</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>K</td>
<td>S.E.</td>
<td>$a_1$</td>
</tr>
<tr>
<td>squash</td>
<td>9.58</td>
<td>0.51</td>
<td>0.10</td>
</tr>
<tr>
<td>corn silage</td>
<td>4.47</td>
<td>0.26</td>
<td>0.15</td>
</tr>
<tr>
<td>corn grain</td>
<td>0.78</td>
<td>0.08</td>
<td>0.05</td>
</tr>
</tbody>
</table>
Table 10 Corn silage profitability. Average net profits after crop establishment and compost costs. Corn revenue is based on a market price for silage of $44/Mg

<table>
<thead>
<tr>
<th>Compost Rate (Mg DM ha⁻¹)</th>
<th>Within year ($)</th>
<th>1 Year ($)</th>
<th>2 Years ($)</th>
<th>3 Years ($)</th>
<th>Total 4-Year Profit ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>347</td>
<td>347</td>
<td>347</td>
<td>347</td>
<td>1388</td>
</tr>
<tr>
<td>10</td>
<td>-596</td>
<td>370</td>
<td>411</td>
<td>851</td>
<td>1035</td>
</tr>
<tr>
<td>20</td>
<td>-1538</td>
<td>418</td>
<td>652</td>
<td>1027</td>
<td>558</td>
</tr>
<tr>
<td>30</td>
<td>-2573</td>
<td>312</td>
<td>538</td>
<td>964</td>
<td>-759</td>
</tr>
<tr>
<td>40</td>
<td>-3478</td>
<td>557</td>
<td>536</td>
<td>890</td>
<td>-1494</td>
</tr>
</tbody>
</table>
Table 11 Squash profitability. Average net profits after crop establishment, maintenance and harvesting costs ($11,749) and compost purchase and application costs ($104 per Mg DM). Squash revenue is based on a market price of $2/kg.

<table>
<thead>
<tr>
<th>Compost Rate (Mg ha⁻¹)</th>
<th>Within year ($)</th>
<th>1 Year ($)</th>
<th>2 Years ($)</th>
<th>3 Years ($)</th>
<th>Total 4-Year Profit ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>15,771</td>
<td>15,771</td>
<td>15,771</td>
<td>15,771</td>
<td>63,084</td>
</tr>
<tr>
<td>10</td>
<td>10,751</td>
<td>18,451</td>
<td>26,811</td>
<td>48,311</td>
<td>104,324</td>
</tr>
<tr>
<td>20</td>
<td>14,651</td>
<td>23,200</td>
<td>30,011</td>
<td>49,511</td>
<td>117,373</td>
</tr>
<tr>
<td>30</td>
<td>17,901</td>
<td>30,780</td>
<td>29,791</td>
<td>50,511</td>
<td>128,988</td>
</tr>
<tr>
<td>40</td>
<td>13,504</td>
<td>23,491</td>
<td>34,651</td>
<td>51,991</td>
<td>123,637</td>
</tr>
</tbody>
</table>
Figure 1 2011 Squash yield. Regression models of yield response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.257x + 25.19$ ($r^2=0.93$), 2009-amended plots $y=0.238x + 25.19$ ($r^2=0.84$), and 2008-amended plots $y=0.201x + 25.19$ ($r^2=0.57$). Error bars indicate ± standard errors.
Figure 2 2011 Corn silage yield. Regression models of yield response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y = -0.0061x^2 + 0.3492x + 9.3$ ($r^2=0.48$), 2009-amended $y=-0.0068x^2 + 0.3478x + 9.3$ ($r^2=0.83$), and 2008-amended plots $y = -0.0088x^2 + 0.4553x + 9.3$ ($r^2=0.99$). Error bars indicate ± standard errors.
Figure 3 2011 Corn grain yield. Regression models of yield response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y = -0.0034x^2 + 0.1522x + 1.73$ ($r^2=0.70$), 2009-amended $y=-0.0028x^2 + 0.1423x + 1.73$ ($r^2=0.85$), and 2008-amended plots $y = -0.0029x^2 + 0.1628x + 1.73$ ($r^2=0.99$). Error bars indicate ± standard errors.
Figure 4 2010 Corn and squash yields. (a) 2010 Squash yield. (b) 2010 Corn silage yield. (c) 2010 Corn grain yield. Response to compost applied in 2008, 2009, and 2010. Data points are the average for each compost level. Error bars indicate ± standard errors.
Figure 5 2009 Squash yield. Regression models of yield response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots $y = 0.1187x + 9.92$ ($r^2=0.87$) and 2008-amended plots $y = 0.0198x + 9.92$ ($r^2=0.13$). Error bars indicate ± standard errors.
Figure 6 2009 Corn silage yield. Regression models of yield response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots $y = 0.127x + 5.09$ ($r^2=0.83$) and 2008-amended plots $y = 0.0295x + 5.09$ ($r^2=0.82$). Error bars indicate ± standard errors.
Figure 7 2009 Corn grain yield. Regression models of yield response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots \( y = 0.0428x + 1 \) \( (r^2=0.91) \) and 2008-amended plots \( y = 0.0066x + 1 \) \( (r^2=0.47) \). Error bars indicate \( \pm \) standard errors.
Soils (0-30 cm) were sampled in July 2011. Electrical conductivity (µS/cm) was not statistically different at any treatment rate (α=0.05). Error bars indicate ± standard errors.

**Figure 8** 2011 Soil electrical conductivity. Soils (0-30 cm) were sampled in July 2011. Electrical conductivity (µS/cm) was not statistically different at any treatment rate (α=0.05). Error bars indicate ± standard errors.
Figure 9 2011 Soil pH. Soils (0-30cm) sampled in July 2011. Soil pH was not statistically different at any treatment rate ($\alpha=0.05$). Error bars indicate ± standard errors.
Figure 10 2009/2010 Wheat cover crop yield. Regression models of yield (Mg DM ha\(^{-1}\)) response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots \( y = 0.0132x + 1.88 \) \((r^2=0.73)\) and 2008-amended plots \( y = 0.0115x + 1.88 \) \((r^2=0.95)\). Error bars indicate ± standard errors. One statistical outlier was identified and removed (40 Mg DM ha\(^{-1}\) rate)
CHAPTER III

NITROGEN CARRYOVER AND FT-IR ANALYSIS OF SOIL ORGANIC MATTER FOLLOWING A ONE-TIME COMPOST APPLICATION IN AN ORGANIC FIELD TRIAL

Abstract

Compost carryover is comprised of nutrient and non-nutrient elements. Both affect crop growth, yet carryover is traditionally described in terms of N-carryover only. This study investigated two key aspects of nutritive carryover (Nitrogen and Phosphorus), as well as a key indicator of non-nutritive carryover (soil organic matter) to determine the residual effect of a one-time compost application. Nitrate ($\text{NO}_3^-$) carryover was determined in an organic corn/squash rotation over a 4-year study with five compost treatment rates. In addition, residual $\text{NO}_3^-$ mineralization was evaluated in a 70-day laboratory incubation trial. Functional groups of compost and soil organic matter following compost addition were further investigated by FT-IR spectroscopy. Overall, soil $\text{NO}_3^-$ levels were low, and additions of compost at any of the treatment rates did not increase soil $\text{NO}_3^-$ beyond the year of application. No $\text{NO}_3^-$ carryover was seen in any year, except in 2011 where it was attributed to a cover crop response rather than as a response to compost. Laboratory incubation also showed no $\text{NO}_3^-$ carryover, with all treatments mineralizing similar amounts of $\text{NO}_3^-$ as control treatments. Soil phosphorus was low in 2011, yet was significantly higher in compost amended soils than in non-amended soils. Pronounced polysaccharide peaks, evident in compost spectra and absent in control soil, were apparent in compost-amended soils one, two and three years after an initial one-time compost treatment. One-time compost applications result in N-mineralization apparent within the year.

Co-authors: Reeve JR and Jacobson A
of application only, however the effects of compost on soil P and organic matter (SOM) are evident even three years after the initial application, suggesting compost is able to influence nutritive and non-nutritive soil properties many years after incorporation.

Introduction

Organic farmers who choose not to use compost often do so because they perceive the costs of purchase, transport and application outweigh the crop benefit in any given year. This is especially the case where farm size is large or where low-value crops are grown. These growers fail to recognize that compost persists in the soil and provides benefits many years beyond the initial year of application. When the long-term carryover of compost is considered, even a one-time application may become economically viable. Studies have shown that a one-time compost treatment can result in increases in SOM, total N, P, K, and water infiltration many years beyond the initial application year (Butler and Muir 2006; Hartl et al. 2003; Ippolito et al. 2010; Reeve et al. 2012).

The persistence of compost in soil gives rise to benefits, both nutritive and non-nutritive, which carryover from year to year enhancing nutrient availability and soil quality beyond the year of application. Nutritive benefits such as N, P, and K are a function of microbially-mediated mineralization dynamics, while non-nutritive components such as improvements to water holding capacity, bulk density, and soil aggregation are more directly attributable to the positive effect compost has on (SOM) accumulation. These non-nutritive aspects are key components of soil quality. Ultimately, crop yields are affected by both nutritive and non-nutritive soil factors. The majority of studies investigating compost carryover are concerned with nutrient carryover, often expressed as N-fertilizer equivalents. Research that
investigates compost carryover on organically certified land, and accounts for more than simply
N carryover, is necessary to enable growers to better manage their compost use.

Compost persistence and stability in soil is predominantly affected by the physical and
chemical composition of the compost itself, as well as the soil type, climate and cropping
system. During composting, easily degradable plant compounds such as carbohydrates and
proteins decompose, and the more recalcitrant plant compounds, such as lignin, together with
microbial products and non-identifiable humic substances are relatively enriched (Leifeld et al.
2002). Most of the easily mineralizable N and C are lost, leaving only more stable forms (Eghball
et al. 1997). These forms are more resistant to decay and thus mineralize gradually over
multiple years.

Both nutritive carryover (N and P), as well as a key indicator of non-nutritive carryover
(SOM), are investigated in this study. Nitrogen, though abundant in the environment, remains
the most frequently limiting nutrient for crop production (Wander et al. 2011). Nearly all of the
N and large proportions of the P found in soils occur as constituents of SOM, which serves as
both the principal long-term storage medium and the primary short-term source of these and
other nutrients (Weil and Magdoff 2004). Composts and manures are commonly used on
organic farms to maintain or improve fertility. Compost-N is predominantly in organic forms and
therefore unavailable for plant uptake. This organic-N must first be mineralized into inorganic
forms ammonium (\(\text{NH}_4^+\)) and nitrate (\(\text{NO}_3^-\)) before it can be assimilated. Depending on initial N
content, C/N ratio and compost maturity, typically only 10% to 30% of total compost-N is plant-
available within the season of application (Amlinger et al. 2003; Eghball et al. 2002; Hadas et al.
1996; Hartz et al. 2000;) and often periods of immobilization occur initially upon incorporation.
The gradual decomposition and chemical transformation of compost over many years results in
a steadily decreasing rate of N mineralization, often described as a decay series in the literature (e.g. Cusick et al. 2006; Klausner et al. 1994). We hypothesize that N mineralization resulting from a one-time compost application will be most evident within the year of application, with following years displaying a gradual, steadily decreasing, N-mineralization trend.

The second most limiting nutrient for plant growth is typically phosphorus. In the calcareous soils of the Intermountain West, P availability is often limited. In these high pH soils, reactive calcium compounds absorb and precipitate P making it unavailable for plant uptake (Reeve et al. 2012). Typically the P/N ratio of manures is greater than that of plants, so growers who base their application rates to achieve an N target often apply P in excess of crop needs (Eghball and Power 1999). Excess P can become an environmental pollutant if it is carried into surface and ground waters where it can cause eutrophication and contamination (Daniel et al. 1994). In addition to accumulation of excess P, concentrations of copper, and zinc, may also accumulate in soils (Wander et al. 2011). Conversely, growers who rely solely on N-fixing cover crops for fertility may deplete their soils of P as well as other nutrients in the long term. Given our calcareous soil, we hypothesize that soil P levels will increase with compost amendment, responding linearly to rate.

In addition to nutrient carryover, non-nutritive carryover also affects crop yield. Yield increases ranging from 5 to 25% have been attributed to non-N sources (Magdoff and Amadon 1980; Schroder and Dilz 1987). Non-nutritive carryover is profoundly influenced by SOM. It is clear that despite comprising less than 5% of a typical soil, SOM exerts a disproportionally large influence on soil properties (Wagner and Wolf 1999). Following compost application, SOM displays enrichment corresponding to the original composition of the compost amendment. Leifeld et al. (2002) noted that changes in the SOM composition after the addition of compost
are characterized mainly by an ongoing decay of the easily degradable OM from the compost, which is mostly polysaccharide and, to a considerable degree, composed of microbial biomass. Polysaccharides, both cellulosic and non-cellulosic, present in composts are primarily of plant and microbial origin and can account for up to 20% of total organic C in compost amended soils (Leifeld et al. 2002). Because polysaccharides are more readily degraded than lignin and other more stable humic materials, their mineralization occurs over shorter time frames.

Soil organic matter is a key indicator of soil processes and properties responsible for many of the non-nutritive benefits ascribed to compost. The enrichment of SOM due to compost can be investigated using infrared spectroscopy, and can provide insights into the value of compost use in the long term.

Fourier transform infrared (FT-IR) spectroscopy is one of the most sensitive infrared techniques and has been used to identify distinct compositional features of compost and SOM (Ellerbrock et al. 1999b, 2001). Infrared wavelengths are known to induce bonding vibrations in a wide range of functional groups, and specific wavelengths can therefore be used to characterize organic and inorganic molecules (Stevenson 1994). Many infrared absorption bands display a higher relative intensity for compost, and are almost absent in spectra for control soils (Soler Rovira et al. 2003). These bands can therefore be used as identifying markers to determine the presence or absence of compost in soil as well as provide insight into the degree of chemical transformation.

Although FT-IR has been employed successfully to study various compositional aspects of compost and SOM, to our knowledge, no studies have investigated compost persistence in soil following a one-time application, within an organic setting. We hypothesize that FT-IR can be used to identify functional groups displaying residual carryover arising from a one-time
application of compost up to three years after incorporation. In particular, we propose that spectra representing the polysaccharide band (1030 – 1150 cm⁻¹) will show decreased absorbance over time as these polysaccharides are preferentially decomposed.

Methods and Materials

Soil Analysis

In 2009, 2010, and 2011 bulk soil samples were collected from both corn and squash plots at a time corresponding to 30 days of corn growth. This 30-day mark is widely used by corn growers to assess early-season soil-N and apply a side-dress fertilizer if required. In each year, five subsamples of 0-30 cm depth were collected from the center of each plot and combined into one representative sample per plot. Soils were passed through a 2-mm sieve and stored at 4°C until analysis. Sub-samples were oven dried for 24 hrs at 105°C and gravimetric moisture content determined. Nitrate (NO₃⁻) and ammonium (NH₄⁺) content was determined by automated colorimetry using a cadmium reduction method (Lachat QuickChem AE) in 5:1 extracts (1M KCl). Electrical Conductivity (EC) and pH were measured in 1:1 soil/water extracts while P was determined in 1:20 soil/NaHCO₃ extracts according to the Olsen method (Olsen et al. 1954).

Corn Leaf Tissue-N

Corn leaf samples were collected and tissue N% determined in 2008, 2009 and 2010. Samples were collected 32, 32, and 34 days after planting in 2008, 2009, and 2010, respectively. Sampling methodology was identical for the three years. Briefly, three whole plants were cut from the center of each plot and pooled to create a representative sample for each plot.
Samples were then oven-dried at 60°C degrees for 48 hours, and tissue-N% concentration determined by dry combustion (LECO TruSpec CN).

**Incubation Trial (NO\textsubscript{3}^- - N and NH\textsubscript{4}^+ - N)**

On August 8, 2011 soil samples were collected from all plots which had received 0, 20, and 40 Mg ha\textsuperscript{-1} compost in 2008, 2009, and 2010. Five subsamples of 0-30 cm depth were collected from the center of each plot and then combined into one representative sample. Soil were passed through a 2-mm sieve and stored at room temperature (25°C) until the experiment was started the following day. Sub-samples were dried for 24 hrs at 105°C and gravimetric moisture content determined. On August 9, 2011 samples were prepared for incubation following methodology described by Sullivan et al. (2011). All samples were brought to 20% moisture content (by weight) and 500g (dry weight equivalent) subsamples were then placed in 0.9L (1 quart) Ziplock freezer bags. The bags were sealed closed except at one end where a drinking straw was inserted into the top to allow for air entry into the bag during incubation. During the course of the incubation, bags were weighed every 14 days and moisture content re-adjusted to 20%. Samples were incubated at 22°C for 70 days. Soil NO\textsubscript{3}^- - N and NH\textsubscript{4}^+ - N was measured at days 0, 7, 14, 21, 28, 35, and 70. A composite sample of 10g of soil was removed from the incubation bags and NO\textsubscript{3}^- and NH\textsubscript{4}^+ analysis was determined by automated colorimetry (Lachat QuickChem AE) in 1M KCl extract as described above.

**Soil Phosphorus**

Soil samples (0-30cm) collected in July 2011 were tested for total available P according to the Olsen-P method (Olsen et al. 1954). All plots which had received 20, and 40 Mg ha\textsuperscript{-1} rates of compost in 2010, as well as all control plots, were tested. Soils were prepared in 1:20
soil/sodium bicarbonate (0.5 M) extracts which had been adjusted to pH 8.50. Extracts were
shaken for 30 minutes in an end-to-end shaker before being filtered (Whatman No. 40 filter).
Sulfuric acid (2N) was added to the filtrate and CO₂ allowed to evolve. Ammonium molybdate
color reagent was then added to the filtrate and P determined at 880nm by spectrophotometer
(Spectronic 20 Genesys 4001/4).

**FT-IR Spectroscopy**

Only the high rate (40Mg DM ha⁻¹) compost, and control treatments were selected for
FT-IR analysis. Soils (0-10cm) were sampled in September 2011 in plots which had received
40Mg ha⁻¹ compost in 2008, 2009, and 2010. Three replicates of each were collected.
Additionally, soils were sampled from plots which had received no compost over the course of
the study. Three replicates of these control soils were collected. Each replicate was comprised of
six soil subsample, collected from the center of each plot and then combined to make one
representative sample. Soils were sieved through a 2-mm screen and air-dried before being
ground with a mortar and pestle.

In addition to soils, compost was also analyzed by FT-IR spectroscopy. A sample from
each of the three composts applied during the experiment was air dried and ground. Two
replicates from each year were scanned, adjusted for background, and their spectra averaged to
depict a representative spectra for each compost.

Individual FT-IR spectra were composed of 333 scans with a resolution of 4cm⁻¹ (Thermo
Scientific Nicolet 6700). For each treatment year, including zero rate control, two samples were
selected and spectra determined for two reps of each sample, for a total of 4 spectra for each
sample year, and 4 spectra for control. Each group of 4 spectra was corrected against the
spectrum for background before being averaged to make one final spectra representative of
each treatment year as well as control. The operating range was 550-3500\textsuperscript{cm}\textsuperscript{-1}. FT-IR spectra were corrected for mineral component by mathematical subtraction based on the FT-IR spectra of the ash from the same sample, as described below.

Compost spectra were interpreted based on the characteristic FT-IR absorption bands for composted manure described by Carballo et al. (2008). Carballo et al. (2008) defined these as; 2960 – 2850\textsuperscript{cm}\textsuperscript{-1} (C-H stretch of aliphatic structures), 1620 – 1660\textsuperscript{cm}\textsuperscript{-1} (C=O vibrations of ketones, quinone, carboxylic acids and esters, as well as C=C vibrations of aromatic components), 1430 – 1455\textsuperscript{cm}\textsuperscript{-1} (O-H in-plane bend of carboxylic acids, CO\textsubscript{2} stretch of carboxylates and aliphatic CH\textsubscript{2} alkanes, and also C-O stretch vibration of carbonates), 1030 – 1150\textsuperscript{cm}\textsuperscript{-1} (polysaccharides), 1404\textsuperscript{cm}\textsuperscript{-1} (weak peak), and 1595\textsuperscript{cm}\textsuperscript{-1} (vibration of the aromatic skeleton of lignin).

**FT-IR -Organic Matter Removal from Soil**

Organic matter was oxidized in a 1:10 soil/sodium hyperchlorite extract. Sodium hyperchlorite (6% NaOCl) was adjusted to pH 9.50. Soils were first allowed to react with the NaOCl for 9 hrs at room temperature (25°C) before being placed in a digestor set at 90°C. Soils were digested for 20 minutes and were agitated every 5 minutes with a vortex mixer. Soils were then allowed to cool to room temperature before being centrifuged for 10 minutes at 5000 rpm. The supernatant was discarded. Heat treatment and centrifuge steps were repeated 4 times until supernatant was transparent.

Soils were then washed in a 1:10 soil/CaCO\textsubscript{3} solution (15mM) before being shaken on an end-to-end shaker for 10 minutes and then centrifuged for 10 minutes at 5000 rpm. Supernatant was discarded and the process repeated for a total of three wash treatments. Soils were then dried at room temperature (25°C) for 12 hrs.
Statistical Analyses

Soil Phosphorus. Statistical analysis was performed using PROC GLIMMIX (SAS version 3.1). Analysis of variance was performed to compare total soil P in 20 Mg DM ha$^{-1}$, 40 Mg DM ha$^{-1}$ and control plots in a single factor design where compost rate was the factor. There was no statistical difference between data for corn and squash plots, so these were combined for analysis. Data were log transformed and all model assumptions met.

Soil Nitrate. Analysis of variance of average NO$_3^-$ concentrations was performed to compare NO$_3^-$ levels between year of compost addition. Statistical analysis by crop type was performed using PROC GLIMMIX (SAS version 3.1). Within year, NO$_3^-$ was averaged across rate. Data were log transformed and all model assumptions met.

Laboratory Soil Incubation. Statistical analysis was performed using PROC GLIMMIX (SAS version 3.1) using a single factor design where year of compost addition was the factor. Analysis of variance was performed to compare total cumulative NO$_3^-$ at 70 days for each treatment (20 Mg DM ha$^{-1}$ and 40 Mg DM ha$^{-1}$) as well as control treatments. Data were log transformed and all model assumptions met.

FT-IR Spectroscopy. Statistical analysis of polysaccharide peak area was conducted. Integration under the peaks was achieved using OMNIC software (Omnic 8.0, Fisher Thermo Scientific Inc.) for all 16 spectra in the range 800-1300cm$^{-1}$. Analysis of variance of peak area was determined using PROC GLIMMIX (SAS version 3.1) in a single factor design where year of compost addition was the only factor. All model assumptions were met.

Determining Carryover. The methodology for determining compost carryover and quantifying decay series terms is based on that developed by Endelman et al. (2010). Central to
this method is an experimental design whereby some plots receive compost for the first time each year and where any given plot receives compost only once over the course of the experiment. By design, in years two and three, some plots received compost for the first time while others had received compost in years previous. Because current-year amended plots have not been fertilized previously they have no carryover and therefore serve as benchmarks from which to determine carryover for plots amended in previous years.

This project was extended by one year because of the poor treatment response seen in year 3. In 2011, our experimental design did not allow for the addition of compost to any plot. Yields for this year, therefore, were dependent upon carryover fertility from compost applications in the one, two, and three years previously. There was no within-year treatment response in 2011, therefore determination of carryover values in that year required a modification to the methodology undertaken in the three years previous. Because there was no carryover coefficient for a within-year response, previous years’ coefficients no longer represented the proportion of current year application that carried over. Instead, carryover was expressed as a proportion of first-year carryover. This enabled an assessment of the overall carryover trends, whether increasing or decreasing over time.

Total soil \(\text{NO}_3^-\) \((T_{\text{NO}_3^-})\) can be expressed as a function of compost \(\text{NO}_3^-\) \((C_{\text{NO}_3^-})\) by Eq. [3.1], where \((m)\) is the slope of the relationship and \((k)\) the intercept. Compost \(\text{NO}_3^-\) \((C_{\text{NO}_3^-})\) is described in units of compost equivalents, a term representing both nutritive and non-nutritive factors.

\[
T_{\text{NO}_3^-} = m(C_{\text{NO}_3^-}) + k
\]  

Total soil \(\text{NO}_3^-\) in a given year is dependent not only upon \(\text{NO}_3^-\) resulting from current year compost applications, but also on \(\text{NO}_3^-\) mineralized from previous years’ applications. Compost
$\text{NO}_3^-$ ($C_{\text{NO}_3}$) is therefore a function of compost carryover and can be expressed as Eq. [3.2] where $\beta$ terms represent carryover coefficients, and $R_t$ terms represent compost application rates at time ‘t’. Note that current-year applied compost serves as a benchmark and is therefore given a coefficient of 1 ($\beta_0=1$). The relationship between total soil $\text{NO}_3^-$ ($T_{\text{NO}_3}$) and compost $\text{NO}_3^-$ ($C_{\text{NO}_3}$) can then be expressed by Eq. [3.3]

\[
C_{\text{NO}_3} = \beta_0( R_t ) + \beta_1( R_{t-1} ) + \beta_2( R_{t-2} )
\]

\[
T_{\text{NO}_3} = k + m [ ( R_t ) + \beta_1( R_{t-1} ) + \beta_2( R_{t-2} ) + \ldots ]
\]

In years where compost was applied (2008-2010), the coefficient ($\beta_x$) terms were determined using PROC NLIN in SAS (version 9.3). Data was transformed where necessary, so that model assumptions were met. One outlier (40 Mg ha$^{-1}$ rep) was removed from 2009 corn data. Initial parameter estimates were generated using PROC REG in SAS (version 9.3).

In 2011, where no compost was applied, the coefficient $\beta_1$ could not be determined. In this case, the carryover coefficients ($\beta_x$) were evaluated in terms of their proportion of first-year carryover. Firstly a new variable ($\alpha$) was defined, where $\alpha = m \times \beta_x$. The $T_{\text{NO}_3}/C_{\text{NO}_3}$ relationship in Eq. [3.3] was then expressed as Eq. [3.4]

\[
T_{\text{NO}_3} = k + \alpha_1( R_t ) + \alpha_2( R_{t-1} ) + \alpha_3( R_{t-2} )
\]

Conclusions about the change in carryover coefficients can now be made by evaluating the ratio of the $\alpha_x$ terms to each other (ie. comparing the ratio $\alpha_3/\alpha_2$ to $\alpha_2/\alpha_1$). Because $\alpha_3/\alpha_2$ and $\alpha_2/\alpha_1$ are numerically equal to $\beta_3/\beta_2$ and $\beta_2/\beta_1$, respectively, these ratios tell us something of the magnitude of the change in carryover each year. These ratios can be further simplified by Eq. [3.5] and the total soil $\text{NO}_3^-$ ($T_{\text{NO}_3}$) model expressed as Eq. [3.6].
\[ d_x = \beta_x / \beta_{x-1} \]  

So that \( d_2 \) represents second year carryover as a proportion of \( \beta_1 \). Likewise, \( d_3 \times d_2 \) represents third year carryover expressed as a proportion of \( \beta_1 \).

\[ T_{\text{NO}_3} = k + \alpha_1 \left[ (R_{t-1}) + (d_2 \times R_{t-2}) + (d_3 \times d_2 \times R_{t-3}) \right] \]  

Terms in this \( T_{\text{NO}_3} \) model Eq. [3.6] were then determined using PROC NLIN in SAS (version 9.3), with initial parameter estimates generated using PROC REG in SAS (version 9.3). Two statistical outliers were removed from the 2011 corn data (plots 124 and 130).

**Results**

Nutritive carryover components, N and P, were evaluated as well as SOM as a key indicator of non-nutritive carryover. In general soil \( \text{NO}_3^- \) increased with increased compost application, though it was responsive within the year of application only. Phosphorus levels were greater in amended soils than in control soils and FT-IR spectra revealed pronounced polysaccharide peak absorbance in treatment soils.

**2011 Season**

No compost was applied during the 2011 growing season, so determination of a within-season \( \text{NO}_3^- \) response was not possible. Compost \( \text{NO}_3^- \) carryover was instead evaluated as a proportion of first year carryover as described above.

Squash soil \( \text{NO}_3^- \) levels (mg/kg) in 2011 displayed a linear response to treatment (Figure 11) and overall, average \( \text{NO}_3^- \) levels were significantly higher in 2011 than in previous years (Figure 12). Considering the error present, \( \text{NO}_3^- \) levels were similar regardless of whether plots were fertilized in 2008, 2009, or 2010. When years are combined, \( \text{NO}_3^- \) was significantly greater
than control for high rate 40 Mg ha\(^{-1}\) only (Figure 11). Ratios of carryover coefficients for, two and three year carryover (d\(_t\) terms) were estimated to be: d\(_2\)=1.17 (±0.40) and d\(_3\)=0.62 (±0.29) with a slope of a\(_1\)=0.15 (±0.09) and an intercept k=9.44 mg/kg (±1.20) (Table 12). The overall 2011 total soil NO\(_3^-\) model for squash plots is shown by Eq. [3.7].

\[
T_{\text{NO3}^-} = 9.44 + 0.15 \left[ (R_{t-1}) + (1.17 \times R_{t-2}) + (0.62 \times 1.17 \times R_{t-3}) \right] \tag{3.7}
\]

These estimates indicate that in 2011, 2009 carryover was 117% of 2010 carryover, while 2008 carryover was 72% of 2010 carryover (1.17 × 0.62). In terms of fertility, the 2009 ratio equates to a 17% increase in fertility compared with 2010-amended plots while the 2008 ratio equates to a 28% decrease (decay) in fertility compared with 2010-amended plots.

Corn soil NO\(_3^-\) levels (mg/kg) in 2011 displayed a linear response to treatment (Figure 13) and overall, average NO\(_3^-\) levels were significantly higher in 2011 than in previous years (p<0.05)(Figure 14). Considering the error present, NO\(_3^-\) levels were very similar regardless of whether plots were fertilized in 2008, 2009, or 2010 (Figure 13). When years are combined, only high rate 40 Mg ha\(^{-1}\) plots showed significantly greater NO\(_3^-\) levels (p=0.06) than control plots, and only at \(\alpha=0.1\) confidence level. Ratios of carryover coefficients for, two and three year carryover (d\(_t\) terms) were estimated to be: d\(_2\)=0.61 (±0.29) and d\(_3\)=0.82 (±0.67) with a slope of a\(_1\)=0.09 (±0.03) and an intercept k=7.91 mg/kg (±0.63) (Table 12). The overall 2011 soil NO\(_3^-\) model for corn plots is shown by Eq. [3.8].

\[
T_{\text{NO3}^-} = 7.91 + 0.09 \left[ (R_{t-1}) + (0.61 \times R_{t-2}) + (0.82 \times 0.61 \times R_{t-3}) \right] \tag{3.8}
\]

These estimates indicate that in 2011, 2009 carryover was 61% of 2010 carryover, while 2008 carryover was 50% of 2010 carryover (0.82 × 0.61). In terms of soil NO\(_3^-\), the 2009 ratio
equates to a 39% decrease (decay) in NO\textsubscript{3} compared with 2010-amended plots while the 2008 ratio equates to a 50% decrease (decay) in soil NO\textsubscript{3} compared with 2010-amended plots.

2010 Season

Squash soil NO\textsubscript{3} levels (mg/kg) in 2010 displayed a linear response to treatment (Figure 15.) and carryover was determined using the method developed by Endelman et al. (2010), and described above. A within-year response to treatment was also reliably measured in 2010. Overall, average NO\textsubscript{3} levels were significantly less in 2010 than those recorded in 2011 (Figure 12).

Estimates of total soil NO\textsubscript{3} model terms revealed within-year response to be slope (a\textsubscript{1})=0.06 (±0.04), while the estimated one-year carryover term was b\textsubscript{1}=-0.24 (±0.66) and the two-year carryover b\textsubscript{2}=-0.59 (±0.80) with a model intercept of (k) was 7.65 mg/kg (±0.77) (Table 13). The overall 2010 total soil NO\textsubscript{3} model for squash is shown by Eq. [3.9].

\[ T_{NO3} = 7.65 + 0.06 \left[ (R_t) - 0.24 \left( R_{t-1} \right) - 0.59 \left( R_{t-2} \right) \right] \]  

In terms of soil NO\textsubscript{3}, the (b\textsubscript{j}) terms indicate that in 2010, there was no NO\textsubscript{3} carryover from either 2009 or 2008-amended plots. In terms of the overall model, and the effect on total soil NO\textsubscript{3}, a one unit change in 2010 compost rate results in a 0.06 unit increase in NO\textsubscript{3}, while compost applied in 2008 or 2009 did not contribute to 2010 NO\textsubscript{3} levels.

Corn soil NO\textsubscript{3} levels (mg/kg) in 2010 displayed a linear response to treatment (Figure 16) and carryover was determined using the method developed by Endelman et al. (2010), and described above. A within-year response to treatment was also reliably measured in 2010. Overall, soil NO\textsubscript{3} concentrations were significantly less in 2010 than those recorded in 2011 (Figure 14).
Estimates of total soil NO$_3^-$ model terms revealed within-year response to be slope ($a_1$) = 0.05 ($\pm 0.02$), while the estimated one-year carryover term was $b_1$ = 0.58 ($\pm 0.35$) and the two-year carryover $b_2$ = -0.48 ($\pm 0.53$) with a model intercept of ($k$) was 4.41 mg/kg ($\pm 0.46$) (Table 13). The overall 2010 total soil NO$_3^-$ model for corn is shown by Eq. [3.10].

\[
T_{NO3} = 4.41 + 0.05 \left[ (R_t) + 0.58 (R_{t-1}) - 0.48 (R_{t-2}) \right]
\]  

[3.10]

In terms of soil NO$_3^-$, the ($b_x$) terms indicate that in 2010, there was a carryover from 2009-amended plots equivalent to 58% of the 2010-amended rate. No carryover from 2008-amended plots was seen. In terms of the overall model, and the effect on total soil NO$_3^-$, a one unit change in 2010 compost rate results in a 0.05 unit increase in NO$_3^-$, plus an additional 0.03 unit increase in NO$_3^-$ (0.05 x 0.58) from every unit of compost applied in 2009, while compost applied in 2008 did not contribute to 2010 total soil NO$_3^-$ levels.

2009 Season

Squash soil NO$_3^-$ levels (mg/kg) in 2009 displayed a linear response to treatment, though there was a great deal of variability between plots particularly at 20 and 40 Mg ha$^{-1}$ rates (Figure 17). Carryover was estimated using the method developed by Endelman et al. (2010), and described above, and a within-year response to treatment was also reliably measured. Overall, average NO$_3^-$ concentrations in 2009 were similar to 2010 levels and significantly less than those recorded in 2011 (Figure 12). Plots amended in 2008 did not show a treatment response, in terms of soil NO$_3^-$, in 2009.

Estimates of NO$_3^-$ model terms revealed within-year response to be slope ($a_1$) = 0.16 ($\pm 0.04$), while the estimated one-year carryover term was $b_1$ = -0.11 ($\pm 0.26$) and model intercept...
(k) was 5.00 Mg DM ha\(^{-1}\) (±0.69) (Table 14). The overall 2009 soil NO\(_3\)\(^-\) model for squash is shown by Eq. [3.11].

\[
T_{\text{NO3}} = 5.00 + 0.16 \left[ (R_t) - 0.11 (R_{t-1}) \right] \quad [3.11]
\]

In terms of soil NO\(_3\)\(^-\), the (b\(_1\)) term indicates that 2008-amended plots contributed no carryover into the 2009 season. In terms of the overall model, and the effect on total soil NO\(_3\)\(^-\), a one unit change in 2009 compost rate results in a 0.16 unit increase in NO\(_3\)\(^-\) while compost applied in 2008 did not contribute to 2009 NO\(_3\)\(^-\) levels.

Corn soil NO\(_3\)\(^-\) levels (mg/kg) in 2009 displayed a linear response to treatment (Figure 18). Carryover was estimated using the method developed by Endelman et al. (2010), and described above, and a within-year response to treatment was also reliably measured. Overall, average NO\(_3\)\(^-\) concentrations in 2009 were similar to 2010 levels and significantly less than those recorded in 2011 (Figure 14). Plots amended in 2008 did not show a treatment response.

Estimates of the total soil NO\(_3\)\(^-\) model terms revealed within-year response to be slope (a\(_1\))= 0.15 (±0.02), while the estimated one-year carryover term was b\(_1\)= -0.03 (±0.11) and model intercept (k) was 3.64 Mg DM ha\(^{-1}\) (±0.28) (Table 14). The overall 2009 total soil NO\(_3\)\(^-\) model for corn is shown by Eq. [3.12].

\[
T_{\text{NO3}} = 3.64 + 0.15 \left[ (R_t) - 0.03 (R_{t-1}) \right] \quad [3.12]
\]

In terms of soil NO\(_3\)\(^-\), the (b\(_1\)) term indicates that 2008-amended plots contributed no carryover into the 2009 season. In terms of total soil NO\(_3\)\(^-\), a one unit change in 2009 compost rate results in a 0.15 unit increase in NO\(_3\)\(^-\) while compost applied in 2008 did not contribute to 2009 NO\(_3\)\(^-\) levels.
70-Day Soil Incubation

Nitrate mineralization in soils which had received 20 Mg DM ha\(^{-1}\) of compost (Figure 19) followed the same trend as those which had received 40 Mg DM ha\(^{-1}\) (Figure 20). Mineralized NO\(_3^-\) exhibited the same response regardless of whether plots had received compost or not, and total cumulative NO\(_3^-\) was not statistically different between treatments or control at 70 days (\(\alpha=0.05\)). Results were similar for corn and squash plots, though squash plots showed higher total cumulative mineralized NO\(_3^-\) at 70 days.

Phosphorus

Composts used in this study contained 0.10%, 0.19% and 0.44% total-P for the years 2008, 2009 and 2010 respectively. Depending on compost rate, P applied per hectare was 10-40kg in 2008, 19-76kg in 2009 and 44-176kg in 2010. Soil P levels measured in 2007 were very low (averaging 2.90 mg/kg (±0.22)). Phosphorus levels measured in 2011 indicated that P was still low though significantly higher in amended plots than in control for both 20 and 40Mg DM ha\(^{-1}\) rates (\(p<0.05\)) indicating that residual P was present in the soil one year after application (Figure 21). Though P was higher in the 40 rate treatment than the 20 rate, this was not statistically significant (\(\alpha=0.05\))(Figure 21).

Corn Tissue-N%

In 2010, corn tissue-N% 30 days after planting displayed a linear response to treatment within year (Figure 22). Tissue-N % was greater in 2010-amended plots than in control plots. Tissue-N % in plots which had received compost in 2009 and 2008 did not show any treatment response.
The estimated decay terms were $b_1 = 0.02 \pm 0.52$, $b_2 = -1.50 \pm 1.14$ with an estimated model slope of $a_1 = 0.07 \pm 0.01$ and an intercept term $k = 3.31\% \pm 0.07$ (Table 15). Decay estimates indicate there was a small (2% of 2010 level) N carryover observed in 2009-amended plots and no carryover observed in 2008-amended plots. In terms of the overall model, and the effect on tissue-N%, a one unit change in 2010 compost rate results in a 0.07 unit increase in tissue-N%, while a similar one unit increase in rate for 2009 and 2008 results in an increase of 0.01 and 0 respectively.

No tissue-N response to treatment was observed in 2009. Both within-year and 2008-amended plots recorded lower tissue-N values than control plots (Figure 23). Decay terms could not be determined for this year.

**FT-IR Spectroscopy in Compost Amended Soil**

Absorption spectra for compost and control soil were very different (Figure 24). The sharp peak recorded at 1027 cm$^{-1}$ (polysaccharides) was only weakly apparent in the bulk soil spectra. A broad peak recorded at 1432 cm$^{-1}$ (O-H in-plane bend of carboxylic acids, CO$_2$ stretch of carboxylates and aliphatic CH$_2$ alkanes, and C-O stretch vibration of carbonates) in the compost spectra was also absent in the control soil. Relative absorption of the band in the region 1640 cm$^{-1}$ (C=O vibration of ketones, quinones, carboxylic acids and esters, as well as C=C vibrations of aromatic components) was more pronounced in the compost than in the control soil. A weak peak in the region 1508 cm$^{-1}$ (vibration of the aromatic skeleton of lignin) was evident in the compost and largely absent in the control soil. The broad absorption band 3200-3500 cm$^{-1}$ (O-H stretching vibrations) displayed a higher relative absorption intensity in the compost compared with the control soil. Two peaks superimposed as a shoulder of the broad O-
H band at 2848\text{cm}^{-1} and 2917\text{cm}^{-1} (C-H stretch of aliphatic structures) were both absent in the control spectra.

Absorption peaks typical of the compost spectra were evident in compost-treated soils, even 3 years after initial application (Figures 25 & 26). The pronounced compost polysaccharide peak at 1027\text{cm}^{-1} was evident in all compost-treated soils and showed reduced intensity as time elapsed since treatment increased. The 2010 treated soil recorded a stronger polysaccharide absorption intensity than both 2009 and 2008-treated soils, which recorded weaker but very similar intensities. Absorption in the region of 1440\text{cm}^{-1} (carboxylic and carbonyl groups) was weaker than compost in the treatment soils, with 2010 and 2008-treated soils showing stronger absorption than both 2009 and control soils. The two sharp peaks seen in the region 2900\text{cm}^{-1} (C-H stretch of aliphatic structures) in the compost spectra, were absent in all treated soils.

Analysis of variance of polysaccharide peak areas determined that the control treatment had a significantly reduced polysaccharide peak area than all treatment plots, regardless of which year those treatments had been applied (p<0.05)(Figure 27). There was no difference in peak area between treatment plots, regardless of year applied (p<0.05).

Discussion

Compost addition to soil promotes SOM accumulation, which in turn affects crop growth and yield through a range of nutrient and non-nutritive properties and processes. While nutritive components such as N, P, and K are a function of microbially-mediated mineralization dynamics, non-nutritive components are more directly attributable to soil physical and chemical properties, affected largely by SOM accumulation and residence in the soil. This study modeled nutrient components of carryover such as N and P, and investigated SOM, a key indicator of non-nutritive carryover, by FT-IR spectroscopic analysis.
Nitrogen

Compost at the rates applied did not result in sufficient NO\textsubscript{3}\textsuperscript{-} mineralization, particularly in corn, for maximum yields to be realized (see Chapter II). Over the course of the study, soil NO\textsubscript{3}\textsuperscript{-} levels were low, and additions of compost at any of the treatment rates did not increase NO\textsubscript{3}\textsuperscript{-} levels beyond the year of application. Overall, 2011 NO\textsubscript{3}\textsuperscript{-} levels were significantly higher than those of 2009 or 2010 for both corn and squash plots (p<0.05) (Figures 12 & 14). However these levels were still below what is typically deemed adequate for corn and squash crops. In corn production it is recommended to apply fertilizer if pre-sidedress soil tests show NO\textsubscript{3}\textsuperscript{-} levels below 25 mg/kg (Heckman et al. 1995; Zebarth et al. 2001). Average NO\textsubscript{3}\textsuperscript{-} levels in our corn plots peaked in 2011 at only 9.98mg/kg. Given these levels, a conventional grower would certainly apply N. Davis and Westfall (2009) suggest a fertilizer application rate of 151kg/ha (135lb/ac) for soil NO\textsubscript{3}\textsuperscript{-} in this range (7-12mg/kg NO\textsubscript{3}\textsuperscript{-} and 1.1-2.0% OM).

Nitrogen mineralization may have also been influenced, to some degree, by the maturity and composition of the compost each year, as well as spring-time temperatures and moisture. Though the composts used each year were purchased from the same facility, they varied somewhat in their chemical composition and degree of maturity (Table 2). The 2010 compost in particular was lower in total N and much lower in NO\textsubscript{3}\textsuperscript{-} than the composts of 2008 and 2009 and had an NH\textsubscript{4}\textsuperscript{+} content which was considerably higher. These factors may have influenced early spring N-mineralization. The cold spring of 2010 may have slowed nitrification, retaining N in NH\textsubscript{4}\textsuperscript{+} form. Brady and Weil (2002), note that ammonia volatilization is more pronounced at high pH and where there are high levels of NH\textsubscript{4}\textsuperscript{+} in the system, and as a result ammonium losses from calcareous soils can be quite large. It is also likely that the particularly wet 2010 spring resulted in a significant proportion of soil NO\textsubscript{3}\textsuperscript{-} leaching and/or undergoing
denitrification, and being lost from the soil as NO, N₂O, and N₂ gasses. Robertson and Groffman (2007) note that denitrification typically starts to occur at water-filled pore space concentrations of 60% and higher, conditions which were quite possible given the wet spring conditions in 2010. Whatever the cause, it is clear that the crop grown in 2010 were N-deficient.

The 2011 year was the only year that exhibited a NO₃⁻ carryover response, with all treatments displaying virtually the same carryover regardless of which year they were treated. Because no within-year compost response was available, as no compost was applied in 2011, the model expressed compost carryover as a proportion of first-year carryover. Model error was great, particularly as time from treatment increased, indicating that there was little difference between the terms. Squash and corn followed a similar trend whereby the carryover coefficients were similar regardless of whether compost had been applied one, two or three years previously. Squash terms were \([d_2 = 1.17] \text{ and } [d_3 \times d_2 = 0.72]\), while corn terms were \([d_2 = 0.61] \text{ and } [d_3 \times d_2 = 0.50]\) (Table 12). Decay ratio terms indicated that NO₃⁻ response two or three years after application was similar to the response seen one year after application. This differs from what we typically find reported in the literature, where studies often describe a steadily decreasing N mineralization trend over time. However, the 2011 year was somewhat of an anomaly and results contrasted with those seen in 2009 and 2010 where no N-carryover was seen. It is therefore possible that we are seeing a response to something other than N in 2011.

In the fall of 2010 it was decided to extend the experiment an additional year, and to plant a hairy vetch cover crop (V. villosa) to overwinter and boost fertility the following spring. Undersander et al. (1990) noted that a well nodulated hairy vetch crop can enrich the soil with 60-120lb/acre of nitrogen (67-134 kg ha⁻¹). Vetch biomass was not recorded in 2011 so it is unclear how much N was added to the system upon incorporation. However, cover crop
biomass collected in 2009 and 2010 showed a linear response to compost treatment, so it is likely that the vetch responded to treatment similarly. The higher NO$_3^-$ levels in 2011 are therefore more likely a response to the legume cover crop planted in the fall of 2010 and allowed to overwinter. Magdoff (1991) noted that a vetch cover crop could provide as much as 80% of the N required by a corn plant over the growing season. Although the vetch crop in our study had not reached its peak biomass before being incorporated, it is clear that it provided a much needed boost to soil NO$_3^-$ reserves. It is likely that the rate response seen in 2011 was due to differing amounts of biomass incorporated as the vetch responded to the compost treatment.

NO$_3^-$ results from 2009 and 2010 are similar in that they both show a within-year response to compost application. However, in both of these years no carryover is apparent in plots amended in previous years. This is true for both corn and squash. In 2010, even though compost had been applied 40 days previously, soil NO$_3^-$ levels were below adequate and the slope of the within-year response was weak ($a_1=0.05$ and $a_1=0.06$ for corn and squash, respectively). Given that total N applied was in the range 160-640 kg N ha$^{-1}$, depending on rate applied, it appears that N-mineralization was low. This is not surprising given the spring weather in 2010. The spring was both colder and wetter than in previous years and cool soil temperatures would certainly have slowed nitrification rates. The 2009 season showed a strong within-year response with slope estimates of $a_1=0.15$ and $a_1=0.16$ indicating that a one unit increase in compost rate resulted in a 0.15 and 0.16 mg/kg increase in soil NO$_3^-$ for corn and squash, respectively. These within-year responses compare favorably with reports in the literature. Klausner et al. (1994) estimated a within-year coefficient of 0.16, Hadas et al. (1996) reported 0.26, and between 0.05-0.15 by Amlinger et al. (2003).
Other factors may have also influenced N mineralization dynamics. Soils were measured at 0-30cm depth, and perhaps a stronger response would have been evident if 0-15cm samples were measured as these may have more closely matched the tillage incorporation zone. Also it is possible that the difference in N response seen in corn and squash was due in part to the black plastic mulch used in the squash plots. This mulch may have influenced mineralization by increasing soil temperatures early in the season though we did not measure this.

**Soil Incubation**

A 70-day incubation conducted at the end of 2011 showed no residual NO$_3^-$ mineralization from compost applied in previous years. Interestingly mineralization in control plots was comparable to that of treatment plots. The laboratory incubation results are in agreement with our field results, and support our finding that no residual N carryover was apparent beyond the season of compost application. Given these findings, it is unlikely that the persistent yield response to compost described in Chapter II can be attributed to N carryover exclusively.

Phosphorus carryover could explain some of the yield response which occurred in the absence of N-carryover. Compost is a significant source of P and composts used in this study contained 1000, 1900 and 4300mg/kg total-P for the years 2008, 2009 and 2010 respectively(Table 2). Depending on compost rate, P applied per hectare was 10-40kg in 2008, 19-76kg in 2009 and 44-176kg in 2010. Soil P levels measured in 2007 were very low (averaging 2.90 mg/kg (±0.22)), so it is likely that any additional P in the system could have produced a yield response. Phosphorus levels measured in 2011 were significantly higher in amended plots than in control for both 20 and 40Mg DM ha$^{-1}$ rates (p<0.05) indicating that residual P was present in
the soil one year after application. Though P was higher in the 40 rate treatment than the 20 rate, this was not statistically significant ($\alpha=0.05$) (Figure 21).

In addition to nutrient carryover, non-nutritive carryover could also be contributing to the yield responses described in Chapter II. Yield increases ranging from 5 and 25% have been attributed to non-N sources (Magdoff and Amadon 1980; Schroder and Dilz 1987). Non-nutritive carryover is profoundly influenced by SOM. It is clear that despite comprising less than 5% of a typical soil, SOM exerts a disproportionally large influence on soil properties (Wagner and Wolf 1999). Following compost application, SOM displays enrichment corresponding to the original composition of the compost amendment. This enrichment can be investigated using infrared spectroscopy. In this study FT-IR spectroscopy was used to investigate SOM which is a key indicator of non-nutritive benefits.

**FT-IR Spectroscopy**

Compost spectra displayed a pronounced polysaccharide peak which was evident in soil even 3 years after incorporation. Leifeld et al. (2002) noted that polysaccharides present in compost are primarily of plant and microbial origin. Compost polysaccharides can be categorized as being either structural or storage in terms of their chemical composition, with structural polysaccharides, such as cellulose, being more resistant to decay than storage polysaccharides, such as starch. The composition of polysaccharides in our compost was not tested, however because our compost was plant derived it is likely that it contained a large proportion of cellulose, and that the more labile polysaccharides were rapidly decomposed within the growing season. Leifeld et al. (2002) noted that changes in the SOM composition after the addition of compost are characterized mainly by an ongoing decay of the easily degradable OM from the compost, which is mostly polysaccharide and, to a considerable
degree, composed of microbial biomass. It is clear that compost persists in soil many years after incorporation contributing distinct functional groups to SOM. Further work is required, however, to determine how these functional groups influence compost carryover.

**Conclusion**

Compost carryover is most commonly described in terms of N-carryover, however other aspects of carryover should also be considered when assessing the benefits of compost use. In this study, compost N-mineralization was evident only within the season of application and was variable dependent upon seasonal variations as well as differences in compost composition. Overall, compost was not an adequate source of NO₃⁻-N for maximum crop growth, and deficiencies were apparent. However, compost did provide a ready source of available P which was still evident in the soil one year after application. Infrared absorbance of compost polysaccharides provided insight into how compost alters the composition of SOM, and indicated that compost persists in the soil at least three years after incorporation. Further work is required to determine the role of polysaccharides in compost carryover.

**References**


In: Magdoff F, and Weil RR (ed) Soil organic matter in sustainable agriculture, CRC Press, Boca Raton, FL

Table 12 2011 Soil nitrate: carryover estimates. Where soil NO$_3^-$ is described by the function: $N = k + a_1 ([u2010] + (d_2 \times u2009) + (d_3 \times d_2 \times u2008])$. The $(d_x)$ terms represent decay as a ratio of first-year decay ($b_x/b_1$). Terms are reported by crop with accompanying standard errors ($\pm$). Two-year ratio ($d_2$), three-year ratio ($d_3$), slope ($a_1$) and intercept ($k$) in (mg kg$^{-1}$), are reported

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept</th>
<th>Slope</th>
<th>$(b_2/b_1)$</th>
<th>$d_2$</th>
<th>$d_3$</th>
<th>$(b_3/b_1)$</th>
<th>$d_2 \times d_3$</th>
</tr>
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<tbody>
<tr>
<td>squash</td>
<td>9.44 (±1.20)</td>
<td>0.15 (±0.06)</td>
<td>1.17 (±0.40)</td>
<td>0.62 (±0.29)</td>
<td>0.72</td>
<td></td>
<td></td>
</tr>
<tr>
<td>corn</td>
<td>7.91 (±0.63)</td>
<td>0.09 (±0.03)</td>
<td>0.61 (±0.29)</td>
<td>0.82 (±0.67)</td>
<td>0.50</td>
<td></td>
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</tbody>
</table>
Table 13 2010 Soil nitrate: carryover estimates. Where soil NO$_3^-$ is described by the function: N = k + [(a$_1$ × u2010) + (a$_1$ × b$_1$ × u2009) + (a$_1$ × b$_2$ × u2008)]. Terms are reported by crop with accompanying standard errors (S.E.). One-year decay term (b$_1$), two-year decay term (b$_2$), slope (a$_1$) and intercept (k) in (mg kg$^{-1}$), are reported

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept</th>
<th>Slope</th>
<th>1 year carryover</th>
<th>2 year carryover</th>
</tr>
</thead>
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<tr>
<td></td>
<td>k</td>
<td>S.E.</td>
<td>a$_1$ S.E. b$_1$ S.E.</td>
<td>b$_2$ S.E.</td>
</tr>
<tr>
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<tr>
<td>corn</td>
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<td>0.46</td>
<td>0.05  0.02 0.58 0.35</td>
<td>-0.48 0.53</td>
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</table>
Table 14  2009 Soil nitrate: carryover estimates. Where soil NO$_3^-$ is described by the function: 

\[ N = k + (a_1 \times u2009) + (a_1 \times b_1 \times u2008) \]. Terms are reported by crop with accompanying standard errors (S.E.). One-year decay term ($b_1$), slope ($a_1$) and intercept ($k$) in (mg kg$^{-1}$), are reported

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept</th>
<th>Slope</th>
<th>1 year carryover</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>k</td>
<td>S.E.</td>
<td>a$_1$</td>
</tr>
<tr>
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<tr>
<td>corn</td>
<td>3.64</td>
<td>0.28</td>
<td>0.15</td>
</tr>
</tbody>
</table>
Table 15 Corn tissue-N% carryover. Decay series estimates of carryover for all years. Where tissue-N is described by the fertility function: \( N = k + [(a_1 \times u_{2010}) + (a_1 \times b_1 \times u_{2009}) + (a_1 \times b_2 \times u_{2008})]. \) Terms are reported with accompanying standard errors (S.E.). One-year decay term \((b_1)\), two-year decay term \((b_2)\), slope \((a_1)\) and intercept \((k)\) in (mg kg\(^{-1}\)), are reported

<table>
<thead>
<tr>
<th>Year</th>
<th>Intercept</th>
<th>S.E.</th>
<th>Slope</th>
<th>1 year carryover</th>
<th>2 year carryover</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( k )</td>
<td>S.E.</td>
<td>( a_1 )</td>
<td>S.E.</td>
<td>( b_1 )</td>
</tr>
<tr>
<td>2010</td>
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<td>0.07</td>
<td>0.01</td>
<td>0.003</td>
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</tr>
<tr>
<td>2009</td>
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<td>0.04</td>
<td>-0.01</td>
<td>0.002</td>
<td>0.20</td>
</tr>
</tbody>
</table>

*There was no 2\(^{nd}\) year carryover in 2009.*
Figure 11 2011 Soil nitrate in squash plots. Regression models of nitrate response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.199x + 8.11$ ($r^2=0.78$), 2009-amended plots $y=0.226x + 8.11$ ($r^2=0.61$), and 2008-amended plots $y=0.130x + 8.11$ ($r^2=0.17$). Error bars indicate ± standard errors. When years are combined, soil NO$_3^-_{\text{e}}$ for the 40 Mg DM Ha$^{-1}$ treatment is significantly greater than control ($p=0.033$)
**Figure 12** Soil nitrate in squash plots in July 2009, 2010 and 2011. Nitrate data (mg/kg) has been averaged across rate for each year. Nitrate levels in 2011 plots were significantly greater than in 2010 or 2009 plots (p<0.05). Error bars indicate ± standard errors.
Figure 13 2011 Soil nitrate in corn plots. Regression models of nitrate response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.130x + 6.79$ ($r^2=0.41$), 2009-amended plots $y=0.096x + 6.79$ ($r^2=0.81$), and 2008-amended plots $y=0.114x + 6.79$ ($r^2=0.66$). Error bars indicate $\pm$ standard errors. Two statistical outliers were identified and removed (plots 124 and 130). When years are combined, soil NO$_3$ for the 40 Mg DM ha$^{-1}$ treatment is significantly greater than control ($p=0.06$) ($\alpha=0.10$).
Figure 14 Soil nitrate in corn plots in July 2009, 2010 and 2011. Nitrate data (mg/kg) has been averaged across rate for each year. Nitrate levels in 2011 plots were significantly greater than in 2010 or 2009 plots (p<0.05). Error bars indicate ± standard errors.
Figure 15: 2010 Soil nitrate in squash plots. Regression models of NO$_3$⁻ response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.093x + 6.73$ ($r^2=0.16$), 2009-amended plots $y=0.016x + 6.73$ ($r^2=0.04$), and 2008-amended plots $y=-0.007x + 6.73$ ($r^2=0.12$). Error bars indicate ± standard errors.
Figure 16 2010 Soil nitrate in corn plots. Regression models of NO$_3^-$ response to compost applied in 2010, 2009, and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.045x + 4.51$ ($r^2=0.90$), 2009-amended plots $y=0.028x + 4.51$ ($r^2=0.25$), and 2008-amended plots $y=-0.029x + 4.51$ ($r^2=0.06$). Error bars indicate ± standard errors.
Figure 17 2009 Soil nitrate in squash plots. Regression models of NO$_3^-$ response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots $y=0.162x + 5.02$ ($r^2=0.57$), and 2008-amended plots $y=-0.019x + 5.002$ ($r^2=0.18$). Error bars indicate ± standard errors.
**Figure 18** 2009 Soil nitrate in corn plots. Regression models of NO$_3^-$ response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots $y=0.149x + 3.47$ ($r^2=0.78$), and 2008-amended plots $y=0.001x + 3.47$ ($r^2=0.01$). Error bars indicate ± standard errors. One statistical outlier was identified and removed (plot 116).
Figure 19 70 Day soil incubation (20Mg DM ha\(^{-1}\) rate). Incubation of (a) Corn and (b) Squash soils. Error bars indicate ± standard errors.
Figure 20 70 Day soil incubation (40Mg DM ha\(^{-1}\) rate). Incubation of (a) Corn and (b) Squash soils. Error bars indicate ± standard errors.
**Figure 21** 2011 Soil phosphorus. Soils (0-30cm) were sampled in July 2011. Soil P (mg/kg) for 2010 treatments rates of 20 and 40 Mg DM ha$^{-1}$ was significantly higher than control plots ($p<0.05$). Error bars indicate ± standard errors.
Figure 22 2010 Corn tissue-N% at 30 days. Regression models of tissue-N% response to compost applied in 2010, 2009 and 2008. Data points are the average for each compost level. Regression models are: 2010-amended plots $y=0.006x + 3.33$ ($r^2=0.27$), 2009-amended plots $y=-0.0005x + 3.33$ ($r^2=0.02$), and 2008-amended plots $y=-0.010x + 3.33$ ($r^2=0.78$). Error bars indicate ± standard errors.
Figure 23 2009 Corn tissue-N% at 30 days. Regression models of tissue-N% response to compost applied in 2009 and 2008. Data points are the average for each compost level. Regression models are: 2009-amended plots $y = -0.010x + 2.12$ ($r^2 = 0.24$), and 2008-amended plots $y = -0.003x + 2.12$ ($r^2 = 0.08$). Error bars indicate ± standard errors. One statistical outlier was identified and removed (plot 139).
Figure 24 FT-IR absorbance spectra of compost and control soil. Compost (blue) and control (red) spectra are shown on a common scale and peaks are labeled with their respective wavelengths for clarity.
Figure 25 FT-IR absorption spectra of compost, treatment and control soils. Compost (purple), 2010-amended soil (green), 2009-amended soil (blue), 2008-amended soil (pink), and control soil (red). Spectra shown on a common scale and spread for visual clarity. Peaks are labeled with their respective wavelengths.
Figure 26 FT-IR absorbance spectra of treatment and control soils. Soils were amended with 40 Mg DM ha$^{-1}$ compost in 2010, 2009 and 2008. Spectra are shown on a common scale.
**Figure 27** Polysaccharide IR peak area. Areas of FT-IR spectra for 40 Mg DM ha$^{-1}$ and control treatment. Treatment peak area was significantly greater than control, regardless of which year the treatment was applied ($p<0.05$). Error bars indicate ± standard errors.
CHAPTER IV

GENERAL CONCLUSIONS

Managing soil fertility remains one of the toughest challenges on organic farms. Although compost has the ability to provide plant nutrients as well as improve soil quality, current methods for determining compost application rates remain tied to conventional farming practices. Alternative management strategies that account for more than simply the nitrogen value of compost and which consider the benefits of compost use over longer time frames may provide growers with a more targeted and cost effective fertility plan.

The benefits of compost use are clear. Improvements to crop yield can be realized, even many years after an initial one-time compost application. Soil nitrate-N benefits from compost amendment within season, while other plant nutrients, such as phosphorus, improve fertility over multiple years. Non-nutritive benefits of compost can potentially influence crop yields for many years. Where crop rotations include a high value crop, such as a vegetable crop, compost use can make economic sense, and can supplement N-fertility measures such as green legume cover cropping, by closing nutrient and organic matter loops.

It was apparent in this study that maximum yield targets did not represent the ideal management goal. Rather, attention to identifying an economically optimal yield may be more relevant in organically managed systems. Although a linear yield response to compost was seen we identified the most economically optimal compost rate as one which was less than the maximum rate and therefore resulted in a less-than-maximum yield. In organic farming systems, the high value nature of compost amendments necessitates a realization that maximum yields do not always provide the greatest economic returns. At some point the cost of each additional unit of compost will exceed the revenue gained from the additional yield. This illustrates the
importance of focusing compost management strategies on realizing economically optimal yields rather than simply aiming for maximum yields targets. We found that crop rotations were an important determinate of overall economic viability. To realize economically optimal compost use, the value of the crop needs to be considered. Having a high value crop within the rotation allowed compost to be applied economically within that season.

This study highlighted the importance of maintaining adequate N fertility in an organic farming system. Nitrogen contributions from compost additions were only evident within the year of application, with no NO$_3^-$ carryover evident in years following. While compost amendments are clearly valuable for their macro and micro-nutrient contribution, they should not be used in isolation. Rather, they should form part of a broader nutrient management strategy which includes N-fixing legume green manures in rotation with other cover/catch crops to improve and maintain N fertility. In short season regions, such as the Inter-Mountain West, crop rotations should be carefully considered so that adequate time is allowed for the production of these N fertility crops.

FT-IR spectroscopy was used to conclude that compost was evident in the soil, as seen by polysaccharide absorption spectra, up to three years after it was initially applied. More work is required, however, to ascertain how much, if any, of this spectral signal was due to the compost or was a result of increased plant biomass returned to the soil in compost-amended plots. Additionally the relationship between these spectral signals and compost carryover needs to be further explored.

Further work is required to partition compost carryover into nutrient, N and P, and non-nutrient effects so that the true value of compost can be more accurately understood. Likewise, our economic analysis would be improved by further work to identify crop yield plateaus and
maximum yields in response to compost treatment. From this, marginal rates of return could more accurately be measured and the economically optimal rate of compost more fully understood.

The results of this study have implications for compost use on organically certified farms by changing the way compost is viewed as a source of plant nutrients, as well as the time frames over which compost is regarded as being beneficial to crops. When compost carryover is considered, growers can more accurately determine the role of compost in their fertility program, and manage its use to optimize economic return.
APPENDIXES
APPENDIX I

PARTICULATE ORGANIC MATTER AS A SINK FOR COMPOST ORGANIC MATTER AND A DETERMINANT OF SHORT-TERM NUTRIENT CARRYOVER

Soil Organic Matter

Soil organic matter is comprised of pools of organic matter that can be classified according to their rate of decomposition and chemical transformation. The active, or labile, pool consists of material in transition between fresh plant residues and stabilized organic matter (Haynes 2005). It has a relatively high average C/N ratio (about 15 to 30) and a short turnover time (less than 10 years) (Janzen et al. 1997). Components include the living microbial biomass, particulate organic matter (POM) that has not been physically protected by aggregates, amino sugars, and other nonhumic substances (Brady and Weil 2002). The most active fraction of SOM largely controls the accumulation of soil C and N, mineralization of N, P and S and the formation of stable aggregates among other important soil processes (Lynch et al. 2005). The passive, or recalcitrant, pool is composed of organic materials that are highly resistant to microbial decomposition because of their chemical structure and/or their association with soil minerals (Haynes 2005). These materials are so resistant to decay that they often remain in the soil for hundreds or even thousands of years. This fraction includes most of the humus physically protected in clay-humus complexes, most of the humin, and much of the humic acids and generally accounts for 60 to 90% of organic matter in the soil (Brady and Weil 2002).

Plant Nutrients and the Active Fraction of SOM

Nearly all of the nitrogen and large proportions of the phosphorus and sulfur found in soils occur as constituents of SOM. The SOM serves as both the principal long-term storage
medium and as the primary short-term source of these and other nutrients (Magdoff and Weil 2004). In terms of N availability for plant needs, the active fraction of SOM is a particularly significant source in the short term. This fraction is typically has turnover times of days to weeks and is dependent on the quality and quantity of the plant residues that it is its source.

In their study of predictors of gross N mineralization and immobilization, Herrmann A M and Witter E (2008) found that gross N mineralization rates were linearly related to the soil organic N content, and that gross N mineralization was not a constant fraction of soil N, rather it was influenced by a range of factors, including soil texture. In evaluating gross N mineralization consideration must be given to the C:N ratio of recently added amendments. Where the C:N ratio is larger than 40:1 immobilization of N usually results as soil microbes use inorganic N from the soil as they decompose the residues. Thus any gross N mineralization calculations must include the potential N immobilization resulting from wide C:N ratios of added residues. The active POM fraction is the site for potential N immobilization and because this fraction is very responsive to management, the C:N ratio of amendments, crop residues and other plant based materials returned to the soil must be considered before an accurate assessment of plant nutrient availability can be made. Although all plant based amendments serve as a source of plant nutrients, the time frame in which these nutrients are mineralized varies widely. The life cycle of the crop and its resulting nutrient demands over time may affect decisions concerning what type of amendment is added to the soil. This is especially true in an organic farming system where there is a greater use of on organic inputs.

Researchers in agricultural settings are most often concerned with the labile pool because it is the site of short term nutrient mineralization and is most responsive to management. This pool is often studied by fractionating it into its constituent parts in order to
better understand their functional dynamics and responses to management. Fractions such as microbial biomass C, soluble substrate C, and particulate organic matter (POM) are often described (Haynes 2005). These selected organic matter fractions have potential as stewardship indices because their characteristics reveal the effects of management accrued over 5-20 years (Wander and Drinkwater 2000).

Particulate organic matter (POM) is composed primarily of plant-derived remains with recognizable cell structure and typically includes fungal spores, hyphae, and charcoal (Spycher et al. 1983; Molloy and Speir 1977; Waters and Oades 1991; Ellert and Gregorich 1996). POM has also been defined as the organic C and N content of primary soil particles in the 53-2000 μm size class (Willson et al. 2001).

The importance of the POM fraction has been recognized recently, with much being written about its value as a measure of labile SOM and its role in nutrient mineralization and C dynamics in the short term. Wander (2004) noted that labile SOM can be assessed effectively by characterizing POM fractions and that the focus on POM, in lieu of other measures of labile SOM, is warranted largely because the POM fraction has a higher proportional response to management than other measures of labile SOM. Christensen (1992) also recognized the strong influence that soil management has on POM.

**Agronomic Management: Effects on POM**

Many land use and agricultural practices affect SOM type and quantity in soils. Crop-rotations, tillage and organic amendments can affect SOM by influencing the quantity and quality of residues that are returned to the soil as well as influencing the rate of decomposition of added residues and native SOM (Gregorich et al. 1995; Haynes and Beare 1996). The active
fraction can be readily increased by the addition of fresh plant and animal residues, but it is also very readily lost when such additions are reduced or tillage is intensified (Brady and Weil 2002).

Because POM is comprised predominantly of plant derived residues, it is most readily influenced when plant residues and other plant derived amendments are added to the soil. For example, additions of plant-derived compost and incorporating green leguminous cover crops are management techniques that both serve increase the size of the POM fraction. Willson et al. (2001) found that the greatest changes in POM were associated with compost inputs and reasoned that this was due to the POM content of the compost itself. Fortuna et al. (2003) noted that 85% of the C in their compost material (diary manure/oak leaf mix) was classified as POM. Nitrogen dynamics of the POM fraction are also affected by compost applications with Willson et al. (2001) noting that 41% of the total N content of compost in their study was contained in the POM fraction and that compost additions increased the POM-C by 25% in the year of application.

SOM is mutable and very responsive to external influences; many indices of soil quality are fixed, but SOM can be altered, particularly in agro-ecosystems (Janzen et al. 1997). Within the SOM pools, the recalcitrant fraction is least subject to alteration by management in agronomic systems. This fraction consists of inert organic matter that is highly resistant to biological oxidation because of its molecular structure or physical protection, resulting in turnover times that are measured in 100’s, if not 1000’s of years (Janzen et al. 1997). The active fraction, on the other hand, is very responsive to alteration by management. Given that this fraction forms the bulk of the plant available mineralizable nutrients, with turnover times of less than a decade, an understanding of the effects of management on this fraction becomes important in an agronomic setting.
Cultivation of topsoils significantly alters the balance of carbon inputs and outputs in a soil. The OM content of cultivated soils is typically about 15 to 30% lower than that of soils under native vegetation (McGill et al. 1998; Anderson 1995; Gregorich et al. 1995; Gregorich and Ellert 1995). This difference is primarily the result of enhanced microbial respiration immediately following cultivation. Because no inputs of carbon accompany this tillage, carbon stores in the soil are diminished. Losses of SOM after cultivation decline over time as the soil ecosystem once again returns to a C/Respiration steady state (often within a few decades) (Janzen et al. 1997). Carbon is also lost from the system when crop biomass is removed from the field, as in the case of a wheat harvest.

Alternatively, any practices that favor C input can positively affect the accumulation of SOM. Reduced tillage and no-till practices have been shown to lead to net accumulation of SOM. For example, Campbell et al. (1995) measured the SOM response to adoption of reduced tillage in a soil that had previously been under a tilled-fallow wheat system for 70 to 80 years. In combination with continuous cropping and enhanced fertilization, the reduced tillage increased the OM content of the 0 to 15cm soil layer by several Mg C ha\(^{-1}\), relative to an estimate of C content at the beginning of the study. Additions of organic amendments, judicious crop rotations, and cover crops/green manures have also been shown to return C, thereby improving SOM stores in the soil system. Because much of the change in SOM occurs in fractions with short turnover times, SOM changes can occur relatively quickly; adoption of revised cropping systems can often measurably benefit SOM content within a few years (Janzen et al. 1997).

**POM: A Source of Plant Available N**

The POM fraction, with a particle size ranging between 50 and 2000 \(\mu\)m diameter, has been linked to short-term nutrient availability (Mapfumo et al. 2007). The readily accessible C
and N in this fraction promote population growth of soil microbes. When C:N ratios are low excess mineralized N is released into the soil solution thereby becoming available for plant uptake. As the microbial population ebbs and flows in response to available substrate, microbial biomass itself is decomposed adding to the pool of plant available N. The POM fraction, with its rapid turnover times, provides a more readily accessible source of energy for the saprotrophic soil organisms responsible for nutrient cycling (Janzen et al. 1992). It is this high turnover rate of readily decomposable plant residues that makes POM an important source of plant nutrients.

The POM fraction is thought to incorporate much of the ‘organic fertilizer property of SOM’ (Swift and Woomer 1993). Not only is the POM fraction an important source of plant available nutrients in the short term, but it can also be used as a predictor of N mineralization as much of the N available for short term mineralization is contained within this fraction. In their study of the biologically active fractions of organic matter, Willson et al. (2001) suggested that POM was a reliable indicator of N mineralization potential (NMP), especially when information about the previous year’s crop production and cover crop inputs are considered. They described a strong correlation between POM and N mineralization in 70 and 150 day aerobic incubations and noted that N mineralization was higher in the months immediately following incorporation of plant residues into the soil, suggesting to them that POM can be used as a reliable estimate of N mineralization. It is not surprising that POM is such an important source of C and N in the short term given the proportion of total soil C and N contained within the POM fraction.

Ouédraogo et al. (2006) noted that particulate organic-C accounted for 47-53% of total organic C, while particulate organic-N contributed 30-37% of total organic N in their study of a Ferric Lixisol. The composition of many soil amendments also contributes directly to the POM fraction. Willson et al. (2001) found that 43% of compost C and 33% of compost N was in the 250-
2000µm size fraction, with an additional 10% of compost C and 8% of compost N separating into the 53-250µm fraction.

The POM fraction can be further subdivided into smaller fractions, each with their own chemical and physical characteristics which affect N mineralization. Mapfumo et al. (2007) described a significant linear relationship between maize yield and the amount of mineralizable N in the macro-POM fraction (250-2000 µm) in a sandy clay loam. Mapfumo further noted that the organo-mineral POM fraction (< 53 µm) was physically protected from mineralization in these soils, however in a coarse sandy soil the organo-mineral fraction became the predominant source of mineralizable N. Soil texture then becomes a factor in nutrient release from the POM fraction. Finer textured soils are able to physically protect the very fine organo-mineral fraction of POM thereby preventing its mineralization in the short term.

**Research Objectives**

Although many studies have investigated the effect of compost on POM fractions, all have been concerned with systems where compost is applied annually. A search of the literature reveals that none describe the affect of a one-time application of compost on POM in the second, third or fourth year after application. Understanding the longer term impact of compost application on POM is important because POM plays such a central role in nutrient mineralization and SOM accumulation in the short term. Knowledge of long-term changes in POM-C can facilitate management of nutrients from organic amendments (Fortuna et al. 2003). The investigation of POM may also help us further explain the crop yield and nutrient carryover results observed in field trials.
Conclusion

An understanding of the role of POM as a source of plant available nutrients in agronomic systems is especially important, particularly because crop management decisions have such a large impact on the size, composition and mineralization rate of the POM fraction. This is especially the case in organically managed settings where the POM becomes the primary source of plant available nutrients.

References


APPENDIX II

FRACTIONATION OF PARTICULATE ORGANIC MATTER FOR FT-IR ANALYSIS

Introduction

This paper is a review of current particulate organic matter (POM) fractionation methodologies. It aims to identify techniques which may be suitable for use with organically managed soils where POM will be further analyzed by FT-IR spectroscopy.

Chemical and Physical Fractionation

The heterogeneous nature of SOM with respect to its chemical, biological and physical composition has resulted in the development of a variety of techniques to extract it from the soil and further isolate its constituent fractions. These techniques can be categorized as being either physical techniques or chemical techniques.

Chemical extraction methods give information on the kinds of organic matter present, while physical methods give information on where the organic matter is located (Elliot and Cambardella 1991). Physical fractionation methods separate soil constituents on the basis of size and/or density, while chemical fractionations distinguish SOM fractions based on their respective solubilities in various extractants, sometimes before and after selected pretreatments of the soil (Olk and Gregorich 2006). However, because chemical methods extract SOM that may be physically protected from microorganisms and therefore not readily available for decomposition, they may not provide an accurate representation of actual organic matter dynamics in soil (Elliot and Cambardella 1991; Elliot et al. 1992). Because they do not distinguish between physically inaccessible material and uncomplexed material, they cannot capture many of the insights gained through physical fractionations (Olk and Gregorich 2006). Physical
fractionation techniques can be less destructive and more selective and results obtained may relate more directly to the structure and function of SOM in situ (Christensen 1992).

Where the resulting POM fractions will be further analyzed using FTIR spectroscopy, the use of harsh reducing chemicals such as sodium iodide (NaI) should be avoided in order to minimize production of chemical artifacts. Such artifacts may give a false representation of the quality of POM within the fractions.

Although chemical extraction methods may have value in identifying kinds of organic matter they may not be useful for isolating fractions for further analysis because some components are transformed, solubilized or oxidized (Elliott and Cambardella 1991). Chemical techniques that use strong reducing agents such as NaI have the potential to create artifacts which may misrepresent the true nature of the SOM in question. In both chemical and physical fractionation, it is not always possible to discern the relationship between organic matter which is in the obtained fractions and that in the undisturbed soil (Elliott and Cambardella 1991).

Due to these concerns physical fractionation techniques may be more suitable for organically managed soils. Physical fractionation techniques are considered chemically less destructive, and the results obtained from physical soil fractions are anticipated to relate more directly to the structure and function of SOM in situ (Christensen 1992). This perhaps explains the popularity of physical extractions in recent years. Physical fractionation according to size and density of soil particles emphasizes the importance between organic and inorganic soil components in the turnover of organic matter (Christensen 2001).

**Size and Density Based Fractionation**

Techniques for extraction are generally size-based, density based, or a combination of the two. The most importantly consideration is the relevance of the resulting fractions to the
study in question. Particulate organic matter may be defined according to size (53-2000µm) or density (<2.0 g cm\(^{-3}\)), with the resulting fractions called coarse fraction (CF), and light fraction (LF) respectively. In addition, POM may be i) free (<1.6 g cm\(^{-3}\)), or without any particular association with soil minerals, or ii) occluded (1.6 – 2.4 g cm\(^{-3}\)), that is, buried within soil aggregates and/or strongly associated with mineral particles (Haynes 2005). The degree to which the soil aggregates are dispersed, either chemically, by sieving, or through sonication, influences the recovery of occluded SOM.

Density fractions are obtained by centrifugation in sequentially heavier liquids with the surface floating material being removed each time as the light fraction (Elliott and Cambardella 1991). The SOM is separated according to age, with the youngest fraction being the lightest. A density liquid exploits the fact that POM is lighter than the average mineral density of 2.6 g cm\(^{-3}\). Typically inorganic suspensions of sodium or potassium iodide, sodium polytungstate (NaPT), or silica gels are used for density-based separations. These solutions can potentially alter the chemical characteristics of SOM fractions: iodide solutions are strong reducing agents and silica gels have a pH of 8 or more and thus can extract humic substances (Wander 2004). In addition, silica gels such as Ludox™ are limited to densities up to 1.4 g cm\(^{-3}\) (Meijboom et al. 1995). Although sodium polytungstate (NaPT) has been shown to have an inhibitory effect on microbial activity in incubation studies (Lutzow et al. 2007), and is difficult – if not impossible – to completely remove from POM (Wander 2004), it is relatively inert and more suitable for use with organically managed soils where FT-IR analysis is conducted.

Choice of density brackets is important, with even small changes of density resulting in vastly different recoveries. Most studies favor a LF critical density of 1.6 g cm\(^{-3}\) rather than 2.0 g cm\(^{-3}\) in order to obtain light fractions with reduced contents of organo-mineral complexes and
ash, and with higher contents of non-complexed macro-OM (Christensen 1992). For the purposes of this study, LF is defined as material with density <1.6 g cm\(^{-3}\) and OLF as 1.6-2.0 g cm\(^{-3}\).

Elliott and Cambardella (1991) note that densities of < 1.6 g cm\(^{-3}\) separate mainly undecomposed organic matter while density of <2.0 g cm\(^{-3}\) can be used to separate organo-mineral complexes. Material with density >2.0 g cm\(^{-3}\) is more tightly mineral-bound, more stabilized, older, and less responsive to short term management. This heavier fraction will not be investigated in this study.

Haynes (2005) found density fractionation to be more effective than particle size separation in separating the labile and non-labile OM fractions. Another advantage is that density fractionation avoids the need for solvent extraction and decreases the possibility of artifact formation (Haynes 2005). Size-based fractionations employ sieving or sedimentation methods to separate POM from the bulk soil. Size based methods are simpler with lower input requirements.

**Aggregate Dispersion Techniques**

To improve SOM recovery, soil aggregates are first dispersed before size or density based separations are performed. The degree to which the aggregates are disrupted directly affects the quantity of recoverable POM. Methods for dispersion include the use of chemical agents such as sodium hexametaphosphate, or physical disruption via shaking or ultrasonic vibration.

Chemical disruption using dispersants such as sodium hexametaphosphate has been shown to influence the chemical properties of SOM (Ahmed and Oades 1984). Christensen (1992) noted that chemical dispersion treatments were not considered feasible for isolation of intact organo-mineral complexes (such as our OFL fraction) and that for this reason most studies
on organo-mineral complexes rely on ultrasonic vibrations to accomplish soil dispersion. Because we are concerned about potentially altering the chemical composition of the POM, which would affect our FTIR analysis, soil dispersion will be achieved by physical method. Sonication rather than shaking is favored because effective dispersion by shaking is best achieved when the soil is shaken in a chemical dispersant.

This is not to say that sonication is without its share of problems, the most documented being the potential for redistribution of organic matter amongst fractions. For example, increasing the intensity of sonication can result in distribution of more organic matter into finer soil fractions compared with a lower intensity of sonication energy (Elliott and Cambardella 1991). It is therefore very important to choose a sonication energy that disrupts the aggregates only enough to recover the POM fraction in question.

Sonication imparts vibrational energy to a soil suspension, causing cavitation or the formation of microscopic bubbles which are produced by a rapid reduction in local pressures. The collapse of the bubbles produces shock waves sufficiently energetic to disrupt the bonds involved in soil aggregation (Gregorich et al. 1988). Sonication without the use of chemical treatment has been used exclusively for disrupting soil while purportedly minimizing the chemical transformations that may occur during chemical extraction (Elliott and Cambardella 1991)

**POM Fractionation Methodology for Organically Managed Soils**

Given the investigation of fractionation methodologies described above, we now identify a method most suitable for organically managed soils. This method is based on the methodology described by Marriott and Wander (2006). Briefly, size and density based separations will be used to obtain three POM fractions. These fractions are; i) coarse fraction
(CF) comprised of particles >53 µm; ii) free light fraction (FLF) (<1.6 g cm\(^{-3}\)); and iii) occluded light fraction (OLF) (1.6-2.0 g cm\(^{-3}\)).

These three POM fractions represent SOM in various states of decomposition and degrees of association with mineral particles. It is generally understood that material closely associated with residue inputs can be concentrated by focusing on the larger or lighter POM constituents (Wander 2004). The CF can be used to assess changes in total POM across treatments, while the FLF and OLF fractions can be used primarily to investigate changes in functional groups by FT-IR spectroscopy. Physical fractionation techniques are recommended with the soils dispersed via ultrasonic vibration.

It is important to consider not only free POM but also that which is associated with mineral particles if we are to make any conclusions regarding C or N transformations in soils. The biological availability of organic substrate in soil is related to the chemical quality of the organic material and to its degree of physical occlusion, which can determine the accessibility of the material to microorganisms (Elliot and Coleman 1988).

Because C:N ratios decrease and H:C ratios increase as labile constituents of SOM are lost and aromaticity increases (Wander 2004), the inclusion of a free and occluded fraction may assist us in identifying changes in functional groups two and three years after a one-time compost application. Accumulation of POM reservoirs that are partially degraded and thus have lower C:N ratios than fresh residues have been cited as evidence of SOM aggrading cropping practices (Wander and Traina 1996a).
Coarse Fraction (>53µm)

Coarse fraction POM has been shown to adequately reflect changes in POM quantity due to management (Wander 2004). This fraction could be most helpful in assessing whether a one-time application of compost increases POM fraction size one and two years after application. Wander (2004) notes that CF methods may not be as sensitive to agronomic treatments as LF methods but they have been effectively used to document changes to labile SOM arising from land use.

Occluded Light Fraction (1.6-2.0g cm⁻³)

Occluded POM has undergone more decomposition during its physical protection within aggregates (Christensen 2001). It is recovered from the soil by more thorough dispersion of aggregates than LF. This can be achieved with greater sonication energy or duration. Sohi et al. (2005) found that the intra-aggregate fraction (OLF) contained a greater proportion of functional
C groups indicative of recalcitrant or microbial C than the free light fraction. They further found that free and intra-aggregate (OLF) organic matter occupy contrasting positions in the decomposition sequence, and are likely to display reactivities sufficiently distinct to operate as discrete pools in new SOM models. This fraction, together with the free light fraction, can be investigated by using FT-IR spectroscopy to assess changes in POM functional groups POM arising from compost addition one and two years after compost application.

In their study of the properties of two SOM fractions, Sohi et al. (2005) found that the intra-aggregate light fraction (defined as <1.80 g cm\(^{-3}\) in that study) contained a greater proportion of microbial products and more resistant C as compared with the free light fraction. They concluded that free and intra-aggregate OM occupy contrasting positions within the decomposition sequence, with the free OM being younger and therefore less decomposed.

**Free light fraction (<1.6-2.0 g cm\(^{-3}\))**

Free light fraction (<1.6-2.0 g cm\(^{-3}\)) represents the fraction of OM that is neither present as readily recognizable litter components (typically >2mm) nor incorporated into organo-mineral complexes (Christensen, 2001). It consists mainly of particulate, partly decomposed plant and animal residues but can also encompass fungal hyphae, spores, faecal pellets, faunal skeletons, root fragments, and seeds (Gregorich and Janzen 1996). Free LF is a transitory pool between litter and mineral-associated OM with a turnover that is slower than that of recently shed litter but faster than that of occluded OM (Christensen 2001). Where inputs of organic manures are significant (such as plant residues, animal manure, composts, or sludge), the nature and seasonal variation in light fraction SOM may be important to carbon turnover and nutrient cycling (Christensen 1992). Free and occluded OM originate from different positions within the
soil with different exposures to microbial decomposition, differ in chemical composition and in
$^{13}$C signature, and exhibit different lability towards microbial turnover (Christensen 2001).

This fraction can be most helpful to investigate questions of short term nutrient cycling
and N mineralization. Hassink et al. (1997) notes that short term changes in SOM may be best
studied by focusing on these labile organic matter fractions. In their review of SOM
fractionation methods Lutzow et al. (2007) noted that the LF (<1.6-2.0 g cm$^3$) best represents
the active SOM with turnover rates of <10 years. Other methods for measuring and
classifying SOM, such as organic C and humic acids, may be of limited use to understand the
link between SOM dynamics and nutrient availability because they do not measure biologically
active fractions (Motavalli et al. 1994).

This labile free light fraction pool encompasses a continuum of organic matter in various
states of decomposition chemical or biological transformation and physical protection. The
labile pool has a disproportionately large effect on nutrient-supplying capacity and structural
stability of soils (Haynes and Beare 1996) and is affected rapidly by management-induced
changes in organic matter inputs or losses (Janzen et al. 1998).

References

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